Atlantic White Cedar: Ecology, Restoration, and Management

Proceedings of the Arlington Echo Symposium

Millersville, Maryland

June 2–4, 2003

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Cover photos: Aerial photos (1,000-foot altitude) of the Penn Swamp clearcut in Wharton State Forest, New Jersey, and the site of the subsequent deer-slash experiment: 1990 (top photo, as clearcut was being finished), 1995 (middle photo), and 2000 (bottom photo).

Photos and cover design by George Zimmermann.

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Preface

I am pleased to write this preface to the proceedings of the symposium on Atlantic white-cedar (AWC) held at the Arlington Echo Outdoor Education Center, Millersville, MD, in June 2003, chaired by Keith Underwood and Philip Sheridan.

The theme of the symposium was “Uniting Forces for Action,” and it was clear that the attendees were indeed united in their desire to study this globally threatened species, gain a more holistic view of the AWC ecosystem, and cooperate to insure that the scientific work so ably done here and elsewhere translates into action to restore and responsibly manage AWC.

More than 15 papers and a number of posters were presented during the symposium, and the attendees were able to view AWC sites, including a number of restoration sites. I was very impressed by the State of Maryland’s use of educational facilities such as Arlington Echo. These facilities have been used to give Maryland’s citizens (especially the children who in turn have educated and motivated their parents) a sense of their connection to the environment and the need for their help and stewardship in restoring AWC. Symposium attendees from other States can take home valuable lessons about environmental education.

Participants in the symposium came from throughout the range of white cedar, from New England to the Gulf coast. There is no doubt that this species, extirpated in many areas, has captured the attention and scrutiny of many researchers and other highly motivated individuals.

This publication includes a representative subset of the papers presented at the symposium. Laderman and Domozych’s paper expands our knowledge of other life forms inhabiting AWC habitats—a fundamental step in understanding the ecosystem as a whole. Papers by Crawford and others, Derby and Hinesley, Mylecraine and others, Hopton and Pederson, and Gengarelly and Lee give us more data on the physical aspects of white-cedar ecosystems and in some instances their interaction with the biological factors. These papers present information that will help us restore and understand AWC and their functioning.

The range, restoration, and stewardship of AWC ecosystems are discussed in papers by McCoy and Keeland, Underwood and others, and Broersma-Cole. We also present the first of numerous papers by Mylecraine and others that have given us vital information on range-wide AWC genetics, and finally a paper by Zimmermann and Mylecraine that discusses the long-term effects of various silvicultural manipulations on the entire vegetation community.

I hope that all of the papers and posters presented at the symposium will eventually find their way into the literature; they all contained information important to our understanding of the species, its continued restoration, and wise management across its range.

George Zimmermann
Professor of Environmental Studies
Richard Stockton College of New Jersey

Contents

CHAMAECYPARIS THYOIDES (ATLANTIC WHITE CEDAR) WETLANDS OF CAPE COD, MASSACHUSETTS, USA: A DESMID DIVERSITY DATABASE

Aimlee D. Laderman and David S. Domozych[1]

Abstract—We are developing a database documenting the microscopic biodiversity of all Atlantic white cedar (AWC) [*Chamaecyparis thyoides* (L.) BSP.] wetland forests. A census of the desmid algal species of seven Cape Cod, MA AWC bogs serves as prototype for the database. Desmids are microscopic unicellular or filamentous green algae (phylum Chlorophyta) in the order Zygnematales, families Mesotaeniaceae (saccoderms) and Desmidiaceae (placoderms). Over one dozen wetland specialists in the AWC biodiversity network contributed Cape Cod samples. Field material is sorted, cultured, photographed or drawn, and preserved. Data is fed into two systems: (1) an all-taxon AWC rangewide biodiversity database and Website, and (2) an all-protist global database "micro*scope". This report describes the first system. Information including images, genus, species, common names, family, and source is entered in a working Filemaker database and converted to Sequel Server format for interactive Web use. The completed database aims to document biodiversity of all taxa in all AWC wetlands across their entire range.

Keywords: Algae, Cape Cod, desmids, diversity, Massachusetts.

INTRODUCTION

The higher flora of acidic wetland forests dominated by Atlantic white cedar [*Chamaecyparis thyoides* (L.) BSP.] (AWC) have a well-defined taxonomic skewing: they contain many species and genera in a few families, but there is a startling absence of the most common plants found in surrounding areas (Laderman 1987, 1989). Little is recorded of the algae of AWC swamps: Hannah Croasdale (1935) included cedar bog sites in "The Freshwater Algae of Woods Hole"; Drouet and Cohen (1935, 1937) studied *Gonyostomum* in a Falmouth, MA, cedar swamp; some New Jersey Pine Barrens algae were described by Moul and Buell (1979). An unpublished survey of herbarium and literature records included cedar-associated algae (Lloyd and others 1980), and Laderman (1980, 1987) reported on the periphytic and planktonic microflora of a Falmouth, MA, bog.

Cedar forests are disappearing in our own time in regions where they were once plentiful. Their historic distribution is indicated in figure 1a. Some go rapidly, replaced by shopping malls, commercial cranberry beds or scenic ponds. Others go slowly, swamped by an abundance of nutrients that favor cedar's competitors. To reverse this trend of habitat loss and to restore *Chamaecyparis* forests, it is essential to improve our understanding of what makes for a healthy cedar swamp. By determining the composition of the producers of the associated microscopic community, we hope to gain insights into what makes AWC communities unique and what allows them to thrive.

Figure 2 summarizes the rigorous environmental factors that are known to affect AWC and other coastally restricted forests and act to exclude most other biota found in similar latitudes (Laderman 1998). The adaptive properties of the dominant trees of coastally restricted forests may have parallels in the microbiota.

These factors have been discussed in detail elsewhere (Laderman 1998, 2000). Figure 3 indicates proposed interrelationships of the physical, chemical and biological properties specific to *C. thyoides* wetland waters that select for extremophiles. Shallow, acid, poorly aerated waters, rich in organic acids and decayed organic matter—all characteristics of cedar bogs—are favorable habitat for an abundance of the green algal species known as desmids. Many desmids are found only in acid bogs, and figure 4 illustrates a few of these saccoderm and placoderm desmids.

This article is a report on the preparation of a database and Website of microscopic diversity in cedar wetlands using the desmid flora of seven Cape Cod bogs as prototype.

Goals of the entire project are:

- To facilitate and stimulate further biodiversity, ecological, genetic and restoration work over the entire range of the keystone canopy dominant, *C. thyoides*, in all its diverse ecotypes
- To lay the groundwork for coordination and support of research and restoration in AWC wetlands. The algae project is part two of an all-taxon rangewide survey of Atlantic white cedar wetlands. Part one, inventory of the higher flora, is now published (Laderman and Ward 1987, 1989) and online http://gosnold.mbl.edu/awc/awcflora.
- To make this information widely available to the scientific community and to the public at large

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Figure 1a—Map of *Chamaecyparis thyoides* distribution. Counties in which AWC has been found are inked in black. Compiled from field observations, herbaria, published material and personal communications (adapted from Laderman 1989).

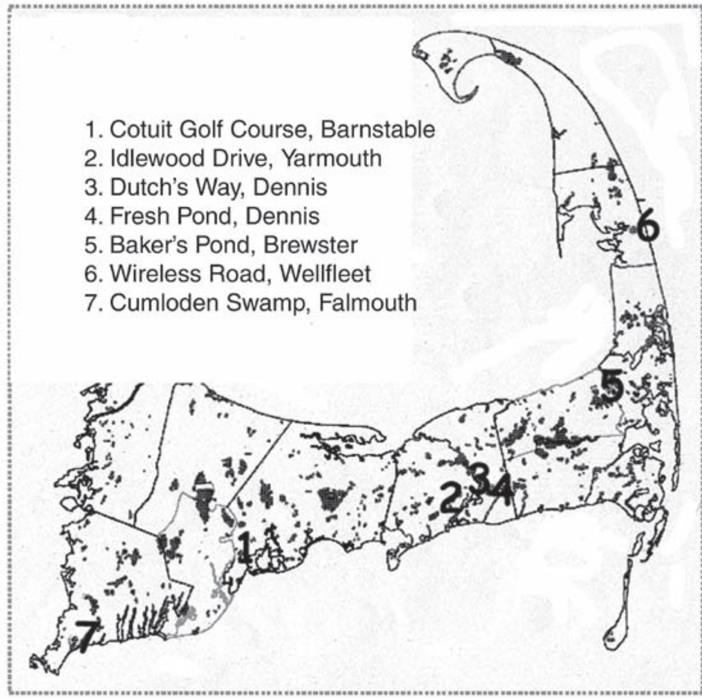

Figure 1b—Map of Cape Cod, Massachusetts, USA, indicating collection sites.

The objectives of the work reported here are:

- To catalog biodiversity of the desmid algae of Atlantic white cedar wetlands
- To analyze their taxonomic distribution
- To define a cadre of indicator species that would reflect the health of an AWC wetland
- To add to the global record of the distribution of microflora

The overall algae project has three major components: (1) field collection and morphological identification of algal flora: water, sediment, submerged vegetation and detritus are collected by a multidisciplinary network of colleagues (the Atlantic White Cedar Network [AWCnet] in AWC sites from Maine to Mississippi, and biogeochemical, ecological and geographic (GIS and USGS quad) data are gathered for each site, (2) archive of ecosystem collections in herbaria at Peabody (Yale University), Smithsonian (Washington, DC) and Gray Museum (Marine Biological Laboratory/Woods Hole Oceanographic Institution, Woods Hole, MA): dried unsorted samples from each site are prepared for ecosystem collections, (3) development of a Website for an interactive database: information for each species and site is stored in an electronic relational database designed to be part of global biodiversity records. The data will be accessible to the scientific community at the herbaria and via a searchable (databased) Website maintained at the Marine Biological Laboratory, Woods Hole, MA.

This report is restricted to the desmids of seven Cape Cod sites, indicated on the Cape Cod map in figure 1b.

PROCEDURES

The AWC biodiversity network specialists (AWCnet) collect water, moss, and detritus from cedar wetlands. After sorting, algae are cultured, photographed, preserved, identified, and this information is entered in a database that is imported into a Website linked to a global database. The AWCnet is composed of over three dozen limnologists, hydrologists, foresters, botanists and mosquito control and soils specialists in Government and private agencies covering the East Coast of the United States.

Field Collection

When possible, all samples are taken from water or saturated areas where cedar trees are the dominant or sole canopy occupant; sampling is done as close to the trees as is practical. The richest harvest is usually found in the photic zone associated with solid substrate, e.g., moss, especially *Sphagnum;* bits of detritus; cedar bark. Water levels in cedar swamps vary from year to year and with the seasons, precipitation, drawdowns from sources outside the swamp but with a contiguous water table, and tree harvest. Many algae are found even when surface water is absent. When water is present, the surface skim is collected. Fresh material is transported in simple plastic snap-tight or zippered bags, kept cool as possible, and hand delivered or shipped to the laboratory in refrigerated insulated containers by overnight mail. Moss, cedar bark, and bits of other submerged or emergent plant material, folded securely into an unsealed paper envelope and allowed to dry, yield excellent specimens. Dry material is not as sensitive to temperature and light variation and may be shipped by regular mail. Procedures may vary as field conditions dictate.

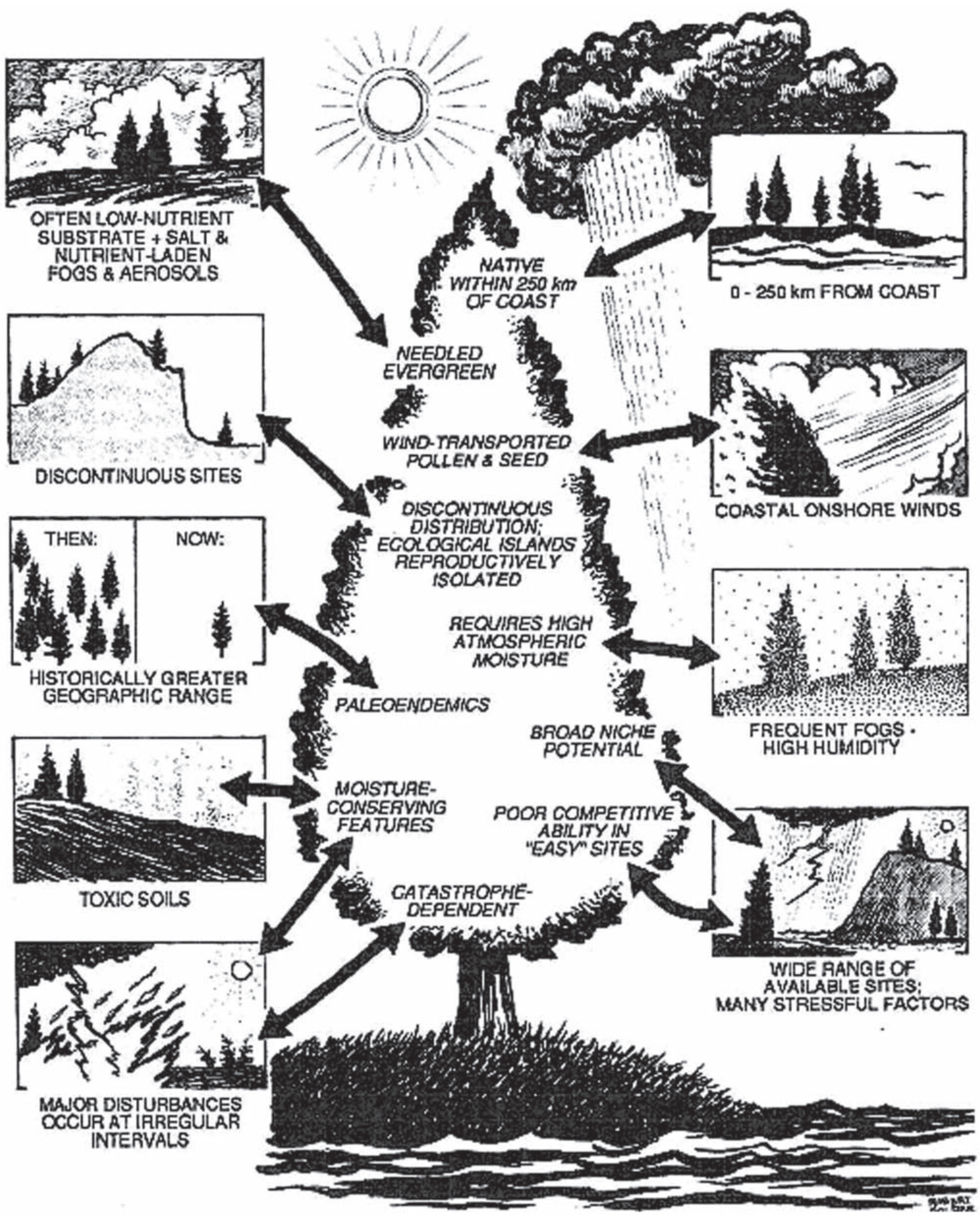

Figure 2—Cartoon illustrates the rigorous nature of the cedar forest environment that selects for extremophiles and related adaptive features of *Chamaecyparis* species (adapted from Laderman 2000).

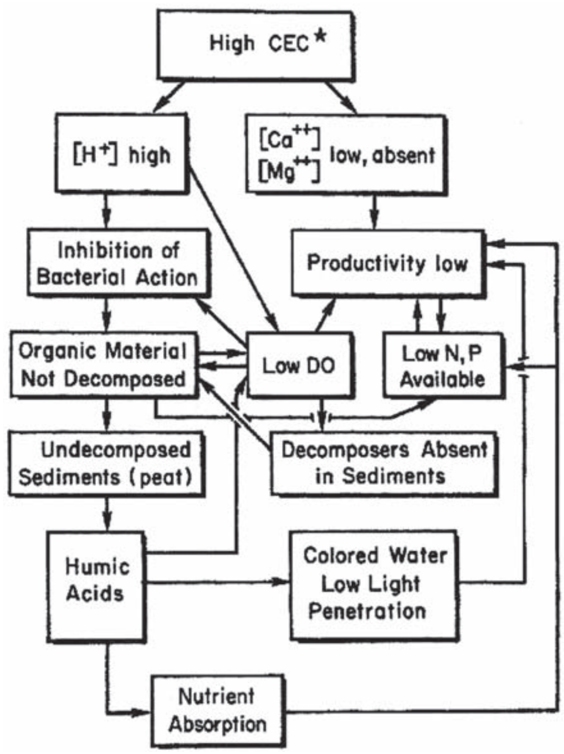

Figure 3—Cedar wetland dynamics. Flow diagram indicates proposed interrelationships of the stringent physical, chemical and biological properties of AWC wetland waters (from Laderman 1989). *CEC = Cation Exchange Capacity.

Processing, Recording, and Data Transfer

It is preferable to study living algae, but portions of each collection are also preserved in three ways: in Lugol's solution (an iodine-based algal preservative), in 70 percent ethanol, and dried in standard herbarium fashion. Samples are cultured (Baylson and others 2001) in three different media: (a) Woods Hole medium, (b) Waris medium, and (c) Desmid medium (Nichols 1973). Media (a) and (b) are supplemented with soil extract.

The photomicrography systems we use are: (a) an Olympus BX60 microscope equipped with Nomarski differential interference contrast (DIC), fluorescence and phase optics; Optronics Magnafire CCD camera, images captured using a Dell computer equipped with Image ProPlus software, and (b) an Olympus CK40 inverted culture microscope equipped with phase optics; Nikon Coolpix 995 digital camera, images captured using Dell computer equipped with Adobe® Photoshop software.

Data for each photograph is maintained in a Microsoft® Excel file recording the specimen's identity, date and site of origin, collector, location of relevant notes in the laboratory notebook, and photographic details. Records are entered in a Filemaker Pro searchable database containing taxonomic (family, genus, species), site and collection data. These data are being imported into an illustrated interactive Website http://gosnold.mbl.edu/AWC/AWCalgae now under construction using SequelServer. Our data are also being entered

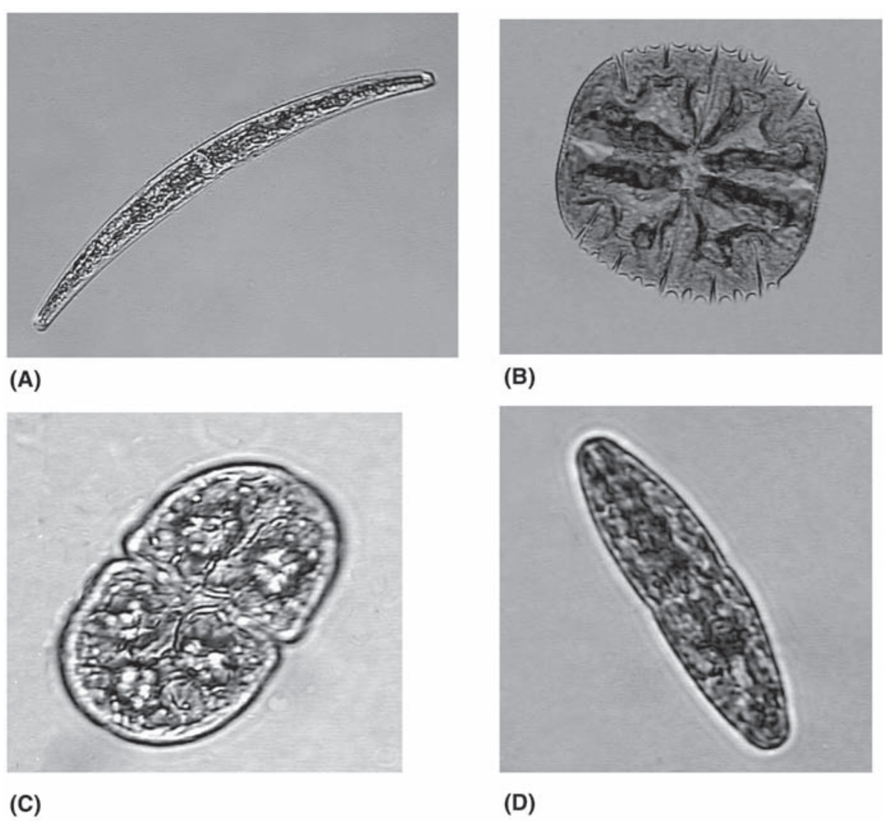

Figure 4—Photomicrographs of living desmids found in Cape Cod AWC wetlands. These algae contain bright green chloroplasts. Magnification 250x-500x. a. *Closterium* b. *Micrasterias* c. *Cosmarium* d. *Tetmemorus*.

into the queryable illustrated protistan global database "micro*scope" http://www.mbl.edu/microscope, which has a separate Atlantic white cedar wetland section. Thus information on cedar wetland microflora will soon be available worldwide online.

Taxonomy, Reproduction, Morphology

Desmids are classed in two families of the Order Zygnematales: Mesotaeniaceae (commonly: saccoderms) with no median constriction, and Desmidiaciae (placoderms) with a midline constriction dividing the cell into identical halves. A distinctive aspect of the life cycle is that there is a complete lack of flagellate stages. The Order Zygnematales usually have amoeboid reproductive cells that fuse within a conjugation tube or gelatinous matrix. They also reproduce by simple cell division or with special structures (akinetes or aplanospores). The vegetative cells may be single or arranged in a simple filament. They may be planktonic, or as epiphytes, attached to submerged or emergent vegetation or debris. This order is found only in fresh water (Bold and Wynne 1978, Pickett-Heaps 1975, Whitford and Schumacher 1973).

OBSERVATIONS

Of the 63 algal species identified in Cape Cod AWC wetlands to date, 22 are desmids (table 1). All these desmids but one, *Netrium digitus,* are in the family Desmidiaceae (placoderms). The 21 placoderms are members of 11 genera. One genus, *Staurastrum*, is represented by 5 forms in 4 species, with one species, *S. dickiei*, found in two forms. One genus, *Cosmarium*, is represented by 4 species. Three genera—*Closterium, Desmidium,* and *Xanthidium*—are represented by 3 species each, and 6 genera—*Euastrum, Hyalotheca, Micrasterias, Pleurotaenium, Spondylosium,* and *Tetmemorus*—are each represented by a single species.

Table 1—Desmid Algae of Cape Cod, Massachusetts

Chlorophyta (Green Algae)
Class Chlorophyceae
Order Zygnematales (Conjugales)
Family Mesotaeniaceae (Saccoderm desmids)
Netrium digitus
Family Desmidiaceae (Placoderm desmids)
Pleurotaenium subcoronulatum v. *detum*
Closterium sp. 1
Closterium sp. 2
Tetmemorus laevis
Euastrum sp.
Micrasterias truncata
Cosmarium margaritatum
C. pachydermum
C. raciborskii
C. trilobulatum v. *depressum*
Xanthidium antilopaeum v. *laeve*
X. cristatum v. *uncinatum*
Staurastrum cingulum v. *floridense*
S. dickiei v. *maximum*
S. dickiei v. *duospinum*
S. gracile
S. quadrispinatum v. *spicatum* f. *furcatum*
Hyalotheca dissiliens
Desmidium swartzii
Desmidium sp.
Spondylosium pulchellum

Organization generally follows Whitford and Schumacher (1973)
Supplementary sources: Bourrelly (1966) and Prescott (1978)

CONCLUSIONS

This preliminary evidence indicates that compared to other algal families, the desmids exhibit high species abundance in the cedar wetlands with very uneven taxonomic distribution (Laderman 1980, 1987). These observations are consistent with previous findings that the desmids form part of a characteristic taxocene favoring undisturbed *Sphagnum*-carpeted cedar swamps (Croasdale 1935, Laderman 1987) and unpublished observations (Smith 1950). Of the 12 families found in our collections to date (unpublished laboratory notes), only those in the order Zygnematales are represented by more than one genus. Species and subtaxa are very unevenly distributed among the 11 placoderm genera, with 5 forms clustered in *Staurastrum* alone. This skewed diversity pattern is similar to that of cedar-associated tracheophytes, with the great majority of plant species clustered in one family, Ericaceae, commonly known as heaths (Laderman and Ward 1987, 1989). Phyletic skewing reflects the rigorous conditions of the ecological-island bog forests. Here most species that are common in surrounding moderate sites cannot survive, and adaptive extremophilic genotypes appear to have radiated rapidly in the cedar bogs' available although restrictive niches. Genetic analysis will be required to indicate patterns of evolution by revealing the true relationships of genera, species, and lower taxonomic forms.

Two morphological features may contribute to the desmids' capacity to thrive and speciate so richly in the inhospitable cedar wetlands: (1) they often form tough, resistant zygotes that remain dormant in adverse conditions (Pickett-Heaps 1975) and (2) vegetative forms characteristically produce a thick mucilage coating (Boney 1981; Gerrath 1993, 2003).

The labor-intensive process of collecting, identifying, and documenting the existence of the extremophilic microinhabitants of the cedar wetlands has just begun. The groundwork is now in place for expansion of the database to other algal taxa, and to cedar sites from Maine to Mississippi. We hope that the Website will facilitate not only research, but stewardship and conservation of the freshwater forests dominated by *Chamaecyparis thyoides*.

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Those who worked with the authors as contributors to the Cape Cod microflora to date are Jennifer Bowen, Jennifer Courcy, Rachel L. Donnette, Lael S. Laderman, David Patterson, Jonathan Polloni, Pamela Polloni, Gabrielle Sakolsky, Donald Schall, and field workers of Cape Cod Mosquito Control. Jessica Tallman, Marjorie Parmenter, and Suzanne Donovan provided valuable technical assistance. Micro*scope is designed and maintained by David Patterson in Melbourne, Australia and the Bay Paul Center, Marine Biological Laboratory, Woods Hole, MA. This paper was greatly improved by comments of the reviewers of this volume.

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PERFORMANCE OF ATLANTIC WHITE-CEDAR PLANTINGS ALONG WATER TABLE GRADIENTS AT TWO SITES IN THE NEW JERSEY PINELANDS

Kristin A. Mylecraine, George L. Zimmermann, and John E. Kuser[1]

Abstract—This study examines the effect of water table depth on Atlantic white-cedar (AWC) [*Chamaecyparis thyoides* (L.) B.S.P.] plantings at two sites in the New Jersey Pinelands. In 1998, rooted cuttings were planted in successive rows along water table gradients, and survival and growth were monitored through two growing seasons. At site 1, there were significant differences in height growth across rows (representing different water table depths), but no significant differences in survival, although there was a trend toward increased mortality at greater water table depths. At site 2, there were significant differences in survival, but not height growth, across rows. Drought conditions in 1999 decreased survival at higher elevations. Slightly higher water tables during 2000 led to significant mortality at lower elevations probably due to long periods of inundation. These results should be considered when choosing sites for future AWC restoration projects.

Keywords: *Chamaecyparis thyoides*, Cupressaceae, moisture requirement, water table, wetland.

INTRODUCTION

Atlantic white-cedar (AWC) [*Chamaecyparis thyoides* (L.) BSP.] is an obligate wetland tree species (Reed 1988), and is restricted to freshwater wetlands along the Atlantic and Gulf coasts of the United States. Its wood is durable, light weight, aromatic, and usually has an even, straight grain, and has been used for a wide variety of timber products (Korstian and Brush 1931, Little 1950, Ward 1989). AWC ecosystems may also provide many ecological benefits including habitat for several plant and animal species, maintenance of water quality, and stabilization of stream flows. Over the past two centuries, there has been a significant decline in the area occupied by AWC in New Jersey (Mylecraine and Zimmermann 2003), as well as throughout its range (Frost 1987, Kuser and Zimmermann 1995). As a result, much recent interest has focused on the species, its management, and restoration.

The occurrence of AWC wetlands may be limited by unfavorable moisture conditions (Little 1950). Further, inadequate hydrologic conditions may be detrimental to AWC regeneration, restoration and growth (Akerman 1923, Harshberger 1916, Little 1950, Pinchot 1899, Zimmermann 1997), and must be considered when choosing sites for these activities. Recent greenhouse (Allison and Ehrenfeld 1999) and field (Harrison and others 2003, Mylecraine and others 2003) studies have begun to quantify the effects of hydrologic variables on AWC regeneration, and have found that both survival and growth of plantings are influenced by water table depth.

This study focused on two highly disturbed, groundwater discharge wetlands in the New Jersey Pinelands. Because of the nature of these sites, many confounding factors could be eliminated allowing us to focus on hydrologic variables. The specific objective was to examine the survival and growth of AWC plantings in relationship to water table depth. Preliminary, first season results of this study have been published (Mylecraine and others 2003). However, hydrologic conditions vary both seasonally and annually, and long-term studies will be necessary to address these issues. Here we present results after two growing seasons.

METHODS

Site Descriptions

Two sites were chosen within the Clayton Sand company property, located in Jackson Township, Ocean County, New Jersey (40° 4' N, 74° 23' W). Both sites were groundwater discharge wetlands with significant elevational and water table gradients over relatively short distances. Site 1 was highly disturbed, lacked vegetative cover, and was located adjacent to a current mining operation on a sterile, sandy soil. Site 2 was also highly disturbed, has previously been used for cranberry production, and had a steep slope leading to a small pond. Detailed site descriptions were previously described by Mylecraine and others (2003).

Experimental Design

AWC stecklings (rooted cuttings) were planted in fall, 1998, along the water table gradients at each site. At site 1, 366 stecklings were planted in 16 rows. At site 2, 125 stecklings were planted in 5 rows. For both sites, stecklings in each row represented replicates at the same position along the water table gradient. Steckling survival and height growth were monitored through two growing seasons.

Water table depth was monitored across each site with a series of test wells. Each well was made from PVC piping, 1.5 m in length and 2.5 cm in diameter. There were 36 wells at site 1 and 14 wells at site 2.

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Relative elevations were measured for all rows of stecklings and test wells, using an optical transit. Three elevations were taken and averaged to calculate the elevation for each steckling row. Test well data were used to establish the relationships between elevation and mean water table depth at each site. These relationships were then used with the row elevations to calculate the mean water table depths for each row of AWC plantings (Mylecraine and others 2003).

All data analyses were conducted using SAS (SAS Institute), and each site was analyzed independently. Chi-square analyses were employed to test for significant differences in survival across rows. Additionally, repeated measures analysis of variance was carried out on steckling height, to test for significant variation in the height response through time, across rows.

RESULTS

Site 1

Both annual and seasonal variation in water table depth was observed throughout the study period. Mean water table depths, by row, are presented in table 1. Average values are presented for the entire monitoring period (1996–2000), as well as for each of the two years included in the current study (1999 and 2000). Averaged across all years (1996–2000), water table depths ranged from -21 cm (row 1) to -141 cm (row 16). Due to drought conditions during 1999, mean values were slightly lower than the 5-year average, while values during 2000 were close to the 5-year average. For both years, mean values during the growing season were only slightly lower than the yearly average (table 1).

Table 1—Mean water table depths (cm) through study period at Site 1, by row. Overall mean includes all data throughout the year, while growing season mean includes all measurements between April and September

	Overall			Growing season	
Row	1996-2000	1999	2000	1999	2000
1	-21	-26	-22	-27	-22
2	-26	-31	-27	-32	-27
3	-25	-29	-25	-31	-26
4	-32	-37	-33	-38	-33
5	-40	-44	-40	-46	-41
6	-42	-46	-42	-48	-43
7	-49	-53	-49	-54	-50
8	-53	-57	-53	-58	-53
9	-57	-61	-57	-62	-57
10	-62	-65	-61	-66	-62
11	-67	-70	-66	-71	-66
12	-72	-76	-72	-77	-73
13	-78	-82	-78	-83	-79
14	-84	-88	-84	-89	-84
15	-97	-101	-97	-102	-97
16	-141	-144	-140	-144	-140

There were no significant differences in steckling survival across rows, representing different water table depths, although there was a trend toward decreased survival at greater water table depths. Percent survival through the first two growing seasons, by row, ranged from 72 to 100 percent (table 2). The row with the greatest water table depth (row 16) also had the lowest survival—72 percent.

There was a significant time*row interaction for steckling height (fig. 1) $p < 0.0001$ suggesting significant variation in performance across rows over time. Mean growth, by row, ranged from 0.2 cm to 4.7 cm in 1999, and from 0.3 to 4.1 cm in 2000 (table 2). Stecklings in rows 1 through 4 exhibited the greatest growth rates. In 1999, mean water tables for these rows ranged from -26 to -37 cm, with a maximum of 4 cm and a minimum of -55. In 2000, mean water tables for

Table 2—Survival and growth of AWC stecklings planted at Site 1, by row. Standard errors are in parentheses

		% Survival		Height Growth (cm)	
Row	N	1999	2000	1999	2000
1	24	100	100	4.7 (0.7)	4.1 (0.3)
2	23	91	91	5.0 (0.6)	4.1 (0.5)
3	23	96	96	3.7 (0.5)	3.7 (0.7)
4	10	100	100	4.6 (0.9)	2.6 (0.4)
5	11	100	100	4.6 (0.9)	2.6 (0.4)
6	11	100	100	1.4 (0.4)	0.7 (0.3)
7	11	100	100	0.8 (0.1)	0.4 (0.2)
8	41	98	85	1.0 (0.3)	0.9 (0.1)
9	28	100	89	1.4 (0.2)	1.0 (0.4)
10	29	90	86	0.9 (0.2)	1.0 (0.3)
11	29	83	79	0.9 (0.2)	0.7 (0.7)
12	39	90	85	0.6 (0.2)	0.8 (0.2)
13	22	86	86	0.4 (0.4)	0.3 (0.3)
14	22	91	91	0.5 (0.6)	1.0 (0.2)
15	21	100	100	0.2 (0.5)	0.7 (0.4)
16	18	78	72	0.5 (0.5)	1.9 (0.4)

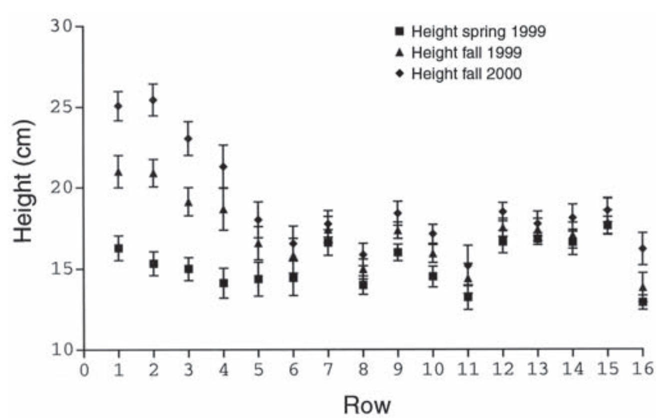

Figure 1—Mean heights of AWC stecklings, planted at site 1, through the study period, by row. Repeated measures ANOVA indicates a significant time*row interaction, suggesting variation in performance among rows.

these rows ranged from -22 to -33 cm, with a maximum of 14 cm and a minimum of -45 cm. Stecklings planted in rows with a mean water table > -50 cm, exhibited little to no growth through the study period.

Site 2

Over 5 years (1996–2000), mean water table depths ranged from -6 cm for row 1 to -130 cm for row 5 (table 3). During 1999, values averaged slightly lower due to drought conditions, ranging from -9 to -137 cm. During 2000, values averaged slightly higher than the 5-year average, ranging from -3 to -137 cm. Water table depths for row 5 (highest elevation) were probably underestimates, as the depth exceeded the well length throughout much of the study.

There were significant differences in survival across rows and years (fig. 2) (Chi-square analysis, $\alpha = 0.05$) in response to water table variation. Survival through the 2-year period ranged from 12 to 96 percent (table 4). Low water tables during 1999, due to drought, decreased survival in row 5—the row with the highest elevation and deepest water table. All stecklings that survived through the first growing season also survived through the 2000 season, with the exception of row 1. Slightly higher water tables at lower elevations, along with periods of inundation, led to significant mortality in row 1.

Table 3—Mean water table depths (cm) at Site 2, by row. Overall mean includes all data throughout the year, while growing season mean includes all measurements between April and September

	Overall			Growing season	
Row	1996-2000	1999	2000	1999	2000
1	-6	-9	-3	-15	-3
2	-25	-29	-24	-34	-24
3	-67	-72	-69	-77	-68
4	-100	-106	-105	-110	-104
5	-130	-137	-137	-141	-136

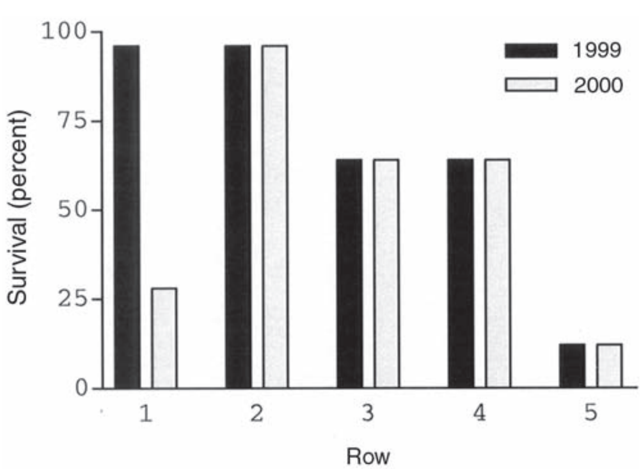

Figure 2—Percent survival of AWC stecklings, planted at site 2, through the study period, by row.

Table 4—Survival and growth of AWC stecklings planted at Site 2, by row. Standard errors are in parentheses

		% Survival		Height Growth (cm)	
Row	N	1999	2000	1999	2000
1	25	96	28	1.1 (0.5)	1.3 (0.5)
2	25	96	96	1.9 (0.3)	7.0 (1.4)
3	25	64	64	0.3 (0.6)	3.8 (0.9)
4	25	64	64	2.1 (0.7)	4.5 (1.1)
5	25	12	12	0.1 (0.1)	0.3 (0.2)

During 1999, mean height growth ranged from 0.1 cm for row 5 to 1.9 cm for row 2. Corresponding values during 2000 ranged from 0.3 cm for row 5 to 7.0 cm for row 2. Although there is a trend toward reduced growth for row 1 (highest water table) and row 5 (lowest water table), and greatest growth at mid-levels, these differences were not significant (repeated measures ANOVA; $p = 0.16$).

DISCUSSION

As the demand for AWC reforestation and restoration increases, it is important to understand the effects of hydrologic variables on such activities, as successful projects will depend on adequate hydrologic conditions. We have examined the effects of water table depth on the performance of planted AWC stecklings at two highly disturbed sites within the New Jersey Pinelands.

Little (1950) observed that first year seedlings grew 2.5 to 25 cm on favorable sites, with subsequent height growth averaging 30 to 46 cm annually. In the current study, observed growth rates at both sites were much lower, probably due to the sterile nature of the sites. However, the sterility of the sites helped to reduce confounding variables and focus on the effects of water table depth on steckling performance under field conditions.

Steckling performance was reduced at the greatest water table depths, as shown by reduced survival and decreased growth rates. At site 1, we found significant differences in the height response across rows (each row representing a different position along the water table gradient), with the greatest growth occurring in rows with an annual mean water table depth no greater than -37 cm (fig. 1). At site 2, we observed reduced survival at the greatest water table depths (row 5; mean water table depth -135 to -140 cm) (fig. 2).

Steckling performance was also reduced at very high water table depths, probably due to long periods of inundation. At site 2, we observed reduced survival among row 1 stecklings during 2000 (fig. 2). Water table depths during 2000 were slightly higher than 1999 and the mean water table depth for row 1 was 3 cm below the ground surface. Flooding duration could not be determined from our data, but may have been responsible for the increased mortality. These results, as well as those of Harrison and others (2003) suggest that newly planted AWC individuals are susceptible to high water table levels and flooding. Harrison and others (2003) observed heavy mortality of seedlings when mean water table depth was above the ground surface and suggest that plantings should not be exposed to inundation for more than three continuous weeks. Mature stands of AWC commonly experience periods of inundation. Golet and Lowry (1987) studied six mature stands in Rhode Island between 1976 and 1982, and found that the mean water table depth ranged from 13 cm above to 11 cm below the ground surface, and the duration of surface flooding varied from 18 to 76 percent of the growing season. These studies suggest that mature individuals may be less susceptible to flooding than young, newly planted stock.

Although we observed variation across sites, similar trends are apparent. Best performance was achieved with a mean water table depth between approximately -10 and -40 cm. Similarly, Harrison and others (2003) found that growth increased as the depth to water table increased to -57 cm, the maximum depth reported in their study.

Previous field studies that have examined the effect of water table depth on performance of Atlantic AWC regeneration have spanned only one season (Harrison and others 2003, Mylecraine and others 2003). Our second season results indicate significant variability in both water table depths and steckling performance across years. This illustrates the importance of taking both seasonal and annual variation into account when choosing sites based on hydrologic variables.

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DECOMPOSITION DYNAMICS IN AN ATLANTIC WHITE CEDAR RESTORATION SITE

Edward R. Crawford, Frank P. Day, and Robert B. Atkinson[1]

Abstract—Decomposition dynamics play an important yet little-studied role in organic matter accumulation in Atlantic white cedar [*Chamaecyparis thyoides* (L.) B.S.P.] wetlands. A root decomposition study using a modified litterbag approach and a leaf litter decay study using standard litterbags, were conducted within restored Atlantic white cedar wetlands in Pocosin Lakes National Wildlife Refuge, North Carolina. The objectives of this study were to quantify aboveground and belowground decomposition rates within a restoration setting. *Chamaecyparis thyoides* (L.) B.S.P. root and leaf litter were deployed as a standard substrate. Native leaf litter decay (k = 0.79 yr^{-1}) was significantly higher relative to *C. thyoides* leaf litter decay (k = 0.36 yr^{-1}). Across a vertical soil profile, native root decay rates were significantly faster compared to *C. thyoides* root decay rates. These results suggest that within this restoration site, the current status of belowground carbon storage function appears to be deficient. Continued long-term study of this and similar restoration sites are need to provide greater insight into appropriate recovery models for various wetland functions of Atlantic white cedar.

Keywords: Carbon sequestration, decomposition, histosols, wetland restoration.

INTRODUCTION

Atlantic white cedar [*Chamaecyparis thyoides* (L.) B.S.P.] distribution is restricted to a narrow band of freshwater wetlands along the Eastern coastal United States ranging from Maine to Mississippi (Laderman 1989). Historically, the largest known stand of Atlantic white cedar, estimated at 26,000 – 45,000 ha, was found in the Great Dismal Swamp of Virginia and North Carolina (Frost 1987, Moore and Allen 1998).

Historically, Atlantic white cedar has been a valuable timber species on the Albemarle-Pamlico peninsula in North Carolina. Elaborate networks of roads, canals and ditches were constructed to provide direct access to these stands (Laderman 1989) and these areas have endured draining, nutrient loading, and fire suppression over the centuries (Richardson 1991). Alligator River and Pocosin Lakes National Wildlife Refuges were established to conserve and manage the area's unique wetlands, including Atlantic white cedar forests (Bryant 1999). They currently contain over 10,000 acres of Atlantic white cedar clearcuts and scattered disjunct remnants of Atlantic white cedar stands (Eagle 1999). The absence of post-harvest forest management coupled with poor logging practices and hydrologic modifications have resulted in poor Atlantic white cedar regeneration in many areas (Eagle 1999).

Typically, Atlantic white cedar is found on deep organic soils, as a result of extended hydroperiods and high litter production rates coupled with slow decomposition rates (Day 1987). Ehrenfeld (1995) has identified the development of a deep peat substrate as a key component of the structure and function of Atlantic white cedar wetlands. Peat deposits strongly affect the hydrology of Atlantic white cedar wetlands by increasing the water holding capacity of the soil (Levy and Walker 1979) and are required for seedling recruitment (Beull and Cain 1943). The wetter hydrologic conditions can lead to anaerobic conditions and reduced soil pH, which can lead to decreasing decay rates (Day 1982, Tupacz and Day 1990) ultimately affecting nutrient availability and production. As a result, hydrologic regimes and carbon cycling are tightly coupled mechanisms that either directly or indirectly affect other ecosystem functions. The regeneration and development of Atlantic white cedar wetlands is dependent on the interplay of several environmental drivers including the catastrophic disturbance of fire, fire intensity, and ground water elevation at the time of fire (Laderman 1989).

While there are many recent studies examining various aspects of Atlantic white cedar restoration (see Shear and Summerville 1999), there are no studies documenting decay processes occurring in Atlantic white cedar restoration sites (Crawford 2002). Soil organic matter accumulation and sequestration are regarded as critical components to ecosystem function and self maintenance of these wetlands. The objectives of this study were to quantify and compare aboveground and belowground decomposition rates within a restoration site at Pocosin Lakes National Wildlife Refuge.

METHODS

Study Site

This study was conducted within a restoration area on a former Atlantic white cedar wetland. The site was 3 km south of Lake Phelps on F2 road within Pocosin Lakes National Wildlife Refuge. The stand was planted with Atlantic white cedar seedlings during the fall of 1998. The site was burned by fire of unknown cause two years prior to planting and was colonized by bracken fern (*Pteridium aquilinum* [L.] Kuhn.) an

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unidentified *Solidago spp.* and barnyard grass (*Echinochloa crusgalli* [L.]). The site is on acidic organic soils, which are ombrotrophic, and classified as typic haplosaprists (U.S. Department of Agriculture 2000).

Study Design

Two 10 m x 15 m plots (each one containing 48 evenly dispersed 1 m X 1 m subplots) were established for above and belowground processes and microenvironmental measurements. Within each subplot, nylon mesh litterbags containing native roots and separate bags containing *C. thyoides* roots were inserted into the soil profile during November 1998. The litterbags were constructed to integrate the decay process over a vertical soil profile (Conn and Day 1997, Tupacz and Day 1990). Belowground plant materials (roots and rhizomes, where applicable) were collected from pit excavations during the summer of 1998 from sites that were representative of the dominant community. At this site, Atlantic white cedar roots (0.03 grams) were placed into each bag to represent the dominant woody component. Upon harvesting, and prior to placing in litterbags, the root material was washed free of peat and air dried. Air dry weights were recorded and converted to oven dry weights using conversion factors obtained from subsamples of the original belowground material.

Roots of *C. thyoides* seedlings donated by Weyerhaeuser Corp. were used as a standard substrate for comparison with decay rates for the native substrate. Within site comparison of *C. thyoides* root decay rates with native root decay rates demonstrated litter quality influences under similar environmental conditions. Known weights of air-dried roots were placed in nylon mesh litterbags constructed of 1 mm pore size nylon mesh and were 40 cm long by 5 cm wide and divided into four 10 cm sections. The litterbags were inserted lengthwise into a vertical slit in the soil with the top of the litterbag positioned at the soil-atmosphere interface. Approximately 3 grams of root material was placed in each of 4 10-cm sections in each litterbag. All roots were between 0.2 and 1.0 cm diameter. Subsamples were oven dried for 48 hours at 75 °C to constant mass for air dry:oven dry mass ratios.

Six root litterbags were collected at each sampling interval from randomly selected subplots (within each plot) throughout the course of the study. Root samples were installed in the field in November 15, 1998 and sampling continued on a regular basis through January 2000. Over the course of the study, sampling intervals were approximately 1 month (when logistically possible). Upon retrieval, the root bags were rinsed with tap water to remove adhering peat and roots growing into the bags were plucked out using forceps. The decomposing substrate was oven dried and weighed to determine mass loss. In order to determine ash free dry weight conversion ratios for the samples, individual samples were ground in a Wiley mill (40 mesh) and ashed in a muffle furnace at 550 °C for 5 hours (Allen and others 1986).

Decay rates of leaf litter were also measured. Recently senesced leaf litter was collected on site, brought back to the lab, and air-dried during the fall of 1998. Approximately 3 grams of the collected air-dried leaves were placed in 20 cm x 20 cm nylon mesh (1 mm hole size) bags. A second group of litterbags that contained *C. thyoides* leaf litter served as site controls. The *C. thyoides* leaf litter was collected during the fall of 1998 from an individual tree that fell during late summer within the Great Dismal Swamp National Wildlife Refuge. Leaf litterbag samples were placed in the field on January 15, 1999 and collection coincided with root sample collection dates from January 1999 through January 2000. Six litterbags of native leaf litter and six litterbags of *C. thyoides* leaf litter were removed from onsite following the same protocol described earlier for the root litterbags. Onsite water table was determined by the oxidation depth on steel rods and soil pH was evaluated over a 0 to 40 cm soil profile. Collection of steel rods and soil materials for pH determination coincided with leaf and root litterbag collection.

Statistical Analyses

The data were evaluated using both linear and exponential decay models to test for the best fit. Relative decomposition rates (-k [yr^{-1}]) for native and standard leaf litter and roots were derived from a fixed-intercept negative exponential decay model (Weider and Lang 1982) according to the following formula:

$$X = e^{-kt} \tag{1}$$

where

X = the proportion of initial mass remaining

k = the decay constant

t = time

Significant differences in decay coefficients between *Chamaecyparis* roots and native roots and *Chamaecyparis* leaf litter and native leaf litter were detected by t-tests (Zar 1998). Using the derived k values, time required to reach 1 percent mass remaining was extrapolated.

RESULTS

Native leaf litter exhibited greater mass loss relative to *Chamaecyparis* leaf litter over the course of this study (fig. 1). Native leaf litter decay rates were twice as fast compared to *Chamaecyparis* litter, with time to reach 1 percent mass remaining also doubled for *Chamaecyparis* litter (table 1).

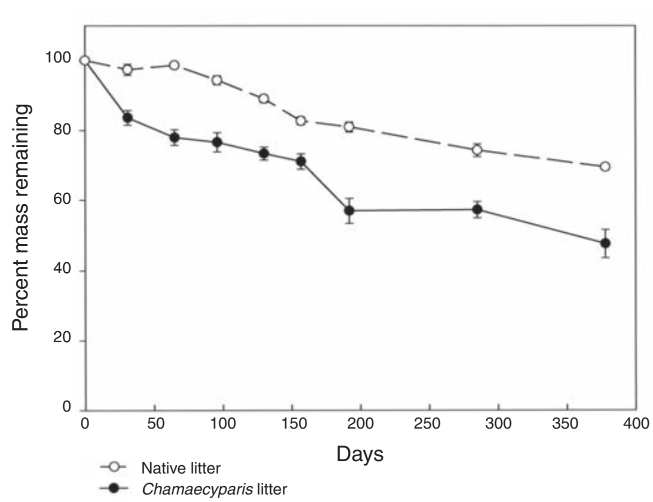

Figure 1—Percent mass remaining for native and *Chamaecyapris thyoides* leaf litter at Pocosin Lakes National Wildlife Refuge. Values represent means with one standard error.

Table 1—Decay rate constants [-k (per year)], coefficient of determiniation (r^2), time to reach 1 percent mass remaining ($t_{0.01}$), and percent mass remaining after 370 days of decay (percent M)[a]

Litter type	Depth class	k	SE	r^2	$t_{0.01}$	Final mass
	cm				*year*	*percent*
			Leaf			
Native		0.79A	0.05	0.96	5.82	47.50
Chamaecyapris		0.36B	0.07	0.98	12.79	69.54
			Root			
Native	0 – 10	0.78A	0.03	0.98	5.82	48.76
	10 – 20	0.57A	0.04	0.95	8.07	62.97
	20 – 30	0.54A	0.02	0.97	8.52	62.59
	30 – 40	0.50A	0.03	0.95	9.21	63.06
Chamaecyparis	0 – 10	0.36B	0.03	0.93	12.79	75.64
	10 – 20	0.34B	0.02	0.95	13.54	76.14
	20 – 30	0.30B	0.01	0.96	15.35	76.60
	30 – 40	0.28B	0.02	0.96	16.44	77.63

[a] All regressions are significant at $p = 0.0001$. Different uppercase letters indicate significant differences ($p = 0.05$) between native and *Chameacyparis* litters.

T-tests between native leaf litter and *Chamaecyparis* leaf litter revealed significant differences ($p = 0.05$) between decay rates (table 1).

Native roots exhibited greater mass loss over time relative to *Chamaecyparis* roots (fig. 2). There was a trend of decreasing mass loss with increasing depth of the vertical soil profile for both native and standard roots (fig. 2). Native root decay rates were more than doubled in the 0 to 10 cm depth interval and nearly doubled in the 10 to 20 cm depth interval relative to *Chamaecyparis* roots (fig. 2). Across all depth intervals, t-tests revealed significant differences ($p = 0.05$) between native root decay and *Chamaecyparis* root decay (table 1).

Mean soil pH was 3.5, and there was little temporal or spatial variation over the soil profile throughout the study (fig. 3). Depth of water table, as indicated by oxidation depth on steel rods, varied seasonally with water table deepest during the summer of 1999 and most shallow during the winter and fall of 1999 (fig. 4).

DISCUSSION

Regeneration of Atlantic white cedar in the mid-Atlantic region has been hindered due to shade intolerance, hydrologic modifications (Hinesley and Wicker 1999) and herbivory by deer, mice, and rabbits (Guidry 1999). These hindrances were present at the Pocosin Lakes restoration site during this study (personal observation).

Decay rates were faster for native leaf litter relative to *Chamaecyparis* leaf litter at Pocosin Lakes National Wildlife Refuge. This was not unexpected as the native leaf litter was dominated by graminoids and forbs and Benner and others (1985) found that the lignocelluloses from herbaceous plants were mineralized several times faster relative to woody species under similar environmental conditions. The decay rate of *Chamaecyparis* litter (k = 0.36 yr^{-1}) was well within the range of values reported for other regional restored and regenerating sites (Crawford 2002) and similar to those reported for Atlantic white cedar by Day (1982) and Yates and Day (1983).

The average soil pH was also within the range of those reported for other restored/regenerating stands in the region (Crawford 2002, Thompson 2001) and also within the range found for other regional Atlantic white cedar stands (Tupacz and Day 1990, Whigham and Richardson 1988). Based on hydrology data inferred from rusting depth on steel rods, with the exception of summer drought during 1999, antecedent and post drought water tables were comparable with other East Coast Atlantic white cedar wetlands (Golet and Lowry 1987).

Both native and *Chamaecyparis* roots had decreasing root decay rates with increasing depth in the soil profile, however, native root decay was significantly faster than *Chamaecyparis* root decay within each depth interval. Similar trends were also exhibited in other Atlantic white cedar restoration sites in the study region (Crawford 2002). Native root decay rates at Pocosin Lakes National Wildlife Refuge were similar to those of other regional restoration sites. Decay rates of *Chamaecyparis* roots were similar to those found in a similar study of stands within Great Dismal Swamp and Alligator River National Wildlife Refuges (Crawford 2002). In that study, the highest rates of both native and *Chamaecyparis* root decay occurred in the 0 to 10 cm depth interval and is likely a result of aerobic conditions found at that interval. Oxidation depth on steel rods within the study site was consistently greater than 10 cm below the soil surface over the course of the study.

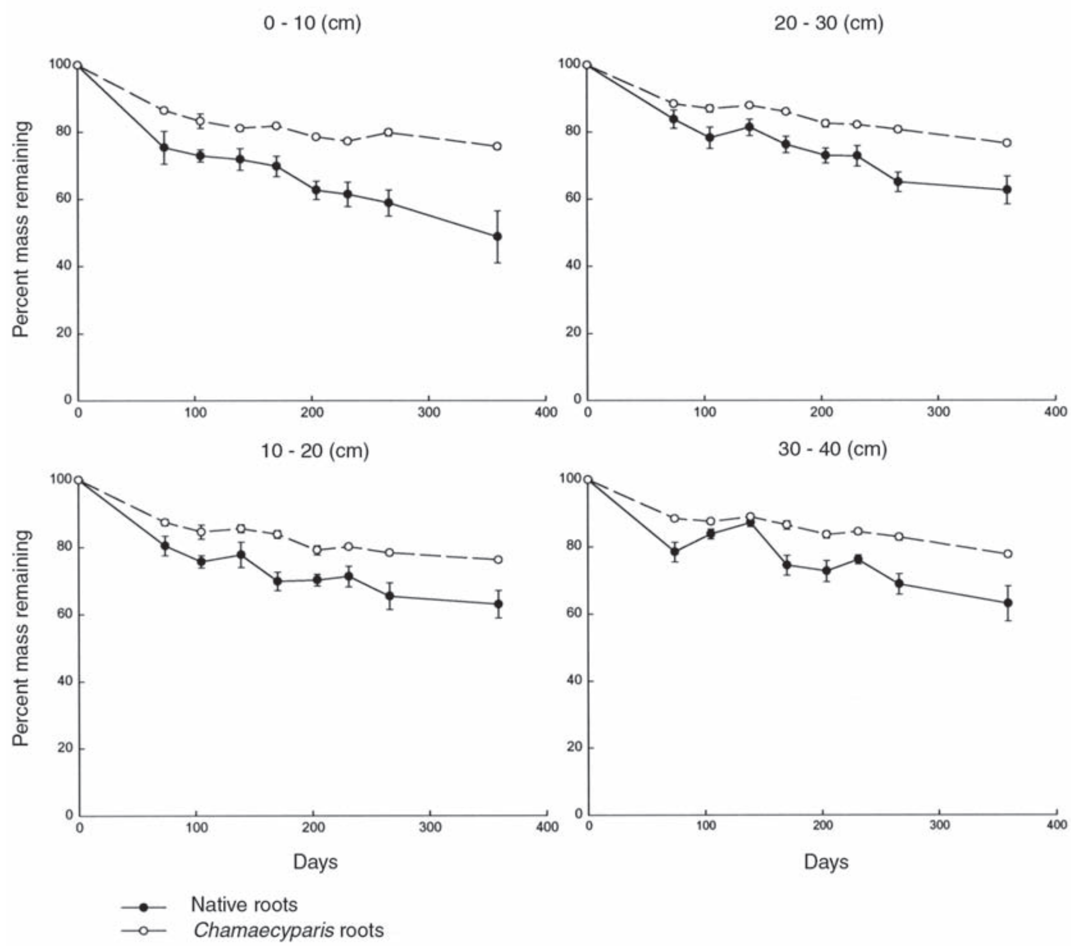

Figure 2—Percent mass remaining of native and *Chamaecyparis thyoides* root litter at Pocosin Lakes National Wildlife Refuge. Values represent means with one standard error.

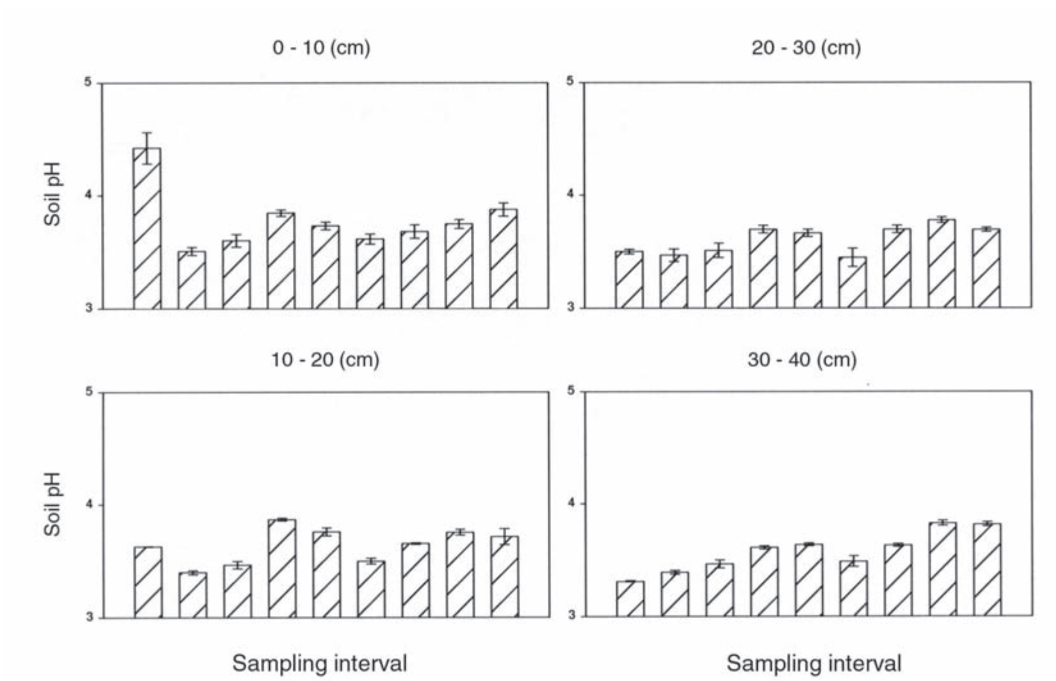

Figure 3—Vertical gradient in soil pH within Pocosin Lakes National Wildlife Refuge. Values represent means with one standard error. Sampling interval refers to nine collection periods between January 1999 and January 2000.

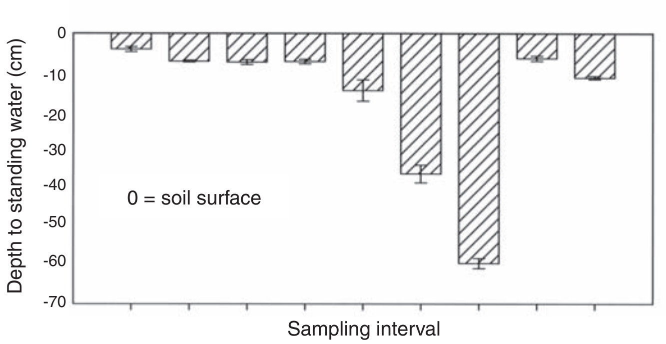

Figure 4—Depth to standing water at Pocosin Lakes National Wildlife Refuge as determined by rusting depth on steel rods. Values represent means with one standard error. Sampling interval refers to nine collection periods between January 1999 and January 2000.

Findings of this study suggest that decomposition of native roots and leaf litter is faster relative to *Chamaecyparis* root and leaf litter decay. Future increases in above and belowground production of Atlantic white cedar coupled with the low decay rates of cedar litter could result in increased contributions to soil organic matter pools and greater carbon sequestration as this restored site matures.

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WATER TABLE AND TEMPERATURE REGIME AFFECT GROWTH OF POTTED ATLANTIC WHITE CEDAR

Scott A. Derby and L. Eric Hinesley[1]

Abstract—Atlantic white cedar [*Chamaecyparis thyoides* (L.) B.S.P.] seedlings were grown for 16 weeks at 18/14 °C, 22/18 °C, or 26/18 °C in controlled-environment chambers. Four water table treatments were used in 20-cm pots (pot-in-pot) containing a pinebark substrate: (1) control (fully drained), (2) one-third full, (3) two-thirds full, and (4) completely full. Pots were watered twice daily with deionized water. They were fertilized once weekly with a modified Hoagland's solution (3 to 4 hours); then, flushed and refilled with deionized water. Growth (height, stem diameter, and dry weight) increased with increasing temperature regime, but temperature effects were minimal for flooded plants. For most variables, the relationship of water table to growth was quadratic; linear for temperature. Maximum height and dry weight occurred when pots were one-third to two-thirds full of water. Flooded plants yielded far less growth and had lower height/diameter ratios compared to other water regimes. Root systems of flooded plants tended to form a mat near the soil surface.

Keywords: *Chamaecyparis thyoides*, containerized plants, controlled-environment, flooded, irrigation, root system, seedlings, wetland restoration.

INTRODUCTION

Atlantic white cedar [*Chamaecyparis thyoides* (L.) B.S.P.] is an evergreen conifer that grows in fresh water swamps and bogs along a narrow coastal belt from southern Maine to northern Florida and west to southern Mississippi (Laderman 1989). The wood is lightweight, fragrant, straight-grained, easily machined, extremely resistant to decay, and is not prone to crack or check (Hinesley and Wicker 2003). All these traits made it highly prized for boats, siding, and shingling in the past. Historically, white cedar was the most valuable tree in the Albemarle Peninsula in the coastal plain of eastern North Carolina (Krinbill 1956). It had a stumpage value up to five times greater than other species, and special effort was made to accurately locate all stands so they could be reached with minimum construction of railroad spurs (Krinbill 1956). Current demands for the lumber include telephone poles, piling, ties, siding, and boat railing. The acreage of Atlantic white cedar (AWC) today is only a small fraction of the original (Davis and others 1997, Frost 1987, Kuser and Zimmermann 1995, Lilly 1981) due not only to logging, but also wildfires and drainage of peatlands for agricultural purposes (Hinesley and Wicker 2003).

Restoration of AWC ecosystems is a high priority in the forested wetlands of eastern North Carolina. However, a serious obstacle in replenishing AWC is lack of planting stock; until that problem is resolved, little progress will be made. AWC is relatively easy to vegetatively propagate from stem cuttings (Boyle and Kuser 1994, Hinesley and others 1994, Hinesley and Snelling 1997), but that method has drawbacks, including high labor intensity and costs. Comparisons of growth for seedlings and rooted cuttings the first few years in the field have yielded mixed results (Kuser and Zimmermann 1995, Phillips and others 1998). In addition, seedlings have more genetic diversity that serves as a buffer against environmental and biotic adversity (Kuser and Zimmermann 1995). However, also unknown is the long-term performance of rooted cuttings.

In North Carolina, production of AWC in outdoor nursery beds has been hampered by several problems including low utilization efficiency of seed, and poor control of seedbed density. Current efforts are focused on production of containerized seedlings and transplants from seed.

Atlantic white cedar is an obligate wetland species (Reed 1988) that can tolerate an elevated water table. An important issue in refining the science of container production is frequency and quantity of irrigation. Too much water is not good, whereas too little water can stress seedlings and reduce their growth. Consequently, our objective was to examine the growth and morphology of white cedar seedlings in response to varying water table depths, at or near saturation, over a range of temperature regimes.

MATERIALS AND METHODS

On January 29, 2002, containerized AWC seedlings were graded by size and planted into standard 1 gallon pots (Nursery Suppliers Classic 300, height = 18.5 cm) containing a substrate of pine bark (filled to depth of 17 cm) with one seedling per pot. The planted pot was then nested inside another Classic 300 pot without drainage holes. Four equally spaced 1.3-cm drainage holes were previously drilled into the outer pot according to assigned water table level: no water table (well drained), one-third full, two-thirds full, and flooded (water at the surface of the substrate). Ten additional seedlings were randomly sampled for initial height (cm), and stem diameter (mm) was determined with a digital caliper near ground line. Roots and shoots were dried to constant weight gram at 65 °C.

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Plants were fumigated overnight in the Southeastern Plant Environment Laboratory at North Carolina State University. The next day, pots were set on mobile carts and placed in three walk-in growth chambers: 18/14 °C, 22/18 °C, and 26/22 °C day/night temperatures, all under long-day conditions. Day lighting occurred for 9 hours and was derived from cool white fluorescent and incandescent lighting providing a photosynthetic photon flux density (PPFD) of 639 µmol m^{-2} s^{-1}. Plants received a 3-hour night interrupt with 58 µmol m^{-2} s^{-1} of incandescent lighting from 2300 to 0200 hours (Downs and Thomas 1991).

The study design was a randomized complete block within each chamber. The four water tables represented treatments, and each replication had four, single-tree plots, i.e., pots. Each cart held two replications (eight pots), and each chamber had five carts for 10 replications (40 plants). Plants were hand watered with deionized water twice daily. Once each week, the outer pot was removed from each container to allow drainage, and the substrate was saturated with a modified Hoagland's nutrient solution (Downs and Thomas 1991). After 1 hour, pots were thoroughly rinsed of the fertilizer solution; then, water tables were restored.

Plants were measured for height and stem diameter once monthly. The study was concluded on May 30, 2002, after 16 weeks. Following final measurement, plants were divided into roots and shoots, and dried to constant weight at 65 °C prior to weighing.

Using SAS (SAS Institute 2000), data were checked for normality and homogeneity of variance, and then analyzed using GLM procedures. Owing to several "temperature x flood treatment" interactions, the analysis was also carried out for each temperature regime separately (table 1). Linear and quadratic effects were tested for temperature, and a cubic effect was included with flood treatments.

Table 1—GLM analysis for growth of potted Atlantic white cedar seedlings subjected to four water table levels and three temperature regimes

					Dry weight		
Source	df	Height	Stem diam.	Ht/diam ratio	Root	Shoot	Total
				18/14 °C			
W	3	**	*	*	**	**	**
lin	1	NS	NS	*	**	**	**
quad	1	**	**	NS	**	**	**
cubic	1	*	NS	NS	NS	NS	NS
Error	27	—	—	—	—	—	—
R^2		0.60	0.49	0.57	0.65	0.51	0.58
				22/18 °C			
W	3	**	NS	**	**	**	**
lin	1	NS	NS	**	**	*	**
quad	1	**	NS	**	**	**	**
cubic	1	NS	NS	NS	NS	NS	NS
Error	27	—	—	—	—	—	—
R^2		0.54	0.39	0.54	0.73	0.54	0.64
				26/22 °C			
W	3	**	**	**	**	**	**
lin	1	**	NS	**	**	**	**
quad	1	**	**	*	**	**	**
cubic	1	NS	NS	NS	NS	*	*
Error	27	—	—	—	—	—	—
R^2		0.60	0.61	0.55	0.87	0.82	0.85
				Combined			
T 2	**	**	**	**	**	**	
lin	1	**	**	**	**	**	**
quad	1	**	NS	NS	NS	*	NS
Rep(T)	27	—	—	—	—	—	—
W	3	**	**	**	**	**	**
T x W	6	**	NS	*	NS	**	**
Error	81	—	—	—	—	—	—
R^2		0.76	0.71	0.61	0.81	0.82	0.83

W = water table level; T = temperature regime; **, *, NS = significant or not significant at $P \leq 0.01$ or 0.05 respectively.

RESULTS

In general, there was a quadratic relationship between water table level and various indices of growth, with maximum values in pots that were one-third to two-thirds full (table 1) (fig. 1). Saturated conditions sharply reduced height and dry weight (fig. 1).

The relationship between temperature and growth was quadratic for shoot weight and height, but linear for stem diameter, root weight, and total weight (table 1) combined analysis. The temperature effect was most evident in unsaturated conditions (pots ≤ one-third or two-thirds full) (fig. 1) (all panels), i.e., going from 22/18 °C to 26/22 °C increased growth more than an increase from 18/14 °C to 22/18 °C.

Under unsaturated conditions, 26/22 °C yielded about 50 percent more shoot weight (fig. 1A) and 30 percent more root weight (fig. 1B) than 22/18 °C. Root weight of flooded plants was 35 to 40 percent of the maximum values for lower water tables (fig. 1B). Plants at 26/22 °C had 50 percent and 80 percent more total weight (roots plus shoot) than those at 22/18 °C and 18/14 °C, respectively (fig. 1C). Total weight of flooded plants was only 36 to 49 percent of the maximum values for other water table levels.

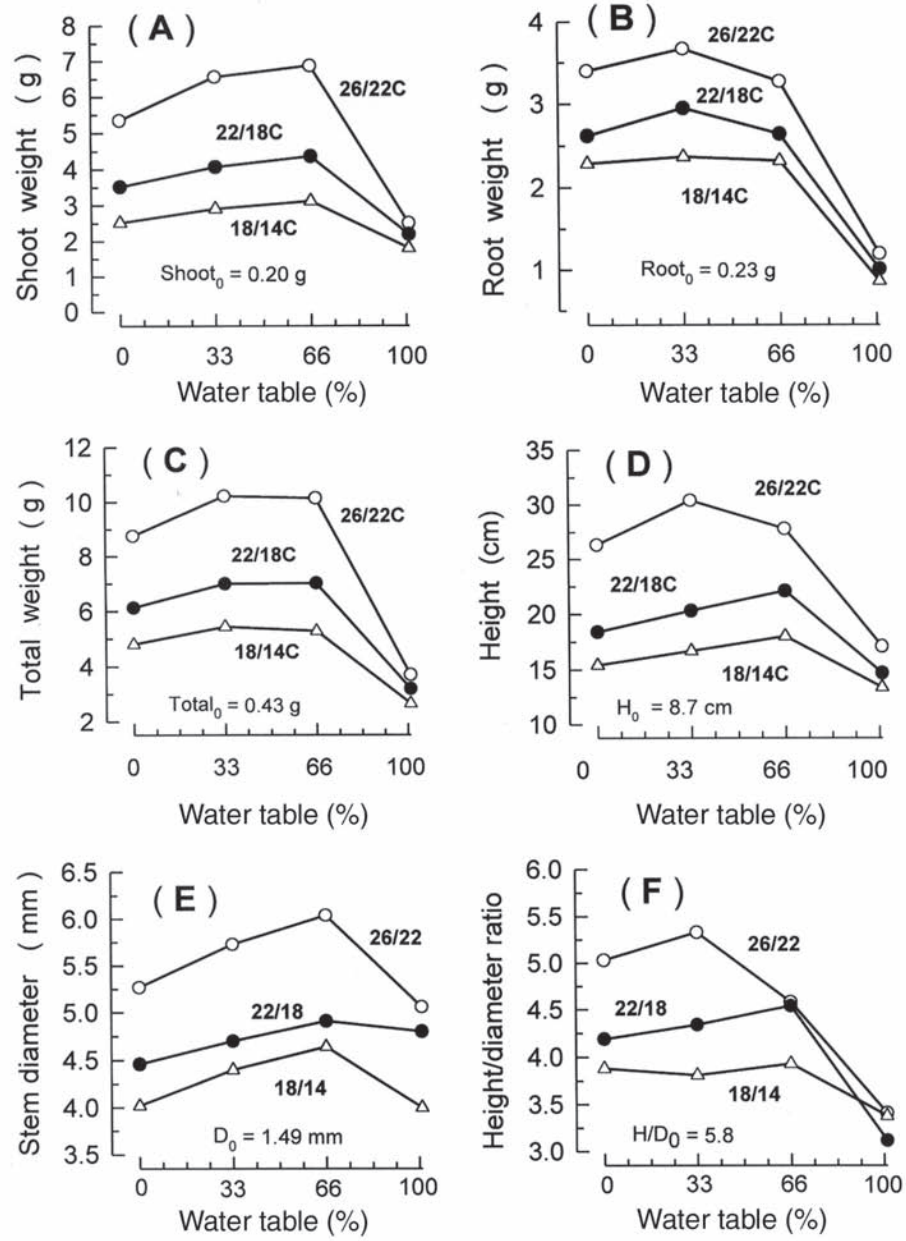

Figure 1—Shoot weight, root weight, total weight, height, stem diameter, and height/diameter ratio for potted Atlantic white cedar seedlings grown for 16 weeks at 18/14 °C, 22/18 °C or 26/22 °C. (A) shoot dry weight, (B) root dry weight, (C) total dry weight, (D) height, (E) stem diameter, and (F) height/diameter ratio. Each "temperature x water table" mean was based on 10 plants, and in general, the standard error of the mean was 2 percent to 5 percent of the mean value.

Maximum height (30 cm) occurred at 26/22 °C (water table = 1/3), 78 percent more than the saturated treatment (17 cm), and 13 percent more than the control treatment (fig. 1D). Height was smallest at 18/14 °C, where plants in pots that were two-thirds full were 34 percent taller than flooded plants (18.0 vs. 13.4 cm, respectively).

Maximum stem diameter occurred in pots that were two-thirds full (fig. 1E). For all water table levels, stem diameters were about 30 percent larger at 26/22 °C compared to 18/14 °C. As with other measures of growth, height/diameter (h/d) ratios were highest for water table levels of one-third and two-thirds (fig. 1F). H/d ratios were sharply lower—and very similar—under saturated conditions, indicating that those plants were stouter than others in relation to their height. By the end of the experiment, all treatments had h/d ratios lower than the initial value of 5.8.

DISCUSSION

In forested wetlands, water tables fluctuate seasonally as well as in response to isolated storm events. The ability of plants to tolerate flooding is affected by many factors, e.g., species, season, water temperature, oxygen content, and water movement. It would likely require a long series of experiments to determine the response of AWC to various combinations of factors bearing on this question. Even though our experiment did not simulate natural conditions, owing to constant water tables, it still allowed inferences about the response of AWC to water tables at or near the soil surface.

In general, the response of various growth indices to water table depth was quadratic (table 1) (fig. 1). The best growth, which occurred in moderately flooded treatments, was consistent with published data stating that water table levels in white cedar stands are typically within 10 to 20 cm of the soil surface during the growing season (Harrison and others 2003, Little 1950, Reynolds and others 1981), though there can be considerable variation from site to site (Atkinson and others 2003, Duttry and others 2003, Mylecraine and others 2003) and year to year (Golet and Lowry 1987).

Saturation resulted in little growth except for stem diameter, with treatment differences most dramatic in root dry weight and total dry weight (fig. 1). This verifies published data in which AWC and nine other taxa were flooded for 28 days during the growing season (Holland and others 2003). Relative root growth declined 10 percent at 22/18 °C and 22 percent at 30/26 °C. In both temperature regimes, AWC was least negatively impacted by flooding. While no plants died in our experiments, Harrison and others (2003) found a negative relationship between flooding and survival in natural stands, where inundation was present for half of the growing season. Growth of AWC tends to decrease on excessively wet sites (Little 1950), likely due to inadequate aeration of the rhizosphere. In our study, plants growing in a saturated substrate developed a mat of roots at the soil surface, suggesting a degree of morphological adaptability in response to waterlogged conditions.

In general, growth of seedlings increased linearly in response to temperature (table 1), confirming previous research with AWC seedlings (Jull and others 1999). It would be interesting to examine the response to temperature regimes of say 30/26 °C or slightly greater, more representative of the plant's southern range mid-growing season temperatures.

We showed that production of containerized AWC can supply planting stock for future regeneration efforts and that white cedar not only tolerates but also thrives with heavy irrigation as long as plants are not flooded. Under experimental conditions, potted plants subjected to shallow water tables yielded as much or more growth than well-drained plants. Clearly, it would be difficult to overwater AWC in drained containers. Heavy watering would not impair growth, although it might lead to excessive leaching of nutrients. Conversely, applying too little water, resulting in a dry substrate, would probably reduce growth.

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CLIMATE SENSITIVITY OF ATLANTIC WHITE CEDAR AT ITS NORTHERN RANGE LIMIT

H. Myvonwynn Hopton and Neil Pederson[1]

Abstract—Atlantic white cedar is a wetland tree species with ecological and commercial importance that is distributed primarily along the Atlantic seaboard to south central Maine. The number of AWC swamps has declined due to human impacts. The potential for rapid climate change and AWC's threatened status make it important to study factors affecting its growth, especially climate. The objectives of this study are to determine the usefulness of AWC for tree-ring analysis, its sensitivity to climate along its northern range limit, and to study its growth rates. Seven AWC populations from northern New Jersey to southern Maine were found to be sensitive to changes in its environment. Growth was most commonly correlated to prior May and June, winter through spring, and current July and August temperatures. Decadal variations in temperature closely mirror variations in AWC growth suggesting that temperature is the primary limiting factor across the region from 1902 to 1995.

Keywords: Climate change, northern range limit, temperature, tree growth, tree productivity.

INTRODUCTION

Atlantic white cedar (AWC), [*Chamaecyparis thyoides* (L.)] ecosystems in the Northeastern United States are deemed threatened because of their rarity in the landscape. New Jersey's AWC population has decreased 74 percent from its estimated historic area (from 47,000 ha down to 12,100 ha) (New Jersey Department of Environmental Protection 2003). In the glaciated Northeast, only 5,300 ha of AWC swamp remain (Motzkin 1991). The primary causes of their net loss over the last two centuries are logging and habitat destruction (Laderman 1989).

Atlantic white cedar is important commercially and has been heavily logged since colonial times because of its workability and resistance to decay and insects. AWC has been historically used for shingles, barrels, and boats. Today it is still an important commercial tree in New Jersey, Virginia, the Carolinas, and Florida, and is often used for telephone poles, piling, ties, and siding (Little and Garrett 1990).

Atlantic white cedar ecosystems are ecologically important because they provide unique cover, habitat, and food for a variety of fauna. For example, a plant survey of a recently discovered AWC community in west central Georgia contributed significantly to the knowledge of rare plant occurrence within the region (Sheridan and Patrick 2003). The larva of the endangered Hessel's Hairstreak butterfly (*Callophrys hesseli* Rawson and Ziegler) feed solely on AWC leaves in Maine (Kluge 1991). White-tailed deer (*Odocoileus virginianus* Boddaert) preferentially eat AWC seedlings as a food source (Dickerson 2002). Therefore, studying, protecting, and managing AWC ecosystems is beneficial both for commercial and ecological reasons.

Atlantic white cedar grows along the Eastern Coast of the United States no more than 130 miles inland, with its northern range limit in south central Maine (Little and Garrett 1990). Because of its economic and ecological value and threatened status as an ecosystem, it is important to understand what factors limit the growth of AWC for future management and conservation. Studying AWC's climatic sensitivity is especially important in the face of potential rapid climate change although the species is not traditionally used in dendrochronological research. Nevertheless, Golet and Lowery (1987) found that changes in measured relative ring width could be explained by variations in water level in several Rhode Island AWC swamps. However, they concluded that their findings were wetland specific without a strong regional climatic signal. Pederson and others (2004) found AWC to be very temperature sensitive in southern New York State and northern New Jersey region. It is not known if this sensitivity can be extended to a regional scale.

The objectives of this study were to: (1) identify how well AWC crossdate (agreement in population's annual radial growth variations), (2) improve our understanding of the climate response of AWC, and (3) study its growth over the last 100 years. Specifically we tested whether the trees along their northern range limit show sensitivity to climate and if so, which climatic variables account for variations in annual ring widths. Although not often tested in temperate regions, it is thought that trees are more sensitive climatically at range limits. Research on loblolly pine (*Pinus taeda* L.) indicates that cool temperatures only became a growth-limiting factor at its northern range limit (Cook and others 1998). Therefore, we hypothesized that radial growth at the northern range limit for AWC was most limited by temperature of the previous and current growing season.

PROCEDURE

Increment cores were collected from seven different sites along the northern edge of Atlantic white cedar range limit:

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Appleton Bog, ME, Saco Heath, ME, Westminster, MA, Monson, MA, North Madison, CT, High Point, NJ, and Uttertown, NJ (fig. 1). To characterize forest composition at each site, four measurements of basal area (BA) using a cruise prism were taken at every fourth or fifth tree cored and averaged. Here we report only those species making up > 10 percent of stand BA (table 1).

Cores were collected and processed using standard dendrochronological techniques (Fritts 1976, Stokes 1968). Dr. Thomas Siccama and his students at Yale University collected cores from North Madison Cedar Swamp, Connecticut in 1988, 1991, and 1992. The cores were loaned to the Lamont-Doherty Tree Ring Lab for this study. Basal area measurements were not available for this site. From each of the other sites, cores were collected from at least 20 different trees using a hand-operated increment borer, except for Monson, MA due to its lack of old trees. Since climate response was the focus of this study, healthy dominant older looking trees were selected for coring so as to maximize the climate response. This non-random selection may not fully represent the stand-level climate response. However, some research suggests that competition can obscure the temperature signal in trees (Cescatti and Piutti 1998). Also, trees in declining health may be unresponsive to climate. Therefore, in keeping with the study's main goal we avoided sampling understory or unhealthy appearing trees. To maximize the geographic coverage while minimizing field and laboratory time, a single core was taken from each tree sampled. A second core was occasionally taken if the tree appeared old to increase sample replication and strengthen the cross-dating. This is less than the typical tree ring protocol of two cores per tree. However, tree replication is more efficient than core replication in reducing estimated mean standard error (Fritts 1976). Once the cores were extracted, they were stored in labeled plastic straws for transport back to the Lamont-Doherty Tree Ring Lab, Palisades, NY.

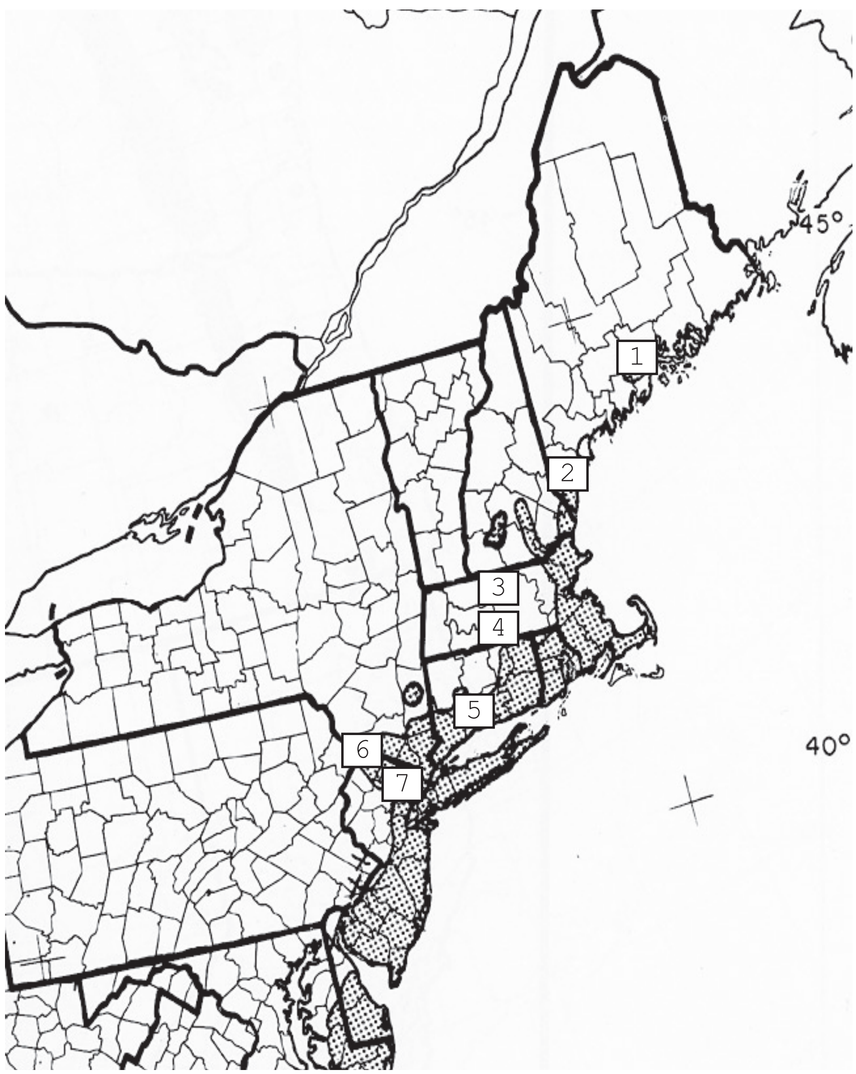

Figure 1—Atlantic white-cedar populations sampled in the Northeastern United States: 1 - Appleton Bog, ME; 2 - Saco Heath, ME; 3 - Westminster, MA; 4 - Monson Cedar Swamp, MA; 5 - North Madison Swamp, CT; 6 - High Point, NJ; 7 - Uttertown Bog, NJ. Stippled area represents the northern distribution of Atlantic white-cedar as adapted from Little (1971).

Table 1—Site information for the Atlantic white cedar stands sampled

Site	County	Latitude/longitude	Elevation	Stand BA (STD)
			m	*m^2/ha*
Appleton Bog, ME	Knox	N43°33' W70°28'	120	66.0 (15.3)
Saco Heath, ME	York	N44°20' W69°16'	40	35.5 (24.0)
Westminster, MA	Worcester	N42°32' W71°57'	250	47.3 (22.8)
Monson, MA	Hampden	N42°03' W72°18'	260	48.0 (6.9)
Madison, CT	New Haven	N41°21' W72°38'	80	—
High Point, NJ	Sussex	N41°38' W74°39'	460	54.8 (17.9)
Uttertown, NJ	Passaic	N41°10' W74°25'	300	79.2 (15.7)

BA = basal area.

At the lab, the cores were air-dried and then glued to wooden mounts. The cores were sanded with increasingly finer sandpaper up to 600 grit. The cores were then examined under a microscope and visually cross-dated. Rings were measured to the nearest 0.001 mm. Visual cross-dating was verified using the program COFECHA (Holmes 1983). Correctly dated time-series of growth were standardized using a double detrending method with the intent to preserve as much low frequency information as possible unrelated to competition (Cook and Kairiukstis 1990). First, a negative exponential curve or linear regression was used to remove geometric growth trends caused by the narrowing of rings as stem diameter increases. If a step change in growth was observed, a second detrending was done using a two-thirds spline to remove increases in growth related to changes in local competition (Lorimer and Frelich 1989). Standardized ring widths were then averaged to create an index of growth for each site.

Chronology signal strength was characterized using series intercorrelation (SNC) and the between tree expressed population signal (EPS). SNC indicates the strength of the common signal within a sample population and is derived from the correlation between all time-series of growth. EPS is a function of the mean correlation of all growth series within a population and sample size (Wigley and others 1984). It describes how well a finite sample size estimates the infinite, hypothetical population. These statistics are among the most commonly used indicators of agreement in year-to-year growth among trees within a population (Cook and Kairiukstis 1990).

The climatic sensitivity of each population was found by correlating the standardized ring index chronology against mean monthly temperature and total monthly precipitation from prior March to October of the current growth year. We chose a 20-month period of climate for correlation analysis as climate of the prior year can influence ring width of the current year (Fritts 1976). Because long-term meteorological records of minimum and maximum temperatures are lacking in the Appleton Bog region of ME, we chose to use gridded meteorological data from the Climatic Research Unit (CRU) of the University of East Anglia, UK (Jones 1994, New and others 2000). Grid points are located every 0.5 degrees and are interpolated climatic data from the eight nearest stations. For our purposes, data from the grid point closest to the sample site was used. Data is available from 1901 to 1995. Correlations were considered significant at $p \leq 0.05$ unless otherwise stated.

Regional Temperature and Growth Trends

To test if the climate signal in AWC is potentially regional, we compared a time-series of regional temperatures with a time-series of regional growth. Using CRU data, the months of temperature most commonly correlated to growth (prior May and June, prior November though current May and current July and August) were averaged using a mean and variance corrected arithmetic average procedure. This time-series covers 1902 to 1995 because 1 year is lost due to the combination of months from the prior and current years (e.g. May, June, November and December 1901 combined with January to May and July and August 1902). An arithmetic average was made of the standardized chronology of each population to create the regional time-series of growth.

RESULTS

Appleton Bog, ME

Appleton Bog is unique in that it is the northernmost known stand of AWC (Stockwell 1999). The forest floor is covered with *Sphagnum* moss and fern species. AWC dominates the canopy, comprising 77.3 percent of the basal area (BA), with occasional red maple [*Acer rubrum* L. (13.6 percent)] and black spruce [*Picea mariana* Mill. (12.5 percent)] present. Maximum tree age was 141 years with more than one-half of the trees older than 119 years (table 2). The SNC was r = 0.616, which is well above the 99 percent confidence level of 0.328 (Holmes 1983). EPS value for the Appleton Bog chronology was 0.929, which is above the accepted level of 0.85 (Wigley and others 1984). The standardized ring index chronology showed below average growth in the early 1920s and 1960s while above average growth occurred in the mid-1920s through the mid-1950s and from the mid-1970s until the mid-1990s (fig. 2A). Appleton Bog AWC was positively correlated with temperature during the prior May, June, September, November, and current January through April (table 3). These months accounted for 29.6 percent of the ring width variations. Only 3 months show positive correlation with precipitation: prior August, current March and July accounted for 8.80 percent of the growth variation.

Saco Heath, ME

Saco Heath is the only known domed-bog to support AWC (Laderman 1989). Saco Heath is composed of scattered aggregations of trees through out a dense shrub layer dominated by blueberry (*Vaccinium* spp. L.). In the forested areas, AWC is the dominant tree comprising 82.9 percent of the BA, while white pine (*Pinus strobus* L.) is present to a lesser extent at 12.7 percent. Maximum tree age at Saco Heath was 129 years with more than one-half of the trees older than 100 years (table 2). The SNC was r = 0.567 while the EPS was 0.933. The ring index chronology had growth trends similar to Appleton Bog. The early 1920s and 1960s were a decade of below average growth (fig. 2A). The low growth in the 1960s was followed by a period of unprecedented above-average growth that lasted to the present. March to August and November to December temperatures of the previous growing season, and January to February and April to August temperatures of the current growing season were positively correlated with growth (table 3) and accounted

Table 2—Statistical characteristics of the final Atlantic white-cedar chronologies[a]

Site	Trees	Cores	Interval	Median age[b]	Rbar	EPS	Number of samples EPS = 0.85	Date when sample depth = 0.85 EPS[c]
	- - - *number* - - -		*years*	*min/max*				
Appleton Bog, ME	21	29	1862 – 2002	119 (63/141)	0.375	0.929	10	1879
Saco Heath, ME	20	30	1874 – 2002	100 (69/129)	0.583	0.933	5	1879
Westminster, MA	26	29	1859 – 2002	111 (82/200)	0.378	0.922	10	1887
Monson, MA	16	17	1865 – 2002	116 (53/138)	0.531	0.926	5	1870
Madison, CT	21	22	1819 – 1992	142 (92/174)	0.608	0.974	4	1831
High Point, NJ	20	31	1807 – 2002	150 (104/196)	0.393	0.936	9	1830
Uttertown, NJ	20	20	1762 – 2002	125 (72/242)	0.257	0.863	17	1897

EPS = expressed population signal.

[a] Rbar is the average correlation between all trees. EPS is the expressed population signal. See text for more details.

[b] Tree ages are likely higher than shown since several sites had some rot, especially Westminster, MA and High Point, NJ.

[c] This column refers to the date when the sample depth equals the number of cores required to reach an EPS value of 0.85 for each chronology. Before this date, sample depth declines and it is expected that the EPS value would drop below 0.85.

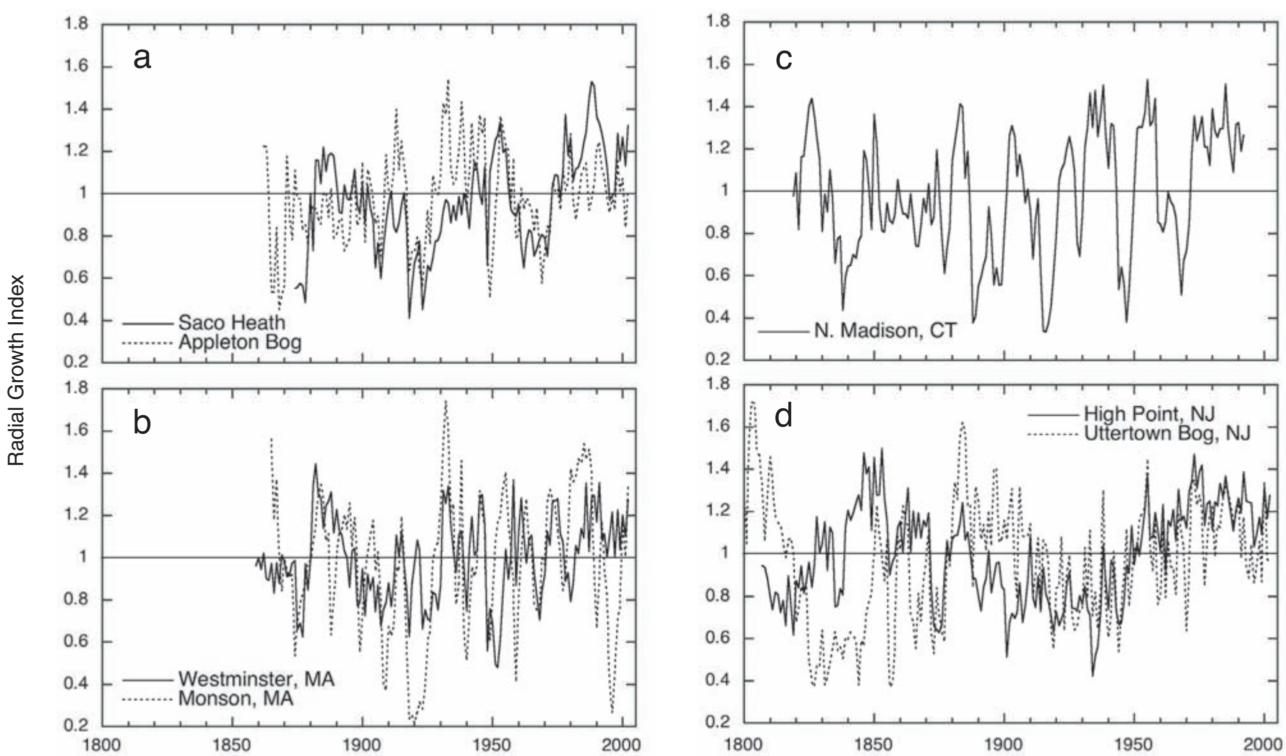

Figure 2—Standardized radial growth chronologies for: (a) Maine (b) Massachusetts (c) Connecticut, and (d) New Jersey. See text for details of each site within each State. Growth is standardized about a dimensionless index of one represented by the horizontal flat line.

Table 3—Significant correlation between Atlantic white cedar growth and mean monthly temperatures

Site	M	A	M	J	J	A	S	O	N	D	J	F	M	A	M	J	J	A	S	O
Appleton Bog, ME			**	**			**		*		**	*	*	*						
Saco Heath, ME	**	***	**	*	**	*			**	**	***	***		**	**	*	**	***		
Westminster, MA			**	**			**		**				*		*			*	*	
Monson, MA			**				*		*			*								
Madison, CT			**	**	*					*		*	*	*	*	*	*			
High Point, NJ		*				**			***	*	*	***		**			*	***		
Uttertown, NJ			*							*			**	*						
Number of sites with significant correlation	1	2	6	4	2	2	3	0	5	4	3	5	4	5	3	2	3	3	1	0

* = a significant relation between 0.01< and < 0.05; ** = significance between 0.001< and ≤ 0.01; *** = significance between < 0.001.

for 44.46 percent of the annual growth variation. Growth at Saco Heath was also positively correlated to mean monthly precipitation data for August and November of the previous growing season, and March to April and August of the current growing season. These months accounted for 27.68 percent of the growth variation.

Westminster, MA

The Westminster forest is comprised of AWC (60.3 percent BA), tamarack [*Larix laricina* (Du Roi) K.Koch (38.1 percent)], red maple (16.9 percent), and red spruce [*Picea rubens* Sarg. (14.8 percent)]. Maximum tree age in this population was 200 years with more than one-half of the trees older than 111 years (table 2). The SNC was r = 0.565 while the EPS was 0.922. Radial growth was generally below the average value between the 1900s and the mid-1950s (fig. 2B) and above the average value after the 1950s. Growth of the Westminster population was positively correlated to the previous year's May to June, September, and November temperatures, as well as current March, May, August, and September temperatures (table 3). Growth was positively correlated with precipitation of the current July and September. Temperature accounted for 30.54 percent of ring width variation, while precipitation accounted for 11.23 percent.

Monson, MA

At Monson, AWC comprises 42.2 percent of the BA, with 25.0 percent red spruce, 25.0 percent eastern hemlock (*Tsuga canadensis* L.), 18.8 percent eastern white pine (*Pinus strobus* L.), and 16.7 percent red maple. Monson has field evidence of logging within the last 70 years. Maximum tree age at Monson was 138 years with more than one-half of the trees older than 116 years (table 2). The SNC was r = 0.622 while the EPS was 0.926. Radial growth was generally below average growth from the 1900s to the mid-1950s (fig. 2B). The radial growth index showed a period of decreasing ring width from the 1890s to the early 1920s. Growth from 1970 through the mid-1980s was above average and has since decreased considerably. Growth in the Monson AWC population was positively correlated to the prior year's March, September, November and current February temperatures (table 3) and accounted for 16.37 percent of the ring width variations. Only a wet prior November was correlated to growth and accounted for 4.02 percent of the annual growth variation.

North Madison Cedar Swamp, CT

The North Madison Cedar Swamp is a late-successional bog-forest dominated by AWC, with scattered red maple (Andrews and Siccama 1995). Maximum tree age in the North Madison Cedar Swamp was 174 years with more than one-half of the trees older than 142 years (table 2). The SNC was r = 0.616 while the EPS was extremely high at 0.974. Radial growth was below average from the 1840s through the early 1920s, after which growth was above average except for a dip in the 1960s (fig. 2C). Growth in the North Madison AWC population was positively correlated to prior May to July and December, and current February to July temperatures (table 3). These months accounted for 19.49 percent of annual growth variations. A wet current October was positively correlated to growth accounting for 5.71 percent variance in ring width.

High Point State Park, NJ

High Point's AWC population is growing at the highest recorded elevation for the species (Laderman 1989). AWC makes up 73.9 percent of the total BA with eastern hemlock (13.7 percent) and red maple (10.9 percent) as the next most dominant trees. Maximum tree age in the High Point was 196 years with more than one-half of the trees older than 150 years (table 2). Several of these trees were hollow, and this prevented analysis of growth in older wood. Hence, age structure is a bit older than what can be reported here. The SNC was r = 0.549 while the EPS was 0.936. The standardized chronology shows a decline in ring width from the 1850s until the 1950s, after which ring width was above average (fig. 2D). This population was very sensitive to temperature (table 3), and growth was positively correlated to prior April, August, November, December, and current January, February, April, July and August. Temperature accounted for 37.54 percent of the ring width variation. Only 2 months, May and November of the prior growing year was positively correlated with precipitation and accounted for 11.43 percent of the ring width variation.

Uttertown Bog, NJ

Uttertown Bog is one of New Jersey's few remaining peatlands (Kuo 2003). AWC represents 80.3 percent of the BA, with eastern hemlock representing 18.0 percent. Maximum tree age in Uttertown Bog was 242 years with a majority of the trees between 110 and 140 years old (table 2). The SNC was r = 0.566 while the EPS was only 0.863. Radial growth increased from the 1860s until the 1910s, followed by a period of decline until the 1950s (fig. 2D). Following a slight dip in the 1960s, growth has increased since the 1970s. Uttertown was one of the least climate-sensitive populations studied here. Prior April and December and current January and April temperatures were positively and significantly correlated to climate (table 3) accounting for only 10.07 percent of the ring width variation. Prior May and December and current May and June precipitation were significant and positively correlated to growth. Precipitation accounted for 13.88 percent of the variation in ring width.

Regional Temperature and Growth Trends

There was a strong agreement between regional temperature and AWC growth (r = 0.66; $p < 0.0001$) from 1902 to 1995 with temperature accounting for 42.9 percent of variation in growth. The agreement is especially evident at decadal time scales (fig. 3).

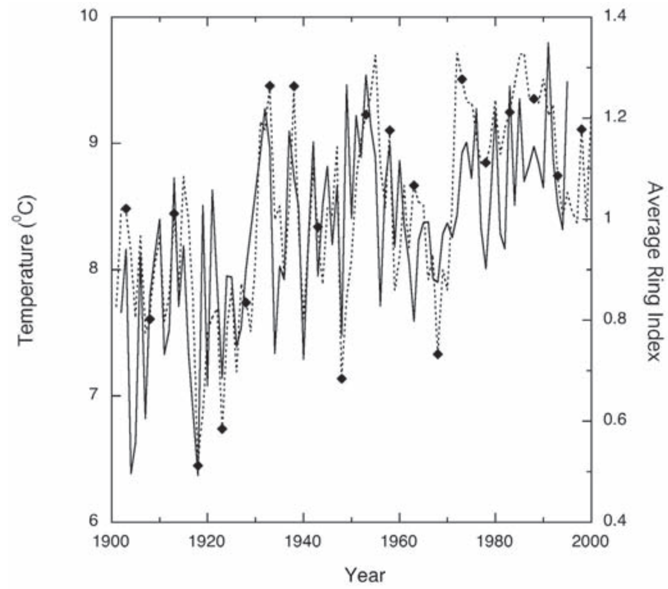

Figure 3—Regional temperatures (solid line) versus regional standardized growth of Atlantic white cedar (dashed line with solid diamond symbols). See text for further details.

DISCUSSION

Usefulness of AWC for Dendrochronology

Our results show that AWC growth is very sensitive to environmental conditions, making it useful for tree-ring analysis. The between-tree EPS was high (> 0.920) in six of the seven populations and suggests a strong common signal. Though not the focus of this study, individual trees from several populations showed possible release from competition that was likely related to canopy disturbance. Standard disturbance detection methods may be used to reconstruct stand history using AWC (Lorimer and Frelich 1989).

The primary limitation for the dendroclimatological use of AWC at the sites studied is its current age structure. Most stands sampled were < 130 years in age. The Uttertown site appears to be essentially an even-aged 120 year-old forest with a few scattered old trees. North Madison, CT and High Point sites had the least disturbed forests of the sites studied and yet no trees older than 200 years were found. If available, sub-fossil or relict wood samples could be analyzed in an attempt to extend stand disturbance history and chronology length. Given AWC's responsiveness to environmental variation, it is likely worth the effort and expense of relict wood recovery. Such a collection would greatly enhance our knowledge of long-term climate variability and AWC ecology in a heavily populated region, which may be useful for restoration of AWC ecosystems.

Climate Response

In general, AWC has a positive correlation to temperature at its northern range limit (fig. 4). Across all sites the monthly temperatures most frequently correlated with growth were prior May and June, prior winter through current spring (November to May), and current July and August temperatures (table 3). The strongest correlations between growth and temperature were during winter months. More than one-half of the sites sampled had levels of temperature sensitivity similar to the trees used for the only Eastern United States temperature reconstruction (Conkey 1982). Temperature accounted for 29.6 percent or more of annual growth variation at four sites making AWC one of the most temperature sensitive trees in the Eastern United States.

Of the sites sampled, correlations were strongest at Saco Heath, Appleton Bog, and Westminster Swamp, indicating that temperature is most important near the northernmost end of AWC's range limit. More southerly populations at North Madison and Uttertown had a lower sensitivity to temperature variation. These results are similar to the temperature

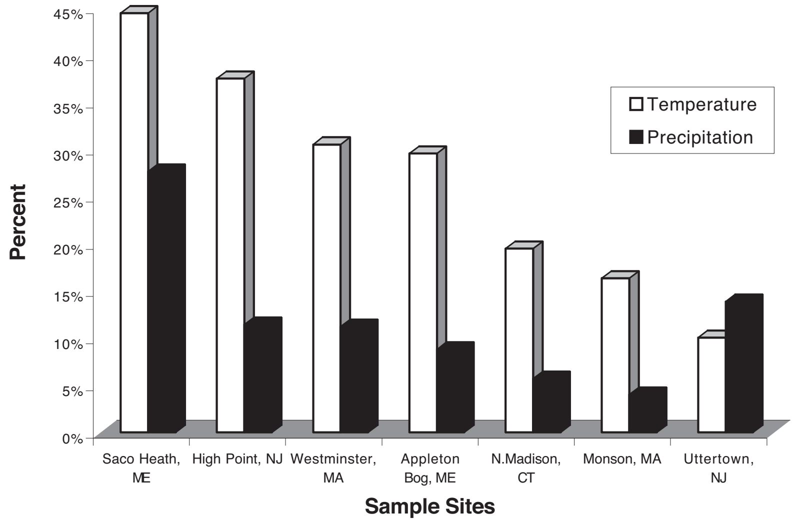

Figure 4—Sites are listed from left to right in the order of decreasing temperature sensitivity. The y-axis represents the amount of annual ring width variation that can be accounted for (r-squared) by temperature (open bars) and precipitation (black bars). See text for further details.

sensitivity for growth documented previously for loblolly pine (Cook and others 1998).

High Point, NJ is an exception to the geographic trend in temperature sensitivity. While High Point was one of the more southern sites sampled, AWC growth was one of the most sensitive to climate, but this population is growing at the highest known elevation for the species and likely experiences lower temperatures than other populations at that latitude. Using the CRU meteorological data, mean annual winter and summer temperatures from 1901 to 1995 for High Point fall close to those for Saco Heath, ME. Average summer and winter temperatures at High Point are 21.06 °C and -1.66 °C, respectively. At Saco Heath summer and winter temperatures averaged 20.06 °C and -3.66 °C. However, there is a significant difference between the elevation of the High Point AWC population (460 m) and the closest meteorological station most likely to have been heavily weighted in the CRU interpolation data (Point Jervis, NY, 143 m). Assuming an average environmental lapse rate of 0.65 °C cooling per 100 m in elevation, the mean summer and winter temperatures at High Point would actually be roughly 19.00 °C and -3.72 °C, respectively. Thus, the temperatures in which the High Point AWC trees live would fall within the range of temperatures of the Maine AWC sites, possibly explaining the exception in the geographic trend.

If AWC responds like other tree species, the reduced temperature sensitivity of populations at more southern sites may be the result of location in the species' range (Cook and others 1998) or greater competition with other plants (Cescatti and Piutti 1998). The current age structure at the Monson site suggests stands are still in the stem exclusion stage during which competition induces self-thinning and excludes tree recruitment (Oliver and Larson 1996). As a result, stand development at Monson may be contributing to AWC's reduced temperature sensitivity at this site.

The Uttertown site was the only site where growth was more sensitive to precipitation than temperature. Also, it was the southernmost site sampled. In addition to a warmer climate than experienced by the more northern populations, increased competition may cause tree growth to be more limited by precipitation as was shown in another tree species by Cescatti and Piutti (1998). While the Uttertown site has the highest estimated BA of the sites sampled, it is similar to the BA of Appleton Bog, near the species' northern range limit. More work is needed to determine if stand densities (a proxy for competition) within one climate region has an influence on AWC's climate sensitivity.

It is interesting to note sites that showed the highest sensitivity to temperature also showed the highest sensitivity to precipitation (fig. 4). The Saco Heath population showed the greatest overall sensitivity to climate. Since Saco Heath is a domed-bog, a unique wetland type for AWC (Laderman 1989), perhaps the site's physical characteristics along with its range position made it the most sensitive to climate. A domed-bog results from the build up of peat over time, which eventually serves to elevate the area from its surroundings so that it is perched above the water table. Surface water does not drain into domed bogs, making precipitation the primary source of water. The physical structure of a domed bog and the resulting hydrological regime would seem to explain Saco Heath's high sensitivity to precipitation. Saco Heath's low stand density (table 1) could also have been a factor in the high temperature sensitivity, following the same logic in the previous paragraph.

Growth Trends

Most sites showed similar decadal variations in growth over the last century, especially over the last 30 years indicating a growth trend for the region (figs. 2 and 3). The recent period of common increased radial growth suggests large-scale influences on growth, such as climate, nitrogen deposition or elevated CO_2. Temperature seems the most plausible explanation for much of the last 100 years. A decline in regional growth trends during the 1960s (fig. 3) in concert with declines in regional and global temperatures (Jones and Moberg 2003) suggest temperature is controlling radial growth. In fact, the CRU meteorological data for our study region and Northern Hemisphere data show that winter and spring temperatures have been rising over the recent decades (Jones and Moberg 2003, Lugina and others 2003) suggesting temperature may be linked to increased radial growth in recent decades.

Minimum temperatures have increased more rapidly than maximum temperatures (Karl and others 1993), which may be an important attributing factor to the increasing growth trend. AWC research of populations in the southern New York State and northern New Jersey region suggests growth is consistently correlated with minimum July and August temperatures (Pederson and others 2004). To date, we don't know if this local sensitivity to growing-season minimum temperature occurs at the regional scale as well.

The Monson site remained the exception to general regional patterns, and had a sharp decline in growth over the last 15 years. The lack of synchronicity in the Monson population may again be attributed to its disturbance history or inadequate sampling for detection of trends.

CONCLUSION

Atlantic white cedar growth is sensitive to changes in its environment. All populations tested showed strong interseries correlation and EPS values, indicating that the trees are responding to a common signal in their environment. More importantly radial growth in AWC at its northern range limit was positively correlated with monthly mean temperature data over the prior and current year. Atlantic white cedar growth also was positively correlated with variations in monthly mean precipitation data, but correlations were not as strong. The sites most sensitive to temperature were also found most sensitive to precipitation.

The forecast of future climatic warming does not appear to pose any immediate threat to growth of this species along its northern range limit. If temperature trends proceed as expected increasing 1.7 to 4.9 °C over the next century (IPCC 2001, Wigley and Raper 2001), growth would likely increase assuming moisture availability does not become a more important limiting factor. This could have implications for future ecosystem management and carbon sequestration modeling. However, how climate change will influence relative competitive ability is unknown.

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DISTRIBUTION OF ATLANTIC WHITE-CEDAR SEEDLINGS IN A NEW HAMPSHIRE SWAMP: ASSOCIATION WITH MICROSITE CHARACTERISTICS

Lara M. Gengarelly and Thomas D. Lee[1]

Abstract—The distribution pattern of Atlantic white-cedar seedlings was studied at Brown Mill Pond in Rye, NH, in order to identify which biological or physical factors were associated with seedling presence. Seedlings occurred on hummocks that rose above the water-filled hollows. However, some hummocks lacked seedlings and most others were only partly covered by seedlings. We chose 171 20 by 20 cm plots in which seedlings were surveyed as either present, absent (not present anywhere on the hummock), or missing (not present in the plot, but present elsewhere on the hummock). Elevation relative to the water table, percent canopy cover, substrate type, herb and shrub density, and distance to nearest probable parent tree were measured in each plot. To determine if seedling presence and absence could be predicted from the measured environmental variables, a standard discriminant function analysis was performed and explained 86 percent of the variance in cedar distribution. The most discriminating factors in explaining seedling presence were (1) moss-litter substrate rather than tussock sedge peat substrate, and (2) elevation within 30 cm of the water table.

Keywords: Atlantic white-cedar, environmental variables, New Hampshire, seedling distribution, substrate type, water table.

INTRODUCTION

Atlantic white-cedar [*Chamaecyparis thyoides* (L.) BSP] populations have decreased in number and size since colonial times, a pattern that has generated much concern for cedar conservation (Laderman and others 1987, Motzkin 1990, Mylecraine and Zimmermann 2000, Sperduto and Ritter 1994). Some of this loss has been attributed to inadequate recruitment under a closed canopy and subsequent successional change (Hickman and Neuhauser 1977, Korstian and Brush 1931, Little 1950, Motzkin 1990, Stoltzfus and Good 1998). However, a closed cedar canopy may not be the primary limiting factor to cedar establishment (Kuser and Zimmermann 1995). In fact, there are conflicting reports concerning the shade tolerance of Atlantic white-cedar (Hickman and Neuhauser 1977, Korstian and Brush 1931, Little 1950, Motzkin 1990, Stoltzfus and Good 1998). Other potentially limiting factors, including microtopography, soil moisture, and substrate may better explain the lack of successful cedar recruitment in some wetlands (Allison and Ehrenfeld 1999, Ehrenfeld 1995a).

As remaining populations of cedar are confined to public and private conservation lands, the decline of cedar due to inadequate seedling recruitment has become a management concern (Allison and Ehrenfeld 1999, Kuser and Zimmermann 1995, Motzkin 1990, Mylecraine and Zimmermann 2000). In fact, recently there has been interest in cedar's recruitment requirements and techniques for regenerating and restoring cedar populations (Ehrenfeld 1995b, Kuser and Zimmermann 1995, Mylecraine and Zimmermann 2000). There is still uncertainty in the habitat requirements for successful restoration.

Typically, natural Atlantic white-cedar swamps are defined by a network of elevated hummocks and frequently water-filled depressions or hollows (Ehrenfeld 1995a, Stoltzfus and Good 1998). Cedar commonly occurs on hummocks and it has been suggested that hummock microtopography, as it affects moisture availability, may be an important factor explaining how cedar seedling distribution occurs on hummocks (Ehrenfeld 1995a, 1995b). According to Ehrenfeld (1995b), cedar seedlings were most common at intermediate elevations on hummocks, avoiding the lowest and highest elevations. Perhaps seedling recruitment is unsuccessful at the top of hummocks and at the lowest elevations in the hollows because of drought and prolonged flooding respectively (Mylecraine and Zimmermann 2000). Moisture is considered one of the critical factors for Atlantic white-cedar regeneration (Laderman 1989, Little 1950). Both excessive and insufficient moisture may prevent germination and seedling growth (Kuser and Zimmermann 1995). According to Little (1950), moisture conditions were optimal for seedling growth if the water table was within 13 cm of the ground surface. In a more recent greenhouse experiment, growth of cedar seedlings was greatest in moist, drained soil; was intermediate in saturated soil; and least in inundated soil conditions (Allison and Ehrenfeld 1999).

In addition, Atlantic white-cedar recruitment is affected by the understory light regime (Little 1950). While cedar's shade tolerance is debated in the literature (Hickman and Neuhauser 1977, Motzkin 1990, Stoltzfus and Good 1998), Little (1950) observed a strong decline in cedar seedling survival when seedlings were beneath the heavy shade of a closed canopy. Other studies indicated that open seed beds free of competing vegetation are necessary for cedar establishment (Buell and Cain 1943, Korstian and Brush 1931).

The frequency of occurrence and growth of cedar seedlings is greater in organic peat soils, i.e., histosols with moss and

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litter substrate compared to other substrates (Allison and Ehrenfeld 1999, Laderman 1989, Little 1950) However, Haas and Kuser (1999) established cedar seedlings on a sandy mineral soil. These conflicting results illustrate that there is still uncertainty in our understanding of cedar germination and seedling establishment requirements with regard to substrate.

The objective of this research was to identify which biological or physical conditions were associated with Atlantic white-cedar seedling presence at Brown Mill Pond in Rye, NH. More specifically, this study investigated the presence and absence of seedlings among microsites in relation to elevation above the water table, percent canopy cover, hummock substrate type, shrub density, and distance to nearest probable parent tree.

METHODS

This study was conducted during the summer of 2000 in a Nature Conservancy-owned cedar wetland at Brown Mill Pond in Rye (Rockingham County), NH. The soils have been classified as a Chocorua mucky peat and hummock-hollow microtopography is well-developed (Kelsea and Gove 1994). The average difference between hummock tops and hollows was 52 cm $\pm$ 7, with a maximum difference of 85 cm (Gengarelly 1999). Cedar dominates some areas of this 45 ha wetland while in others it mixes with *Acer rubrum* L. (red maple), *Tsuga canadensis* (L.) Carr. (eastern hemlock), and *Picea rubens* Sarg. (red spruce). In previous fieldwork (Gengarelly 1999), the site was divided into five communities or stands based on tree species composition, cedar diameter, and cedar height.

The field survey of seedlings was conducted in the pond edge community, which borders Brown Mill Pond and its tributary, Bailey Brook (Gengarelly 1999). This area was selected because it was characterized by an uneven-aged cedar stand with continuous cedar establishment, a discontinuous cedar-red maple canopy and the highest water table on the study site (Gengarelly 1999). Two types of hummocks were present, categorized by the dominant surface ground cover (substrates). Tussock sedge hummocks were characterized by a tussock sedge (*Carex stricta* Lam.) substrate consisting of a network of vertical rhizomes intertwined with fine roots and decomposing organics, such as leaf litter (Lord and Lee 2001). Moss-litter hummocks were characterized by a carpet of mosses, including *Sphagnum* spp., *Dicranu*m spp., and other taxa, and areas lacking mosses; i.e., litter-covered substrate). Moss-litter covered hummocks are commonly described in other cedar wetlands, especially in New Jersey, and are referred to simply as peat hummocks (Ehrenfeld 1995a).

The sampling design included thirteen transects (~ 15 m in length), randomly located in the pond edge community such that they extended perpendicular to and away from the pond and brook. Hollows were not sampled because seedlings were never found in them. All hummocks with surfaces > 15 cm in elevation above the water level and within 2 m of either side of the transect were mapped, numbered, characterized for substrate type, surveyed for cedar seedlings, and the area was measured. Hummock area was determined by measuring the length and width of a hummock and using the ellipse formula (Area = π Length*Width/4). Any cedar displaying some scale-like foliage (suggesting that seedlings were at least in their second growing season) and a height between 5 and 30 cm was considered a seedling. The number of seedlings and substrate type was recorded for each hummock.

Along each transect approximately 15 randomly located 20 by 20 cm plots were established on the mapped hummocks. Plots fell into one of three categories: (1) seedling absent, (2) seedling missing, and (3) seedling present. New random locations on hummocks were generated until an equal number of plots of each type was sampled. *Seedling absent* plots were situated on hummocks that did not contain cedar seedlings. *Seedling missing* plots, while not containing any seedlings, were located on hummocks that supported seedlings elsewhere. All seedling missing plots were a minimum of 30 cm from any cedar seedling. *Seedling present* plots contained at least three cedar seedlings. Plots with one or two seedlings were omitted. In the end, 57 of each type of sampling plot; i.e., absent, missing, or present were observed.

Microsite Characteristics

Within each plot, selected environmental variables were measured. Percent cover of substrate types—tussock sedge, leaf litter, moss—was determined by projecting 100 dots over the 20 by 20 cm plot and recording the substrate intercepted by each dot. Densities of all herbaceous and shrub species were measured. In order to determine elevation, the vertical distance from the center of the plot to the water table was measured using a line level and meter stick. Elevations were adjusted to a single water table height in July (July 3, 2000). Thus, the July 2000 water table was used as the reference elevation. Percent open canopy was quantified using a digital camera with a fish-eye lens and images were processed with Gap Light Analyzer (Frazer and others 1999). Parent tree proximity was the mean distance between the center of the plot and the two closest reproductive adult cedars.

Statistical Analysis

In order to determine if plots with cedar seedlings, without seedlings, and missing seedlings could be predicted from the measured environmental variables, a standard discriminant function analysis was performed; i.e., all predictors entered in one step. The analysis was performed with SPSS 9.0 for PC (Norusis 1999). The seven predictor variables were: (1) elevation relative to the water table, (2) percent open canopy, (3) percent moss substrate, (4) percent leaf litter substrate, (5) percent tussock sedge substrate, (6) herbaceous and shrub density, and (7) distance to nearest probable parent. Three seedling groups were tested, with group membership established prior to the analysis and based on seedling plot type. Group 1 consisted of the seedling present plots (n = 57), group 2 consisted of the seedling missing plots (n = 57), and group 3 consisted of seedling absent plots (n = 57).

A standard multiple regression was performed between cedar seedling number per hummock as the dependent variable and hummock substrate type (tussock sedge vs. moss-litter), hummock area, and the appropriate interaction term; i.e., substrate type x area, as independent variables. In order to improve the normality and linearity of the residuals, square root transformations were used on the seedling number and hummock area measures. This analysis was made using SYSTAT 5.2 for PC (Wilkinson and others 1992).

RESULTS

Discriminant Function Analysis

The three seedling groups—present, absent, missing—differed in their relationship with the environmental variables as described by the discriminant functions. As there were three groups (seedling present, seedling missing, and seedling absent) two discriminant functions were created in this analysis. Together, these discrimant functions successfully explained 86 percent of the variance in the discriminant scores. Furthermore, each discriminant function alone explained a significant proportion of the variance in the discriminant scores ($r_c = 0.85$ and 0.70, respectively, $p < 0.001$).

The first discriminant function (DF1) discriminated among seedling distribution class based on plot substrate type: seedling-absent plots occurred most often on hummocks with tussock sedge substrate while the other two kinds of plots occurred most often on hummocks with either moss or litter substrate. High scores on DF1 were associated with both tussock sedge substrate (table 1) and the seedling absent group (fig. 1). Low scores were associated with moss or litter substrate (table 1) and both the seedling present and seedling missing groups (fig. 1). The second discriminant function (DF2) showed that on hummocks with moss-litter substrate, seedlings were more likely to be found at elevations within 30 cm of the water table than at higher elevations (table 1). High scores on this function were associated with greater elevations while low scores were associated with lower elevations. Points with high scores on DF2 and thus high scores on elevation were predicted to be in the seedling missing group (fig. 1).

Table 1—Discriminant function analysis of seedling presence[a]

Predictor variable	Standardized canonical discriminant function coefficients: Function 1	Function 2
Elevation relative to water table (*cm*)	0.170	0.958
Open canopy (*percent*)	0.016	-0.099
Moss substrate (*percent*)	-0.471	-0.251
Litter substrate (*percent*)	-0.482	0.103
Tussock substrate (*percent*)	0.378	0.043
Herb and shrub density	0.254	0.109
Distance to nearest parent tree (*cm*)	0.356	0.180

[a] Results of discriminant function analysis that predicted Atlantic white-cedar seedling groups (seedling present, seedling absent, seedling missing) at Brown Mill Pond, Rye, New Hampshire, based on several environmental predictor variables (July 2000). The standardized canonical discriminant function coefficients for each predictor variable and each discriminant function are presented.

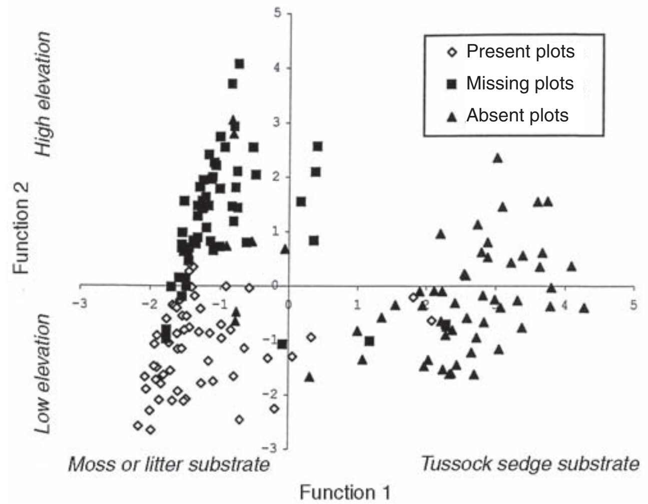

Figure 1—Discriminant function analysis testing predictability of Atlantic white-cedar seedling groups (seedling present, seedling absent, seedling missing) at Brown Mill Pond, Rye, NH based on several environmental variables (July 2000). Discriminant axes scores for all plots in each membership group (n = 57). The x-axis represents scores on discriminant function one. The y-axis represents scores on discriminant function two. Italicized labels indicate the predictor variable most strongly related to low or high scores on each function.

The discriminant analysis results were confirmed with univariate one-way analyses of variance (ANOVA) that tested each substrate variable across all three seedling groups (table 2). Seedling absent plots had significantly greater percent tussock substrate (mean = 76 percent) than either seedling present plots (mean = 3 percent) or seedling missing plots (mean = 3 percent) and mean percent cover of both moss and litter substrate were significantly lower for the seedling absent group than for the other seedling groups (table 2).

Univariate tests also confirmed that elevation was a predictor of seedling presence at Brown Mill Pond. The mean elevation of seedling missing plots was significantly greater (mean = 33.9 cm) than the elevations of the seedling present (mean = 17.6 cm) or seedling absent plots (mean = 19.0 cm) (table 2). A larger number of plots with cedar seedlings was located at low to intermediate elevations (10 to 25 cm) than at either the lowest (< 5 cm) or highest (> 30 cm) elevations on hummocks with moss-litter substrate (fig. 2A). While plots were less common at elevations < 10 cm, all four of these plots contained cedar seedlings (fig. 2B).

Regression Analysis

Only one of the independent variables contributed significantly to prediction of number of seedlings per hummock. Specifically, square root of hummock area had a standardized regression coefficient that differed significantly from zero (Beta = 0.68, t = 2.15, $p = 0.03$) (table 3) while coefficients associated with substrate type and the interaction did not differ significantly from zero ($p > 0.05$) (table 3). Mean area of moss-litter hummocks was 1.104 m^2, while mean area of tussock sedge hummocks was 0.349 m^2.

Table 2—Mean differences in predictors among seedling presence classes[a]

Predictor variable	Classification group mean: Seedling present (n = 57)	Seedling missing (n = 57)	Seedling absent (n = 57)	F	p
Elevation relative to water table (*cm*)	17.61a (5.30)	33.91b (8.47)	19.01a (9.32)	74.77	< 0.001
Open canopy (*percent*)	10.39a (3.32)	8.28b (2.38)	11.71a (3.78)	16.49	< 0.001
Moss substrate (*percent*)	35.14a (26.89)	20.21b (24.04)	2.25c (6.14)	34.66	< 0.001
Litter substrate (*percent*)	47.26a (30.26)	74.79b (28.44)	10.96c (28.47)	69.11	< 0.001
Tussock substrate (*percent*)	3.32a (17.35)	3.53a (15.98)	76.33b (38.15)	150.59	< 0.001
Herb and shrub density	2.93a (6.53)	4.46a (9.08)	51.11b (38.11)	81.29	< 0.001
Distance to nearest parent tree (*cm*)	64.61a (59.31)	82.55a (101.55)	203.02b (102.39)	39.84	< 0.001

[a] The group mean for each predictor variable used in a standard discriminant analysis that tested the predictability of Atlantic white-cedar seedling group membership at Brown Mill Pond, Rye, New Hampshire (July 2000). Standard deviations are reported in parentheses. F and p values are for the main effect of one-way analyses of variance comparing a predictor variable across all 3 seedling classification groups. Means with the same letter are not significantly different according to a Tukey's multiple comparison test (p < 0.05).

Table 3—Multiple regression of hummock area and substrate type[a]

Independent variable	Unstandardized coefficient (b)	Standardized coefficient (ß)	t-ratio	p
Square root of hummock area	0.024	0.682	2.15	0.03
Hummock substrate type	-0.678	-0.178	-1.08	0.29
Square root area x substrate type	-0.006	-0.188	-0.63	0.53
Constant	1.1663	0	1.29	0.20

[a] Results of standard multiple regression performed between Atlantic white-cedar seedling number per hummock as the dependent variable and hummock substrate type, area, and their interaction as the independent variables at Brown Mill Pond, Rye, New Hampshire (July 2000). Hummock area and seedling number were square root transformed prior to analysis.

DISCUSSION

Cedar seedling distribution was not random at Brown Mill Pond. Seedlings were absent from hummocks with tussock sedge substrate and present on hummocks with some alternative substrate such as moss or leaf litter. On the moss-litter hummocks, seedlings were present most often 10 to 25 cm above the water table and missing from the highest elevations (> 30 cm).

Substrate Type

In agreement with the work by Allison and Ehrenfeld (1999) cedar seedlings in this study appeared to prefer a peat-based substrate with overlying moss or litter more than a graminoid-based substrate with overlying sedge and grass. Historically, organic peat has been considered a suitable site for cedar seedlings (Little 1950) but few seedling surveys have addressed the distribution of established seedlings

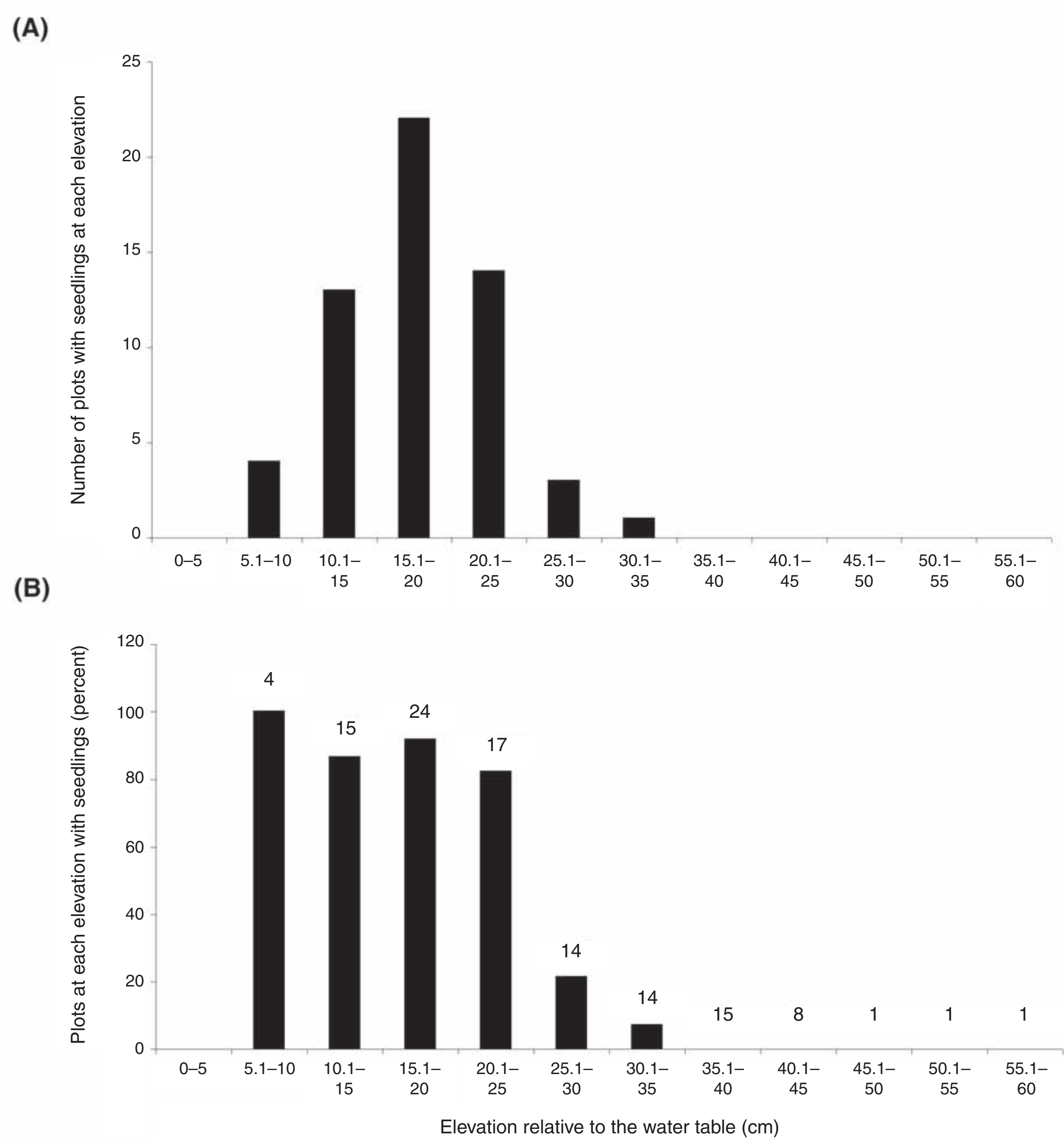

Figure 2—Results of Atlantic white-cedar seedling survey conducted in the pond edge community at Brown Mill Pond, Rye, NH. (A) The number of plots with cedar seedlings for each elevation class on moss-litter hummocks. Elevations were adjusted to a single water table height in July (July 3, 2000). (B) Percent of plots at each elevation with cedar seedlings. Number above elevations gives sample size of that class.

in relation to soil type. Although germination studies alone may not determine long-term establishment requirements, greenhouse experiments indicate stronger germination on peat moss than sand (Mylecraine and Zimmermann 2000). The factor explaining this difference in germination is still unknown (Mylecraine and Zimmermann 2000). Even though cedar seedlings appear to grow best in peat substrate, sphagnum moss, moist mineral soil, and rotten wood have also been reported as suitable cedar seedbeds (Mylecraine and Zimmermann 2000).

There are three possible explanations for a lack of cedar establishment on tussock hummocks relative to moss-litter hummocks. First, tussock hummocks may differ from moss-litter hummocks in elevation relative to the water table. However, elevations of tussock hummocks were statistically similar to elevations on moss-litter hummocks where seedlings were established.

Second, the unsuitability of tussock substrate for germination and growth was discounted during specific experimental tests in the field (Gengarelly 2003).

The third possible explanation is the most plausible: the lack of cedar on tussocks may be due to the relatively small size of tussock hummocks, which on average were 32 percent as large as moss-litter hummocks. Wind-dispersed cedar seed is more likely to encounter a larger hummock than a small one. The multiple regression analysis including hummock substrate type and hummock area offered support for this hypothesis, as substrate was not a significant predictor of seedling number per hummock when hummock area was included in the model (table 3). This analysis overall only explained 40 percent of the variance in seedling number per hummock, so hummock area may only partially explain the lack of seedlings on tussocks at Brown Mill Pond. Additional factors limiting cedar establishment on tussock hummocks remain unclear. The multivariate analysis suggested that neither density of competing herbs and shrubs nor percent canopy cover played a strong determining role in cedar seedling distribution.

Elevation Relative to the Water Table

At Brown Mill Pond, seedlings were less likely to occur at the lowest and highest elevations of the moss-litter hummocks. Most seedlings were found 10 to 25 cm above the water table. Similar to the pattern at Brown Mill Pond, Ehrenfeld (1995b) suggested that cedar seedlings were absent from lowest microsites, especially the bottom 20 cm of hummocks relative to the hollow surface, studied in New Jersey. In fact, a band of cedar seedlings at the intermediate zone was reported in New Jersey by Ehrenfeld (1995b) (Personal communication. 2001. J.G. Ehrenfeld, Professor, 126 Environment and Natural Resource Sciences Building, Cook Campus, 14 College Farm Road, New Brunswick, NJ 08901-8551) in sites where hummock height was large enough to include an intermediate elevation (~ 35 to 55 cm above the hollow surface). Similarly, Akerman (1923) indicated cedar survival was best at the mid-section of rotting stumps.

Ehrenfeld (1995a, 1995b) suggested that microtopography may affect cedar seedling distribution on hummocks through its effects on water availability. The lack of seedlings in the hollows is attributed to frequent flooding in these depressions (Allison and Ehrenfeld 1999, Mylecraine and Zimmermann 2000). In general, wetland woody species establish on elevated microsites, avoiding standing water (Huenneke and Sharitz 1990, Titus 1990). Conversely, the highest elevations of hummocks at Brown Mill Pond may be too dry for establishment of cedar, which requires sufficient moisture for survival (Allison and Ehrenfeld 1999, Little 1950).

Other Factors

Previous research offers conflicting evidence regarding competing vegetation and light requirements (Buell and Cain 1943, Hickman and Neuhauser 1977, Korstian and Brush 1931, Little 1950, Motzkin 1990, Mylecraine and Zimmermann 2000). According to Buell and Cain (1943) open seedbeds free of competing vegetation in North Carolina were optimal for cedar establishment. However, Korstian and Brush (1931) found that seedlings become established under the shade of shrubs. The present survey was conducted in a habitat where light probably is not limited.

Herbivory may also limit seedling presence and absence in certain parts of cedar's geographic range. In New Jersey deer browse has contributed to great losses of cedar seedling presence in many wetlands (Kuser and Zimmermann 1995, Zimmermann 1997). Although herbivory was not measured at this site herbivory appeared to be rare or uncommon at Brown Mill Pond during the course of this study.

Distance to the nearest probable parent tree was not as strong a factor as elevation and substrate in determining cedar seedling presence. Perhaps the proximity of seed source was not as important as other factors because the pond edge canopy was dominated by both Atlantic white-cedar and red maple (Gengarelly 1999). However, in a study that compared six wetlands, Allison and Ehrenfeld (1999) suggest that cedar establishment was associated with a cedar canopy that served as a dependable cedar seed source.

In general, disturbance (e.g., windthrow, commercial harvesting, fire, drought, or flooding) is expected to alter the available habitats in a wetland. For instance, extensive drought during the growing season may permit seedling establishment in the lowest elevations, typically areas devoid of seedlings due to standing water (Allison and Ehrenfeld 1999). Ehrenfeld (1995b) showed that sites with a history of more frequent windthrow had taller hummocks and the distribution of tree seedlings, including cedar, shifted to slightly higher elevations in these wetlands. Thus, if Brown Mill Pond experiences a disturbance in the near future, then the current seedling distribution patterns are likely to change.

CONCLUSION

This study describes the microsite conditions associated with the cedar seedling distribution at Brown Mill Pond. Specifically, cedar seedlings occurred in scattered clumps on moss-litter hummocks typically 10 to 25 cm above the water table. Seedlings were absent from tussock sedge hummocks; however, tussock sedge hummocks were smaller than moss-litter hummocks. These patterns suggested that seedling distribution may be controlled by moisture as a function of elevation and edaphic conditions associated with different substrates. However, these relationships are statistical associations and only manipulative field experiments that rigorously test the exact microhabitat conditions will determine the causal factors in cedar establishment.

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GEOGRAPHIC VARIATION IN ATLANTIC WHITE-CEDAR: PRELIMINARY PROVENANCE RESULTS

Kristin A. Mylecraine, John E. Kuser, George L. Zimmermann, and Peter E. Smouse[1]

Abstract—We present first year results from rangewide provenance plantations of Atlantic white-cedar [*Chamaecyparis thyoides* (L.) B.S.P.], established at (1) Richard Stockton College of New Jersey, (2) Brendan T. Byrne State Forest, New Jersey, and (3) Hofmann Forest, North Carolina. Each plantation includes rooted cuttings from thirty different source populations. During the first year, we observed reduced winter survival among extreme southern provenances (Georgia, Florida, and Mississippi populations) planted at Richard Stockton college. We also observed significant differences in height growth among planting sites and provenances, with stecklings planted at Hofmann Forest growing significantly more than those at the two New Jersey locations. Provenances originating from mid-latitudes performed best at all three planting locations. These results should be taken into account when choosing stock materials for regeneration and restoration throughout the species range.

Keywords: Adaptive variation, *Chamaecyparis thyoides*, Cupressaceae, growth, phenotypic variation.

INTRODUCTION

Significant decline of Atlantic white-cedar [*Chamaecyparis thyoides* (L.) B.S.P.; AWC] over the past two centuries (Frost 1987, Harshberger 1916, Kuser and Zimmermann 1995, Little 1950, Mylecraine and Zimmermann 2003) has led to much recent interest in its management and restoration in New Jersey (Haas and Kuser 1999, Mylecraine and Zimmermann 2003, Mylecraine and others 2003b) and throughout the species range (Bryant 1999, Eagle 1999, Hinesley and Wicker 1999, Laderman 1989). Several important factors need to be considered, including optimal site conditions to promote cedar regeneration, appropriate restoration techniques, and the type and source of propagules used. The geographic source material for introduction should be a major consideration (Handel and others 1994), as the ultimate success of a project may depend on choosing genetically adapted material (Lesica and Allendorf 1999).

Patterns of genetic variation and the degree of localized adaptation vary among species, and must be determined by analyzing both genetic and phenotypic data. Genetic markers, such as allozymes, microsatellites, and AFLPs (amplified fragment length polymorphisms) may be used to determine the amount of population and regional differentiation in a species, and we have used allozyme data to establish that significant regional variation is present within white-cedar (Mylecraine and others 2003a). It is well known, however, that neutral genetic markers may underestimate the degree of variation in morphological, growth and/or reproductive traits, as these latter traits will likely be more influenced by natural selection pressures (Lewontin 1984, Pfrender and others 2000, Wheeler and Guries 1982). Analysis of phenotypic variation and field performance of different provenances in different localities would be more relevant for determining potential restoration success.

Provenance variation is evident in AWC, although relatively few populations have been examined to date, and the extent and pattern of the differences remain unclear. Haas and Kuser (1999) found first year growth of the high elevation (457 m) stand at High Point, NJ, to be significantly lower than that of other source populations in the State, when grown in a common garden experiment in central New Jersey. A North Carolina population grew about the same as two of the New Jersey populations in the first year (Haas and Kuser 1999), but exceeded the New Jersey populations in height in subsequent years (Dugan and Kuser 2003). Jull and others (1999) examined growth rates of seedlings from six AWC provenances under controlled conditions and found differences in temperature optima among provenances. In another study, stratification requirements for seed germination varied among provenances (Jull and Blazich 2000). Summerville and others (1999) examined variation among several North Carolina populations from three different soil types and found that seedlings from parent trees occupying wet mineral soils were frequently the best performers on mineral sites. Conversely, seedlings from parent trees occupying organic soils were frequently among the best performers on organic sites, indicating some association between environmental and genetic factors within the species. Prior to the current study, no rangewide provenance trials had been established for AWC.

To examine geographic patterns of adaptive variation in AWC, we have established three rangewide provenance plantations, two in New Jersey and one in North Carolina.

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Our specific objectives are (1) to quantify the extent and pattern of provenance variation within AWC, and (2) to identify appropriate source populations for restoration in North Carolina and New Jersey.

METHODS

Populations

Between August 1999 and January 2001, we collected foliage material from approximately 20 individuals from each of 34 populations, throughout the entire range of Atlantic white-cedar. The locations of these populations are listed in table 1.

Propagation

From each individual, we made approximately 10 cm foliage cuttings, stripped the bark from the lower 2 cm, along one side of the branch, and dipped the branches in rooting hormone (Hormodin number 3, Geiger companies). We rooted these cuttings in pine cells (1.0 inch diameter by 6.3 inches long; Stuewe & Sons, Inc.) in a mistbed at the New Jersey Agricultural Experiment Station research greenhouse (Rutgers University), where they received mist for 6 seconds every 6 minutes until roots developed. After cuttings were successfully rooted, we applied a 20–20–20 fertilizer formulation about once a month. The cuttings grew fast enough so that we subsequently had to transplant many of them into larger pots (D40 Deepot Cells, 2.5 inch diameter by 10 inches long; Stuewe & Sons, Inc.), before outplanting.

Plantations

We have established three provenance plantations to date (fig. 1). We planted the first, located at the Natural Sciences and Mathematics (NAMS) arboretum at the Richard Stockton College of New Jersey (39°29 N, 74°33 W), during fall 2001. This planting is located on an upland site, with sandy soil, and is irrigated during the growing season. To deter white-tailed deer browsing, the site is surrounded by an electric fence, and individual cuttings are protected with individual tree shelters (12 inches wide by 36 inches tall; Terra Tech).

Table 1—AWC populations included in rangewide provenance plantations at Richard Stockton College of New Jersey (SC), Brendan T. Byrne State Forest, NJ (BB) and Hofmann Forest, NC (HF)

Abbr.	Population	Lat.	Long.	SC	BB	HF
ME-AB	Appleton Bog TNC Preserve, ME	44º20'N	69º16'W	X	X	X
ME-SH	Saco Heath TNC Preserve, ME	43º33'N	70º28'W	X	X	X
NH-RYE	Rye Cedar Stand, NH	42º59'N	70º47'W	X	X	X
MA-WH	Woods Hole, MA	41º32'N	70º39'W	X	X	X
CT-PF	Pachaug State Forest, CT	41º35'N	71º53'W	X	X	X
NY-CB	Cranberry Bog Preserve, NY	40º54'N	72º41'W	X	X	X
NY-SF	Sterling Forest, NY	41º11'N	74º17'W	X	X	X
NJ-BB	Brendan T. Byrne State Forest, NJ	39º53'N	74º30'W	X	X	X
NJ-BR	Bass River State Forest, NJ	39º38'N	74º26'W	X	X	X
NJ-BP	Belleplain State Forest, NJ	39º11'N	74º51'W	X	X	X
DE-MN	Middleford North TNC Preserve, DE	38º40'N	75º33'W	X	X	X
MD-AE	Arlington Echo, MD	39º04'N	76º37'W	X	X	X
MD-CC	Cypress Creek, MD	39º04'N	76º32'W	X	X	
MD-SC	Sullivan's Cove, MD	39º04'N	76º33'W	X	X	
VA-IP	Franklin, VA	36º35'N	76º54'W		X	
NC-DS	Great Dismal Swamp NWR, NC	36º32'N	76º28'W	X	X	X
NC-D	Alligator River NWR, NC	35º48'N	75º54'W	X	X	X
NC-BC	Bladen, Cumberland Counties, NC	34º50'N	78º44'W	X	X	X
NC-ML	Moore, Lee Counties, NC	35º19'N	79º11'W	X	X	X
NC-BR	Brunswick County, NC	34º05'N	78º20'W	X	X	X
SC-W	Waccamaw River, SC	33º54'N	78º43'W	X		X
SC-SH	Sand Hills State Forest, SC	34º33'N	80º04'W		X	X
SC-SP	Shealy's Pond Heritage Preserve, SC	33º52'N	81º14'W	X	X	X
SC-C	Cheraw State Park, SC	34º38'N	79º54'W			X
SC-G	The Bishop Gravatt Center, SC	33º44'N	81º35'W	X		X
SC-P	Poinsett Electr. Bombing Range, SC	33º47'N	80º28'W	X		X
GA-WC	Whitewater Creek, GA	32º28'N	84º16'W	X	X	X
GA-JC	Juniper Creek, GA	32º31'N	84º32'W		X	
FL-ONF	Ocala National Forest, FL	29º12'N	81º39'W	X	X	X
FL-ANF	Apalachicola National Forest, FL	30º08'N	84º53'W	X	X	X
FL-TH	Tates Hell State Forest, FL	29º51'N	84º51'W	X	X	X
FL-YR	Yellow River, FL	30º34'N	86º57'W	X	X	X
FL-BW	Blackwater River State Forest, FL	30º51'N	86º51'W	X	X	X
MS-BC	Bluff Creek, MS	30º32'N	88º41'W	X	X	X

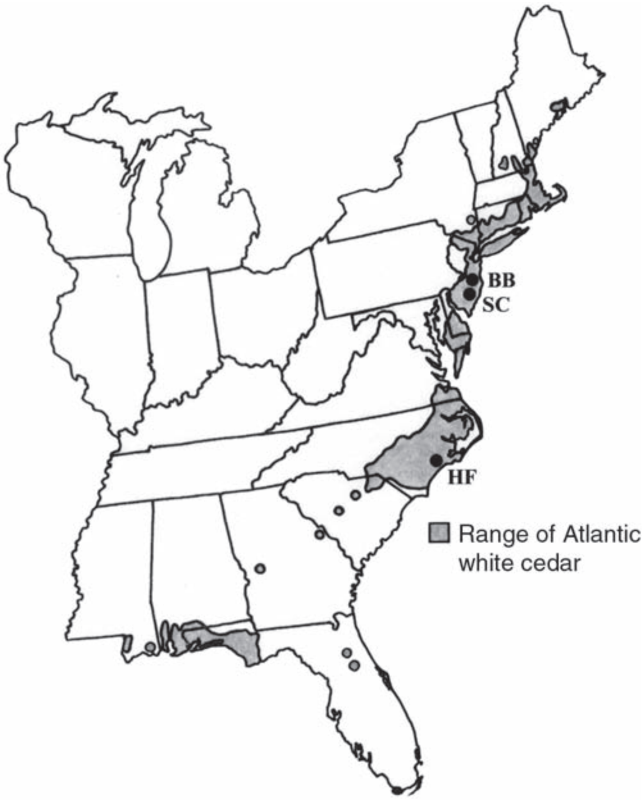

Figure 1—Range of AWC, redrawn and modified from Little (1971), showing the locations of three provenance plantations, at Richard Stockton College of New Jersey (SC), Brendan T. Byrne State Forest, NJ (BB), and Hofmann Forest, NC (HF).

We planted the second plantation within Hofmann Forest, North Carolina (34°52 N, 77°18 W), during spring 2002. This is a lowland site, with organic soil, and is surrounded by loblolly pine plantations. Prior to planting, all existing vegetation was removed. Although there is little evidence of deer damage in the area, we have again protected the plantings with individual tree shelters. We established the third plantation at Brendan T. Byrne State Forest, New Jersey (39°56 N, 74°28 W), in spring 2002. This plantation is embedded within a 20 ha Atlantic white-cedar mitigation project (Mylecraine and others 2003b) and occupies an organic sandy soil.

Experimental Design

For each of the three planting locations, we selected a total of 30 provenances to include in the design (table 1). Twenty-five provenances were included at all three planting locations, but the remaining five provenances varied across sites due to the availability of sufficient rooted material. Each site was divided into three replicates, within each of which we established six blocks. Within a single replicate, we planted cuttings into a randomized block design, with each block containing one steckling from each of 30 populations, randomized (as single-tree plots) within the block. We planted ramets from six different genets from each provenance, a different genet for each of the six blocks. Thus, we had 180 genotypes (6 from each of 30 provenances) planted within a replicate, each genotype replicated once. The entire design was then replicated three times per site. Overall, each of 6 genets is represented 3 times at each site, for a total of 18 stecklings from each provenance. We planted stecklings at 1.8 m by 1.8 m spacing, with 2.4 m between blocks. The physical layouts of the planting designs varied slightly among the three locations due to site-specific spatial configurations.

Survival and Growth Measurements

For the Richard Stockton college plantation, planted during fall 2001, we were able to assess first winter survival. The other two plantations were planted in spring, so we were not yet able to assess winter hardiness. For all three plantations, we monitored first season survival and growth. We measured cutting height at the time of planting and again at the end of the first growing season (2002), to calculate first season height growth. We performed four separate analyses of variance (ANOVA) using SAS (SAS Institute) to test for differences in growth among planting sites and provenances. If the data did not meet the normality assumption for ANOVA, we used a log transformation. First, we performed a two-factor, combined site analysis to test the following effects: site, provenance, and site*provenance. For this analysis, we included only the 25 provenances that were planted at all three sites. We treated site as a fixed effect and provenance as a random effect, and conducted the analysis using PROC MIXED. To further investigate site differences, we performed two orthogonal contrasts, comparing (1) Hofmann Forest vs. the two New Jersey plantations, and (2) Brendan T. Byrne State Forest vs. Richard Stockton college. Second, we conducted three individual site one-factor ANOVAs to assess provenance differences within each site. To examine the relationship between mean height growth and the latitude of provenance origin, we performed regression analyses for each planting location.

RESULTS

Survival

Over the first winter (2001–2002), we observed variation in survival rates between provenances at the Richard Stockton college plantation (fig. 2). Survival among northern and central provenances (Maine to Maryland) was high, ranging from

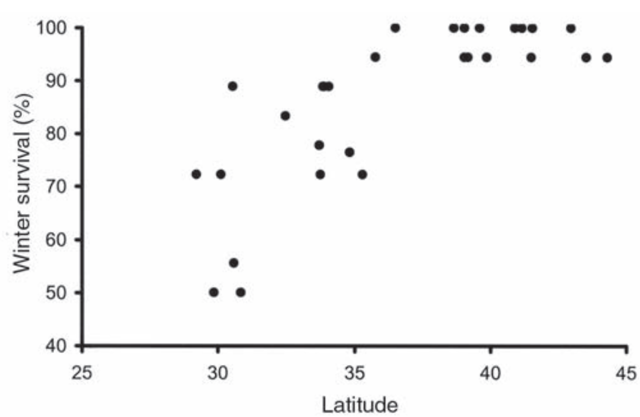

Figure 2—Percentage of cuttings surviving through the first winter (2001–2002) at Richard Stockton College of New Jersey, for 30 AWC provenances, plotted by latitude of source population.

94 to 100 percent. There was considerable variation in survival rates among North Carolina and South Carolina provenances, ranging from 66.7 to 100 percent. Among southern provenances (western Georgia, peninsular Florida and Gulf Coast populations), survival was reduced, ranging from 44 to 83 percent. These data suggest a non-linear relationship between latitude and first season survival (fig. 2). We did not observe any additional mortality through the first growing season. The remaining two plantations (Brendan T. Byrne State Forest and Hofmann Forest) were planted in the spring, so we were not able to assess winter survival. However, survival rates through the first growing season were high for all provenances (ranging from 89 to100 percent) at both sites.

First Season Height Growth

Results of the combined site ANOVA indicate significant planting site, provenance, and site*provenance effects (table 2). All populations grew substantially more at the Hofmann Forest, North Carolina site than at the two New Jersey sites, as indicated by the highly significant NJ vs. NC contrast. The significant site*provenance (genotype by environment) interaction suggests that the relative performances of provenances varied across sites, and requires that we examine the provenance pattern at each site individually. At all three sites, we found highly significant ($p < 0.0001$) within-site differences among provenances. Figure 3 depicts mean provenance growth at each site, plotted by latitude of provenance origin. The overall growth pattern was similar across sites. Height growth was greatest for populations from the southern and mid-Atlantic Coastal Plain, with North Carolina populations among the best performers at all three sites. Florida and Gulf coast populations exhibited reduced height growth, particularly at the two New Jersey plantations. New England populations also exhibited reduced growth, compared to those from the southern Atlantic Coastal Plain. We found significant polynomial relationships between growth and source latitude at both Brendan T. Byrne State Forest and Hofmann Forest, and a similar, but non-significant ($p = 0.072$) trend at Richard Stockton college (fig. 3).

DISCUSSION

This study represents the first rangewide provenance test of Atlantic white-cedar. The newly established provenance plantings are obviously in their early stages of development, and must therefore be viewed as a “work in progress”; this project will continue for several years to come, because geographic variation in growth and survival patterns is influenced by inter-annual climatic variability and may take years to manifest itself in long-lived tree species.

We did, however, find significant variation in both first season survival and growth among different white-cedar provenances. Survival through the first winter at Richard Stockton College of New Jersey was much reduced for southern populations, indicating that either the site or the climatic factors were not optimal for these populations. Survival in subsequent years, at all three planting locations, will be important in making the distinction between these two possibilities. We also observed significant variation in height growth among planting sites and source populations. North Carolina populations were among the best performers at all three sites. The growth patterns agree with the earlier work by Haas and Kuser (1999) that North Carolina populations may outgrow native New Jersey populations in common gardens. However, the growth differences among provenances were only slight after the first year, and it may take several years for these patterns to fully develop.

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Table 2—Results of the combined site Analysis of Variance (ANOVA) and orthogonal contrasts, conducted on log-transformed steckling height growth at all three planting sites. Site is treated as a fixed effect, and provenance is treated as a random effect

Source	df	SS	MS	F	P
Site	2	635.6	317.8	662.9	<0.0001
NJ vs. NC	1	630.6	630.6	1309.6	<0.0001
RS vs. BB	1	1.0	1.0	2.1	0.1545
Provenance	24	52.1	2.2	4.5	<0.0001
Site*Provenance	48	23.1	0.5	1.5	0.0179
Error	1189	384.4	0.3	—	—

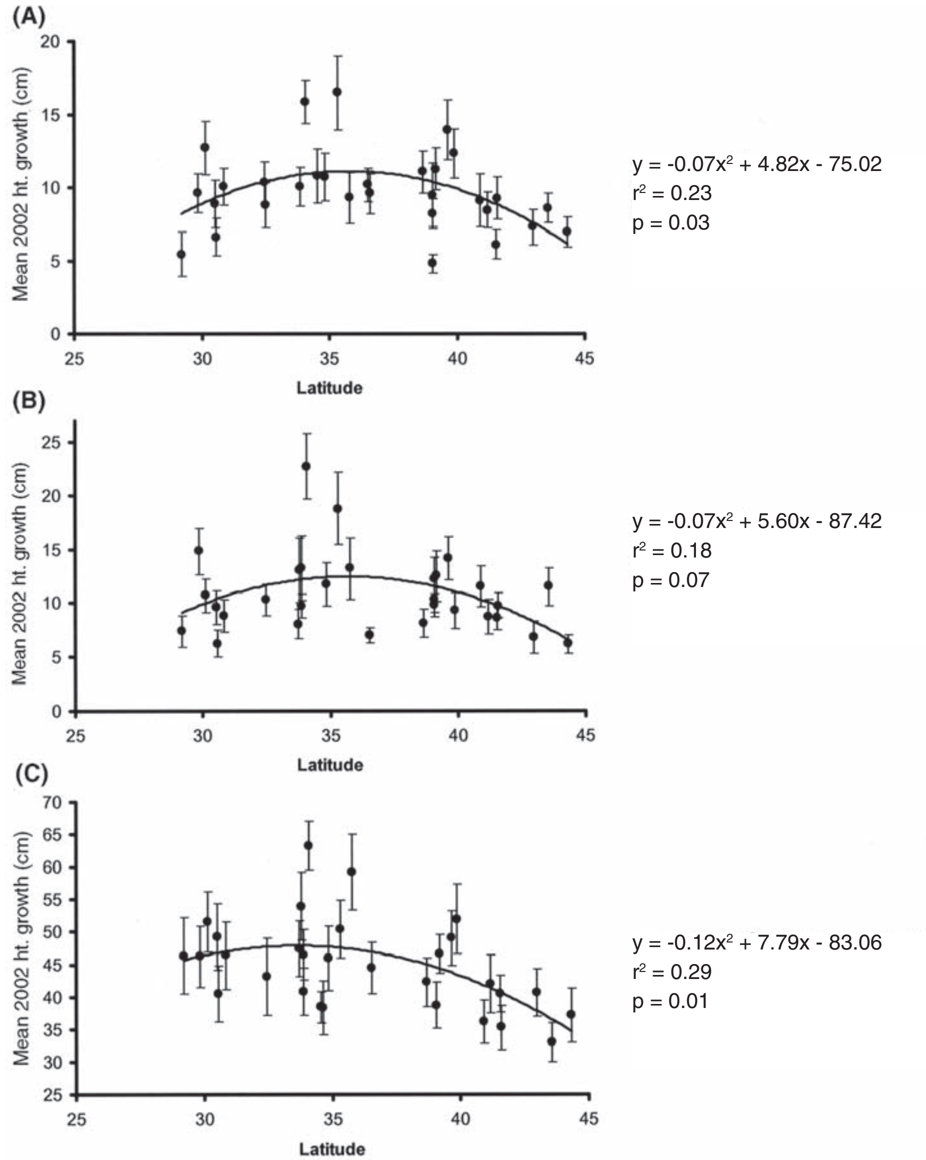

Figure 3—Mean first season height growth for 30 AWC provenances planted at (a) Brendan T. Byrne State Forest, NJ, (b) Richard Stockton College of New Jersey, and (c) Hofmann Forest, NC (HF), plotted by latitude of origin.

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LOCATIONS OF ATLANTIC WHITE CEDAR IN THE COASTAL ZONE OF MISSISSIPPI

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Abstract—Atlantic white cedar [*Chamaecyparis thyoides* (L.) B.S.P. cedar] is a tree species found in freshwater wetlands along a coastal band from Maine to Mississippi. Numerous studies have documented the occurrence of this species along the Atlantic seaboard, but relatively few studies have been conducted along the Gulf of Mexico coast. Nine stands in Florida have been quantitatively described in the literature, but only one stand in Mississippi and none in Alabama have been so described. Other cedar locations in Alabama and Mississippi have been mentioned, but these sites have not been inventoried. In Mississippi, cedar is considered imperiled, but it is currently unknown whether the existing populations are increasing or decreasing. The one stand described for Mississippi, located near Vancleave, in Jackson County, is degrading as a result of hydrologic regime modification and urbanization. Through contact with the Mississippi Natural Heritage Program, limited fieldwork and herbarium searches, we have identified an additional 22 sites where cedar occurs as at least individual trees. However, the size and species composition of these additional cedar stands is unknown. The discovery of these additional sites is encouraging and may suggest that the species is more common than previously thought.

Keywords: Atlantic white cedar, *Chamaecyparis thyoides*, Gulf Coastal Plain, Mississippi, plant community composition.

INTRODUCTION

The native range of Atlantic white cedar [*Chamaecyparis thyoides* (L.) B.S.P.] extends from Maine down the Atlantic coast to Florida and west along the Gulf of Mexico coast to Mississippi (Little 1950, Laderman 1987, 1989). The majority of cedar stands are found along the Atlantic coast, with fewer stands along the Gulf of Mexico coast, and a very few disjunct individuals or stands in Illinois, Indiana, and Ohio (Coulter, n.d., Patterson 1876, Schaffner 1932, Mohr 1901). The origin of these disjunct stands is unknown, as is whether they are natural populations or were planted by early settlers. Also unknown is whether cedars are regenerating at those sites.

Korstian and Brush (1931) observed large stands of cedar in Alabama, Florida, Georgia, and Mississippi, but these stands were not quantitatively described and subsequent logging may have degraded or eliminated some of the stands. More recently, one stand in Mississippi (Eleuterius and Jones 1972) along Bluff Creek near Vancleave, Jackson County and nine stands in Florida (Ward and Clewell 1989) have been quantitatively studied and published in the literature. No stands in Alabama have been so studied. Ward and Clewell (1989) presented the most comprehensive description of known cedar locations along the Gulf of Mexico coast, but the majority of stands they discussed included only a general location and little community composition data. In addition to the stand along Bluff Creek, Ward and Clewell (1989, page 11) report that in Mississippi cedar occurs along "...tributaries of the Escatawpa River and Pascagoula River, Jackson County; and branches of the Catahoula River, Pearl River County." Ward and Clewell (1989, pages 10-11) further list cedar as occurring in "...headwater extensions of the Escambia River and Perdido River into Escambia County, Alabama; and tributaries entering the Perdido River, Perdido Bay, and Bon Secour Bay, Baldwin County, Alabama." This sparse occurrence of gulf coast cedar may reflect how few large stands remain. Since little is known about cedar along its southern range, documenting the extent, health, and environmental condition of these populations is important.

Habitat degradation and conversion to other uses, such as agriculture, are serious problems across the entire range of cedar (Little 1950). Many stands identified during the early part of the 20th century were cut for lumber production (Korstian and Brush 1931). In Mississippi, some stands have been degraded by urbanization, erosion and/or land use changes (Personal communication. 2000. Mr. Joseph Cruthirds, Professor, Delgado Community College, 615 City Park Ave., New Orleans, LA 70119). Also, degradation of cedar swamps can result from manipulations of the surrounding habitat, which is known to have significant effects on nutrient cycling (Zhu and Ehrenfeld 1999) and plant community composition (Ehrenfeld and Schneider 1991) in these stands.

Information about management of cedar stands in the Southeastern United States is lacking. Although some information on regeneration of cedar is available for the Atlantic coast (Boyle and Kuser 1994, Carter 1987, Kuser and Zimmermann 1995, Little 1950), we cannot assume that gulf coast cedar swamps will function in a like manner because of differences in species composition, soils, climate, and a host of other factors. For example, Eleuterius and Jones (1972) found only 20 percent of the plant species in a Mississippi cedar stand to be in common with Atlantic coast stands. Cedar stands in Florida had 20 percent of the plant species in common with Mississippi and 16 percent in common with Atlantic coast stands. This lack of similarity in species composition between regions indicates functional and physical differences that

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emphasize the need for more study of gulf coast cedar populations.

The future health and sustainability of cedar populations in the Southeastern United States are dependent upon achieving a greater understanding of the distribution, landscape position, abundance, and population dynamics of the existing stands. This is especially true for the western extent of its range. The objective of this manuscript is to take the first step and report on the known locations of cedar along the Mississippi Gulf Coast.

METHODS

Locations of individual cedar trees or stands along the Gulf of Mexico Coast were determined from a variety of sources. The Mississippi Museum of Natural Science database provided the greatest number of locations in the form of field survey summaries (Personal communication. 2003. Ron Wieland, Botanist, Mississippi Museum of Natural Science, 2148 Riverside Drive, Jackson, MS 39202). For each stand listed in the database the location, dominant and codominant species and evidence of regeneration were listed. A few additional stands were located through contacts with numerous private individuals and Federal/State/NGO offices. Potential sites were visited and preliminary data gathered through casual observation. Lastly, some locations were identified through boat and land surveys conducted by the authors during 2002 and 2003.

RESULTS AND DISCUSSION

Historic Distribution

The natural distribution of cedar does not currently extend west of the Pearl River, even though environmental and soil conditions between the Pearl and Mississippi rivers appear to be appropriate for the successful growth and regeneration of the species (McCoy and others 1999, 2003). Apparently a population of cedar existed along Little Bayou Sara in West Feliciana Parish, LA, when warm temperate deciduous forest species dominated the area between 7,600 and 3,200 years B.P. (Brown 1938, Givens and Givens 1987). This site is about 135 km west of the Pearl River. The cause of cedar extinction in this area is unknown.

Current Stand Locations

A list of locations from the literature or unpublished data for Mississippi and Louisiana is included in table 1 and mapped on figures 1 and 2. A preliminary list of co-occurring tree and shrub species, family, and common names is presented in table 2 and cross referenced to locations given in table 1.

We found numerous sites with scattered cedar trees along the lower portions of both the Escatawpa and Pascagoula rivers in Jackson County, Mississippi. A small stand can be found along the lower Escatawpa River, on the north side of Interstate 10. The stand is located on Grand Bay National Wildlife Refuge and is between the Mississippi Welcome Center and the Escatawpa River bridge. Tree and shrub species associated with this cedar stand include sweetgum, live oak, baldcypress, red maple, Eastern red cedar, Chinese tallow tree, American witchhazel, yaupon, American holly, blueberry, and Hercules' club (see table 2 for scientific names). Both sensitive and royal ferns were also observed at this site. A portion of the stand was damaged by a fire in 2001 and several cedar trees were killed. The most damage occurred on higher ground that grades into a longleaf pine stand.

We found an additional stand along the bank of Bluff Creek, downstream of the Vancleave site, on the Mississippi Sandhill Crane National Wildlife Refuge, in Jackson County. Plant species associated with cedar were slash pine, black tupelo, water oak, sweetgum, red maple, Florida anise tree, swamp titi, large gallberry, deerberry, yaupon, clammy azalea, dwarf palmetto, summer grape, pitcherplant, and cinnamon fern. The stand consisted of about 50 cedar trees that ranged from 2 to 30 m in height. The largest cedar was 48.2 cm diameter at breast height (d.b.h.). The northern end of this small stand supported numerous seedlings and a few suppressed saplings.

Another new cedar stand, located in the Black Creek Swamp, near Hurley, Jackson County, Mississippi, is part of a wetland mitigation bank being developed by Wetlands Solutions, LLC under regulatory authority of the U.S. Army Corps of Engineers. Although cedar is a dominant canopy species in portions of this site, often with trees up to 35 m in height, the majority of the area is dominated by slash pine, canopy-sized black titi, swamp bay and sweet bay. Big-leaf gallberry was a common shrub in the area while sphagnum moss (*Sphagnum* sp.) and various ferns (royal, cinnamon and chain ferns) were important in the ground layer. Some visibly stressed and a few standing dead cedar were noted in the area. During our preliminary field trip in April 2002, we noted only two or three small clusters (< 100/m^2) of first-year cedar seedlings. Although no attempt was made to quantify shrub-sized cedars, few < 2 m tall were noted. This lack of regeneration is expected under a dense canopy (Korstian and Brush 1931, Laderman 1989). Sphagnum was present over an organic soil, but the leaf litter was primarily broadleaf.

We have not yet been able to verify the stands reported along branches of Catahoula Creek by Ward and Clewell (1989), but we have located a series of small stands scattered along Juniper Creek, a tributary of the East Hobolochitto Creek and the Pearl River in Pearl River County, just south of Poplarville, MS. These stands are just northwest of the headwaters of Catahoula Creek and may be the stands referred to by Ward and Clewell (1989). The limited portion of these stands that we have seen consists of bottomland hardwood forest with scattered cedar in low, wet areas, on slightly elevated areas within a braided stream area, or along the toe of the slope at the edge of the floodplain (see Messina and Conner 1998 for a description of toe-of-the-slope communities). Additional species include sweetbay, redbay, loblolly pine, black tupelo, baldcypress, swamp tupelo, and swamp titi. These stands are protected at the discretion of the forest manager for Weyerhaeuser, Inc. (Personal communication. 2003. Tina Knoll, Forester, Weyerhaeuser Inc., 211 Armstrong Road, Columbia, MS 39429). However, even without activity within the cedar stands, silvicultural activity in the surrounding slash pine plantations may impact the cedar stands: degradation of cedar stands may occur as a result of fertilizer and other chemicals draining into the bogs, and a changed hydrology from drainage to enhance slash pine growth.

Table 1—Locations of Atlantic white cedar individuals and stands in Mississippi (including data on Louisiana plantations). Many of these locations represent old collections with the current status of the tree or site unknown. Most of the data was obtained from the Mississippi Natural Heritage Program (MNHP) or observed during 2002. The general location of these sites is shown on figure 2. Data in the last column cross-references to species lists for each location, given in Table 2

Site location	Notes	Citation and/or last observation noted	Table 2
Forrest County, MS	Camp Shelby training site; up to 29.6 ha in size; seedlings present; gum-bay-pine site; extends into private land near Davis Creek a tributary of Black Creek.	MNHP 1997,1998 Observed 2002	L,M
George County, MS	Not common; .5 km west of Howell, MS, west of MS/AL State line in Juniper bay.	MNHP 1978	O
Hancock County, MS	Near Hwy 603 over the Jourdan River.	MNHP 1967	
Jackson County, MS	[1]Beside Bluff Creek on natural levee. [2]Largest cedar stand in Mississippi; up to 60 feet above creek; stand ½ mile wide. [3]Quantitative description of the stand. [4]Cedars extend 11 km with a maximum width of 0.8 km.	Observed 2002 [1]MNHP 1968 [2]Jones 1967 [3]Eluterius and Jones 1972 [4]Ward and Clewell 1989	B
Jackson County, MS	A nice stand of cedar at Vancleave, MS, along a small tributary of Bluff Creek.	Jones 1967 MNHP 2002	C
Jackson County, MS	Moss Point Lumber Mill on Escatawpa River at pipeline crossing (Levee).	MNHP 2002 Observed 2002	D
Jackson County, MS	Moss Point Lumber Mill on Escatawpa at pipeline crossing (Marsh).	MNHP 2002 Observed 2002	E
Jackson County, MS	Escatawpa River 1.6 km above Moss Point Lumber Mill (Marsh).	MNHP 2002 Observed 2002	F
Jackson County, MS	Escatawpa River 1.6 km above Moss Point Lumber Mill (Levee).	MNHP 2002 Observed 2002	G
Jackson County, MS	Escatawpa River at Moss Point Lumber Mill (Levee along SW—most pipeline).	MNHP 2002 Observed 2002	H
Jackson County, MS	East bend on Bluff Creek just West of Paige Lake, Gautier Unit—MS Sandhill Crane NWR.	MNHP 2002 Observed 2002	I
Jackson County, MS	West bend of Bluff Creek .8 km west of Paige Lake, Gautier Unit—MS Sandill Crane NWR.	MNHP 2002 Observed 2002	J
Jackson County, MS	Hook on Bluff Creek at north boundary of Sec 34, Gautier Unit—MS Sandhill Crane NWR.	MNHP 2002 Observed 2002	K
Jackson County, MS	Welcome center nature trail at I-10. About 100 cedars with some dead from a recent fire. Few seedlings or saplings. Narrow habitat between upland and swamp.	MNHP 2002 Observed 2002	M
Jackson County, MS	Growing along Brickyard Bayou 3.2 km south of George County line and Hwy 63.	MNHP 1966	
Jackson County, MS	Occasional in baldcypress/tupelo swamp 8-9 miles north of Escatawpa River along Jackson Creek.	MNHP 1951	
Jackson County, MS	On Jackson Creek in sandy soil 4.1 km north of Interstate 10.	MNHP 1979	
Jackson County, MS	12.8 km south of George/Jackson County line along Hwy 57.	MNHP 1966	
Jackson County, MS	East bank of Escatawpa River in mesic woods with baldcypress 1.7 km from Mississippi/Alabama border.	MNHP 1978	
Jackson County, MS	In sandy soil along Escatawpa River 1.7 km NE of Orange Grove 2.5 km south of Interstate 10.	MNHP 1980 Observed 2002	

(continued)

Table 1—Locations of Atlantic white cedar individuals and stands in Mississippi (including data on Louisiana plantations). Many of these locations represent old collections with the current status of the tree or site unknown. Most of the data was obtained from the Mississippi Natural Heritage Program (MNHP) or observed during 2002. The general location of these sites is shown on figure 2. Data in the last column cross-references to species lists for each location, given in Table 2 (continued)

Site location	Notes	Citation and/or last observation noted	Table 2
Jackson County, MS	Shrubby 2-m tall in savannah on Saracenia Road 1.5 km north of Interstate 10.	MNHP 1977	
Jackson County, MS	Scattered young trees in mucky bayhead between Escatawpa River and Big Creek 3.4 km south of Hwy 614, 1.6 km west of Mississippi/Alabama border.	MNHP 1983	
Jackson County, MS	Up and down stream from Interstate 10 crossing of Escatawpa River.	MNHP 1983	
Jackson County, MS	North end of Goodes Lake at Jackson Creek 2.6 km north of Interstate 10.	MNHP 1983	
Jackson County, MS	Along sandy banks of Escatawpa River at Big Creek.	MNHP 1983 Observed 2002	
Jackson County, MS	Several 15-25 cm dbh trees 4.2 km east of Hurley, MS, along Spring Creek.	MNHP 1985	
Jackson County, MS	Frequent near Black Creek 4.8 km north of Moss Point along Interstate 10.	MNHP 1984	
Jackson County, MS	Moss point, along east bank of the East Pascagoula River in pine woods.	MNHP 1983	
Jackson County, MS	South of Vancleave, Herbarium Collection.	Li 1962 (Li 1962)	
Jackson County, MS	Black Creek Swamp near Hurley, MS, 7.5 km west of the Escatawpa River. Many large and small cedar, a few dead or dying.	Observed 2002	
Juniper Creek, Pearl River County, MS	[1]Scattered patches along banks and in swampy area. [2]Disturbed and logged area; is the western most extension known at the time; [3]Possible reference.	[1]MNHP 1973 [2]Eluterius and Jones 1972 [3]Korstian 1931 [3]Little 1950 Observed 2002	A
Pearl River County, MS	A number of trees in swamp up to 7.6-m tall and 25-cm diameter. Juniper Swamp along Moren Creek 9.6 km SE of Poplarville along Hwy 53.	MNHP 1968	
Pearl River County, MS	Trial plantings of cedar 1989, Pink Smith Road off Hwy 43.	McCoy and others 2003	
West Feliciana Parish, LA	Holocene fossils of cedar twigs.	Givens and Givens 1987	
West Feliciana Parish, LA	Pleistocene fossils of cedar twigs.	Brown 1938	
St. Tammany Parish, LA	Trial plantings of cedar 1990, John Bennett Road off Hwy 36.	McCoy and others 2003	

Another new stand was found along Davis Creek on the DeSoto National Forest, just west of the Camp Shelby Army National Guard training site in Forrest County. This site, just south of Hattiesburg, MS, is characterized by a past blow-down event of unknown timing that uprooted or snapped many cedar trees. Extensive regeneration in the form of cedar seedlings to saplings can be found throughout the canopy gap area caused by the storm. Blowdown events are important in the regeneration of cedar stands. In fact, Ward and Clewell (1989) noted that cedar seedlings tend to become established in cohorts when conditions are correct, such as after a major blowdown that removes at least part of the canopy. Cedar seedlings and saplings outside of the disturbed area on the Camp Shelby site were less common.

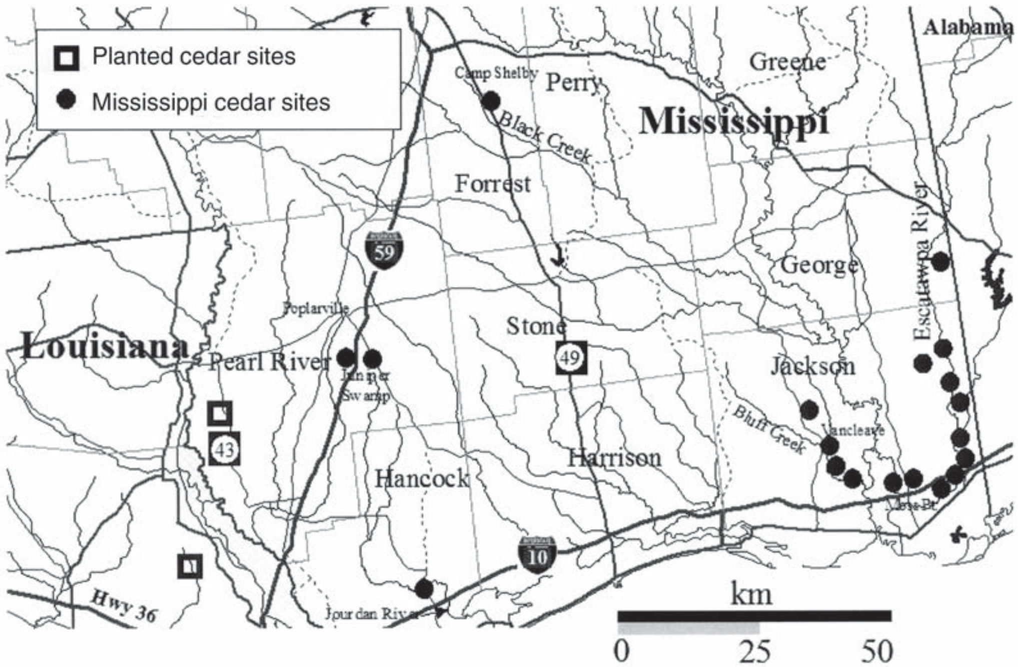

Figure 1—General locations of known sites of Atlantic white cedar listed in table 1. Some dots may cover more than one site.

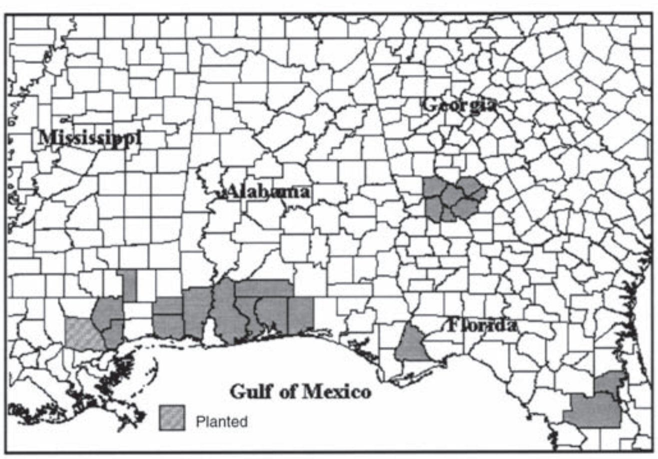

Figure 2—Range map of counties with cedar within Mississippi, Alabama, Georgia, Florida and Louisiana cedar plantations. This map shows the counties, in gray, associated with the cedar locations described in table 1.

Although cedar seedlings can survive a few years in shaded conditions, they will eventually die if they are not in a canopy gap and exposed to direct sunlight (Korstian and Bush 1931, Laderman 1989).

At the Camp Shelby site, competing woody vegetation included loblolly pine, slash pine, red maple, redbay, large gallberry, and sweetleaf. Loblolly pine and cedar were the tallest trees in the canopy at about 25 to 35 m tall. A thick shrub layer and extensive growth of large smilax vines made walking difficult in some areas. Cinnamon and royal ferns were present in the herbaceous layer. This site had an organic soil and a well developed litter layer of mixed needles and broad leaves. The stream channel was braided with very moist soils and pooled water. Like many other cedar sites we have visited, there was evidence of shallow flooding with standing water in depressions. One part of the site was subjected to long periods of standing water and was characterized by numerous dead cedars, many still standing, and extensive amounts of cedar regeneration. Many of the cedar trees occurred in small clusters of variable size. This clustering of cedar suggests gap phase dynamics occurred on this site, probably as a result of the dying trees creating canopy gaps. Two clusters of cedar about 15 m apart consist of 6 to 10 individuals each. The diameters of the individual cedars ranged from 7.7 to 62.8 cm d.b.h. and they were 25 to 30 m tall. The largest cedars noted during this visit may represent the largest cedars in Mississippi.

The Camp Shelby stand represents our northernmost known location for cedar in Mississippi. The stand is unusual in that it is so far inland, about 60 km more than the other Mississippi stands. Five cedar stands in Georgia are also far inland at > 225 km from the coast. These stands are found along the fall line in the sandhills of western Georgia (Sheridan and others 1999, Sheridan and Patrick 2003). The presence of these stands at such great distances from the coast is encouraging and suggests that other cedar stands may exist farther north in Alabama and Mississippi. Another stand of cedar on Camp Shelby was rumored to have existed, but it was probably destroyed by the installation of a shooting range.

Two planted populations of cedar are known to exist along the gulf coast (McCoy and others 1999, 2003). These stands were planted to study the restoration and growth potential of

Table 2—Plants observed at individual Mississippi locations from table 1. These observations are compiled from Mississippi Museum of Natural History unpublished data, journal articles or personal observations as noted in table 1. The time of each observation varies seasonally as well as by year. No comparison between sites is intended. Data in columns A through O refers to specific locations, given in table 1, where each listed species was found

Group and family	Species	Common name	A	B	C	D	E	F	G	H	I	J	K	L	M	N	O
FORBS																	
Alismataceae	*Sagittaria lancifolia* L.	Bull tongue arrowhead								X							
Alismataceae	*Sagittaria latifolia* Willd.	Duck-potato							X			X					
Asclepiadaceae	*Cynanchum angustifolium*	Gulf coast swallow					X										
Asteraceae	*Eupatorium* L.	Boneset								X							
Asteraceae	*Carphephorus odoratissimus* (J.F. Gmel.) Herbert	Vanilla leaf			X												
Asteraceae	*Solidago sempervirens* L.	Seaside goldenrod					X	X	X								
Bromeliaceae	*Tillandsia usneoides* (L.) L.	Spanish moss							X								
Iridaceae	*Iris virginica* L.	Virginia blueflag								X							
Sarraceniaceae	*Sarracenia* L.	Pitcherplant											X				
Saururaceae	*Saururus cernuus* L.	Lizard's tail								X							
GRASSES AND SEDGES																	
Cyperaceae	*Cladium mariscus ssp. Jamaicense* (Crantz) Kükenth.	Jamaican grass				X	X	X	X	X	X	X	X				
Cyperaceae	*Scleria* Berg.	Nut rush			X												
Juncaceae	*Juncus roemerianus* Scheele	Needle grass rush										X					
Poaceae	*Arundinaria gigantea* (Walt.) Muhl.	Giant cane		X													
Poaceae	*Dichanthelium* (A.S. Hitchc. & Chase) Gould	Rosette grass			X												
Poaceae	*Panicum virgatum* L. Pers.	Wand panic grass wort									X	X					
SHRUBS																	
Aquifoliaceae	*Ilex coriacea* (Pursh) Chapman	Large gallberry		X	X								X	X		X	
Aquifoliaceae	*Ilex glabra* (L.) Gray	Inkberry		X													
Aquifoliaceae	*Ilex vomitoria* Ait.	Yaupon		X			X	X	X	X		X	X				
Arecaceae	*Sabal minor* (Jacq.) Pers.	Dwarf palmetto							X	X	X						
Arecaceae	*Serenoa repens* (Bartr.) Small	Saw palmetto			X							X					
Asteraceae	*Baccharis halimifolia* L.	Eastern baccharis									X						
Betulaceae	*Alnus serrulata* (Ait.) Willd.	Brookside alder		X													
Betulaceae	*Alnus* P. Mill.	Alder										X					
Calycanthaceae	Calycanthus floridus L.	Sweetshrub															
Calycanthaceae	*Calycanthus floridus* L.	Eastern sweetshrub		X								X					
Caprifoliaceae	*Viburnum nudum* L.	Possumhaw														X	X
Clethraceae	*Clethra alnifolia* L.	Coastal sweet pepperbush		X													
Cyrillaceae	*Cliftonia monophylla* (Lam.) Britt. ex Sarg.	Buckwheat tree, Black titi		X	X							X	X				
Cyrillaceae	*Cyrilla racemiflora* L.	Swamp titi	X		X							X				X	X

(continued)

Table 2—Plants observed at individual Mississippi locations from table 1. These observations are compiled from Mississippi Museum of Natural History unpublished data, journal articles or personal observations as noted in table 1. The time of each observation varies seasonally as well as by year. No comparison between sites is intended. Data in columns A through O refers to specific locations, given in table 1, where each listed species was found (continued)

Group & family	Species	Common name	A	B	C	D	E	F	G	H	I	J	K	L	M	N	O
SHRUBS (cont.)																	
Ericaceae	*Lyonia lucida (Lam.) K. Koch*	Shinyleaf		X													
Ericaceae	*Gaylussacia mosieri* Small	Wolly huckleberry															
Ericaceae	*Kalmia latifolia* L.	Mountain-Laurel		X	X												
Ericaceae	*Lyonia* Nutt.	Staggerbush															
Ericaceae	*Rhododendron* L.	Azalea		X								X	X			X	
Ericaceae	*Vaccinium arboreum* Marsh.	Tree Sparkleberry		X									X				
Ericaceae	*Vaccinium elliottii Chapman*	Elliott's blueberry		X			X						X				
Ericaceae	*Vaccinium* L.	Blueberry			X												
Ericaceae	*Vaccinium stamineum* L.	Deerberry					X						X				
Fabaceae	*Amorpha fruticosa* L.	False indigo bush									X						
Grossulariaceae	*Itea virginica* L.	Virginia sweetspire		X								X					
Hamamelidaceae	*Hamamelis virginiana* L.	American witch hazel		X	X												
Illiciaceae	*Illicium floridanum* Ellis	Florida anisetree															
Lauraceae	*Cinnamomum camphora* (L.) J. Presl	Camphor tree					X										
Myricaceae	Morella inodora *(Bartr.)*	Small scentless bayberry												X		X	
Myricaceae	Morella cerifera *(L.) Small*	Wax myrtle		X				X	X	X	X	X					
Rosaceae	*Photinia pyrifolia* (Lam.) Robertson & Phipps	Red chokeberry							X	X							
Styracaceae	*Styrax americanus Lam.*	American snowbell		X												X	
TREES																	
Aceraceae	*Acer rubrum* L.	Red maple		X													
Aceraceae	*Acer rubrum* var. *drummondii* (Hook. & Arn. ex Nutt.) Sarg.	Swamp red maple				X		X	X	X	X	X	X			X	
Anacardiaceae	Toxicodendron vernix (L.) Kuntze	Poison Sumac							X							X	
Aquifoliaceae	*Ilex Opaca* Ait.	American holly		X													
Arecaceae	*Sabal palmetto* (Walt.) Lodd. Ex J.A. & J.H. Schultes	Cabbage palmetto		X													
Cornaceae	*Nyssa biflora* Walt.	Swamp tupelo	X	X						X		X	X	X	X		X
Cornaceae	*Nyssa sylvatica* Marsh.	Black tupelo	X														
Cupressaceae	*Chamaecyparis thyoides* (L.) B.S.P.	Atlantic white cedar	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X
Cupressaceae	*Taxodium ascendens* Brongn.	Pond cypress						X	X	X	X	X			X		
Cupressaceae	*Taxodium distichum* (L.) L.C. Rich.	Baldcypress	X	X													
Euphorbiaceae	*Triadica sebifera* (L.) Small	Chinese tallowtree					X		X	X							
Fagaceae	*Castanea* P. Mill.	Chestnut															

(continued)

Table 2—Plants observed at individual Mississippi locations from table 1. These observations are compiled from Mississippi Museum of Natural History unpublished data, journal articles or personal observations as noted in table 1. The time of each observation varies seasonally as well as by year. No comparison between sites is intended. Data in columns A through O refers to specific locations, given in table 1, where each listed species was found (continued)

Group and family	Species	Common name	A	B	C	D	E	F	G	H	I	J	K	L	M	N	O
TREES (cont.)																	
Fagaceae	*Quercus hemisphaerica* Bartr. ex Willd.	Darlington's oak											X				
Fagaceae	*Quercus minima* (Sarg.) Small	Dwarf live oak			X												
Fagaceae	*Quercus nigra* L.	Water oak		X	X			X				X	X				
Fagaceae	*Quercus pumila* Walt.	Running oak			X												
Hamamelidaceae	*Liquidambar styraciflua* L.	Sweetgum		X													
Juglandaceae	*Carya Nutt.*	Hickory		X													
Magnoliaceae	*Liriodendron tulipifera* L.	Tulip tree		X													
Magnoliaceae	*Magnolia grandiflora* L.	Southern magnolia		X			X						X				
Magnoliaceae	*Magnolia virginiana* L.	Sweet bay	X		X									X			
Oleaceae	*Fraxinus caroliniana* P. Mill.	Carolina Ash					X			X							
Oleaceae	*Osmanthus americanus* (L.) Benth. & Hook. f. ex Gray	Devilwood		X	X												
Pinaceae	*Pinus elliottii* Engelm.	Slash pine		X				X	X		X	X				X	
Pinaceae	*Pinus palustris* P. Mill.	Longleaf pine													X	X	
Pinaceae	*Pinus taeda* L.	Loblolly pine											X			X	
Rutaceae	*Zanthoxylum clava-herculis* L.	Hercules'club													X		
Salicaceae	*Salix nigra* Marsh.	Black willow								X							
VINES																	
Anacardiaceae	*Toxicodendron radicans* (L.) Kuntze	Eastern poison ivy					X	X	X	X		X				X	
Caprifoliaceae	*Lonicera japonica* Thunb.	Japanese honeysuckle								X							
Fabaceae	*Wisteria frutescens* (L.) Poir.	American wisteria									X						
Rhamnaceae	*Berchemia scandens* (Hill) K. Koch	Alabama supplejack									X						
Smilacaceae	*Smilax glauca* Walt.	Sawbrier										X					
Smilacaceae	*Smilax* L.	Greenbrier			X											X	
Smilacaceae	*Smilax rotundifolia* L.	Horsebrier						X	X								
Vitaceae	*Vitis aestivalis* Michx.	Summer grape											X				
Vitaceae	*Ampelopsis arborea* (L.) Koehne	Peppervine								X	X						
Vitaceae	*Parthenocissus quinquefolia* (L.) Planch.	Virginia creeper								X							
Vitaceae	*Vitis rotundifolia* Michx.	Muscadine			X							X					
FERNS																	
Blechnaceae	*Woodwardia areolata* (L.) T. Moore	Netted chain fern														X	
Dennstaedtiaceae	*Pteridium aquilinum* (L.) Kuhn	Northern bracken fern			X												
Osmundaceae	*Osmunda cinnamomea* L.	Cinnamon fern						X	X		X	X	X			X	
Osmundaceae	*Osmunda regalis* L.	Royal fern				X	X		X	X	X	X	X			X	
Shagnaceae	*Sphagnum*	Sphagnum								X		X	X				

this species. One stand is located in a bayhead on the Bogue Chitto National Wildlife Refuge, northwest of Picayune, Pearl River County, MS. At this location, two small stands were established using wildlings collected at the Bluff Creek stand near Vancleave. After 11 years, survival at this site was only 56 percent, but most of the mortality was in one stand that was situated along a small creek. The other cedar plantation, in three small stands established during the spring of 1990 in the wetter areas of a slash pine plantation, is located near Abita Springs, St. Tammany Parish, LA. Survival rate was very high at this site with 91 percent of the trees living through 10 growing seasons. The presence of cones as early as 1997 and of numerous seedlings in 2000 suggested vigorous stand development. Although most of the seedlings were small (< 25 cm), a few were 60 cm in height and were more than 8 m from the nearest possible parent tree.

Several additional locations of cedar have been mentioned in manuscripts, herbarium labels, field notes and/or the Mississippi Museum of Natural Science database http://www.mdwfp.com/museum/ and range from single trees to small stands. Most of these sites are in Jackson County and include numerous scattered trees along the lower reaches of the Escatawpa and Pascagoula rivers and Bluff Creek. Generally, there were fewer than 25 individual cedar stems noted at these locations. Cedars at these sites range in height from 7 to 20 m. Competing vegetation included slash pine, red maple, redbay, baldcypress, Carolina ash, wax myrtle, yaupon, deerberry, and camphor tree (table 2). Some sites near the coast were adjacent to brackish water; where trees were perched on river banks above the water level. Additional Jackson County sites that the authors have not visited include Big Cedar Creek, the Escatawpa River at Highway 614, Goodes Mill Lake, and along Saracenia Road (table 1).

Cedar sites in other counties include Juniper Grove (south of the Poplarville-Pearl River Airport, in Pearl River County), near the Highway 603 bridge over the Jourdan River (in Hancock County), and Juniper Bay and along the Escatawpa River near Highway 96 (in George County). These data indicate that at least five counties in southern Mississippi support individual trees or stands of cedar (fig. 2).

PLANT ASSOCIATIONS

Species listed in table 2 and noted in the text are not meant to be a comprehensive list of plant species found in the cedar stands discussed. As such, the list of plants for any given site should not be used to determine similarity or differences between locations. Table 2 represents a subset of plants that were casually observed during one visit to some Mississippi cedar sites, or from data taken from the Mississippi Museum of Natural History database, herbarium labels, and published papers. Species names are directly from the source (field notes, database listings, herbarium sheets, etc.), or in the case of our collections, from Kartesz and Meacham (1999). At this time, little significance can be given to plant associations listed in this paper. These results are presented as preliminary data for information purposes only. Only the study of Eleuterius and Jones (1972) consisted of a quantitative description of plant species associated with cedar, and that study was conducted only at Bluff Creek. More complete plant listings will be possible after a more complete inventory and analysis of these stands.

CONCLUSIONS

Little is known about the cedar stands along the gulf coast. With only one stand described in detail and just a few additional locations known for this species, the future of cedar in coastal Alabama and Mississippi is uncertain. This study has highlighted the location of several additional stands and many isolated trees. It appears that the species is not as rare along the gulf coast as previously thought. Many of the known stands, however, are degrading as a result of urbanization, erosion, land use and hydrologic changes in the surrounding landscape. As we continue to characterize selected stands, our knowledge of and our ability to predict the sustainability of the gulf coast cedar stands will improve.

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ATLANTIC WHITE CEDAR SPECIES RECOVERY AND WETLAND ENHANCEMENT PROJECT AT HOWARD'S BRANCH, ANNE ARUNDEL COUNTY, MARYLAND

Keith R. Underwood, William B. Moulden, Dennis C. McMonigle and David J. Wallace[1]

Abstract—The creation of a new site for Atlantic white cedar [*Chamaecyparis thyoides* (L.) B.S.P.] was the driving force behind a stream stabilization and wetland enhancement project in Anne Arundel County, MD. The Howard's Branch project was designed to create peatland ecosystems in a highly degraded stream valley flood plain by creating a functional seepage wetland supporting a sustainable Atlantic white cedar community. A series of cobble weirs and a network of sand berms were placed over a dry lakebed to simulate the geology and hydrology found in natural Atlantic white cedar sites. In April 2001, construction of the project was completed with the planting of 1,000 Atlantic white cedars propagated from the 10 remaining stands of the species on the western Coastal Plain of Maryland. This paper reviews the procedures developed for the Howard's Branch project, reports on the status of the constructed wetland, and discusses the regulatory hurdles that were encountered.

Keywords: Bog, *Chamaecyparis thyoides*, forested wetlands, seepage wetlands, stream restoration.

INTRODUCTION

Atlantic white cedar [*Chamaecyparis thyoides* (L.) B.S.P.] (AWC) is an evergreen, wetland tree species that inhabits peatlands along the Atlantic and Gulf coasts of the United States. In the past 200 years most of the historic peatlands along the eastern seaboard of the United States have been destroyed. The remaining peatlands and their adjacent uplands harbor an inordinately large number of rare, threatened and endangered plant and animal species (Laderman 1989). Some of these species are found only in cedar swamps (Cryan 1985). AWC may be considered globally threatened as a species or as a community type (McAvoy and Clancy 1993), and has been reduced to < 2 percent of its original acreage (Noss and others 1995).

AWC survives only under a very narrow set of environmental conditions (Laderman 1989, 2003). Rainwater that falls in the sandy uplands in landscapes above AWC wetlands is slowly released over confining soil layers, or at sea level as ex filtrating groundwater seepage. Plants such as Sphagnum mosses and emergent hydrophytes colonize these wet, sandy, mineral soils and over a number of years form an organic peat substrate. Wetland vegetation associated with these areas is typically well adapted to low pH conditions. Peat will also frequently accumulate along the edges of adjacent open water habitats. Associated water impoundments, such as spring-fed ponds, sometimes develop floating mats of peat as water lilies (*Nymphaea odorata* Ait.) and other species create organic material faster than it decomposes. Eventually, species including sedges and Sphagnum mosses colonize these peat mats and form hummocks that are suitable micro-habitats for the establishment of AWC seedlings. Close investigation of sites now containing AWC on organic soils revealed that seepage wetlands are associated with suitable habitat for this species.

In Maryland, AWC occurs on peat that has formed on sandy seepage wetlands along the edges of bogs, ponds, streams, and tidal headwaters of the Chesapeake Bay (Hull and Whigham 1987). The species was once abundant within the predominantly deciduous tidal swamps along the major rivers of the eastern shore of the Chesapeake Bay (Dill and others 1987, Shreve and others 1910). In Anne Arundel County, Maryland, a series of peatland complexes occurs along a narrow band of sandy, acidic soils of the Magothy Geologic Formation (Kirby and Matthews 1973). Where these soils surface at or near sea level along the shores of the Severn and Magothy Rivers, groundwater discharges and peatlands develop. Ten of these peatlands contain the only known remaining wild populations of AWC on the western Coastal Plain of Maryland (WCPMD) and represent the western edge of the range of the species in Maryland. Sheridan and others (1999) found that AWC occurred in nine sites containing a cumulative total of 1,214 living trees > 1.2 m in height remaining on the WCPMD. A 10th site containing < 100 trees was found in 2000 and has subsequently been reduced to less than 20 trees. These inventories indicated declining populations and resulted in investigations of potential sites for AWC conservation, restoration, and recovery opportunities to ensure the continued existence of AWC and other peatland species in Anne Arundel County.

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METHODS

Project Site Selection

In the fall of 1997 sites were evaluated for their potential for the establishment of a new AWC population to make up for historic and recent losses. The leading experts in AWC restoration were consulted before and during the undertaking of this project. A 3 acre dry lakebed on a tributary of the Severn River known as Howard's Branch was selected due to factors weighing in its favor to provide a suitable base for creation of a seepage wetland with AWC, including: its general topography, the constant water flow through the site, its proximity to remnant AWC populations (fig. 1), its position in the landscape, the accessibility for equipment and the willingness of landowners to host the project.

Site History

In 1930, a forested stream valley flood plain was flooded to a depth of -4 feet with the construction of an earthen dam across a small stream known as Howard's Branch. The resulting lake was the sole drinking water supply for the community of Sherwood Forest until 1970. In 1980 the dam failed and the lake drained, exposing sediments that had accumulated behind the dam over 50 years. The stream subsequently cut through those sediments, which were then transported downstream to tidal waters, damaging tidal and sub-tidal ecosystems, and resulting in disturbance regime plants such as common reed (*Phragmites australis*) in the tidal wetlands and Eurasian water-milfoil (*Myriophyllum spicatum*) in the adjacent shallow tidal waters.

Physical Conditions

Stream characteristics—Howard's Branch is approximately one mile long from its headwaters to its outflow into Brewer Creek. The stream begins as toe slope seepage from two upper reaches of the ravine system, wherein it converges into one main stream channel just above the project site. The remnants of the dam are located 1,000 feet upstream of a pond at the tidal interface with Brewer Creek, a tributary of the Severn River. The Howard's Branch stream valley floodplain (former impoundment) within the project site is approximately 737 feet long and 120 feet wide. The floodplain ranges in elevation between 10 and 15 feet above sea level. The drainage area to the project site is a total of 231 acres, or 0.4 square miles, and is comprised of a mix of forested open space and low density residential. The base flow of the stream through the project site is about 2 cubic feet per second.

Soils—Surface soils within the project area are generally comprised of mixed alluvial deposits that range from clay to sand (Kirby and Matthews 1973). These soils have been deposited in the former impoundment from various upstream eroding soils. The stream valley flood plain is generally level, with a slope of < 1 percent. The soils are poorly drained and remained wet even in dry periods. The soils surrounding the project area are Monmouth fine sandy loam, with a slope of 15 to 40 percent, which overlays the white sands of the Magothy Formation. Within the Monmouth mapping unit are some deep gullies that have a very sandy or silty surface layer, and the slopes are highly susceptible to erosion when existing vegetation is removed (Kirby and Matthews 1973).

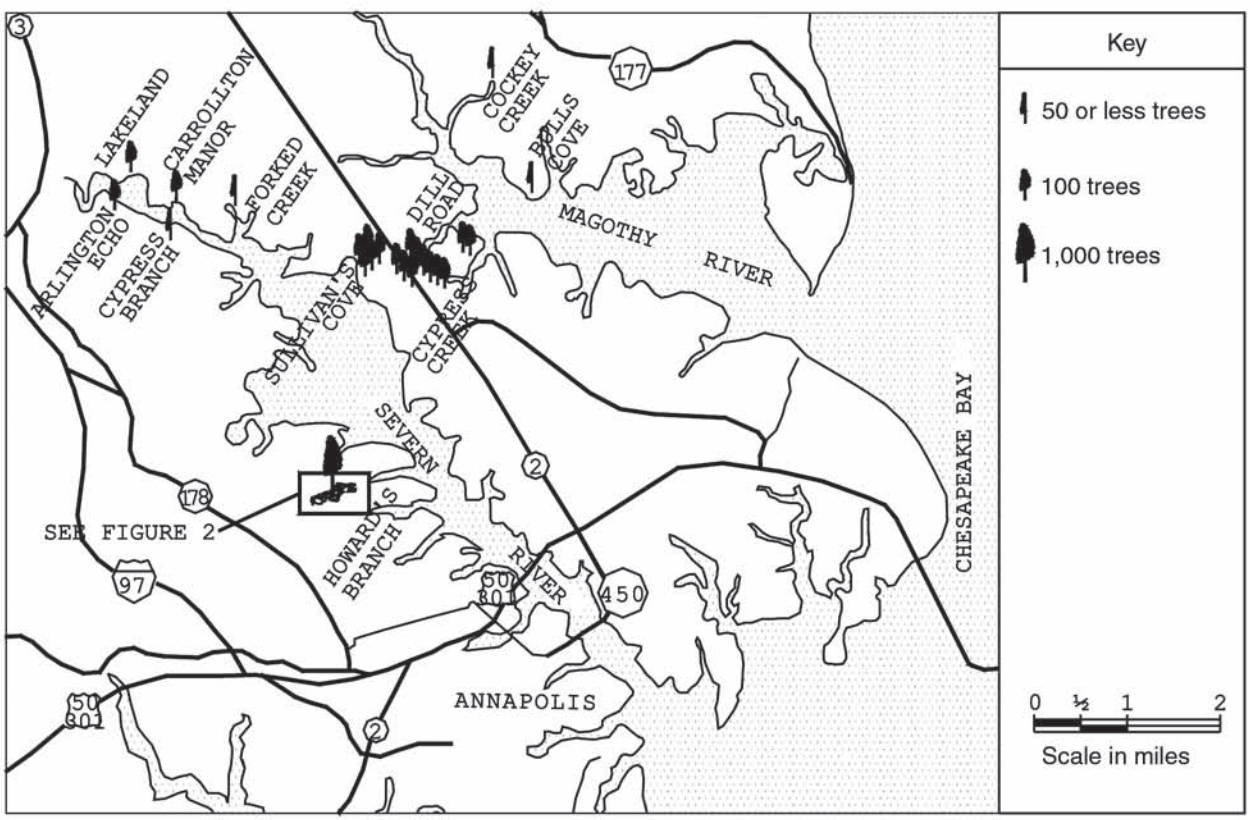

Figure 1—Locations of all known Atlantic white cedar populations west of the Chesapeake Bay.

Preproject Environmental Conditions

Wetlands—The project area was dominated by nontidal wetlands surrounded by steep slopes. The site was predominantly characterized as an open wet meadow dominated by rice cut-grass (*Leersia oryzoides*), Asiatic tearthumb/mile-a-minute vine (*Polygonum perfoliatum* L.), joe-pye weed (*Eupatorium dubium* Willd.), boneset (*Eupatorium perfoliatum* L.), and goldenrod (*Solidago sp.*). A small amount of sphagnum (*Sphagnum sp.*) was present.

Woody species included smooth alder (*Alnus serrulata* Ait. Willd), which formed a mature stand consisting of about 20 large individuals standing 30 feet apart at the upper end of the project site. Sweet gum (*Liquidambar styraciflua* L.) formed a small stand of perhaps 50 individuals up to 4 inches diameter at breast height (DBH) on the north side of the site at the confluence of a forested ravine. Eleven mature red maples (*Acer rubrum* L.) up to 11 inches DBH were mapped at the uppermost end of the flood plain and also formed a small stand of trees up to 4 inches DBH at the lower end of the project site. The site was generally wetter near the upper end and drier near the downstream end in the vicinity of the remaining portion of the earthen dam.

Uplands—The watershed contains a low density residential community on the south side and a 100 year old forest under a conservation easement on the north side. The drainage area consists primarily of steep slopes that grade up to 120 feet in elevation. About 80 percent of the uplands are currently forested with a mixed hardwood canopy, including oak (*Quercus sp.*) and American beech (*Fagus grandifolia* Ehrh.), with a minor Virginia pine (*Pinus virginiana*) softwood component and a mountain laurel (*Kalmia latifolia* L.) understory.

Planning and Permits

The project was designed to simulate the geology and hydrology found in the remaining native AWC sites. A fill operation was designed and permitted, resulting in the establishment of a seepage wetland that supports a viable population of AWC.

Flow rates for various storm events at Howard's Branch were determined using calculations derived from the Soil Conservation Service "Urban Hydrology for Small Watersheds" Technical Release (TR) 55 (U.S. Department of Agriculture 1986). The Anne Arundel County soil survey maps (Kirby and Matthews 1973) were also consulted to develop the calculations. Current land use results in a runoff curve number (RCN) of 70, while the ultimate development based on existing zoning results in a RCN of 77. The time of concentration—the time for rainwater runoff to travel from the hydraulically most distant point of the watershed to a point of interest within the watershed—for both existing conditions and ultimate build-out based on current zoning was calculated to equal 1.15 hours. A conservative approach from the Seelye (1960) design manual and Harr (1990) was used to determine the capability for the capillary potential of sand to wick water toward the surface. This information was used to design the project features.

Plans were designed to physically alter the lakebed and incised stream channel in relationship to existing hydrologic characteristics. The project was designed to capture and redirect base flows to maximize irrigation of the berms while redirecting and reducing the energy associated with large storm flows allowing excess water to pass harmlessly through the site. The plans were submitted to the regulatory authorities, and the U.S. Army Corps of Engineers and Maryland Department of the Environment issued permits for temporary impacts to 125,389 square feet (2.88 acres) of existing nontidal wetlands and 737 linear feet of stream channel.

Construction

A series of seven cobble weirs were constructed across the main stream channel about 100 feet apart in 1 foot lifts as grade controls. Each weir flooded the soils above it, including the incised eroding stream channel, resulting in water retention, creation of sheet flow, and reduction in velocities of storm water passing through the project site. The weir structures were reinforced with the placement of sandstone boulders at the toe of each weir. The subsequent colonization of the weirs by vegetation was intended to further improve their stability.

A network of berms, comprised of sand, gravel, and wood chips about 3 feet thick and 40 feet wide, was combined with the cobble weirs to form a new surface topography that would control surface and subsurface hydrology (fig. 2). Hydrology for the sand berms was provided by lateral seepage of water from the moats and capillary action, resulting in increased and stabilized soil moisture levels. The berms were designed to serve as temporary haul roads by placing a single 12 foot wide strip of poly-woven geotextile in the design locations of the future berms.

The sand berms were placed about ten feet from the toes of the adjacent steep slopes flanking the northern and southern sides of the project site. The resulting depressions between the tops of the berms and the adjacent side slopes serve to capture surface water and ground water seepage from the side slopes, which formed long pools (moats) that surrounded most of the site (fig. 3) .The water surface elevation in the moats was designed to be up to 3 feet higher than the water surface elevation in the channel. Water captured in the moats would then move laterally and irrigate the sand berms. As water slowly filters through the sand berms to lower elevations, sandy seepage slopes are created similar to those found in other AWC sites. The sand berms were placed to meet the edge of water impoundments created by the weirs in the stream channel.

Six thousand tons of white, bank-run silica sand and gravel were used to form the berms and 1,000 tons of sandstone (ferracrete) boulders were used as grade controls for the weirs and the lower stream channel. One hundred cubic yards of Virginia pine and red maple wood chips were trucked into the site. One hundred four cubic foot bales of Canadian peat were placed on the site and exposed to rain a month before planting of the trees. One thousand tons of processed white silica sand (white play sand) was placed on the surface of the berms to preclude the establishment of red maple and sweet gum, undesirable plant species that require nutrients not available in pure sand.

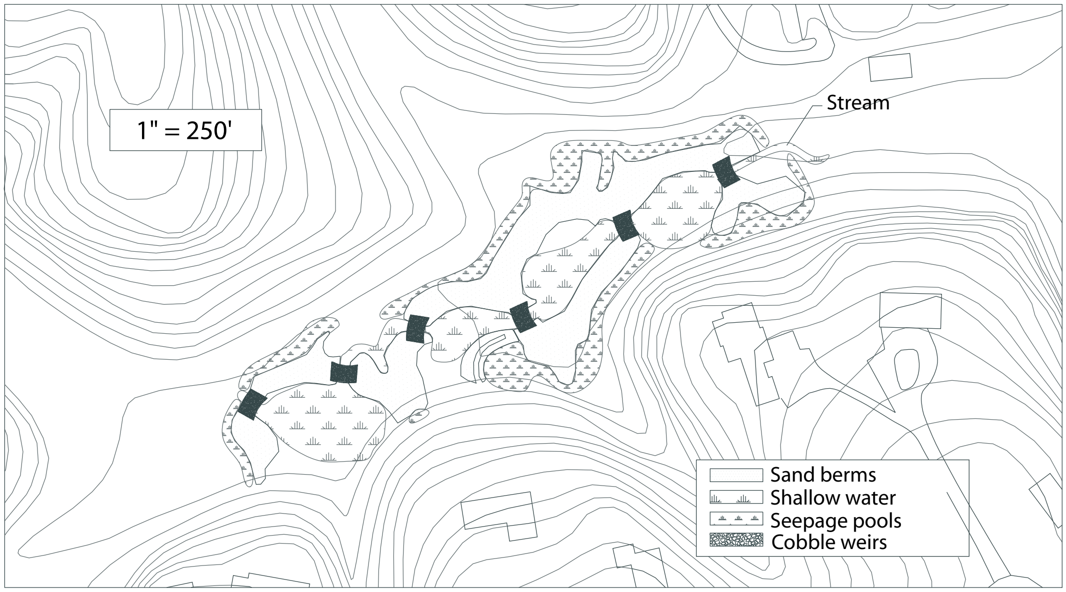

Figure 2—Howard's Branch as-built drawing.

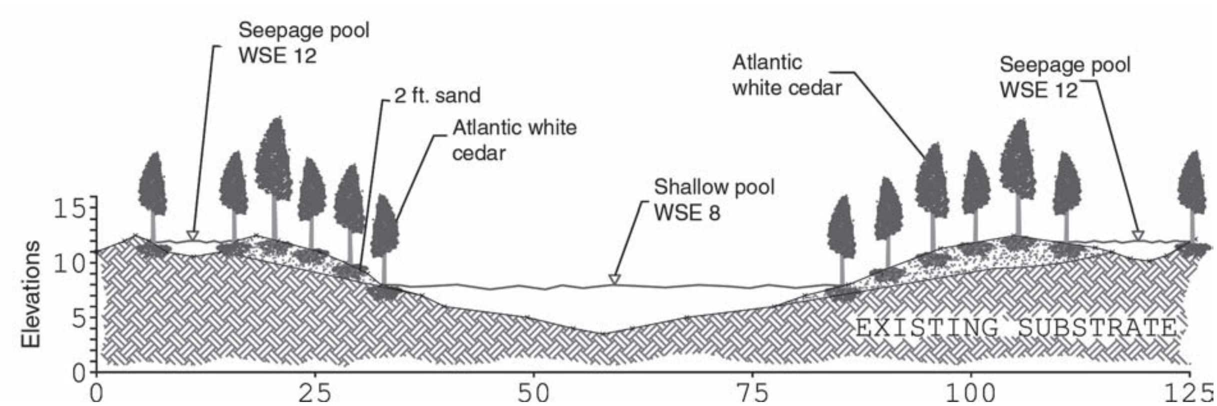

Figure 3—Howard's Branch typical cross section drawing.

The last 50 feet of the lower end of the stream channel needed to match up with the level of the undisturbed channel on the adjacent downstream property. The stream was incised 5 feet below the adjacent flood plain with eroding vertical banks. The channel was filled with bank-run gravel. Geo-textile was then placed over the gravel, and native sandstone boulders weighing up to 2 tons were used to line the channel to create a stable configuration for the new, steeper gradient.

Construction access into the project site was achieved by shaping a wedge of bank-run gravel to form a ramp into the project site over a storm drain pipe system. The drain pipe and three drop structures were filled with gravel, which would treat storm water from a 10 acre drainage area that had previously discharged as untreated storm water into the site. This technique allowed some storm water to filter into the site as ground water. Upon departure from the site, the access ramp was reshaped to form a pedestrian pathway.

Vegetation

Two schoolyard wetlands were constructed specifically to propagate indigenous AWC stock for this project. Seed and cuttings were collected from all ten of the remaining wild populations west of the Chesapeake Bay. The seed and cuttings were used to propagate trees in the school-yard wetlands and the Anne Arundel County greenhouse. Meadowview Biological Research Station in Woodford, VA also generated seedlings from the Arlington Echo native population as part of another study. The schoolyard wetlands were then used as grow out areas for the potted cedars.

Following construction in April, 2001, one thousand Anne Arundel County-native AWC were planted as containerized saplings up to 48 inches in height on the sand berms by volunteers including school children, community members, politicians, and representatives of universities, research organizations, regulatory agencies and civic associations. Some plant species associated with AWC were subsequently introduced to the site from local native sources.

In a variation on techniques used by Haas and Kuser (1999) that resulted in the best growth of AWC in a sterile sand site with appropriate hydrology, we used peat, wood chips and Osmocote® as soil amendments during planting. Subsequently, the cedars were fertilized with half strength Miracid® three times in 2001 and twice in 2002. One handful of Holly tone® was also applied to the base of each tree in August of 2002.

Regeneration of AWC is hindered by browsing of white-tailed deer (*Odocoileus virginiana*) and other herbivores, a problem that is widely recognized throughout the range of AWC (Hinesley and others 2003, Kuser and Zimmermann 1995, Laderman 1989, Little and Somes 1965). Herbivore exclusion cages were constructed of 12 gauge, 4-inch grid, galvanized wire fencing, secured into the ground with a single piece of 6 foot long No. 6 rebar woven through the fencing and driven 2 feet deep into the soil, resulting in a 48 inch high cage. Cages were placed around approximately 80 percent (800) of the planted AWC saplings.

Groundwater monitoring wells were installed at various locations across the project site prior to construction. Twelve 36-inch lengths of PVC pipe were installed to depths of about 24 inches. Photographic documentation of the project features and some of the organisms as well as groundwater to surface depth data was collected once every 2 weeks during the growing season prior to construction and each year following construction of the project.

Grab samples of water leaving the site were collected once each year and sent to Phase Separation Sciences laboratory in Baltimore, MD for analysis of pH, dissolved oxygen, nitrogen, nitrate, ammonia, phosphorus and turbidity concentrations. A field run topographic survey of the entire project site was also conducted each year. Soil profiles from borings near the groundwater monitoring wells were described using the Munsell soils charts.

RESULTS

General Description

We converted an eroding wetland dominated by non-native and invasive plant species to a stable seepage wetland complex supporting a number of threatened plant species. Construction of the project resulted in temporary impacts to 89,810 square feet (2.06 acres) of regulated, existing nontidal wetlands and 737 linear feet of stream channel. Wetland enhancement was achieved on 47,266 square feet of emergent wetlands in shallow flooded areas and 42,264 square feet of forested wetlands on the sand berms. An additional 280 square feet of stream channel was stabilized at the outfall. The area that was left undisturbed was 35,079 square feet.

Topography

The new topography formed by capping the flood plain with sand berms and cobble weirs remains stable in its designed position. Erosion of the stream banks was prevented, as was the subsequent loading of sediments into the downstream tidal water ecosystem of the Chesapeake Bay.

Hydrology and Hydrogeology

The constructed berms and weirs slow and retain base flow surface waters while allowing a non-erosive course for surface waters generated by storm events. Vegetation growing on the weirs has restricted the channel width over the weirs and created water depths of approximately 6 inches, which is adequate for passage of local fish species. Surface water flows are directed into a broad, flat, and gentle meandering pattern. Water captured in the moats and retention of water above the weirs serves both to raise and to stabilize the groundwater table and irrigate the berms.

Groundwater has been maintained within 12 inches of the surface of the sand berms at all monitoring well locations for 3 years with the exception of two wells in 2002 following a 3-year drought. The irrigation moats (seepage pools) flanking the berms remained full with the exception of the highest pool, which was dry for a week in 2002 during a drought. The sand berms are wet to the surface through capillarity and the lateral seepage of water from the irrigation moats through the sand. Most of the former lake bottom is now perennially submerged under 1 to 18 inches of water.

The base flow of the stream is now slowed and distributed to maximize the irrigation of the sand berms, while energies associated with storm flows are adequately dissipated by the project features to allow the water to pass harmlessly through the site. The project will also reduce the peak flows associated with storm events and will slowly release the storm water as base flow to the stream.

Vegetation

One hundred seventy-seven vascular plant species were identified in the site by William S. Sipple in 2003 (appendix 1). Using Reed (1988) to determine the wetland indicator status, 34 percent of the identified species are obligate wetland plants and 72 percent are facultative or wetter. AWC is now the dominant tree on the project site. The average DBH was 1 inch in 2003, and the canopy diameter averaged 34.6 inches. The planted cedars have grown from an average of 2 feet in height to a range of between 6 and 12 feet tall. More than 50 percent of the trees produced seed in 2003. Thousands of AWC seedlings resulting from natural recruitment range up to 16 inches in height. *Utricularia sp.* and American bur-reed (*Sparganium americanum* Nutt.) are dominant in the flooded areas. The herbaceous layer is dominated by yellow-eyed grass (*Xyris torta* Sm.) and the groundcover is dominated by American cranberry (*Vaccinium macrocarpon* Ait.) and *Sphagnum sp.* Algal mats are present on the surface of the wet sand.

Soil Characteristics

Soil characteristics have varied only slightly since the completion of the project although some wetness characteristics and redoximorphic features are evident in the created sand berms. Hydric soils require long periods of time for the development of wetness characteristics (Environmental Laboratory 1987).

Water Quality

Grab samples were collected just below the project site each year and sent for lab analysis. The pH of the water collected just below the project site has decreased from 6.54 in 2001 to 5.60 in 2003. This may in part be due to the release of organic acids provided by decomposing vegetation. Pre- and post-construction nitrogen levels were not detectable at a concentration greater than or equal to the practical quantitative limit. The post construction level of phosphorus was 0.10 ppm.

Fauna

Deer browse has not yet been a problem, and only a few trees have been damaged each year by deer scraping. In addition, over the 3 year period since construction, approximately 10 trees have been damaged or destroyed by beaver (*Castor canadensis*). Bullfrogs (*Rana catesbiana*), pickerel frogs (*Rana palustris*), green frogs (*Rana clamitans*), wood frogs (*Rana sylvatica*), American toads (*Bufo americanus*), spring peepers (*Pseudacris crucifer*), *gray* treefrogs (*Hyla sp.*) and spotted salamanders (*Ambystoma maculatum*) use the site for reproduction. Bluegills (*Lepomis macrochirus*) and pumpkinseeds (*Lepomis gibbosus*) have invaded the main channel from downstream, but have not made it into the moats.

DISCUSSION

We have observed considerable losses of dozens of peatland species in the Atlantic Coastal Plain. These peatland ecosystems need active management, preservation and restoration. Damaged ecosystems, e.g., Howard's Branch, provide sites that "could be restored in such a way as to enhance the chances of survival for one or more rare, endangered, or threatened species" (Cairns 1986). Given the recent scientific documentation of the immense benefits provided by peatland ecosystems to tidal estuaries (Hinesley and Wicker 1999), restoration or establishment of new peatlands in created environments to make up for historic losses takes on a fresh urgency.

Peatlands, including AWC swamps, become degraded when development and/or agriculture occurs within their drainage areas (Ehrenfeld and Schneider 1990, Laidig and Zampella 1999). Without immediate efforts to appropriate habitat, manage storm water, and restore populations of peatland species, they will continue to disappear along with significant genetic variation within a number of species throughout their range. For instance, Benedict (in press) predicts the extinction of the fourth largest remaining AWC stand on the WCPMD in 2015.

The resultant substrate for AWC at this site was most similar to the "extremely barren sand locations" at the lakeside site of Haas and Kuser (1999). It differed from other projects that attempted to restore AWC on former and/or degraded peatlands (Hinesley and Wicker 1999, Smith 1999). It is anticipated that the course growing medium, coupled with the creation of suitable habitat with relation to shade tolerance (Belcher and others 2003), planting survival (Brown and Atkinson 1999, McCoy and others 2003), natural regeneration (Eagle 1999, Zimmermann and others 1999) and competition with hardwoods (Eagle 1999) will produce the correct habitat for colonization by early successional species such as AWC. Accordingly, a peatland should develop as the AWC stand and its associated species produce organic material more rapidly than it decomposes. The novel approach of creating seepage wetlands at Howard's Branch could be used in other geographic areas to enhance the sustainability of other rare species dependent on this geomorphic setting.

Aspects of the Howard's Branch project were contested by some individuals within the regulatory community. Key regulators stated that the site in its pre-project condition represented a stable, appropriately vegetated wetland, and further stated that the site represented the premium reference model for wetlands on the WCPMD. However, the authors documented erosion of the banks of the stream channel breaking away in series and falling into the channel at a rate of 1 cubic yard per week throughout the pre-construction monitoring period. The exposed sediments in the former impoundment allowed the colonization of invasive disturbance regime vegetation, mostly Asiatic tearthumb and rice cutgrass. Both are common species in disturbed wetland soils, and the tearthumb is a non-native weed. The few tree species on the site were locally abundant, including red maple and sweet gum. Some of the red maple, sweet gum, and alder died or showed stress from inundation, but alders in general have benefited from the project and increased in number and area covered. The subsequent erosion and deposition downstream also dictated occupation by the invasive species common reed at the tidal interface and Eurasian water- milfoil in the adjacent shallow tidal water. Downstream reaches are more stable now, and the AWC complex, which includes other rare plants, is a higher quality wetland than the system that was in the site.

There was also a dispute as to whether the depth of sand placed to form the berms, and the amount of capillary movement of water through the sand, would result in wetlands or uplands. The depth to groundwater is an important aspect of creating appropriate habitat for AWC (Atkinson 2001, Harrison and others 2003, Mylecraine and others 2003). A conservative approach from the Seelye (1960) design manual and Harr (1990) was used to determine the capability for the capillary potential of sand to wick water toward the surface. However, the primary hydrology source for the berms was not only groundwater rising from below, but also water moving laterally and downward from the higher elevation moats. Constant seepage through the sand berms creates similar irrigation to highly organic sites mentioned by Atkinson (2001), where the organic content modulated water table fluctuations. The sand berms on the Howard's Branch site are wet enough to support algae growth on the surface and recruitment of obligate wetland plants is occurring. We not only believe that the sand berms will meet the definition of wetlands (Environmental Laboratory 1987), but that they will function as high quality wetlands for the long term.

Finally, fisheries biologists opposed the issuance of the permits, despite the fact that electro-shocking showed that there were no fish in Howard's Branch before the project was undertaken. They argued that the weirs would produce fish blockages and, therefore, the project would not benefit any fisheries resource. However, rains shortly after the project was completed raised the water level in the stream, and sunfish moved through the weirs and invaded the site. The fish have not invaded the moats, where they could reduce amphibian populations.

Although AWC wetlands are a globally threatened ecosystem with < 2 percent of its historic acreage remaining and not a single intact AWC ecosystem remains in the State, it is listed by the Maryland Department of Natural Resources' Heritage Program as an S-3 (State watch list) species and given no protection. State ranking is determined by the number of occurrences of any given species within the State. Historical evidence of former abundance exists, including excavated stumps and logs, local lore, site names and numerous personal communications, that indicates that the AWC forest dominated a large part of the landscape of the Cypress Creek watershed 100 years ago. Dissection of that population occurred with the construction of a highway. The ensuing development adjacent to the highway caused the species to retreat to three small isolated locations. These remnants are now erroneously considered as three occurrences by the Maryland Department of Natural Resources' heritage program. This splitting of sites does not accurately represent the current rarity or the former importance of the AWC ecosystem type.

A significant obstacle to ecological restoration can be the lack of ecological understanding by policymakers. Regulatory arguments against this project reflected limited knowledge of restoration ecology or habitat requirements for these organisms. For example, the reference sites used were highly degraded and represented refugia, not optimal habitat. The pre-restoration analysis in this project served to remind restoration ecologists and regulators of the importance of appropriately evaluating reference systems as models. There is a need to transfer technology such as that which was developed in this project to illustrate the value of restored wetlands and to distinguish between high quality and low quality wetlands.

CONCLUSIONS

The successful establishment of viable, reproducing populations of several rare wetland species at Howard's Branch is an example of what can be accomplished given the will to act now while opportunities still exist. This 3 acre project at Howard's Branch, from concept to construction, was achieved at a cost of less than $350,000.

This project has demonstrated the feasibility of restoring and enhancing rare ecosystems using damaged sites; e.g., former impoundments, abandoned sand mines, stormwater management facilities and degraded wetlands, to create seepage wetlands in the absence of existing peatlands; of establishing a viable, reproducing population of AWC and associated species (a rare plant community) in a created seepage wetland; of designing criteria for establishment of functional AWC wetlands within the historic range; and of "Uniting Forces For Action" by actively engaging the public and promoting interest, awareness, education and stewardship. Similar projects should be undertaken wherever possible.

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REGULATION AND PROTECTION OF PEATLANDS IN ANNE ARUNDEL COUNTY, MARYLAND

Judy Broersma-Cole[1]

Abstract—In the late 1990s several previously unknown peatlands and one population of Atlantic white cedar [*Chamaecyparis thyoides* (L.) BSP], were discovered on a peninsula in Anne Arundel County, Maryland. At the request of the Anne Arundel County legislative delegation, the Maryland Department of the Environment revised its regulations to include the new peatlands on the list of Nontidal Wetlands of Special State Concern (NTWSSC), a special category of wetland. These wetlands have the highest degree of State regulatory protection, including an expanded 100-foot buffer. Subsequently, the Anne Arundel County bog committee (Committee) helped develop a county protection program and regulations were implemented including 100-foot, 300-foot and 1,000-foot limited development buffers in the contributory drainage areas. The Committee continues working to protect, preserve and restore the peatlands of Anne Arundel County, Maryland. This paper presents the new State and county regulations that attempt to protect these peatlands and focuses on the recent, rapid development of these regulations.

Keywords: Anne Arundel County Bog Committee, Anne Arundel County, MD, Atlantic white cedar, Maryland Department of the Environment, peatlands, protection, regulation.

INTRODUCTION

In the late 1990s several previously unknown peatlands and one population of Atlantic white cedar [*Chamaecyparis thyoides* (L.) BSP], (AWC) were discovered on a peninsula in Anne Arundel County, Maryland. The recently found peatlands include Main Creek Bog, discovered in 1998, and located in a millpond just above tidal waters. It supports a plant community that includes leatherleaf [*Chamaedaphne calyculata* (L.) Moench], round-leaved (*Drosera rotundifolia* L.) and spatulate-leaved (*Drosera intermedia* Hayne) sundews, white beakrush [*Rhynchospora alba* (L.) Vahl], and yellow-eyed grass (*Xyris* sp). The Maryland Avenue bog, discovered in 1999, resembles a "pitch pine pocosin" and includes sweet bay magnolia (*Magnolia virginiana* L.), pitch pine (*Pinus rigida* Miller), leatherleaf, pitcher plants (*Sarracenia purpurea* L.), cotton-grass (*Eriophorum virginicum* L.), and large cranberry (*Vaccinium macrocarpon* Aiton). The Dill Road AWC swamp was previously known as a part of the Cypress Creek AWC swamp. Recent documentation (Sipple 1999) now shows it as a fragmented, separate peatland. Unfortunately, this fragmentation and stormwater pollution have destroyed the rare species (pitcher plant, round-leaved sundew, and leatherleaf) and many of the large AWC trees.

Description of Newly Found Bogs

In Anne Arundel County, there are 4,083 acres of watersheds that contribute to bogs. These bogs and their watersheds have been identified and surveyed (fig. 1) and their ecological characteristics have been documented. All occur on the exposed sands of the Magothy geological formation (Kirby and Matthews 1973). These cretaceous sediments are composed of light colored, highly acidic sands interspersed with shallow aquacludes of clay or sandstone cemented with iron oxides and carbonates (Vokes 1957). The aquacludes direct rainfall through the sands into depressional areas. The resulting seepage has a pH of 5.5 to 3.7 and limits growth to plants that tolerate highly acidic conditions.

These peatlands commonly have histic soils; permanent, shallow inundation; a living Sphagnum (*Sphagnum* sp) layer; and a community of acidophilic plants including sweet bay magnolia, pitcher plants, large cranberry, round-leaved and spatulate-leaved sundews, leatherleaf, giant cane [*Arundinaria gigantea* (Walter) Chapman], cotton-grass, white beakrush, and yellow-eyed grass.

Regulation of the Peat Bogs

The State of Maryland protects nontidal wetlands with the Nontidal Wetlands Protection Act of 1989 (Act), implemented on January 1, 1991. The Act and its code of Maryland regulations (Code of Maryland Regulations 2001) provide hierarchical levels of review and protection to nontidal wetlands based on their functions and values and the extent of the authorized impacts.

Regulations stipulate that a 25-foot buffer must completely surround all nontidal wetlands. Alterations of < 5,000 square feet of nontidal wetlands with no significant plant or wildlife value receive an expedited review and authorization process. Nontidal wetlands with significant plant or wildlife value receive a higher level of regulatory protection, including the opportunity for the public to provide input. These wetlands include those located in cold water fishery watersheds; wetlands containing State and/or Federal rare, threatened or endangered plant and/or animal species; large, relatively intact wetlands; Atlantic white cedar, American larch [*Larix laricina* (Duroi) K. Koch], bald cypress [*Taxodium distichum*

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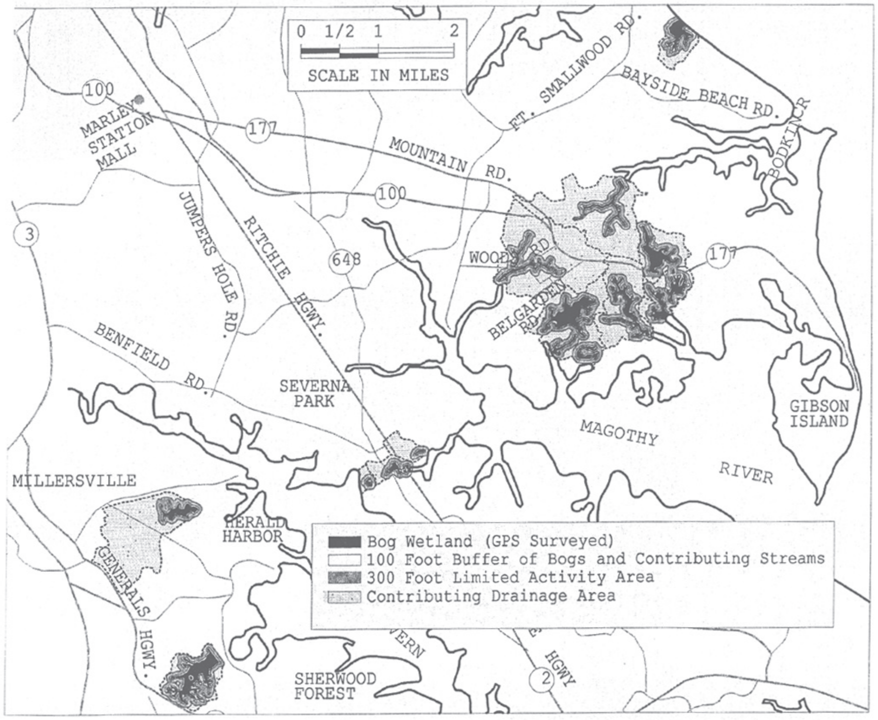

Figure 1—Anne Arundel County bog protection area guidance map.

(L.) Rich.], balsam fir [*Abies balsamea* (L.) Miller], and red spruce (*Picea rubens* Sarg.) swamps; vernal pools; Delmarva Bays; and peatlands, also known as bogs. The highest level of regulatory protection is assigned to Nontidal Wetlands of Special State Concern (NTWSSC). NTWSSC are defined in regulation (Code of Maryland Regulations 2001) as "…having exceptional ecological or educational value of Statewide significance." If the wetland is a NTWSSC, the regulated buffer surrounding the wetland is expanded to 100 feet, affording additional protection for the rare species and uncommon, unique and/or unusual habitats. Alterations to the 100-foot wetland buffer require avoidance where feasible; minimization to the extent practicable; and the implementation of specific best management practices (BMPs) for sediment control and stormwater management where applicable. Because the newly discovered peatlands were not classified as NTWSSC, they did not have the maximum 100-foot expanded buffer.

Many NTWSSC have complexes of rare, threatened or endangered species and/or are uncommon wetland ecotypes including peatlands and AWC swamps. These NTWSSC are specifically listed, by name and United States Geological Survey quadrangle map, in the State regulations. Ten of the Anne Arundel County peatlands were included in the original regulations as NTWSSC. Two of these ten peatlands include populations of AWC. However, AWC is not listed as a rare species in Maryland and, therefore, wetlands with AWC are not necessarily NTWSSC.

These newly discovered bogs were not listed as NTWSSC in regulation, and had only a 25-foot buffer and not the maximum allowed. In December of 1999, at the request of the Anne Arundel County legislative delegation, the Maryland Department of the Environment was asked to include the newly discovered peatlands in the list of NTWSSC already in regulation. Subsequently, a group of State, and county agencies, nongovernmental organizations, and interested citizens formed the Anne Arundel County bog committee (Committee).

The Committee was tasked with the identification, description, mapping, and naming of the peatlands. The Committee meets regularly to address peatland issues. It includes representatives from the Maryland Department of the Environment, Maryland Department of Natural Resources, Chesapeake Bay Critical Areas Commission, U.S. Fish and Wildlife Service, U.S. Army Corps of Engineers, Anne Arundel Community College, Magothy River Land Trust, Severn River Association, Anne Arundel County legislative delegation, Anne Arundel County council, Anne Arundel County government, Atlantic White Cedar Alliance, and local concerned citizens.

As a result of regular meetings, workshops, and fieldwork, State emergency regulations were developed to include all of the nontidal wetlands on the Mountain Road peninsula as a system of interconnected peatlands. The regulations took effect in September of 2000 and mandated a 100-foot regulated buffer and specific BMPs for watersheds contributing to NTWSSC. The emergency regulations were formally adopted by the State legislature in January of 2001.

As a result of the Committee activities, the Anne Arundel County council passed legislation in 2002 to regulate 100-foot, 300-foot, and 1,000-foot buffers in all known bog watersheds (County Council of Anne Arundel County, Maryland 2002). Future development in these buffer areas will be limited. No new development is permitted within the 100-foot buffer except minor (< 150 square feet) accessory structures to existing houses. No new development is permitted within the 300-foot regulated buffer except on existing legal lots; subdivision, stormwater management, and sewage treatment facilities are not permitted. New development is allowed within the 1,000-foot contributory drainage area under certain conditions. Development in that area is permitted provided non-structural stormwater management practices with no direct discharge into a peatland are utilized; impervious surfaces are generally < 10 percent of the site area; < 20 percent of forested areas are cleared; and established BMPs such as strict sediment controls and control of invasive plant species and yard waste are implemented.

The decision process for determining the appropriate State procedures for the regulation of nontidal wetlands in Anne Arundel County are outlined in Table 1. Regulations are from the Nontidal Wetland Protection Act of 1989, State of Maryland, and were adopted in 1991.

Additional Initiatives

The Committee is currently prioritizing properties for acquisition and looking for sources of funding. The State and county purchased 400 acres of Mountain Road property in 2002 for the Magothy River Greenway hub. This acquisition preserves two of the peatlands, South Gray's and Blackhole Creek bogs. Also, the State of Maryland has purchased two lots that were to be developed with single family homes in the Maryland Avenue bog complex. The purchases not only help preserve the bogs but also their essential ground water recharge areas. Landowners have protected additional building lots platted within the bogs for use as forest preservation credit within the Chesapeake Bay Critical Area.

The Committee has undertaken the restoration of a 3-acre sea level fen that had been buried under material dredged for a nearby community marina in the 1970s. The restoration area is located directly downstream of the North Grays Creek bog complex. The dredged material and its common reed [*Phragmites australis* (Cav.) Trin.] mat will be removed and replaced with clean Magothy formation sands dredged from North Gray's Creek tidal channels. In addition, the Anne Arundel County Department of Public Works has undertaken restoration and/or enhancement projects at several other peatlands including stormwater retrofits, fill removal, nonpoint source water quality improvements, and restoration of hydrology. When appropriate, AWC seedlings from local provenances will be planted in restoration and enhancement projects to augment existing populations.

Table 1—Hierarchical decision tree for regulating State nontidal wetlands in Anne Arundel County, Maryland

1a. The nontidal wetland impacts are minor (< 5,000 square feet with no significant plant or wildlife value 2

- 2a. The impacts are determined to be minimally adverse — activity exempt
- 2b. The impacts to nontidal wetlands are < 5,000 square feet — 25-foot regulated buffer and an expedited review w/o public notice

1b. The nontidal wetland impacts are major (> 5,000 square feet or with significant plant or wildlife value) 3

- 3a. Full permit review with public notice — 25-foot regulated buffer
- 3b. Nontidal Wetlands of Special State Concern listed in regulation — 100-foot regulated buffer

The Maryland Department of the Environment is currently in the process of revising its nontidal wetland regulations to include other Anne Arundel County peatlands that are not listed in regulation as NTWSSC. This will extend the added State regulatory protections and a 100-foot buffer to all of the known bogs, peatlands and AWC forests located in Anne Arundel County.

CONCLUSION

As a result of the intensive efforts of the Committee, new State and County regulations have been developed and implemented for protecting the peatlands of Anne Arundel County. Known peatlands have been delineated, named, and mapped. The regulatory buffers have been expanded to protect sensitive areas including upslope drainage and groundwater recharge areas. However, these regulations are not a panacea and do not prohibit all development in the watersheds feeding the peat bogs. In fact, some property owners have lobbied for a relaxation of the protection mechanisms. The committee is concerned that additional peatland discoveries in Anne Arundel County may be difficult, if not impossible, to add to the county protection program. As population growth pressures continue to escalate in the Chesapeake Bay watershed, more applications for permits to disturb sensitive areas are anticipated by the regulatory agencies. Several properties containing peatlands are for sale at exorbitant waterfront prices which governmental agencies cannot afford to purchase and preserve. However, the Committee is dedicated to continuing its work to protect these valuable wetland resources.

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ATLANTIC WHITE-CEDAR REGENERATION AND VEGETATION DYNAMICS AT PENN SWAMP, NEW JERSEY: TEN YEARS OF CHANGE

George L. Zimmermann and Kristin A. Mylecraine[1]

Abstract—Penn Swamp is a 55-ha Atlantic white-cedar [*Chamaecyparis thyoides* (L.) B.S.P.] stand in southern New Jersey. In 1989–1990, a 3.4 ha area was clearcut and a woven fence (> 3 m tall) was erected around 3-ha, leaving 0.4 ha unprotected from deer. Within both protected and unprotected areas, three logging slash treatments were delineated: no slash, slash, and double slash in a Latin square design. Permanent plots were established to monitor white-cedar regeneration and long-term vegetation dynamics. All vegetation was identified and measured by height class (< 0.3 m, 0.3 to < 0.6 m, 0.6 to < 1.3 m, and ≥ 1.3 m), annually between 1990 and 1995, and again in 2000. The effect of slash on cedar was statistically though probably not biologically significant in one instance (cedar over 1.3 m inside the fence). Deer impact remains dramatic, and manifested by inadequate cedar stocking, over dominance of shrubs, and few cedar reaching the 0.3 to 0.6 m height class in unprotected areas.

Keywords: Atlantic white-cedar, browse, deer, long-term effects, vegetation impacts.

INTRODUCTION

The decline of Atlantic white-cedar [*Chamaecyparis thyoides* (L.) B.S.P.], abbreviated AWC, during the last few centuries is supported by scientific and historical accounts (Cottrell 1929, Vermeule and Pinchot 1900). With the acknowledged value of wetlands, it has become increasingly important to maintain and restore wetland communities whenever possible. Many studies focus only on the first few years following restoration. Therefore, long-term data, especially as they relate to the effect of various treatments, are needed. This study examines vegetational changes over a 10-year period at an AWC regeneration site in southern New Jersey, and examines the long-term effects of slash and deer protection treatments on AWC regeneration.

In 1990, seven AWC restoration and regeneration study sites were established from the Jackson State Forestry Resource Education Center south to Belleplain State Forest in New Jersey. Several different cedar loss scenarios were studied, and the major factors thought to be limiting cedar regeneration were experimentally manipulated. These limiting factors included deer browsing, hardwood competition, logging slash (plant materials left after logging) and availability and type of cedar propagules. Penn Swamp, located in Wharton State Forest, was one of these study sites: a clearcut cedar harvest that focused on deer and logging slash effects. Following yearly measurements between 1990 and 1995, four of the seven sites, including Penn Swamp, were designated as long-term sites to be remeasured periodically until 2010. All vegetation would be remeasured on permanent plots to characterize long term community dynamics.

METHODS

Site

Penn Swamp is a 55-ha AWC stand located in Wharton State Forest in the southern New Jersey Pine Barrens. Delays from spiking AWC trees by unknown environmentalists caused the clearcutting of 3.4-ha to extend from late 1989 to early 1991. This did not affect the areas where the data were collected which all “started” at the same time. A pre-clearcut survey was done in 1989 on 34 randomly selected plots. The point sampling technique, which employs an angle gauge or prism was used to sample canopy trees, since this method tallies trees based on their sizes rather than their frequencies (Avery and Burkhart 1994). Table 1 gives a summary of the point sampled canopy composition. AWC dominated the overstory of the pre-cut forest in average diameter, frequency, and number of stems. The pre-cut understory included *Vaccinium corymbosum* L.*, Rhododendron viscosum* (L.) Torr.*, Leucothoe racemosa* (L.) Gray*, Gaylussaccia frondosa* (L.) T.&G.*, Acer rubrum* L.*, Nyssa sylvatica* Marshall*,* and AWC (Zimmermann 1992).

Zimmermann and others (1999) found the soils to be Typic histosols, varying from zero thickness near the edges of the swamp to 193 cm deep near an old stream channel that ran through the swamp. The peat at 186 cm was radiocarbon dated to 9980 ± 240 YBP. Charcoal, evidence of fire, was found throughout the extracted peat cores. Aerial photographs of Penn Swamp taken in 1940, 1962, 1974, and 1986 showed a dynamic ecosystem with different scales and levels of disturbance producing a multi-age AWC stand (Zimmermann and others 1999).

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Table 1—Pre-clearcut point sample of Penn Swamp canopy done in 1989 (n=34)

Species	Mean density	Standard error	Mean DBH	Frequency
	No./ha		*cm*	*percent*
Acer rubrum	246.3	68.9	20.1	44.1
Betula populifolia	15.7	15.7	11.6	2.9
Chamaecyparis thyoides	1189.9	123.6	25.0	97.0
Magnolia virginiana	324.7	208.1	7.6	11.7
Nyssa sylvatica	22.8	20.4	22.4	5.8

Experimental Design

A 3.05 m high woven deer fence was erected, encompassing 3.0 of the 3.4-ha clearcut area. The fenced portion of the site served as the treatment (deer exclusion) and the unfenced area (0.4 ha) served as the control. Within each of these protection levels (fenced and unfenced), two Latin squares were established to partition potential confounding variation in the x and y dimensions (fig. 1). In this case, the partitioned variations that were considered potential problems were: moisture gradients and residual stand effects. The Latin squares contained three slash treatments: no slash (N), normal slash (S), and double slash (D). Logging slash was composed of cedar branches and undesirable cut trees such as swamp hardwoods left on the site after logging. The slash treatments were established shortly after most of the stand was clearcut (November 1990). Due to delays by the State contractor, the deer fence was not erected until after the first winter had passed (May 1991). The fence has not been maintained since 2001, and gaps now exist in some places along its perimeter.

Vegetation Surveys

Data were collected on the site before and after the clearcut. Full surveys (n = 72) were conducted in 1989, 1991, 1992, 1993, 1994, 1995, and 2000. The full surveys included (1) a point sample of canopy trees, (2) a five square m plot to record vegetation by height classes 0.3 to < 0.6 meters, 0.6 to < 1.3 meters and ≥ 1.3 meters tall, (3) a 1 square m plot to record vegetation < 0.3 meters tall and to establish percent ground cover, and (4) a 2.5-m transect to record interception lengths of downed debris. The point sample has not been used since the mature trees were cut in 1989–90, and the debris transect was not conducted in the 2000 survey since no more information was needed on conditions for seed germination and establishment. Detailed plot instructions and a sample plot card can be found in earlier reports (Zimmermann 1992, 1993, 1995).

Data Analysis

The vegetation data generated consisted of density (number per hectare), percent browsed (number of total stems showing 15 percent or more of their branches clipped), and percent frequency (the percentage of total plots on which the species is present). The data here were analyzed by treatment, species and height classes. Density and percent browse (not presented here due to space limitations) were modeled to test the effects of the deer and slash treatments. Models were run by species and height classes. The statistical analyses for all sites followed the same procedure: analysis of variance (ANOVA) were run using PROC General Linear Models (GLM) on SAS (SAS Institute 2003). If data met ANOVA assumptions and statistically significant models were found, two multiple comparison tests (Tukey and Ryan-Einot-Gabriel-Welsch) were performed to determine treatment differences—if there was a rare disagreement among the two tests, Tukey was preferred. If data did not meet ANOVA assumptions, even after transformations, then nonparametric SAS procedures (NPAR1WAY) were performed.

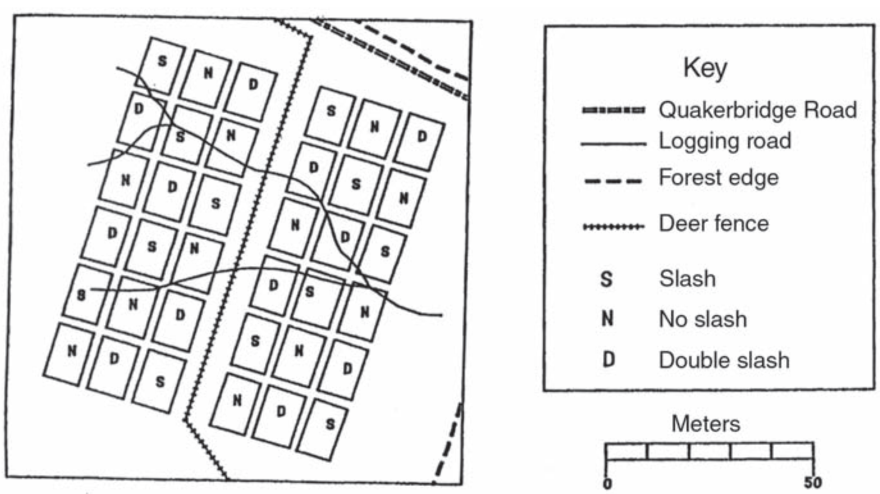

Figure 1—Close-up diagram of the experimental setup at Penn Swamp: deer exclosure and logging slash treatments.

Table 2—Vegetation at Penn Swamp in the year 2000 by treatment. mdens = are mean densities per hectare, sen = the standard error of the mean for mean density, mper = mean percent of stems browsed, sfreq = the percent of plots a species was found on. 3 = the < .3 m height class, 36 = the .3 to < .6 m height class, 63 = the .6 to < 1.3 m height class, 13 = ≥ 1.3 m height class. The sample size per treatment was 12, a total of 72 sample points. Statistical significance (prob < .05) for protection from deer is indicated for density and percent browse data by a superscript of "*". Slash was only significant for white cedar density in the ≥ 1.3 m height category.

Species	mdens3	sen3	mper3	sfreq3	mdens36	sen36	mper36	sfreq36	mdens63	sen63	mper63	sfreq63	mdens13	sen13	sfreq13
Fence—No Slash															
Acer rubrum	5000.0*	2302.8	0.0	33.3	0.0	0.0	0.0	0.0	666.7*	284.3	0.0	33.3	2166.7*	796.1	50.0
Chamaecyparis thyoides	74166.7	17772.1	20.8*	91.7	6666.7*	1620.6	26.8*	83.3	6666.7*	2035.0	10.0	58.3	46500.0*	7683.6	100.0
Clethra alnifolia	27500.0	12560.5	0.0	50.0	333.3*	224.7	0.0	16.7	666.7*	376.1	0.0	25.0	333.3*	224.7	16.7
FERNS	20000.0	10000.0	0.0	16.7	.	.	.	.	.	.	.	.	.	.	.
Gaylussacia frondosa	0.0	0.0	.	0.0	166.7	166.7	0.0	8.3	333.3*	224.7	0.0	16.7	2166.7*	1641.5	33.3
Kalmia angustifolia	3333.3*	3333.3	0.0	8.3	0.0*	0.0	.	0.0	0.0	0.0	.	0.0	0.0	0.0	0.0
Leucothoe racemosa	2500.0	1305.6	0.0	25.0	333.3	224.7	0.0	16.7	1333.3*	710.7	0.0	33.3	6000.0	3256.7	41.7
Lyonia ligustina	0.0	0.0	.	0.0	166.7*	166.7	0.0	8.3	166.7	166.7	0.0	8.3	666.7*	666.7	8.3
Magnolia virginiana	0.0	0.0	.	0.0	0.0	0.0	.	0.0	0.0	0.0	.	0.0	166.7	166.7	8.3
Nyssa sylvatica	0.0	0.0	.	0.0	0.0	0.0	.	0.0	0.0	0.0	.	0.0	666.7	512.5	16.7
Rhexia virginica	2500.0	2500.0	0.0	8.3	0.0	0.0	.	0.0	0.0	0.0	.	0.0	0.0	0.0	0.0
Rhododendron viscosum	27500.0	10597.7	0.0	50.0	1500.0*	857.2	0.0	33.3	2166.7*	625.6	0.0	66.7	8000.0*	2215.6	75.0
Rubus hispidus	19166.7	17427.7	0.0	16.7	0.0	0.0	.	0.0	0.0	0.0	.	0.0	0.0	0.0	0.0
Sedge or Carex spp	84166.7	23043.4	20.0	83.3	333.3*	333.3	0.0	8.3	666.7	666.7	0.0	8.3	0.0	0.0	0.0
Smilax spp.	1666.7	1666.7	0.0	8.3	0.0	0.0	.	0.0	0.0	0.0	.	0.0	166.7*	166.7	8.3
Toxicodendron radicans	10000.0	4438.1	0.0	33.3	0.0	0.0	.	0.0	0.0	0.0	.	0.0	0.0	0.0	0.0
Vaccinium corymbosum	50833.3	12089.2	0.0	100.0	4000.0*	1128.2	0.0	83.3	11833.3	2194.5	0.0	91.7	22833.3*	2865.2	100.0
Vaccinium vacillans	23333.3	11827.9	0.0	33.3	0.0	0.0	.	0.0	0.0	0.0	.	0.0	0.0	0.0	0.0
Fence—Slash															
Acer rubrum	2500.0*	1305.6	0.0	25.0	333.3	333.3	0.0	8.3	500.0*	261.1	0.0	25.0	1500.0*	609.3	41.7
Chamaecyparis thyoides	35000.0	8211.2	29.9*	83.3	5000.0*	1141.5	31.1*	75.0	6000.0*	2229.3	0.0	66.7	33500.0*	6015.8	100.0
Clethra alnifolia	17500.0	5919.3	20.0	50.0	1500.0*	1018.8	0.0	25.0	2333.3*	1150.3	0.0	33.3	2666.7*	1214.4	41.7
FERNS	20000.0	10000.0	0.0	16.7	.	.	.	.	.	.	.	.	.	.	.
Gaylussacia frondosa	0.0	0.0	.	0.0	0.0*	0.0	0.0	0.0	333.3*	224.7	0.0	16.7	1666.7*	948.2	25.0
Leucothoe racemosa	2500.0	2500.0	0.0	8.3	500.0	358.9	0.0	16.7	1333.3*	864.6	0.0	25.0	11500.0	4697.7	66.7
Lyonia ligustina	0.0	0.0	.	0.0	0.0*	0.0	0.0	0.0	166.7	166.7	0.0	8.3	166.7*	166.7	8.3
Nyssa sylvatica	0.0	0.0	.	0.0	0.0	0.0	0.0	0.0	166.7	166.7	0.0	8.3	666.7	376.1	25.0
Pinus rigida	833.3	833.3	0.0	8.3	0.0	0.0	.	0.0	0.0	0.0	.	0.0	0.0	0.0	0.0
Rhododendron viscosum	10833.3	4993.7	0.0	41.7	1000.0*	522.2	0.0	33.3	2333.3*	594.6	0.0	66.7	6000.0*	1633.0	66.7
Rubus hispidus	5000.0	5000.0	0.0	8.3	0.0	0.0	.	0.0	0.0	0.0	.	0.0	0.0	0.0	0.0
Sedge or Carex spp	89166.7	34519.6	0.0	83.3	0.0*	0.0	.	0.0	0.0	0.0	.	0.0	0.0	0.0	0.0
Smilax spp.	833.3	833.3	0.0	8.3	0.0	0.0	.	0.0	0.0	0.0	.	0.0	0.0*	0.0	0.0
Toxicodendron radicans	16666.7	10323.1	0.0	33.3	0.0	0.0	0.0	0.0	500.0	500.0	0.0	8.3	166.7	166.7	8.3
Vaccinium corymbosum	37500.0	8971.8	0.0	100.0	5000.0*	1446.0	0.0	75.0	10500.0	2271.4	0.0	91.7	20000.0*	3266.0	100.0
Vaccinium vacillans	2500.0	1794.4	0.0	16.7	166.7	166.7	0.0	8.3	166.7	166.7	0.0	8.3	0.0	0.0	0.0

(continued)

Table 2—Vegetation at Penn Swamp in the year 2000 by treatment. mdens = are mean densities per hectare, sen = the standard error of the mean for mean density, mper = mean percent of stems browsed, sfreq = the percent of plots a species was found on. 3 = the < .3 m height class, 36 = the .3 to < .6 m height class, 63 = the .6 to < 1.3 m height class, 13 = ≥ 1.3 m height class. The sample size per treatment was 12, a total of 72 sample points. Statistical significance (prob < .05) for protection from deer is indicated for density and percent browse data by a superscript of "*". Slash was only significant for white cedar density in the ≥ 1.3 m height category. (continued)

Species	mdens3	sen3	mper3	sfreq3	mdens36	sen36	mper36	sfreq36	mdens63	sen63	mper63	sfreq63	mdens13	sen13	sfreq13
Fence—Double Slash															
Acer rubrum	4166.7*	3361.6	0.0	16.7	0.0	0.0	0.0	0.0	166.7*	166.7	0.0	8.3	4833.3*	3415.3	50.0
Aronia spp.	0.0	0.0	.	0.0	0.0	0.0	.	0.0	0.0	0.0	.	0.0	166.7	166.7	8.3
Chamaecyparis thyoides	28333.3	9987.4	11.2*	58.3	2500.0*	783.3	12.5*	58.3	4833.3*	1266.3	20.0	83.3	24000.0*	4052.7	100.0
Clethra alnifolia	20000.0	12851.1	0.0	33.3	2500.0*	1234.0	0.0	41.7	2000.0*	984.7	0.0	41.7	2500.0*	2147.9	25.0
Fern spp.	140000.0	.	0.0	8.3	.	.	.	.	.	.	.	.	.	.	.
Gaylussacia frondosa	833.3	833.3	0.0	8.3	0.0*	0.0	0.0	0.0	666.7*	512.5	0.0	16.7	2500.0*	1578.7	25.0
Leucothoe racemosa	1666.7	1123.7	0.0	16.7	333.3	224.7	0.0	16.7	333.3*	333.3	0.0	8.3	4166.7	2194.5	41.7
Pinus rigida	833.3	833.3	0.0	8.3	0.0	0.0	.	0.0	0.0	0.0	.	0.0	0.0	0.0	0.0
Rhododendron viscosum	15833.3	4839.6	0.0	66.7	3000.0*	937.4	0.0	58.3	4333.3*	1226.8	0.0	75.0	10833.3*	3554.4	100.0
Rubus hispidus	833.3	833.3	0.0	8.3	0.0	0.0	.	0.0	0.0	0.0	.	0.0	0.0	0.0	0.0
Sedge or Carex spp	64166.7	16070.8	0.0	75.0	0.0*	0.0	.	0.0	0.0	0.0	.	0.0	0.0	0.0	0.0
Smilax spp.	5000.0	5000.0	0.0	8.3	0.0	0.0	.	0.0	0.0	0.0	.	0.0	500.0*	261.1	25.0
Toxicodendron radicans	4166.7	3361.6	0.0	16.7	0.0	0.0	.	0.0	0.0	0.0	.	0.0	0.0	0.0	0.0
Vaccinium corymbosum	41666.7	10998.2	0.0	75.0	4833.3*	1526.7	0.0	66.7	9333.3	1263.3	0.0	100.0	28000.0*	5087.1	100.0
Vaccinium vacillans	2500.0	2500.0	0.0	8.3	0.0	0.0	.	0.0	0.0	0.0	.	0.0	0.0	0.0	0.0
No Fence—No Slash															
Acer rubrum	23333.3*	8378.7	0.0	75.0	0.0	0.0	.	0.0	0.0*	0.0	.	0.0	166.7*	166.7	8.3
Betula populifolia	0.0	0.0	.	0.0	0.0	0.0	0.0	0.0	166.7	166.7	0.0	8.3	166.7	166.7	8.3
Chamaecyparis thyoides	12500.0	3508.6	96.9*	66.7	0.0*	0.0	0.0*	0.0	0.0*	0.0	.	0.0	0.0*	0.0	0.0
Clethra alnifolia	0.0	0.0	.	0.0	166.7*	166.7	0.0	8.3	0.0*	0.0	0.0	0.0	0.0*	0.0	0.0
Gaylussacia frondosa	0.0	0.0	.	0.0	1333.3*	666.7	0.0	33.3	5666.7*	2384.7	0.0	50.0	8500.0*	2641.5	66.7
Grass spp.	0.0	0.0	.	0.0	3333.3*	3333.3	0.0	8.3	2500.0*	1725.5	0.0	16.7	0.0	0.0	0.0
Ilex verticillata	10000.0	10000.0	0.0	8.3	0.0	0.0	.	0.0	0.0	0.0	.	0.0	0.0	0.0	0.0
Kalmia angustifolia	17500.0*	12500.0	0.0	33.3	1166.7*	833.3	0.0	25.0	333.3	224.7	0.0	16.7	0.0	0.0	0.0
Leucothoe racemosa	2500.0	1305.6	0.0	25.0	2833.3	1242.1	0.0	50.0	7333.3*	2260.8	0.0	66.7	11166.7	3613.6	75.0
Lyonia ligustina	10000.0	4438.1	0.0	50.0	2666.7*	1639.2	0.0	41.7	500.0	358.9	0.0	16.7	1333.3*	864.6	25.0
Mitchella repens	14166.7*	9728.0	0.0	16.7	0.0	0.0	.	0.0	0.0	0.0	.	0.0	0.0	0.0	0.0
Nyssa sylvatica	0.0	0.0	.	0.0	0.0	0.0	.	0.0	0.0	0.0	.	0.0	333.3*	333.3	8.3
Pinus rigida	0.0	0.0	.	0.0	333.3	333.3	100.0	8.3	0.0	0.0	0.0	0.0	333.3	333.3	8.3
Rhododendron viscosum	25000.0	8393.7	0.0	75.0	166.7*	166.7	0.0	8.3	333.3*	224.7	0.0	16.7	333.3	224.7	16.7
Rhus copallina	0.0	0.0	.	0.0	0.0	0.0	.	0.0	0.0	0.0	.	0.0	166.7	166.7	8.3
Rubus hispidus	28333.3	16914.8	0.0	25.0	0.0	0.0	.	0.0	0.0	0.0	.	0.0	0.0	0.0	0.0
Sedge or Carex spp	22500.0	13434.7	0.0	25.0	666.7*	449.5	0.0	16.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Smilax spp.	2500.0	1305.6	0.0	25.0	666.7	512.5	0.0	16.7	500.0	500.0	0.0	8.3	4500.0*	2536.1	33.3
Toxicodendron radicans	2500.0	2500.0	0.0	8.3	0.0	0.0	.	0.0	0.0	0.0	.	0.0	0.0	0.0	0.0
Vaccinium corymbosum	25833.3	7732.1	0.0	75.0	2833.3*	1057.7	0.0	58.3	10166.7	1833.3	0.0	100.0	34666.7*	5664.0	100.0
Vaccinium vacillans	2500.0	2500.0	0.0	8.3	0.0	0.0	.	0.0	0.0	0.0	.	0.0	0.0	0.0	0.0

(continued)

Table 2—Vegetation at Penn Swamp in the year 2000 by treatment. mdens = are mean densities per hectare, sen = the standard error of the mean for mean density, mper = mean percent of stems browsed, sfreq = the percent of plots a species was found on. 3 = the < .3 m height class, 36 = the .3 to < .6 m height class, 63 = the .6 to < 1.3 m height class, 13 = ≥ 1.3 m height class. The sample size per treatment was 12, a total of 72 sample points. Statistical significance (prob < .05) for protection from deer is indicated for density and percent browse data by a superscript of "*". Slash was only significant for white cedar density in the ≥ 1.3 m height category. (continued)

Species	mdens3	sen3	mper3	sfreq3	mdens36	sen36	mper36	sfreq36	mdens63	sen63	mper63	sfreq63	mdens13	sen13	sfreq13
								No Fence—Slash							
Acer rubrum	42500.0*	24187.6	0.0	66.7	166.7	166.7	0.0	8.3	500.0*	500.0	0.0	8.3	0.0*	0.0	0.0
Aronia spp.	0.0	0.0	.	0.0	0.0	0.0	.	0.0	0.0	0.0	.	0.0	166.7	166.7	8.3
Betula populifolia	0.0	0.0	.	0.0	166.7	166.7	0.0	8.3	0.0	0.0	0.0	0.0	166.7	166.7	8.3
Chamaecyparis thyoides	19166.7	5288.4	72.2*	75.0	0.0*	0.0	0.0*	0.0	0.0*	0.0	.	0.0	166.7*	166.7	8.3
Chamaedaphne calyculata	8333.3	8333.3	0.0	8.3	1500.0	1500.0	0.0	8.3	333.3	333.3	0.0	8.3	0.0	0.0	0.0
Clethra alnifolia	1666.7	1666.7	0.0	8.3	0.0*	0.0	.	0.0	0.0*	0.0	.	0.0	0.0*	0.0	0.0
FERNS	25000.0	5000.0	0.0	16.7	8000.0	0.0	0.0	16.7	0.0	0.0	0.0	0.0	.	.	.
Gaylussacia frondosa	2500.0	1305.6	0.0	25.0	333.3*	224.7	0.0	16.7	2333.3*	846.9	0.0	50.0	12166.7*	4549.1	75.0
Grass spp.	0.0	0.0	.	0.0	500.0*	500.0	0.0	8.3	0.0*	0.0	0.0	0.0	0.0	0.0	0.0
Ilex verticillata	39166.7	39166.7	0.0	8.3	0.0	0.0	.	0.0	0.0	0.0	.	0.0	0.0	0.0	0.0
Kalmia angustifolia	3333.3*	3333.3	0.0	8.3	500.0*	358.9	0.0	16.7	500.0	500.0	0.0	8.3	0.0	0.0	0.0
Leucothoe racemosa	1666.7	1123.7	0.0	16.7	666.7	376.1	0.0	25.0	3833.3*	716.0	0.0	83.3	7500.0	3403.4	58.3
Lyonia ligustina	5000.0	2302.8	0.0	33.3	166.7*	166.7	0.0	8.3	500.0	358.9	0.0	16.7	666.7*	512.5	16.7
Mitchella repens	54166.7*	54166.7	0.0	8.3	0.0	0.0	.	0.0	0.0	0.0	.	0.0	0.0	0.0	0.0
Nyssa sylvatica	0.0	0.0	.	0.0	0.0	0.0	.	0.0	0.0	0.0	.	0.0	500.0	261.1	25.0
Pinus rigida	0.0	0.0	.	0.0	0.0	0.0	.	0.0	0.0	0.0	.	0.0	166.7	166.7	8.3
Rhododendron viscosum	5833.3	3128.2	0.0	25.0	166.7*	166.7	0.0	8.3	0.0*	0.0	0.0	0.0	166.7*	166.7	8.3
Rubus hispidus	46666.7	18020.8	0.0	50.0	0.0	0.0	.	0.0	0.0	0.0	.	0.0	0.0	0.0	0.0
Sedge or Carex spp	39166.7	12459.5	0.0	58.3	4833.3*	2599.0	0.0	25.0	0.0	0.0	0.0	0.0	1666.7	1666.7	8.3
Smilax spp.	4166.7	4166.7	0.0	8.3	166.7	166.7	0.0	8.3	0.0	0.0	0.0	0.0	1500.0*	821.1	25.0
Toxicodendron radicans	1666.7	1666.7	0.0	8.3	0.0	0.0	.	0.0	0.0	0.0	.	0.0	0.0	0.0	0.0
Vaccinium corymbosum	29166.7	7120.0	0.0	75.0	2166.7*	716.0	0.0	58.3	14166.7	3785.6	0.0	91.7	46000.0*	6372.3	91.7
Vaccinium vacillans	6666.7	3097.7	0.0	33.3	0.0	0.0	.	0.0	0.0	0.0	.	0.0	0.0	0.0	0.0
								No Fence—Double Slash							
Acer rubrum	36666.7*	25055.5	0.0	75.0	166.7	166.7	0.0	8.3	0.0*	0.0	0.0	0.0	166.7*	166.7	8.3
Chamaecyparis thyoides	20000.0	6396.0	93.3*	66.7	1000.0*	674.2	50.0*	16.7	166.7*	166.7	0.0	8.3	166.7*	166.7	8.3
Chamaedaphne calyculata	12500.0	12500.0	0.0	8.3	0.0	0.0	.	0.0	0.0	0.0	.	0.0	0.0	0.0	0.0
Clethra alnifolia	2500.0	1794.4	0.0	16.7	166.7*	166.7	0.0	8.3	166.7*	166.7	0.0	8.3	0.0*	0.0	0.0
FERNS	30000.0	.	0.0	8.3	5000.0	1000.0	0.0	16.7	0.0	0.0	0.0	0.0	.	.	.
Gaylussacia frondosa	2500.0	1305.6	0.0	25.0	1000.0*	522.2	0.0	33.3	2333.3*	731.7	0.0	58.3	9833.3*	2467.5	83.3
Grass spp.	0.0	0.0	.	0.0	4000.0*	2984.8	0.0	25.0	2500.0*	1635.3	0.0	25.0	0.0	0.0	0.0
Ilex glabra	833.3	833.3	0.0	8.3	0.0	0.0	.	0.0	0.0	0.0	.	0.0	0.0	0.0	0.0
Ilex verticillata	3333.3	3333.3	0.0	8.3	0.0	0.0	.	0.0	0.0	0.0	.	0.0	0.0	0.0	0.0
Kalmia angustifolia	10000.0*	8348.5	0.0	16.7	1500.0*	1183.9	0.0	16.7	1000.0	1000.0	0.0	8.3	0.0	0.0	0.0
Leucothoe racemosa	2500.0	2500.0	0.0	8.3	333.3	333.3	0.0	8.3	3166.7*	903.1	0.0	66.7	15000.0	5357.0	75.0
Lyonia ligustina	4166.7	2599.0	0.0	25.0	666.7*	666.7	0.0	8.3	333.3	224.7	0.0	16.7	1333.3*	791.4	25.0

(continued)

Table 2—Vegetation at Penn Swamp in the year 2000 by treatment. mdens = are mean densities per hectare, sen = the standard error of the mean for mean density, mper = mean percent of stems browsed, sfreq = the percent of plots a species was found on. 3 = the < .3 m height class, 36 = the .3 to < .6 m height class, 63 = the .6 to < 1.3 m height class, 13 = ≥ 1.3 m height class. The sample size per treatment was 12, a total of 72 sample points. Statistical significance (prob < .05) for protection from deer is indicated for density and percent browse data by a superscript of "*". Slash was only significant for white cedar density in the ≥ 1.3 m height category. (continued)

Species	mdens3	sen3	mper3	sfreq3	mdens36	sen36	mper36	sfreq36	mdens63	sen63	mper63	sfreq63	mdens13	sen13	sfreq13
Magnolia virginiana	833.3	833.3	0.0	8.3	0.0	0.0	.	0.0	0.0	0.0	.	0.0	0.0	0.0	0.0
Mitchella repens	11666.7*	11666.7	0.0	8.3	0.0	0.0	.	0.0	0.0	0.0	.	0.0	0.0	0.0	0.0
Nyssa sylvatica	0.0	0.0	.	0.0	0.0	0.0	.	0.0	0.0	0.0	.	0.0	166.7	166.7	8.3
Pinus rigida	2500.0	1305.6	0.0	25.0	0.0	0.0	0.0	0.0	166.7	166.7	0.0	8.3	166.7	166.7	8.3
Rhododendron viscosum	13333.3	10894.4	0.0	16.7	333.3*	224.7	0.0	16.7	666.7*	449.5	0.0	16.7	500.0*	358.9	16.7
Sedge or Carex spp	44166.7	20205.7	0.0	41.7	9500.0*	6652.1	0.0	33.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Smilax spp.	6666.7	6666.7	0.0	8.3	0.0	0.0	0.0	0.0	166.7	166.7	0.0	8.3	1666.7*	1344.6	16.7
Vaccinium corymbosum	24166.7	7432.4	0.0	66.7	1666.7*	689.0	0.0	41.7	13333.3	2538.6	0.0	83.3	31666.7*	6256.3	100.0
Vaccinium vacillans	833.3	833.3	0.0	8.3	0.0	0.0	.	0.0	0.0	0.0	.	0.0	0.0	0.0	0.0

RESULTS AND DISCUSSION

The first 5 years of measurements showed the severe impact of white tail deer on AWC regeneration (Zimmermann 1992, 1993, 1995, 1997). This effect is still obvious 10 years later (year 2000). Table 2 shows year 2000 vegetation density and frequency by treatment combinations. AWC generally are denser and more frequent inside the fence than outside.

In 2000, protection from deer was not statistically significant for cedar under 0.3 m but was significant in all larger height classes. For AWC ≥ 1.3 m, there was also a statistically significant interaction between protection from deer and slash treatment ($p < .017$). Deer protection was also significant for a number of other species: statistically significant differences for density were found for *Acer rubrum, Clethra alnifolia* L.*, Gaylussacia frondosa, Lyonia lingustrina* (L.) DC*, Rhododendron viscosum*, and *Vaccinium corybosum.* Figure 2 shows the treatment effects in 2000 by vegetation classes. Deer impact on cedar and the shrub community is profound. In 2001, Dr. Gerry Moore, Director, the Department of Science at the Brooklyn Botanical garden in New York conducted three surveys (spring, summer, and fall) of the Penn Swamp study area to document any other plant species not found by the permanent plots. Table 3 presents those species by protection treatment, not already listed in Table 2 and/or those found in the 2001 survey. Species diversity was richer outside the fence where there was inadequate cedar regeneration. The dense, uniform, and closed cedar canopy inside the fence probably created less environmental heterogeneity and opportunities for other plant species.

In the first 5 years of the study, slash appeared to only affect cedar in the lowest height class with little impact in the higher height classes (Zimmermann 1992, 1993, 1995, 1997). In 2000, the only statistically significant impact on vegetation from slash occurred on the > 1.3 m AWC inside the fence. However, the difference between the highest AWC density inside the fence, 46,500 cedar/ha (> 1.3 m), found in the no slash treatment, is probably not biologically different from the lowest density (24,000/ha in double slash, inside the fence treatment for > 1.3 m stems). There will probably be more than adequate AWC even at the double slash levels (inside the fence) to bring the stand to a 'normal' stocking level. The reduced impact of slash found in 2000 could be due to the low inherent biological impact of slash at the levels studied here, but could also be due to the degradation of the slash, whose layering has decreased as well as decomposing over 10 years. Korstian and Bush (1931) and Little (1950) maintain that slash has a negative impact on AWC regeneration, while Cottrell (1929) thought differently. It is easy to see how our results could support either view: double slash did reduce cedar density but the densities are still high enough to maintain "normal" or fully stocked levels provided the cedar remain dominant. Because logging slash can vary from site to site, more studies are needed to examine these issues in depth.

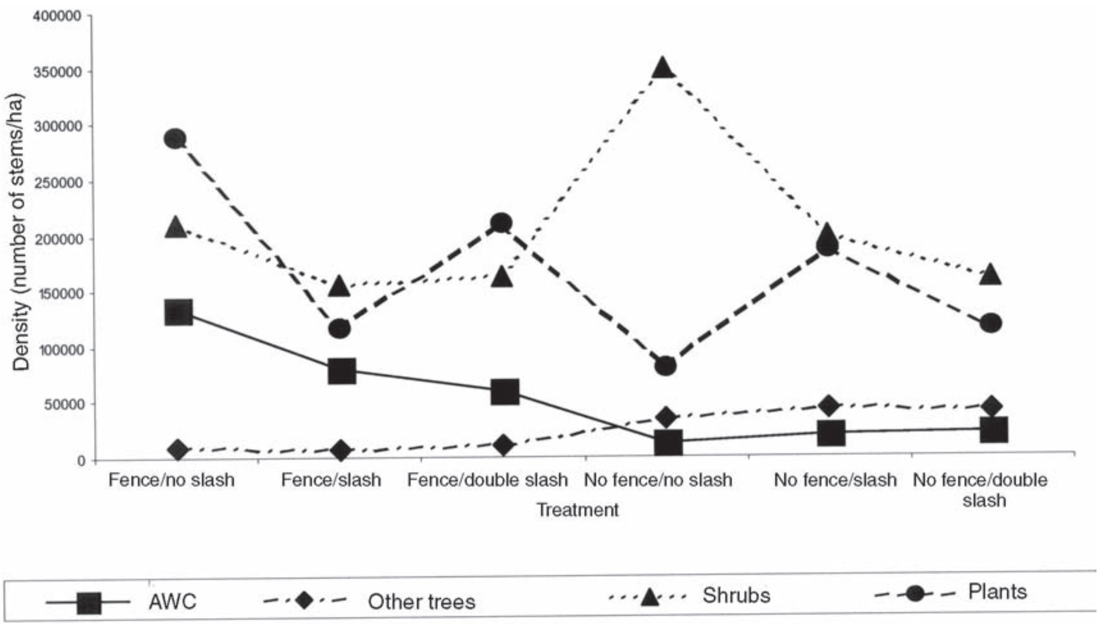

Figure 2—The overall effect of treatments on vegetation by category (all height classes combined) in the year 2000 at Penn Swamp. AWC = Atlantic white-cedar, other trees = all other tree species combined, shrubs = all multi-stemmed woody plants combined, and plants = all herbaceous and vine-like plants combined.

Table 3—Species listed are those not already found in table 2 (because their densities were not statistically significant between fenced and unfenced), and/or species found in spring, summer, and fall 2001 floral surveys conducted in Penn swamp treatments

Inside the deer fence	Outside the deer fence
Juncus effusus (L.)	*Andropogon glomeratus (Walt.) B.S.P.*
Woodwardia virginica (L.) Sm	*Andropogon virginicus* (L.)
	Carex atlantica (Bailey)
	Carex striata var. brevis (Bailey)
	Drosera intermedia (Hayne)
	Drosera rotundifolia (L.)
	Dulichium arundinaceum (L.) Britt.
	Juncus effusus (L.)
	Magnolia virginiana (L.)
	Muhlenbergia uniflora (Muhl.) Fern
	Thelypteris simulata (Davenport) Nieuwl.
	Triadenum virginicum (L.) Raf.
	Woodwardia areolata (L.) T. Moore
	Woodwardia virginica (L.) Sm.
	Xyris difformis Chapman

CONCLUSION

Various auto and allogenic forces over centuries have created a complex, multi-aged forest at Penn Swamp. This heterogeniety demands a management plan that takes this variability into consideration. Regeneration over the last 10 years has been heavily impacted by white tail deer. These effects are profound and mirror similar past findings in New Jersey (Little and others 1958). The slash load at Penn Swamp, while negatively affecting AWC regeneration in only the double slash treatment, will still probably allow a 'full' or normal stocking of AWC to occur, provided they are protected from deer. Like all natural systems, many forces are at play at Penn Swamp; that is why this and similar vegetation studies will be continued for 10 more years. More long-term data are needed from many more sites so management decisions can be adequately assessed.

ACKNOWLEDGMENTS

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Conference Attendees

1 *Paul Ericksson*
2 *Kenneth O. Summerville*
3 *Bill Pickens*
4 *Eric Hinesley*
5 *Darren Rickwood*
6 *Steve Barry*
7 *Anne Hairston-Strang*
8 *Jennifer Cure*
9 *Phil Sheridan*
10 *Steve Whittemore*
11 *Mike Petruncio*
12 *Aimlee Laderman*
13 *Scott Smith*
14 *Robert Williams*
15 *Keith Underwood*
16 *Doug Back*
17 *Roger Stallard*
18 *Johnathan Boynton*
19 *Bill Sipple*
20 *Robert Heeren*
21 *Myvonwynn Hopton*
22 *Neil Pederson*
23 *Brian Martin*
24 *Kevin Thompson*
25 *David Walbeck*
26 *Bill Cure*
27 *Judy Broersma-Cole*
28 *Lara Gengarelly*
29 *Lisa Hartrick*
30 *Keith Benedict*
31 *Jennifer Johnson*
32 *Mike Hart*
33 *William Zipse*
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35 *Joe Swartz*
36 *Kristin Mylecraine*
37 *Bob Belcher*
38 *Ed Crawford*
39 *Craig Clarke*
40 *George Zimmermann*
41 *Marianne Burke*

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Burke, Marianne K.; Sheridan, Philip, eds. 2005. Atlantic white cedar: ecology, restoration, and management: Proceedings of the Arlington Echo symposium. Gen. Tech. Rep. SRS-91. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station. 74 p.

A symposium was held on the globally threatened and coastally restricted tree species, Atlantic white-cedar (Chamaecyparis thyoides (L) B.S.P.) at the Arlington Echo Outdoor Education Center, Millersville, MD, in June 2003. The theme of the symposium was "Uniting Forces for Action," and participants in the symposium came from throughout the range of this species, from New England to the Gulf Coast. More than 15 papers and posters were presented addressing topics on community and ecosystem ecology of natural Atlantic white cedar (AWC) habitats, ecosystem restoration and stewardship efforts, the current range of the species, information on range-wide AWC genetics, and the long-term effects of various silvicultural manipulations on the entire vegetation community in AWC habitat.

Keywords: Atlantic white-cedar, Chamaecyparis thyoides, coastally restricted, ecosystem restoration, genetic variation.

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