Appendix A
Species Specific Removal Methods

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Tree of Heaven

ELEMENT STEWARDSHIP ABSTRACT
for Ailanthus altissima
Tree-of-Heaven
To the User:

Element Stewardship Abstracts (ESAs) are prepared to provide The Nature Conservancy's Stewardship staff and other land managers with current management-related information on those species and communities that are most important to protect, or most important to control. The abstracts organize and summarize data from numerous sources including literature and researchers and managers actively working with the species or community.

We hope, by providing this abstract free of charge, to encourage users to contribute their information to the abstract. This sharing of information will benefit all land managers by ensuring the availability of an abstract that contains up-to-date information on management techniques and knowledgeable contacts. Contributors of information will be acknowledged within the abstract and receive updated editions. To contribute information, contact the editor whose address is listed at the end of the document.

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The Nature Conservancy
Element Stewardship Abstract
For Ailanthus altissima

I. IDENTIFIERS

Common Name: TREE-OF-HEAVEN Global Rank: G?

General Description:
Deciduous trees in the family Simaroubaceae, native to China.

II. STEWARDSHIP SUMMARY

Ailanthus is a fast growing tree, a prolific seed producer, a persistant stump and root sprouter and an aggressive competitor with respect to the surrounding vegetation. It occurs primarily in disturbed areas, though it may invade undisturbed habitats. It was brought into California mainly by the Chinese who came to California during the goldrush in the 1890's, and frequently occurs in abandoned mining sites. Little work has been done on developing biological or chemical control methods. The most effective means of control may be to pull seedlings by hand before the tap root develops.

III. NATURAL HISTORY

Range:
In the Americas, Ailanthus occurs from Canada to Argentina. Native to China. Also escaped in Europe.

Habitat:
Ailanthus is native to central China, where its history is as old as the written language of the country (Hu 1979). Little information is available on its ecology in China, although Hu (1979) reviews its cultural importance and value for wood products and medicine.

The species was apparently introduced into America by two different routes. The first route began with Pierre d'Incarville mistaking it for the lacquer tree in China and sending seeds to England around 1751 (Feret and Bryant 1974, Hu 1979). It was then introduced to America by a Philadelphia gardener in 1784 (Hu 1979). Because of its rapid growth and ability to grow in unfavorable conditions with little care, it became a common stock in eastern nurseries by 1840. The second route was through Chinese miners. During the days of the California gold rush, many Chinese miners brought ailanthus seeds with them as they settled in California, probably because of its medicinal and cultural importance to them.

Escaping from cultivation and quickly becoming established on both coasts, ailanthus has expanded its range considerably since its initial introductions. Specimens from the Harvard University Herbarium indicate that it "runs wild from Massachusetts...to Oregon ... and from Toronto, Canada ... to Argentina ..." (Hu 1979). In some localities ailanthus is so well established that it appears to be a part of the native flora (Little 1974).

In the eastern United States, the frequency of ailanthus occurrences increases as one nears the cities. In neglected urban areas, ailanthus grows "as trees close to buildings, as hedges, or as bushy aggregates along railroad tracks, highway embankments, walls at the ends of bridges and overpasses, or in cracks of sidewalks and along fences" (Hu 1979). Although it is usually found in disturbed areas, it occasionally spreads to undisturbed areas. Kowarik (1983) views human settlements as centers of its distribution and roads as migration routes.

In California ailanthus is widely naturalized in cismontane areas, especially around old dwellings and mining settlements (Munz and Keck 1973). It has become established in Pleasants Valley, Solano and Marin counties, Berkeley, Vacaville, Petaluma, San Andreas, Angel's Camp, Columbia, and in various places in the Sacramento Valley (Robbins et al. 1951).

Ecology:
Although ailanthus is sensitive to frost damage during its early years (Adamik and Brauns 1957), 6-year-old trees have survived winters of -33 centigrades accompanied by high winds (Zelenin 1976). Although Koffer (1895) suggested that ailanthus was unable to withstand the prolonged dry seasons of the Midwest, Dubroca and Bory (1981) commented on the "drought resistance" of the species. Dry soils are probably more suitable for its growth than wet soils (Adamik and Brauns 1957).

Ailanthus does well on very poor soils. Adamik and Brauns (1957) cultivated the species on rather thin topsoil and it "thrives even on stony ground." The tree has been used in revegetating acid mine spoils, tolerating a pH of less than 4.1, soluble salt concentrations up to 0.25 mmhos/cm and phosphorus levels as low as 1.8 ppm (Plass 1975). The tolerance of ailanthus to soil salinity is a disputed point in the literature. Opinions range from "salty soils not suitable for growth" (Adamik and Brauns 1957) to ailanthus "growing well on very saline shell sands (Lavrinenko and Volkov 1973). Intermediate views are expressed by Brogowski et al. (1977), Semoradova and Materna (1982) and Zelenin (1976).

Ailanthus has been planted widely in urban areas because of its ability to tolerate atmospheric pollution. Its ability to adapt to "the dirt and smoke, the dust and drought of cities" was recognized nearly 100 years ago (Sargent 1888). More recently ailanthus has been observed to survive cement dust near cement and lime works (Klincsek 1976); it is moderately resistant to fumes produced by the coke and coal-tar industry (Kozyukina and Obraztsova 1971); its leaves absorb significant amounts of sulfur in areas of high traffic flow (Kim 1975); it can accumulate high levels of mercury in its tissues (Smith 1972); and it is somewhat resistant to ozone exposure (Davis et al. 1978).

Although ailanthus may suffer from root competition by other trees already established in an area (Cozzo 1972), usually it competes successfully with other plants (Cozzo 1972, Hu 1979) and is considered a "dangerous weed" in forest plantations (Magic 1974). A high degree of shade tolerance gives ailanthus a competitive edge over other plant species (Grime 1965). The production of toxic chemicals by ailanthus may also explain the success of this plant. An aqueous extract of ailanthus leaves has been shown to be toxic to 35 species of gymnosperms and 10 species of angiosperms (Mergen 1959). This may be important in limiting natural succession in ailanthus stands. The toxicity levels are highest in the leaves during the early part of the growing season and are maintained at high levels at least until October (Voigt and Mergen 1962).

Reproduction:
Ailanthus reproduces both sexually and asexually. Asexual reproduction is by vegetative sprouting from stumps or root portions (Hu 1979). Flowering occurs rather late in the spring (June). Ailanthus has the longest winter dormancy of all the trees in its native Chinese habitat (Hu 1979). Precocious flowering is not a rare occurrence in this species and has been observed in seedlings only 6 weeks after germination (Feret 1973).

Seeds ripen in large crowded clusters from September to October of the same year and may persist on the tree through the following winter (Little 1974, Hu 1979). An individual tree can produce 325,000 seeds per year which are easily wind-dispersed (Bory and Clair-Maczulajtys 1980). These seeds average over 30,000 per kilogram. This amount yields up to 6-7000 "usable plants" (Little 1974). Limited testing of ailanthus seeds indicate that they have dormant embryos, and that germination is benefited by stratification on moist sand for 60 days at 41 F (Little 1974).

Seedlings establish themselves rapidly by producing a well formed tap root in less than three months (Adamik and Brauns 1957). In more compacted soils these seedlings put forth long rope-like lateral roots to exploit a greater soil volume (Rabe and Bassuk 1984). Ailanthus grows quickly in full sunlight and averages a meter of growth in height per year for at least the first 4 years (Adamik and Brauns 1957). The trees may grow to 15-20 meters tall but have a rather short lifespan of less than 50 years (Adamik 1955).

Impacts:
Although only occasionally found in nondisturbed areas (Kowarik 1983), Ailanthus is a prolific seed producer, grows rapidly and can successfully compete with the native vegetation. It produces toxins which prevent the establishment of other species (Mergen 1959). The root system is aggressive enough to cause damage to sewers and foundations (Hu 1979).

Ailanthus was not nominated by any specific preserve manager, but is recognized by TNC staff as an important exotic weed. A recent survey (2 March 1985) of CNPS members showed a wide distribution of this tree throughout California. Members of both the Mt. Lassen and Sequoia chapters consider it a major pest at low elevations. There are also reports of it growing in Santa Cruz, Riverside, San Bernardino, Los Angeles and San Diego counties.

IV. CONDITION

Threats:
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Trend:
Trend in native range in China not known, but the species has become much more abundant globally in the past century.

V. MANAGEMENT/MONITORING

Management Requirements:
Weed control involves three fundamental objectives: prevention, eradication and control.

From a practical viewpoint, methods of weed management are commonly categorized under the following categories: physical, thermal, managerial, biological, and chemical (Watson 1977). Physical methods include both manual and mechanical methods. Thermal methods include both broadcast burning or spot treatment with a flame thrower. Managerial methods include the encouragement of competitive displacement by native plants and prescribed grazing. Biological control is usually interpreted as the introduction of insects or pathogens which are highly selective for a particular weed species. Chemical control includes both broadcast and spot application.

The most desirable approach is that of an integrated pest management plan. This involves the optimum use of all control strategies to control weeds. This approach is generally accepted as the most effective, economical, and environmentally sound long-term pest control strategy (Watson 1977). In cases where more than one control technique is used, the various techniques should be compatible with one another. Broadcast herbicide application, for example, may not work well with certain managerial techniques (i.e., plant competition).

PHYSICAL CONTROL. The two types of physical control methods discussed below, manual and mechanical, produce slash (i.e., cutting debris) that can be disposed of by several techniques. If cut before seeds

are produced it may be piled and left for enhancement of wildlife habitat (i.e., cover for small mammals). Debris may be fed through a mechanical chipper and used as mulch during revegetation procedures. Care should be taken to prevent vegetative reproduction from cuttings. Burning the slash piles is also effective in disposing of slash.

MANUAL CONTROL. Manual methods use hand labor to remove undesirable vegetation. These methods are highly selective and permit weeds to be removed without damage to surrounding native vegetation.

Hand Pulling: Ailanthus is probably best controlled by manual removal of young seedlings. Seedlings are best pulled after a rain when the soil is loose. This facilitates removal of the rooting system, which may resprout if left in the ground. After the tap root has developed, this would be extremely difficult. Plants should be pulled as soon as they are large enough to grasp but before they produce seeds.

The Bradley Method is one sensible approach to manual control of weeds (Fuller and Barbe 1985). This method consists of hand weeding selected small areas of infestation in a specific sequence, starting with the best stands of native vegetation (those with the least extent of weed infestation) and working towards those stands with the worst weed infestation. Initially, weeds that occur singly or in small groups should be eliminated from the extreme edges of the infestation. The next areas to work on are those with a ratio of at least two natives to every weed. As the native plant stabilizes in each cleared area, work deeper into the center of the most dense weed patches. This method has great promise on nature reserves with low budgets and with sensitive plant populations. More detailed information is contained in Fuller and Barbe (1985).

Cutting: Manually operated tools such as brush cutters, power saws, axes, machetes, loppers and clippers can be used to cut ailanthus. This is an important step before many other methods are tried, as it removes the above-ground portion of the plant. For thickly growing, multi-stemmed shrubs and trees, access to the base of the plant may not only be difficult but dangerous where footing is uncertain.

Hand Digging: The removal of rootstocks by hand digging is a slow but sure way of destroying weeds which resprout from their roots. The work must be thorough to be effective as every piece of root that breaks off and remains in the soil may produce a new plant. Such a technique is only suitable for small infestations and around trees and shrubs where other methods are not practical.

Girdling: Girdling involves manually cutting away bark and cambial tissues around the trunks of undesirable trees such as ailanthus. This is a relatively inexpensive method and is done with an ordinary ax in the spring when the trees are actively growing. Hardwoods are known to resprout below the girdle unless the cut is treated with herbicides. Although it may be undesirable to leave standing dead trees in an area, this technique has been shown to reduce stump sprouting in live oaks, and may be a useful technique for controlling ailanthus.

MECHANICAL CONTROL. Mechanical methods use mechanized equipment to remove above ground vegetation. These methods are often non-selective in that all vegetation on a treated site is affected. Mechanical control is highly effective at controlling woody vegetation on gentle topography with few site obstacles such as rocks, stumps or logs. Most mechanical equipment is not safe to operate on slopes over 30 percent. It is also of limited use where soils are highly susceptible to compaction or erosion or where excessive soil moisture is present. Site obstacles such as rocks, stumps or logs also reduce efficiency.

Chopping, Cutting or Mowing: Saplings may be trimmed back by tractor-mounted mowers on even ground or by scythes on rough or stony ground. Unwanted vegetation can be removed faster and more economically in these ways than by manual means and with less soil disturbance than with scarification. However, these methods are non-selective weed eradication techniques. They reduce the potential for biological control through plant competition and open up new niches for undesirable vegetation. In addition, wildlife forage is eliminated.

Saplings usually require several cuttings before the underground parts exhaust their reserve food supply. If only a single cutting can be made, the best time is when the plants begin to flower. At this stage the reserve food supply in the roots has been nearly exhausted, and new seeds have not yet been produced. After cutting or chopping with mechanical equipment, ailanthus resprouts from root crowns in greater density if not treated with herbicides.
PRESCRIBED BURNING. A flame thrower or weed burner device can be used as a spot treatment to heat-girdle the lower stems of small trees. This technique has advantages of being less costly than basal and stem herbicide treatments and is suitable for use during wet weather and snow cover. Ailanthus resprouts after heat-girdling (Cozzo 1972).

MANAGERIAL CONTROL. In most cases ailanthus prevents the establishment of other native plants and must be initially removed. Following physical or thermal removal of mature plants, root crowns must be treated to prevent resprouting. Seedlings of native plant species usually cannot establish fast enough to compete with sprout growth from untreated stumps. Ailanthus is shade tolerant, so presumably can and will sprout under other plants.

Prescribed grazing: The continued removal of the tops of seedlings and resprouts by grazing animals prevents seed formation and also gradually weakens the underground parts. Grazing must be continued until the seedbank is eliminated, as the suppressed plants return quickly after livestock is removed and begin to dominate pastures again.

BIOLOGICAL CONTROL. The term "biological control" is used here to refer to the use of insects or pathogens to control weeds. The introduction of exotic natural enemies to control plants is a complex process and must be thoroughly researched before implementation to prevent biological disasters. Such tools are not normally suitable for preserve managers to implement.

Biological control of ailanthus has not been addressed to any extent beyond the anecdotal stage. No susceptibilty of ailanthus to parasites was found or noticed in Austrian nurseries (Adamik and Brauns 1957). French (1972) notes that the zonate leafspot (Cristulariella pyramidalis) causes defoliation of ailanthus in Florida. In India, Atteva fabricella is considered an ailanthus defoliator (Misra 1978) and Italian seedlings, weakened by cold, were weakly parasitized by the fungus Placosphaeria spp. (Magnani 1975).

Please notify the California Field office of The Nature Conservancy of any field observations in which a native insect or pathogen is seen to have detrimental effects on ailanthus. These reports will be used to update this Element Stewardship Abstract. Management techniques which may encourage the spread of species-specific agents may be desirable in controlling ailanthus.

CHEMICAL CONTROL. Methods of chemical control of ailanthus are poorly explored in the literature. Detailed information on herbicides in general is available in such publications as Weed Science Society of America (1983). The Weed Science Society reference gives specific or USDA (1984) information on nomenclature, chemical and physical properties of the pure chemical, use recommendations and precautions, physiological and biochemical behavior, behavior in or on soils and toxological properties for several hundred chemicals. Comprehensive coverage of this information will not be presented in this Element Stewardship Abstract. In applying herbicides it is recommended that a dye be used in the chemical mixture to mark the treated plants and thus minimize waste.

The following discussion highlights herbicide application methods which may be useful in controlling ailanthus. Herbicides may be applied non-selectively (i.e., broadcast applications) or selectively (i.e., spot applications). Both types of applications have advantages and disadvantages and will be discussed separately.

Broadcast Herbicide Application: In general, when using broadcast application methods, plants should be sprayed only when in full leaf. Results are poor prior to full leaf development. The best results have been obtained when plants are in the fruiting stage in late summer or early autumn (Mathews 1960).

Kolarvskij (1967) reports that 2,4-D can stop seedling growth in alianthus, and Sterrett et al. (1971) found that a mixture of 2- chloroethyl phosphoric acid and potassium iodide gives 80-100% defoliation of ailanthus within 3 weeks.

Spot Chemical Methods: Spot chemical methods consist of various techniques for manually applying herbicides to individual plants or small clumps of plants (such as stump resprouts). These methods are highly selective as only specific plants are treated. They are most efficient when the density of stems to be treated is low.

Jones and Stokes Associates (1984) reviewed a variety of spot chemical techniques. The following is an excerpt from this report, listing techniques in order of increasing possibility of herbicide exposure to the environment or to humans in the vicinity of treated plants.

1) Stem injection: Herbicides are injected into wounds or cuts in the stems or trunks of plants to be killed. The herbicide must penetrate to the cambial tissue and be water-soluble to be effective. The chemical is then translocated throughout the tree and can provide good root-kill, and thus prevents resprouting.

2) Cut stump treatment: Herbicides are directly applied to the cambial area around the edges of freshly cut stumps. Application must occur within 5-20 minutes of cutting to ensure effectiveness. McHenry (1985) suggests late spring as the best season to do this. In early spring sap may flow to the surface of the cut and rinse the chemical off. At other times of the year translocation is too poor to adequately distribute the chemical. Applications may be made with backpack sprayers, sprinkling cans, brush and pail, or squeeze bottles. This treatment is effective in killing root systems of sprouting hardwoods. Picloram should not be used for this technique as it is known to "flashback" through root grafts between treated and untreated plants and may damage the untreated individuals.

Tre-Hold, an asphalt based formulation containing 1% NAA ethylester has been used as a sprout retardant on ailanthus with varying degrees of effectiveness (Amchem Products 1967).

3) Basal/Stem sprays: High concentrations of herbicides in oil or other penetrating carriers are applied, using backpack sprayers, to the basal portion of stems to be killed. The oil carrier is necessary for the mixture to penetrate bark and enter the vascular system. This method gives good root kill, especially in the fall when vascular fluids are moving toward the roots. This method may be easier to use with small diameter stems than the two previous techniques.

4) Herbicide pellets: Pelletized or granular herbicides are scattered at the bases of unwanted plants. Subsequent rainfall dissolves the pellets and leaches the herbicide down to the root system. Optimal time for treatment is towards the end of the rainy season to prevent leaching beyond the root zone.

Management Programs:
Tim Thomas (1985) has removed a small stand of ailanthus in the Santa Monica Mountains National Recreation Area, by pulling up young saplings and is currently monitoring the site to see if it resprouts.

Contact: Tim Thomas, Park Ranger, Santa Monica Mountains National Recreation Area 22900 Ventura Blvd. Woodland Hills, CA 91364, (213) 888-3440.

Monitoring Requirements:
Monitoring is needed to determine the presence of ailanthus on or near preserves.

VI. RESEARCH

Management Research Needs:
What types of undisturbed habitats does it invade? How do native species respond to ailanthus toxins, and how is recovery potential of an area previously occupied by ailanthus affected by these toxins? What is the chemical make-up of these toxins? What can be used to buffer the effects of the toxins so that understory native seedling growth is encouraged? At what age is the tap root so long that it precludes ailanthus removal by hand?

VII. ADDITIONAL TOPICS

VIII. INFORMATION SOURCES

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IX. DOCUMENT PREPARATION & MAINTENANCE

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Marc C. Hoshovsky, CAFO (1988)

PROCELAIN-BERRY

Ampelopsis brevipedunculata (Maxim.) Trautv.

Grape family (Vitaceae)

NATIVE RANGE

Northeast Asia - China, Korea, Japan, and Russian Far East

DESCRIPTION

Porcelain-berry is a deciduous, woody, perennial vine. It twines with the help of non-adhesive tendrils that occur opposite the leaves and closely resembles native grapes in the genus Vitis. The stem pith of porcelain-berry is white (grape is brown) and continuous across the nodes (grape is not), the bark has lenticels (grape does not), and the bark does not peel (grape bark peels or shreds). The Ieaves are alternate, broadly ovate with a heart-shaped base, palmately 3-5 lobed or more deeply dissected, and have coarsely toothed margins. The inconspicuous, greenish-white flowers with "free" petals occur in cymes opposite the leaves from June through August (in contrast to grape species that have flowers with petals that touch at tips and occur in panicles. The fruits appear in September-October and are colorful, changing from pale lilac, to green, to a bright blue. Porcelain-berry is often confused with species of grape (Vitis) and may be confused with several native species of Ampelopsis -- Ampelopsis arborea and Ampelopsis cordata.

ECOLOGICAL THREAT

Porcelain-berry is a vigorous invader of open and wooded habitats. It grows and spreads quickly in areas with high to moderate light. As it spreads, it climbs over shrubs and other vegetation, shading out native plants and consuming habitat.

DISTRIBUTION IN THE UNITED STATES

Porcelain-berry is found from New England to North Carolina and west to Michigan (USDA Plants) and is reported to be invasive in twelve states in the Northeast: Connecticut, Delaware, Massachusetts, Maryland, New Jersey, New York, Pennsylvania, Rhode Island, Virginia, Washington D.C., West Virginia, and Wisconsin.

HABITAT IN THE UNITED STATES

Porcelain-berry grows well in most soils, especially forest edges, pond margins, stream banks, thickets, and waste places, where there is full sunlight to partial shade, and where it is not permanently wet. Porcelain-berry appears to be less tolerant of heavily shaded areas, such as that found in mature forest interiors.

BACKGROUND

Porcelain-berry was originally cultivated around the 1870s as a bedding and landscape plant. In spite of its aggressiveness in some areas, it is still used in the horticultural trade (for example, the ornamental A. brevipedunculata 'Elegans' is often recommended as a landscape plant with a cautionary note that "care must be taken to keep it from overtaking and shading out small plants"). The same characteristics that make porcelain-berry a desirable plant for the garden -- its colorful berries, good ground coverage, trellis-climbing vines, pest-resistance, and tolerance of adverse conditions -- are responsible for its presence in the United States as an undesirable invader.

BIOLOGY & SPREAD

Porcelain-berry spreads by seed and through vegetative means. The colorful fruits, each with two to four seeds, attract birds and other small animals that eat the berries and disperse the seeds in their droppings. The seeds of porcelain-berry germinate readily to start new infestations. Porcelain-berry is often found growing in riparian areas downstream from established patches, suggesting they may be dispersed by water also. The taproot of porcelain-berry is large and vigorous. Resprouting will occur in response to cutting of above-ground portions.

MANAGEMENT OPTIONS

Because porcelain-berry vines can grow up to 15 ft. in a single growing season, especially when rainfall is abundant, and seed may be viable in the soil for several years, effective control requires dedicated followup. Treatment measures often must be repeated during the growing season and for several years afterwards to fully eradicate the plant. Prevention of flowering, fruiting and production of mature seeds will help reduce its spread.

Manual

Hand pulling of vines in the fall or spring will prevent flower buds from forming the following season. Where feasible, plants should be pulled up by hand before fruiting to prevent the production and dispersal of seeds. If the plants are pulled while in fruit, the fruits should be bagged and disposed of in a landfill. For vines too large to pull out, cut them near the ground and either treat cut stems with systemic herbicide or repeat cutting of regrowth as needed.

Chemical

Chemical control in combination with manual and mechanical methods is effective and likely to be necessary for large infestations. The systemic herbicides triclopyr (e.g., Garlon® 3A and Garlon® 4) and glyphosate (e.g., Roundup® and Rodeo®) have been used successfully by many practitioners.

Foliar applications The most effective control has been achieved using triclopyr formulations. From summer to fall, apply a water-based solution of 2.5% Garlon® 3A (triclopyr amine) to foliage or cut plants first, allow time for regrowth and then apply the mixture. Smaller infestations can be controlled to some extent with spot applications of glyphosate to leaves, used sparingly to avoid contact of desirable plants with spray. Cut the vines back during the summer and allow to resprout before applying herbicide, or apply glyphosate to leaves in early autumn, just prior to senescence. Basal bark applications

Apply a mixture of 20-30% Garlon® 4 (triclopyr ester) mixed with commercially available basal oil, horticultural oil, diesel fuel, No. 1 or No. 2 fuel oil, or kerosene, to 2 - 3 ft. long sections of stem near the base of the vines.

USE PESTICIDES WISELY: Always read the entire pesticide label carefully, follow all mixing and application instructions and wear all recommended personal protective gear and clothing. Contact your state department of agriculture for any additional pesticide use requirements, restrictions or recommendations.

NOTICE: mention of pesticide products on this page does not constitute endorsement of any material.

CONTACT

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SUGGESTED ALTERNATIVE PLANTS

Many lovely non-invasive vines are available. Some native substitutes to consider include trumpet honeysuckle (Lonicera sempervirens), trumpet creeper (Campsis radicans), American wisteria (Wisteria frutescens)*, Virginia creeper (Parthenocissus quinquefolia), and goldflame honeysuckle (Lonicera heckrottii). In the southeast, several species of native Ampelopsis occur and should be considered if the habitat is appropriate. Please consult the native plant society in your state for more suggestions and information on sources of native plants. *NOTE: If you wish to plant wisteria, make certain that it is the native species. Two commonly planted ornamental wisterias, Chinese wisteria (Wisteria sinensis) and Japanese wisteria (Wisteria floribunda), are exotic and aggressive invaders.

OTHER LINKS

• http://www.invasive.org/search/action.cfm?q=Ampelopsis%20brevipedunculata

• http://www.lib.uconn.edu/webapps/ipane/browsing.cfm?descriptionid=38

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http://www.nps.gov/plants/alien/
USDA, ARS, National Genetic Resources Program. Germplasm Resources Information Network - (GRIN) [Online Database]. National Germplasm Resources Laboratory, Beltsville, Maryland. URL: http://www.arsgrin. gov/var/apache/cgi-bin/npgs/html/taxon.pl?2964 (01 September 2004).
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Mimosa

Albizia julibrissin Durz.
Pea family (Fabaceae)
NATIVE RANGE
Iran to Japan (according to Hortus 3rd. ed.)
DESCRIPTION
Silk tree, also known as mimosa, or silky acacia, is a small to medium-sized tree that can grow up to 20-40 feet tall. The bark is light brown, nearly smooth, and generally thin with lens shaped areas along the stem. The attractive fernlike leaves of mimosa are finely divided, 5-8 inches long by about 3-4 inches wide, and alternate along the stems. Silk tree has showy and fragrant pink flowers, about 1½ inches long, that resemble pom-poms and are arranged in panicles at the ends of branches. Fruits are flat, straw-colored pods about 6 inches long containing light brown oval-shaped seeds about ½ inch in length. Pods ripen in August to September and begin to disintegrate soon after, but remain on the trees into winter.
ECOLOGICAL THREAT
Because silk tree can grow in a variety of soils, produce large seed crops, and resprout when damaged, it is a strong competitor to native trees and shrubs in open areas or forest edges. Dense stands of mimosa severely reduce the sunlight and nutrients available for other plants.
DISTRIBUTION IN THE UNITED STATES
Silk tree is naturalized from New Jersey to Louisiana and in California.
HABITAT IN THE UNITED STATES
Silk tree takes advantage of disturbed areas, often spreading by seed from nearby ornamentals or from contaminated fill dirt. It prefers full sun and is often seen along roadsides and open vacant lots in urban/suburban areas. Silk tree can tolerate partial shade but is seldom found in forests with full canopy cover, or at higher elevations (above 900 m or 3,000 ft), where cold hardiness is a limiting factor. It can, however, become a serious problem along riparian areas, where it becomes established along scoured shores and where its seeds are easily transported in water. Like many successful exotics, it is capable of growing in a wide range of soil condtions.
BACKGROUND
Silk tree was introduced to the U.S. in 1745. Silk tree continues to be a popular ornamental because of its fragrant and showy flowers.
BIOLOGY & SPREAD
Silk tree reproduces both vegetatively and by seed. Silk tree seeds have impermeable seed coats that allow them to remain dormant for many years. One study showed that 90% of the seeds were viable after five years and, for another species of mimosa, a third of its seeds germinated after 50 years in open storage. Seeds are mostly dispersed below or around the parent plant, but can be dispersed further by water. Silk trees grow rapidly under good conditions but are short-lived and have weak, brittle wood. If cut or top-killed, trees resprout quickly and sprouts can grow over three feet in a season.
MANAGEMENT OPTIONS
Silk tree can be controlled using a variety of mechanical and chemical controls.
Mechanical
Trees can be cut at ground level with power or manual saws. Cutting is most effective when trees have begun to flower to prevent seed production. Because mimosa spreads by suckering, resprouts are common after treatment. Cutting is an initial control measure and will require either an herbicidal control or repeated cutting for resprouts.
Girdling is effective on large trees where the use of herbicides is impractical. Using a hatchet, make a cut through the bark encircling the base of the tree, approximately six inches above the ground. Be sure that the cut goes well below the bark. This method will kill the top of the tree but resprouts are common and may require a follow-up treatment with a foliar herbicide. Hand pulling will effectively control young seedlings. Plants should be pulled as soon as they are large enough to grasp, but before they are old enough to flower. Seedlings are best pulled after a rain when the soil is loose. The entire root must be removed since broken

fragments may resprout.

Chemical

Silk tree seedlings and small trees can be controlled by applying a 2% solution of glyphosate (e.g., Round-up®) or triclopyr (e.g., Garlon®) and water plus a 0.5% non-ionic surfactant to thoroughly wet all leaves. Systemic herbicides such as glyphosate and triclopyr can kill entire plants because the chemicals travel through a plant from the leaves and stems to the actively growing roots, where they prevent further cell growth. Use a low pressure and a coarse spray pattern toreduce damage from spray drift on non-target species. Use caution when applying these products, as glyphosate is a nonselective herbicide that may kill non-target plants that are only partially contacted. Triclopyr is a selective herbicide for many broadleaf plant species. Triclopyr is a selective herbicide for many broad-leaved plant species and should be considered for sites where native or other desirable grasses are meant to be conserved.

The cut-stump and basal bark herbicidal methods should be considered when treating individual trees or where the presence of desirable species preclude foliar application. Stump treatments can be used as long as the ground is not frozen. Horizontally cut stems at or near ground level. Immediately apply a 25% solution of glyphosate or triclopyr and water to the cut stump making sure to cover the outer 20% of the stump. Basal bark applications are effective throughout the year as long as the ground is not frozen. Apply a mixture of 25% triclopyr and 75% horticultural oil to the base of the tree trunk to a height of 12-15 inches from the ground. Thorough wetting is necessary for good control; spray until run-off is noticeable at the ground line.

USE PESTICIDES WISELY: Always read the entire pesticide label carefully, follow all mixing and application instructions and wear all recommended personal protective gear and clothing. Contact your state department of agriculture for any additional pesticide use requirements, restrictions or recommendations.

NOTICE: mention of pesticide products on this page does not constitute endorsement of any material.

CONTACT

For more information on the management of silk tree, please contact:

• Kris Johnson, Great Smoky Mountains National Park, Gatlinburg, TN

SUGGESTED ALTERNATIVE PLANTS

Many small to medium-sized trees make excellent alternatives to silk tree. A few examples include serviceberry (Amelanchier arborea), redbud (Cercis canadensis), flowering dogwood (Cornus florida), river birch (Betula nigra), fringe tree (Chionanthus virginicus), American holly (Ilex opaca), and sweetgum (Liquidambar styraciflua). Check with the native plant society in your state for plant recommendations for your particular area.

OTHER LINKS

• http://www.invasive.org/search/action.cfm?q=Albizia%20julibrissin

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Garlic Mustard

ELEMENT STEWARDSHIP ABSTRACT
for Alliaria petiolata
(Alliaria officinalis)
Garlic Mustard

To the User:

Element Stewardship Abstracts (ESAs) are prepared to provide The Nature Conservancy's Stewardship staff and other land managers with current management related information on those species and communities that are most important to protect, or most important to control. The abstracts organize and summarize data from numerous sources including literature and researchers and managers actively working with the species or community.

We hope, by providing this abstract free of charge, to encourage users to contribute their information to the abstract. This sharing of information will benefit all land managers by ensuring the availability of an abstract that contains up to date information on management techniques and knowledgeable contacts. Contributors of information will be acknowledged within the abstract and receive updated editions.

For ease of update and retrievability, the abstracts are stored on computer at the national office of The Nature Conservancy. Each abstract has a Nature Conservancy office or program responsible for its updating and maintenance. The address and telephone number of the office is recorded on the first page of the abstract. Anyone with comments, questions, or information on current or past monitoring, research, or management programs for the species or community described in an abstract should contact the Land Steward in the office responsible for that abstract.

This abstract is a compilation of available information and is not an endorsement of particular practices or products.

Please do not remove this cover statement from the attached abstract.

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SPECIES CODE

PD.BRA01010

SCIENTIFIC NAME

Alliaria petiolata (M. Bieb.) Cavara and Grande

Alliaria petiolata is the correct name for this species. In American floras this species is often referred to as Alliaria petiolata Andrz. Names used in older floras include Alliaria alliaria L. (Britton), Sisymbrium alliaria Scop., Sisymbrium officinalis DC (not S. officinale), and Erysimum alliaria L.

The genus name Alliaria refers to the garlic or Allium like fragrance of the crushed leaves, an unusual odor for the mustard family. The species name petiolata refers to the petiolate leaves.

COMMON NAMES

GARLIC MUSTARD is the most widely and virtually only common name used in North America. Additional common names, used primarily in England and old pharmacopiaeas, include Hedge garlic, Sauce alone, Jack by the Hedge, and Poor man's mustard, and less frequently Jack in the Bush, Garlic Root, Garlicwort, and Mustard root (de Bray 1978, Brooks 1983). Common names reference the European use of this plant as a potherb (Fernald et al. 1958), and the edge habitat in which it frequently occurs.

DESCRIPTION and DIAGNOSTIC CHARACTERISTICS

Alliaria petiolata is an obligate biennial herb of the mustard family (Brassicaceae). Seedlings emerge in spring and form basal rosettes by midsummer. Immature plants overwinter as basal rosettes. In the spring of the second year the rosettes (now adult plants) produce flower stalks, set seed, and subsequently die.

Basal leaves are dark green and kidney shaped with scalloped edges, 6 10 cm diameter. Stem leaves are alternate, sharply toothed, triangular or deltoid, and average 3 8 cm long and wide, gradually reducing in size towards the top of the stem. All leaves have pubescent petioles 1 5+ cm long. New leaves produce a distinct garlic odor when crushed. The fragrance fades as leaves age, and is virtually non existent by fall.

Plants usually produce a single unbranched or few branched flower stalk, although robust plants have been recorded with up to 12 separate flowering stalks. Flowers are produced in spring (usually April to May) in terminal racemes, and occasionally in short axillary racemes. Some plants produce additional axillary racemes in mid summer. Flowers are typical of the mustard family, consisting of four white petals that narrow abruptly at the base, and 6 stamens, two short and four long. Flowers average 6 7mm in diameter, with petals 3 6mm long. Fruits are linear siliques, 2.5 6cm long and 2mm wide, held erect on short (5mm), stout, widely divergent pedicels. Individual plants produce an average of 4-16 siliques. Siliques contain an average of 10-20 seeds, arranged alternately on both sides of a papery sinus. Seeds are black, cylindrical (3mm x 1mm) and transversely ridged, and range in weight from 1.62 2.84mg.

Adult plants range in height from 0.05m to 1.9m, and average 1.0m, at the time of flowering.

Immature plants can be confused with other rosette-forming species, especially violets (Viola sp.), white avens (Geum canadense), and Cardamine sp in midwestern and northeastern states, and fringecup (Tellima grandiflora) and piggy-back plant (Tolmiea menziesii) in western states. Alliaria petiolata can be distinguished from these plants by the strong garlic odor in spring and summer. In fall and winter Alliaria can be distinguished by examining the root system. Alliaria petiolata has a slender, white, taproot, with a distinctive "s" curve at the top of the root, just below the root crown (Nuzzo, personal observation). Axillary buds are produced at the root crown and along the upper part of the "s".

Chromosome number of 2n=36 has been recorded for European material, and 2n=24 for North American and European material (Cavers et al. 1979). An analysis of genetic variation indicated that North American

populations may have originated from multiple introductions from Europe, most likely the British Isles, as well as Belgium and The Netherlands (Meekins 2000).

Excellent illustrations are contained in Cavers et al. (1979). Descriptive characteristics derived from Cavers et al. (1979) and Gleason and Cronquist (1991) except where otherwise noted. There is one other species in this genus (Gleason and Cronquist 1991).

STEWARDSHIP SUMMARY

Alliaria petiolata invades forested communities and edge habitats. The plant has no known natural enemies in North America, is self-fertile, and is difficult to eradicate once established. Thus, the best and most effective control method for Alliaria petiolata is to prevent its initial establishment.

In shaded and partially shaded communities lacking Alliaria petiolata the preferred method is to monitor annually, and remove all Alliaria petiolata plants prior to seed production. Once it is established, the management goal is to prevent seed production until the seed bank is depleted, potentially 2 5 years. Cutting of flowering stems at ground level provides the most effective control with minimal or no side effects, but has a high labor cost. Burning and herbicide application both provide control at a lower labor cost, but each has potential drawbacks: Fire may increase total presence of Alliaria petiolata, and may alter ground layer composition; and herbicides may negatively impact some native ground layer species. The method of choice depends on the size of infestation, the type of community invaded, and the work force available. In all cases, control must be continued annually until the seedbank is exhausted. Frequency and abundance of Alliaria petiolata in unmanaged sites limit effectiveness of single site management, as seeds are continually imported into the managed site. Once Alliaria petiolata is well established, successful control is unlikely without considerable expenditures of labor and money over an extended period of time. Biological control for this species is in development.

IMPACTS (THREATS POSED BY THIS SPECIES)

Alliaria petiolata is one of the few alien herbaceous species that invades and dominates the understory of forested areas in North America. Its phenology is typical of cool season European plants, and Alliaria grows during early spring and late fall when native species are dormant.

Alliaria petiolata dominated sites frequently have low native herbaceous richness and cover, and it has been implicated as the cause of this low diversity (Anderson et al. 1996, McCarthy 1997, White et al. 1993). However, little research has been conducted to document this assumption. In a laboratory experiment Meekins and McCarthy (1999) documented that Alliaria petiolata outcompeted seedlings of chestnut oak (Quercus prinus) but was in turn outcompeted by seedlings of boxelder (Acer negundo) and the annual jewelweed (Impatiens capensis). In a field experiment, McCarthy (1997) found that removing garlic mustard resulted in greater relative cover of annual species, although actual percent cover of annuals was equal or greater in plots with garlic mustard. Nuzzo (unpublished) conducted an 8-year monitoring study in Illinois and found that in areas with garlic mustard, cover of native perennial herbaceous species declined significantly but species richness did not change. In Ohio, McCarthy (1987) found no correlation between species diversity and Alliaria petiolata biomass, and determined that species richness was similar in plots with and without Alliaria petiolata.

Garlic mustard may threaten some butterfly species. Adults of several native butterfly species (Pieris napi oleracea, P.n. marginata, P. virginiensis) lay eggs on garlic mustard, but many or all of the larvae die before completing development (Bowden 1971). Thus, garlic mustard serves as a population sink for these species. This is of particular concern with the rare West Virginia White Butterfly (Pieris virginiensis) which lays eggs on garlic mustard in the absence of the related native host plant, Dentaria [Cardamine] diphylla (Porter 1994).

Alliaria petiolata is most widespread in the midwestern and northeastern United States and in southern Ontario. Canada lists Alliaria petiolata as one of four principal invasive aliens in upland habitats (White et al. 1993). Connecticut, Illinois, Indiana, Missouri, and Wisconsin have developed vegetation management guidelines for Alliaria petiolata (Brunelle 1996, Nuzzo et al. 1990, Smith 1993). Vermont ranks Alliaria petiolata as a category 1 species ("highly invasive")(Vermont DEC 1998), and Minnesota ranks Alliaria petiolata as a "moderate" threat ("species shows invasive behavior, and known to impact native species, or has a wide distribution and statewide abundance") (Minnesota DNR 1991).

Alliaria petiolata appears to alter habitat suitability for native birds, mammals, and amphibians, and may affect populations of these species. No studies have been conducted of the interaction between Alliaria petiolata and these native animals, however. Alliaria petiolata is avoided as a food plant by white tailed deer in Illinois (Nuzzo personal observation), but may be consumed by deer in Ontario (Cavers et al. 1979). White tailed deer may enhance spread of this species by distributing seed short distances, by exposing bare soil through hoof action, and by consuming native herbaceous species, thereby reducing native competition and increasing available habitat for Alliaria petiolata. No deer Alliaria petiolata studies have been conducted to substantiate these observations.

GLOBAL RANGE

Alliaria petiolata is native "throughout Europe from about 68o north southwards, but less common in the extreme south" (Tutin et al. 1964), occurring from England (Martin 1982) east to Czechoslovakia (Lhotska 1975), and from Sweden and Germany south to Italy, but is noticeably absent from Iceland, the Azores, Sardinia and Spitsbergen (Tutin et al. 1964). Alliaria petiolata also occurs in North Africa, India, Sri Lanka (Cavers et al. 1979), and has spread to New Zealand (Bangerter 1985) and North America (Cavers et al. 1979Gleason and Cronquist 1991, Nuzzo 1993a).

The North American range extends from British Columbia (Cavers et al. 1979, White et al. 1993), Washington, and Oregon in the west, and in the east from New England (Gleason and Cronquist 1991), and Ontario (Cavers et al. 1979) to Tennessee (Nuzzo 1993a), Georgia (B.Blossey, p.c.) and westward to Arkansas, Knasas and South Dakota (herbariums specimens). Alliaria petiolata was first recorded in North America in 1868 on Long Island NY, and by 2000 had spread to 34 states and 4 Canadian provinces. This plant has spread exponentially since introduction (Nuzzo 1992b, Nuzzo 1993a).

In the United States Alliaria petiolata is most abundant in the New England and Midwestern states, but also has populations established as far west as North Dakota and Kansas, and south to Georgia and Arkansas. Infrequent collections from mountain states indicate the plant may be a sporadic rather than established component of the regional flora, and/or in the process of becoming established in Utah (1971, 1983, 1984) and eastern Colorado (1952, 1958, 1966) (dates of herbarium collections). In the western states, Alliaria petiolata is established in Portland, Oregon, and several locations in Seattle WA.

In Canada Alliaria petiolata is well established in Victoria and Vancouver, British Columbia (Cavers et al. 1979, White et al. 1993), and in the St. Lawrence Valley from Point Pelee in Ontario to Quebec City in Quebec (Cavers et al. 1979). Alliaria petiolata is especially abundant in southwestern Ontario, and near Toronto and Ottawa (White et al. 1993). White et al. (1993) recorded the plant as common in deciduous woods on the Canadian Shield, although 25 years earlier Cavers et al. (1979) stated the plant was noticeably absent from the region.

HABITAT

In its native Europe, Alliaria petiolata is an edge species, growing in hedges and fencerows (Fitter et al. 1974, Martin 1982) and in open woods (Wilmanns and Bogenrieder 1988). It is disturbance adapted, and

is frequently a component of ruderal communities (Swies and Kucharczyk 1982), including open, highly disturbed forests (Klauck 1986) in Europe.

In North America, Alliaria petiolata is most common in deciduous forest (Cavers et al. 1979, Nuzzo 1992, 1993a, Trimbur 1973, Byers and Quinn 1998), and also occurs in the partial shade characteristic of oak savanna, forest edges, hedgerows, shaded roadsides, and urban areas, and occasionally in full sun (Nuzzo 1991). It grows best in partial or filtered light; Dhillion and Anderson (1999) found that both biomass and photosynthetic rates increase as light levels increase, except that at full light photosynthesis declines. Thus, Alliaria petiolata is less successful in full sun and full shade than in partial shade (Dhillion and Anderson 1999). Garlic mustard is common in low-quality forests (Hawkes and Abrahamson 1994, Schwartz and Heim 1996) and less frequent in isolated woodlots (forest 'islands' surrounded by crop 'oceans') (Brothers and Springarn 1992). It is rarely found under coniferous trees in the Midwest, but has been reported from under seven species of coniferous trees in Ontario (Cavers et al. 1979).

Alliaria petiolata grows on sand, loam, and clay soils, and on both limestone and sandstone substrates, and is rarely found on peat or muck soils. It frequently grows in well fertilized sites (Cavers et al. 1979), and is described as a nitrophile by Passarge (1976) and Wilmanns and Bogenrieder (1988). In Europe, Alliaria petiolata increased in cover with deposition of air borne industrial emissions, which increased soil nitrogen, nitrate, phosphorous and pH (Wilmanns et al. 1986, Wilmanns and Bogenrieder 1988). In the British Isles, Alliaria petiolata is associated with calcareous soils (Clapham et al. 1962). In the U.S., it has been recorded growing on soils with ph 5.0 to 7.2 (Byers and Quinn 1998, Anderson and Kelley 1995), and may be less common on soils of lower pH (Anderson and Kelley 1995, Grime et al. 1988). Under laboratory conditions, Alliaria petiolata biomass increased as pH increased from 4.2 to 6.3 (Anderson and Kelley 1995).

Alliaria petiolata is common in river associated habitat, particularly in the Northeast (Nuzzo 1993a). It may preferentially invade drier forest communities in the Midwest than it does in the northeast (Nuzzo 1993a). This is supported by the higher presence along railroads in the Midwest (Nuzzo 1993a), which are generally indicative of drier habitats. Byers and Quinn (1987) reported that Alliaria petiolata, once considered a plant of floodplains and moist woods in New Jersey, had become common in a wider range of habitats. In the Great Plains garlic mustard is most frequently recorded from moist, usually riverine, habitat and waste ground (Kansas and Oklahoma), while on the eastern edge of the Rocky Mountains it has been recorded along hiking trails (Utah), and on the grounds of a hotel and around a beaver pond (Colorado).

BIOLOGY ECOLOGY

Alliaria petiolata seeds germinate in early spring, beginning in late February or early March, and concluding by mid May in northern states and Canada (Cavers et al. 1979, Roberts and Boddrell 1983). In northern Illinois, germination coincides with emergence of spring beauty (Claytonia virginica) and false mermaid weed (Floerkea proserpinacoides).

Seedling density in heavily infested forests was recorded at 5,080/m2 at the cotyledon stage, and 2,235/m2 at the 2 3 leaf stage, in northern Illinois (Nuzzo unpublished), at 830-1800/m2 in central Illinois (Anderson et al., 1996), and approximated at 20,000/m2 in Ohio (Trimbur 1973). Seedlings undergo high mortality, declining by 30% to more than 50% (Trimbur 1973, Cavers et al. 1979, Byers and Quinn 1998) by late spring.

By June seedlings develop the characteristic rosette of first year plants. First year rosettes are sensitive to summer drought (Byers and Quinn 1998, MacKenzie 1995) and 60-90% die by fall (Byers and Quinn 1998, Anderson et al. 1996). By mid fall rosettes average 4 10 cm diameter (range 1 15 cm) and are dark green to purplish in color. The rosettes continue to grow in winter during snow free periods when temperatures are above freezing (Cavers et al. 1979, Anderson et al. 1996). Natural mortality continues through winter: in northern Illinois rosette density in November averaged 186.4/m2 (range 50 466/m2), and declined signifi-

cantly to an average of 39.9/m2 (range 4 102/m2) by the following spring (Nuzzo 1993b).

Total survival rate from seedling to adult stage varies from 1.4% to 42.3% (Meekins 2000, Anderson et al. 1996, Byers and Quinn 1998, Cavers et al. 1979) and is strongly influenced by weather, with lower survival in years with dry summers (Meekins 2000).

Alliaria petiolata is an obligate biennial: all plants that survive the winter produce flowers in their second year, regardless of size, and subsequently die (Cavers et al. 1979, Byers and Quinn 1998, Bloom et al. 1990, Meekins 2000). Plants only 5cm tall, with 3 4 leaves, have been observed with flowers and seeds. The majority of plants are taller, averaging 0.7 to 1.0 meters when in flower. Plant height is influenced by available light; plants in deep shade averaged 46 cm tall while plants in partial shade averaged 77.5 cm tall (Meekins 2000). Maximum growth occurs at 50% of full light (Meekins 2000). Flower stalks begin to elongate in March or April, growing at the rate of 1.9cm/day (Anderson et al. 1996), and flowers open early April through May. Stem elongation and flowering activity are presumably temperature-related; in England, long-term studies have documented that Alliaria petiolata flowers open 4 to 6.6 days earlier with each 1 degree C increase in temperature (Fitter et al. 1995, Sparks and Yates 1997). Alliaria petiolata flowering period occurs some 6 10 weeks after new seedlings germinate; in established populations in northern locations generations overlap, and two cohorts can be seen from March through June. Most flowers open by 1100 hr (Cruden et al. 1996). Flowers remain open for 2 (3) days, but nectar production and insect visitation occur primarily on the first day of flowering (Cruden et al. 1996). Alliaria petiolata flowers can be self or cross pollinated (Cavers et al. 1979, Babonjo et al. 1990, Anderson et al. 1996). Flowers that are not insect pollinated automatically self-pollinate (Cruden et al. 1996). Pollinators include medium and small sized solitary bees (primarily Andrenidae and Halictidae), medium sized flies (primarily Syrphidae) and rarely a honey bee or bumblebee (Cruden et al. 1996). Midges visit flowers and may effect pollination (Cavers et al. 1979). Seed production is similar for both self-pollinated and cross-pollinated flowers (Anderson et al. 1996). Whether in bred or out bred, Alliaria petiolata plants maintain substantial genetic variation within populations (Byers 1988). Because Alliaria petiolata is self fertile, a single individual is sufficient to populate or repopulate a site.

Plants usually produce 1 2 flowering stems, although a single individual may produce up to 12 separate stems. Damage to the primary flower stem stimulates growth of additional stems (Cavers et al. 1979) from axillary buds at the stem base and along the root crown, although such damage is not a prerequisite for development of multi stem plants (Nuzzo personal observation). Some plants continue to produce flowers through August in small axillary inflorescences. Plants with large rosettes produce flowers earlier and for a longer time period, and consequently produce significantly more seeds, than plants with small rosettes (Byers 1988).

Seeds develop in a linear silique, with siliques forming on the lower part of the inflorescence while flowers are still opening on the upper part. Seeds ripen and disperse between mid June and late September (Cavers et al. 1979). Seed production varies by site and year (Trimbur 1973, Byers and Quinn 1998, Meekins 2000). Alliaria produces an average of 11.5-19.5 seeds/silique (range 3 to 28), and 8.9-18.8 siliques/plant (range 2 to 422)(Trimbur 1973, Nuzzo 1991, Nuzzo 1999, Cruden et al. 1996, Byers and Quinn 1998, Susko and Lovett-Doust 1998, Cavers et al. 1979). Actual production varies significantly within and between communities, with plants in drier communities tending to produce fewer seeds than plants in mesic and wet communities (Byers and Quinn 1998), and isolated plants producing more seeds than plants growing in high densities (Meekins 2000). Plants produce an average of 136-295 seeds (Byers and Quinn 1998, Nuzzo 1999, Trimbur 1973), and up to 2421 seeds under lab conditions (Meekins 2000). Maximum production per plant is estimated at 7,900 seeds on a plant with 12 stems, while minimum production is 14 seeds on a plant with 2 siliques (Nuzzo unpublished). Seed production is density dependent, with each plant producing fewer seeds as density increases, but total seed production increasing with increasing density (Trimbur 1973, Meekins 2000). Seed production within dense patches of Alliaria ranges from 3,607/m2 to 45,018/m2 (Nuzzo unpublished, Anderson et al. 1996, Byers and Quinn 1998, Trimbur 1973) and has been

estimated as high as 107,580/m2 (Cavers et al. 1979). At low density, Alliaria seed production varies from 168-8034/m2 (Meekins 2000).

Seeds are dormant at maturity and require 50 to 105 days of cold stratification to come out of dormancy (Byers 1988, Lhotska 1975, Baskin and Baskin 1992, Meekins and McCarthy 1999). The dormancy period lasts eight months in southern locales (Baskin and Baskin 1992, Byers 1988) and eight to 22 months in northern areas (Cavers et al. 1979, Solis 1998). Alliaria seeds break dormancy more rapidly when exposed to low temperatures that fluctuate around freezing (0.5 to 10 C, as occurs in central states such as Kentucky) than under a constant temperature regime well below freezing (as occurs in northern states and Canada). This is likely a physiological rather than genetic response, as Ontario seed germinated at 20% and 50% in 3 months when moist stratified at 5 and 2 degrees C, respectively (Cavers et al. 1979).

Unlike some forest crucifers that fail to germinate under leaf cover, Alliaria petiolata seeds germinate in both light and dark after dormancy is broken (Bloom et al. 1990, Byers 1988). Light alone will not stimulate germination during cold stratification (Byers 1988). Germination rates of 12 100% have been reported (Baskin and Baskin 1992, Byers 1988, Cavers et al. 1979, Anderson et al. 1996, Roberts and Boddrell 1983), but rates vary greatly within and between populations and habitats (Byers 1988, Cavers et al. 1979). Interestingly, substrate affects germination rate: Baskin and Baskin (1992) reported lower germination on sand substrates than on finer soils, as seeds on sand failed to afterripen (possibly due to water relations at the seed:soil interface). The majority of seeds germinate as soon as dormancy is broken (Roberts and Boddrell 1983, Baskin and Baskin 1992). A small percentage of seed remains viable in the seed bank for up to four years (Roberts and Boddrell 1983, Baskin and Baskin 1992). In a three years study, Meekins (2000) determined that Alliaria petiolata stands produce numerous viable seeds but relatively few seedlings germinate, and speculated that the majority of seeds in the seedbank never germinate.

In 4 sites in Ohio the seedbank averaged 936 seeds/m2 (range 305-1780) in the top 10 cm of soil (Byers and Quinn 1998). Byers (1998) determined that seeds were concentrated in the upper 5cm of soil, and that three of four populations maintained a seed bank after germination. The fourth population, located in a floodplain, lacked a seedbank due to flooding and scouring of the surface, but was expected to gain new seeds during flood deposition.

Alliaria petiolata spreads exclusively by seed (Cavers et al. 1979). Seeds typically fall within a few meters radius of the plant. Wind dispersal is limited, and seeds purportedly do not float well, although seeds readily attach to moist surfaces (Cavers et al. 1979). Anthropogenic distribution appears to be the primary dispersal mechanism (Lhotska 1975, Nuzzo 1992, 1993a). Seeds are transported by natural area visitors on boots and in pant cuffs, pockets and hair, and by roadside mowing, automobiles and trains (Nuzzo 1992). Seeds are widely dispersed in floodwaters. Seeds may be dispersed by rodents or birds; isolated plants are frequently found at the bases of large trees in forest interiors. Seeds may possibly be distributed directly or indirectly by white tailed deer (Odocoileus virginianus).

In southern locales Alliaria petiolata populations are even aged, alternating annually between immature plants and adult plants (Baskin and Baskin 1992), probably due to the 8 month seed dormancy. In northern climates, garlic mustard populations can be even aged in early stages of invasion, and then become multi aged as the seed bank builds up. In many stands Alliaria petiolata cycles between large populations in one year and small populations the next, often with the majority of plants in the rosette stage one year and the flowering stage the next year (Meekins 2000, Nuzzo 1999, Nuzzo et al. 1996). Meekins (2000) speculated this may be due to establishment of two subpopulations from two or more separate colonization events occurring in different years.

Alliaria petiolata is disturbance adapted, and is frequent in sites subjected to continued or repeated disturbance, such as floodplains (Pyle 1995, Byers and Quinn 1998), and in early successional or low quality

communities (Hawkes and Abrahamson 1994, Luken et al. 1997). The greatest increases in presence occur in sites subjected to large scale natural disturbances (Nuzzo 1999). Byers and Quinn (1998) found that Alliaria petiolata resource (biomass) allocation to reproduction was greatest in the most disturbed site. By implication, continued disturbance promotes greater seed production which in turn promotes larger populations. Once garlic mustard enters a forest it becomes a permanent part of the community, increasing in presence each year (Nuzzo 1999). In the absence of disturbance, Alliaria petiolata gradually declines to a low stable level. This strategy of increased presence and low but continuous abundance allows garlic mustard to rapidly expand when disturbance occurs (Nuzzo 1999).

At any given site Alliaria petiolata cover and density fluctuate annually, reflecting the biennial nature of the plant (Byers 1988, Nuzzo 1999). These annual fluctuations are deceptive, as Alliaria petiolata consistently occurs with increasing frequency through time, on average doubling in four years and tripling in eight years (Nuzzo 1999). Garlic mustard is frequently overlooked at low density levels. In many sites it can be present for a number of years before appearing to "explode" in favorable years. Once it reaches this level of abundance control is difficult to achieve. Alliaria petiolata invades densely vegetated quadrats as readily as sparsely vegetated quadrats, and species-rich quadrats more readily than species-poor quadrats (Nuzzo unpublished).

McCarthy (1997) observed that Alliaria petiolata spreads in a demic pattern, i.e.; through establishment of multiple small populations. Within 7 Illinois forests, garlic mustard spread at an average rate of 5.4 m/year, although in any given site rate-of-spread varied substantially, increasing up to 36 m and decreasing as much as 18 m between years (Nuzzo 1999). This "advance-retreat" pattern is typical of garlic mustard, and explains the "sudden" appearance of a dense garlic mustard stands in a forest where few plants were seen the year before. The general pattern of spread is a ragged advancing front, supplemented by establishment of satellite populations 6-30m ahead of the front. After a few years, the front coalesces with the satellite populations to form an extensive area of garlic mustard (Nuzzo 1999).

Because sites dominated by Alliaria petiolata frequently have lower species diversity than similar uninvaded sites, and invaded sites often appear to lose species diversity, the assumption has been made that garlic mustard outcompetes native groundlayer flora (McCarthy 1997, Anderson et al. 1996, White et al. 1993.). Meekins and McCarthy (1999) experimentally demonstrated that seedlings of chestnut oak (Quercus prinus) had reduced growth when grown with Alliaria petiolata, while jewelweed (Impatiens capensis) and boxelder (Acer negundo) had increased growth; i.e.; Alliaria petiolata was competitively superior to Q. prinus, but I. capensis and A. negundo were competitively superior to Alliaria petiolata. McCarthy (1997) found that annual species had higher relative cover in quadrats where garlic mustard was removed than in paired quadrats containing garlic mustard, and concluded that garlic mustard outcompeted annual species (although, actual cover of annual species was similar or greater in plots with garlic mustard).

Alliaria petiolata produces several phytotoxic chemicals that may interfere with native plant species (Vaughn and Berhow 1999). The roots contain sinigrin (and its breakdown product allyl isothiocyanate (AITC)), and glucotopaeolin (and its breakdown product benzyl isothiocyanate (BzITC)) (Vaughn and Berhow 1999).

Alliaria petiolata may also inhibit mycorrhizal activity in native plants (Roberts and Anderson 1998, Vaughn and Berhow 1999). Under field conditions, density of garlic mustard was significantly negatively correlated with soil mycorrhizal potential (Roberts and Anderson 1998). Like other members of the Brassicaceae, Alliaria petiolata is nonmycorrhizal, while some 75% of native groundlayer plants are mycorrhizal (Harley 1969). AITC and BzITC are known to restrict growth of mycorrhizal fungi (Gamliel and Stapleton 1993), and water leachates of Alliaria petiolata prevented germination of a mycorrhizal fungi (Roberts and Anderson 1998).

Alliaria petiolata is rarely if ever browsed by deer or other large herbivores in the U.S., although Cavers

et al. (1979) reported occasional browsing by deer. Garlic mustard is occasionally browsed by slugs and snails, which remove small amounts of leaf tissue (Nuzzo, personal observation), and sometimes attacked by an unidentified flea beetle (B. Blossey, personal observation).

In Europe Alliaria petiolata is fed upon by 69 species of insects, including at least 5 that are monophagous (feed exclusively upon Alliaria) (Szentesi 1991, Hinz and Gerber 1998). In its native range, Alliaria petiolata is a preferred host plant for Pieridae butterflies (Forsberg and Wiklund 1989, Courtney and Duggan 1983, Remorov 1987, Kuijken 1987). In the Netherlands Alliaria petiolata is targeted by the orange tip butterfly Anthocharis cardamines (Pieridae) when the preferred host species Arabis glabra is unavailable (Kuijken 1987). In eastern Europe Alliaria petiolata is utilized by butterflies that feed on commercial crucifers (Remorov 1987) and thus may be a threat to commercial production of cabbage. However, macerates of Alliaria petiolata leaves sprayed on cauliflower deterred oviposition by the garden pebble moth (Jones and Finch 1987). s
At least one native butterfly (spring azure: Celastrina ladon) uses Alliaria petiolata as a nectar source in Pennsylvania (Yahner 1998). Three native butterflies (including Pieris virginiensis and Pieris napa oleracea) sometimes use Alliaria petiolata for ovipositing, but the larvae rarely survive (Bowden 1971, Huang et al. 1995, Courant et al. 1994). Pieris virginiensis, a rare butterfly specific to toothwort (Dentaria [Cardamine] diphylla and D. laciniata), is stimulated to oviposit on Alliaria petiolata because it is chemically similar to the native toothworts; all contain the glucosinolate sinigrin, a strong attractant that the native butterflies use to cue in on the appropriate host plant (Huang et al. 1995). Larvae of these native butterflies feed on Alliaria petiolata leaves, but most die by the 4th instar (Haribal and Renwick 1998). Courant et al. (1994) speculate that some P.n. oleracea may be adapting to Alliaria petiolata, as 14 of 34 larvae raised on this plant developed into adults. In general, garlic mustard serves as a population sink for native butterflies (Bowden 1971, Huang et al. 1995). It is taller than the native Dentaria (the sole plant host of P. virginiensis), has a longer growing season, and is often more abundant than the native Dentaria; thus, native butterflies presumably have difficulty locating the native host when both plant species are present, and confuse Alliaria petiolata for Dentaria based on the chemical similarity (Huang et al. 1995, Porter 1994).

In Illinois a root rot (Fusarium solani) has been observed to kill 80-90% of Alliaria petiolata in laboratory conditions (Chen 1996). In Ontario an unidentified virus (or several viruses) has been observed to kill flowering plants and prevent them from ripening viable seeds (Cavers personal communication 1989). Alliaria petiolata is frequently infected with a strain of turnip mosaic virus (TuMV Al) in both Ontario and Europe, with infected plants developing a mosaic leaf pattern (Stobbs and Van Schagen 1987). The virus does not affect total seed production or seed germination, but does reduce diameter of individual seeds and average silique length (Stobbs and Van Schagen 1987). Although closely related to TuMV Br, a virus that infects crops in the Brassicaceae, the two viruses are mutually exclusive: the Alliaria virus is not transmissible to commercial Brassicaceae species, specifically rutabagas and canola, nor does TuMV Br infect Alliaria petiolata (Stobbs and Van Schagen 1987). In Europe Alliaria petiolata is a host plant for seven fungi (Hinz and Gerber 1998) and a number of viruses, including cucumber mosaic virus (CMV) and turnip mosaic virus (TuMV), that infect commercially propagated crucifers (Polak 1985). Alliaria is host for an isolate of turnip yellow mosaic virus (TYMV A) that induces systemic infection in broccoli, turnip, and other crucifers grown in Europe (Pelikanova et al. 1990). This was the first finding of TYMV A virus in wild growing vegetation in the former Czechoslovakia (Pelikanova 1990).

Garlic mustard was historically eaten as a potherb, particularly in winter and early spring when few greens were unavailable (Georgia 1920). There is no direct evidence that Alliaria petiolata was specifically imported for garden or medicinal use, although Fernald et al. (1958) state that this “old fashioned garden plant...has spread somewhat to roadsides and borders of groves”, and cite earlier authors who describe the use of garlic mustard as a salad plant. Zennie and Ogzewalla (1977) promote eating garlic mustard for it’s high Vitamin A content (8,600 units/100g in young leaves, 19,000 in basal leaves) and Vitamin C content (190mg/100g in young leaves), both substantially higher than levels in commercially grown fruits and vegetables.

Alliaria petiolata is an ingredient in several 'gourmet' recipes (Shufer 1999, Couplan 1999). This has prompted a warning by editors of HORTIDEAS newsletter to "avoid planting this herb" (Hortideas, 1999). At least one U.S. seed company (Canterbury Farms) offers Alliaria petiolata seeds for sale ($1.00/package).

RECOVERY POTENTIAL

Recovery potential of areas cleared of Alliaria petiolata has not been determined. Communities in good natural quality should recover well and without assistance if Alliaria petiolata is removed before a large population develops. Few native species occur in dense garlic mustard infestation, and recovery of heavily infested communities may require replanting. Allelopathic chemicals, if present, may inhibit growth of some native species. Low quality forests, which lack many native species and community structure, will require replanting regardless of the impact of Alliaria petiolata.

Luken et al. (1997) removed Amur honeysuckle (Lonicera maackii) from an early successional forest, and found that Alliaria petiolata rapidly increased in abundance, becoming the dominant plant species in the resulting gaps. They caution that management for one nonindigenous species may result in an increased abundance of another nonindigenous species. McCarthy (1997) experimentally removed Alliaria petiolata and found that species richness did not change over a three year period.

MONITORING REQUIREMENTS

Qualitative:
Monitor once or twice annually for Alliaria petiolata presence in a site: Monitor in late fall or very early spring for immature rosettes, and in early to mid spring for flowering adults. The purpose of this type of monitoring is to locate and remove plants before seeds are produced. Once Alliaria petiolata is established in a natural area, efforts should be made to eradicate, or at least to contain, the plant rather than just monitor its spread. Walk through potential habitat: parking lot edges, trails, riverbanks, floodplains, and streamsides. If time allows monitor all forest perimeters (Alliaria petiolata usually enters along disturbed edges), edge communities, and dry mesic and floodplain forest interiors (Alliaria petiolata occasionally enters at the bases of large trees). Look for basal rosettes in late summer through very early spring; look for flowering plants in mid to late spring. Flowering plants are easier to locate, but must be removed as soon as seen. Basal rosettes are less noticeable, but may be removed at any time prior to flowering.

Quantitative:
Monitor annually for presence/absence, frequency, cover, density, and seed production on a square meter basis, as appropriate for meeting management objectives. Use a quadrat size and shape that mimizes variance in abundance between quadrats within a given management area or treatment plot.

MONITORING PROGRAMS

A monitoring protocol is being developed to assess growth of Alliaria petiolata in the U.S. and Europe, with and without natural enemies (V.Nuzzo and B. Blossey).

MANAGEMENT

Alliaria petiolata should be removed from natural areas before it sets seed, as one plant can populate or repopulate a site. It appears to threaten the structure of forest communities, and should be given high management priority as soon as it is observed in or near a preserve.

The goal of Alliaria petiolata management is to prevent seed production. The primary management objec-

tive in areas lacking this plant is to prevent establishment, by annually monitoring for and removing all Alliaria petiolata plants. The primary management objective in infested sites is to prevent seed production. Cutting flowerstalks is effective in small populations. Fire and herbicide may be useful for larger populations but both have potential side effects. No method provides 100% control.

Growing season mortality reduces Alliaria petiolata seedling populations by 80%-90% between spring and late fall (Anderson et al. 1996, Byers and Quinn 1998); hence, control is most economical when undertaken in late fall or early spring prior to flower production. Late fall is usually the preferred season for control, as native plants are dormant and management can be conducted until snow covers the ground. If weather is unfavorable in fall, control can still be conducted in early spring. Delaying control until spring can be risky, as native herbs may begin growth earlier than anticipated, and weather may limit or prevent management activities.

Management priority should be given to small populations of Alliaria petiolata, and to high quality natural areas. Low quality natural areas, and sites with established large Alliaria petiolata populations, have low management priority.

BIOLOGICAL CONTROL

Biological weed control — the use of a plant's specialized "natural enemies" to control a nonnative plant — may be the only effective way to reduce large infestations Alliaria petiolata in North America. B. Blossey of Cornell University is coordinating a program to investigate the potential for biological control of Alliaria. At least 69 phytophagous insect species and seven fungi are associated with Alliaria petiolata in Europe (Hinz and Gerber 1998). Of these, four monophagous weevils (two shoot-miners, one seed-feeder, and one root feeder; all Ceutorhynchus species) are being investigated as potential biocontrols. Ceutorhynchus alliariae and C. robertii are sibling species that mine shoots and petioles of rosettes and bolting plants of garlic mustard; heavily attacked plants die without producing seeds (Hinz and Gerber 1998). C. constrictus larvae develop in Alliaria petiolata seeds; each larva destroys one-two seeds (Hinz and Gerber 1998). C. scrobicollis larvae feed in the root crown and occasionally the lower parts of shoots; heavily attacked plants die without producing seeds (Hinz and Gerber 1999). Adults of all 4 Ceutorhynchus species feed on Alliaria petiolata leaves and cause some damage, but the majority of damage is caused by the larvae (Hinz and Gerber 1999). A fifth species, the flea beetle Phyllotreta ochripes, preferentially feeds on Alliaria petiolata, but adults have been found on other Cruciferae, including Rorippa, Brassica, and Sinapsis species (Hinz and Gerber 1999). All five insects will be studied for several years in Europe to document their impact on Alliaria petiolata, and to verify that Alliaria petiolata is their exclusive food source. If one or more insects pass all safety and specificity tests required by the USDA, they will be brought to North America for field releases.

An unidentified virus (or several viruses) can kill a flowering plant and prevent it from ripening viable seeds in Ontario (Cavers personal communication 1989), and may provide some control. Alliaria petiolata developed severe mosaic symptoms when infected with CMV and TuMV, but only when grown in ruderal soil; plants in a natural environment had mostly symptomless infection (Polak 1985).

BURNING

Prescribed burning provides inconsistent management results; Alliaria petiolata populations may remain stable, decrease, or increase following one or more fires. In Kentucky, Alliaria petiolata populations had no significant change following three years of repeated dormant season burns (Luken and Shea 2000). In Illinois, Alliaria petiolata populations were maintained at a low level when fires burned completely through the affected area, and were conducted for at least two consecutive years (Nuzzo 1991, Nuzzo et al. 1996, Schwartz and Heim 1996). Fires that left a thin litter layer (1-2 cm) did not kill Alliaria petiolata root crowns, which subsequently produced multiple flower stalks from axillary buds (Nuzzo et al. 1996). Fires that removed the litter layer may enhance survival of seedlings that germinate after burning (Nuzzo et al.

1996). Thus, after a single fire, total Alliaria petiolata cover can increase due to survival of adult plants, and/or to enhanced seedling survival.

Spring and fall fires are equally successful in reducing cover of Alliaria petiolata rosettes (Nuzzo 1991, Nuzzo et al. 1996). Spring fires also reduce seedling presence if conducted during the germination period (Nuzzo 1991), although native herbaceous species are damaged by a growing season burn (Schwartz and Heim 1996).

Use of fire as a management tool should be tailored to the specific community. Removal of the litter layer may facilitate invasion by disturbance adapted species, including Alliaria petiolata, particularly if there is little native ground layer present at the site. Fires should only be conducted when at least two or more consecutive fires can be scheduled; burning only once may increase Alliaria petiolata abundance. Impact of consecutive fires on the community should be considered, including changes in ground layer composition. While fire can maintain Alliaria petiolata at low abundance, it does not eliminate this plant (Nuzzo et al. 1996).

CHEMICAL

Dormant season herbicide application can provide effective control of Alliaria petiolata, but poses a potential threat to native herbaceous and graminoid species. Glyphosate (Roundup, Rodeo, Accord) produces a high degree of control, but results in some native herb loss and significantly reduces graminoid cover. Bentazon (Basagran SG, Basagran T/O) produces nearly equal control, with much lower impact on graminoid species, and little or no impact on native herbs. Glyphosate is suitable for use in forest communities that have few semi evergreen herbaceous or graminoid species. Bentazon appears suitable for use in many forest communities, but should be subjected to further tests before widespread use. The effectiveness of chemical control is dependent on the chemical contacting all plants; under field conditions this rarely or never happens.

Glyphosate (Roundup, Rodeo, Accord)

Glyphosate (Roundup) applied at 1%, 2%, and 3% concentrations to dormant rosettes in late fall or early spring reduced adult cover by >95% (Nuzzo 1991, 1996). Control was slightly greater with higher glyphosate concentrations. Seedlings that germinate after application are not affected by the herbicide because glyphosate is inactivated when it contacts soil. Glyphosate applied after germination will significantly reduce seedling populations (Nuzzo 1991).

Glyphosate is a non-selective herbicide and will kill native species, particularly when applied in spring. At the community level, glyphosate did not affect mean species richness or total mean herbaceous cover, but did significantly reduce cover of both sedges and grasses, at both 0.5% and 1% concentrations (Nuzzo 1996).

2,4 D (sold under many brand names)

2,4 D is not recommended for control of Alliaria petiolata. An ester formulation of 2,4 D applied during the growing season reduced Alliaria petiolata cover by 10% at 0.25 lb/acre, 70% at 0.50 lb /acre, and 45% at 1.0 lb /acre. Dormant season application at very low concentration had no effect on Alliaria petiolata (Nuzzo unpublished). Ortho Weed B Gone (a 2,4 D formulation available at many hardware and garden stores) at 0.4% had no effect on Alliaria petiolata adult plants (Rich Dunbar personal communication 1990).

Mixing 2,4 D with other chemicals provides a more effective kill, based on visual assessment. A 1% solution of Mecamine (2,4 D plus Dicamba) reduced Alliaria petiolata (Bill McClain personal communication 1990). Treatment with Kilmore (2,4 D, MCPP and Dicamba) applied at the rate of 1.1 liter/hectare killed all

flowering Alliaria petiolata (R.H. Brown personal communication in Cavers et al. 1979).

Triclopyr (Garlon 3A, Garlon4, Pathfinder)

A spring application of triclopyr amine (Garlon 3A) mixed at 7oz/5gallon water (just over 1%) killed 92% of Alliaria petiolata rosettes in a limited test (Rich Dunbar personal communication 1990).

Bentazon (Basagran SG and Basagran T/O)

Growing season application of Bentazon (Basagran) at 0.50 1.0 lb /acre reduced Alliaria petiolata rosette cover by 90 95% (Nuzzo 1994). Impacts of dormant season application could not be determined by this study, as severe drought killed rosettes in both control and treatment plots (Nuzzo 1996). Bentazon did not affect species richness or herb cover, and had minimal effect on graminoid cover (Nuzzo 1996). Alliaria petiolata seedlings were not affected by treatment (Nuzzo 1996). Bentazon is a post emergent contact herbicide that kills dicots (broadleaf plants) and some sedges but will not kill most grasses. It is used to control mustards in agricultural fields and is primarily a contact herbicide, meaning it kills or injures portions of the plant that it lands on and little else. It blocks photosynthesis by binding with the D1 protein of the photosystem II complex. This stops CO2 fixation and production of ATP and NADPH2 which are needed for plant maintenance and growth. In most cases, however, the plants are actually killed because the blockage in the photosystem promotes formation of oxidized molecules that cause chain reactions destroying chlorophyll, carotenoids and cellular membranes. Bentazon's biggest drawback appears to be that it is very soluble in water and does not bind to soil particles which suggests a high potential for groundwater contamination. It has a short half life and breaks down rapidly in soil (average half-life 20 days), however, and estimates based on an EPA survey indicate it is found in about 0.1% of rural drinking water wells nationwide although it is widely used. It was not detected in any US community water systems nor was it detected at concentrations above 0.02 mg/L in any well. Bentazon is rapidly broken down in non-susceptible plants and has little effect on germinating seeds. The brands Basagran SG and Basagran T/O are registered for use in non-crops sites, roadways and other rights of way and thus can be legally used in natural areas in at least some states. The labels state that this herbicide should not be applied under sycamores or rhododendrons and that it may kill trees and shrubs if applied to soil beneath their canopies.

Acifluorfen (Blazer)

Acifluorfen is not recommended for control of Alliaria petiolata. Like Bentazon, Aciflourfen is used to control mustards in agricultural fields. Warm season application of Acifluorfen (Blazer) at 0.25, 0.37 and 0.5 lb /acre reduced Alliaria petiolata cover by 30 40% (Nuzzo 1994). Dormant season treatment eliminated Alliaria petiolata rosettes, and prevented seedling germination the next spring (no seedlings were observed 7 months after fall treatment, and 2 months after spring treatment) (Nuzzo 1996). Aciflourfen reduced species richness and graminoid cover, and significantly reduced herbaceous cover in treated plots (Nuzzo 1996). Aciflourfen has a soil residual, and treated plots had low herbaceous cover through the first growing season after treatment (Nuzzo 1996).

Mecoprop (Mecomec, many other brand names)

Alliaria petiolata was heavily damaged by aerial application of 2.4 kg /acre of mecoprop (Birnie 1984).

CUTTING

Cutting flowering Alliaria petiolata plants at ground level results in 99% mortality, and eliminates seed production. Cutting at 10 cm above ground level results in 71% mortality and reduces seed production by 98% (Nuzzo 1991). Cutting is most effective when plants are in full bloom and/or have developed siliques; plants cut earlier in the flowering period may have sufficient resources to produce additional flowerstems

from buds on the root crown (Nuzzo, personal observation).

Cut flower stems may form viable seed. Solis (1998) pulled Alliaria petiolata plants at four flowering stages (flowerbud, flowering, newly formed siliques, well-formed siliques) and piled each flowering stage in a separate fenced plot; the following spring Alliaria petiolata seedlings were abundant in all plots and absent from the empty control plot, indicating that pulled (and by implication, cut) Alliaria petiolata flower stems produce viable seed. Cavers et al. (1979) suggested that vivipary (germination of seeds while still in the silique) does not occur, although all seeds remained viable during the observation period. Until more information is available, cut or pulled stems should be removed from the site whenever feasible.

Cutting with a weed whip provides quick removal of flowering stems, but may remove other desirable species. Some native species, such as Trillium, are severely impacted if cut. Most other species are not substantially damaged, and the benefits of removing Alliaria petiolata should be weighed against the temporary reduction in growth and reproduction of native groundcover species. Cutting with a weed whip may distribute cut stems (and hence seeds) across the site; care should be taken to collect and remove cut material whenever possible.

GRAZING, DREDGING, AND DRAINING

Not tested.

MANIPULATION OF WATER LEVEL AND SALINITY

Not tested. Short-term growing season inundation does not harm Alliaria petiolata rosettes (Byers and Quinn 1998). Extended mid summer flooding resulted in 100% mortality of immature rosettes, but also distributed Alliaria petiolata seeds across the floodplain (Nuzzo 1999).

MOWING, DISCING AND PULLING

Mowing may have an effect comparable to cutting, but the equipment could severely disturb soil and native vegetation. Mowing would be useful in full sun situations such as roadsides, but may distribute seeds. In forested locations a weed whip is more practical and less damaging. Discing is not an acceptable method of control due to the destruction of the community. Root cutting is a very effective although labor intensive means of control.

Pulling is very labor intensive but effective if the upper half of the root is removed. Alliaria petiolata frequently snaps off at or just below the root crown when the flower stalk is pulled, leaving adventitious buds which send up new flower stalks. Pulling can result in substantial soil disturbance, damaging desirable species and bringing up Alliaria petiolata seeds from the seedbank. Soil should be thoroughly tamped after pulling to minimize chances for re establishment of garlic mustard or other weedy species. In general, cutting is a less destructive method of control than pulling but is effective only when the flower stalk is elongating, whereas pulling can be conducted throughout the growing season.

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BITTERSWEET

ELEMENT STEWARDSHIP ABSTRACT
for Celastrus orbiculata
Asiatic Bittersweet

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The Nature Conservancy
Element Stewardship Abstract
For Celastrus orbiculatus Thunb. (C. articulatus)

I. IDENTIFIERS

Common Name: asiatic bittersweet, oriental bittersweet

General Description:
Celastrus orbiculatus is a deciduous woody vine which climbs by means of twining about a support. The branches are round, glabrous, light to darker brown, usually with noticeable lenticels. The outer surface of its roots are characteristically bright orange. Individuals have been found climbing to heights up to 18 m in the Great Smoky mountains (Langdon 1993). Plants with stems 5 cm diameter at breast height are common and some reach 13 cm dbh.
Axillary buds are 1 3 mm long, rounded, with outer scales sometimes becoming spine like. Leaves are glabrous, alternate in arrangement and extremely variable in size and shape, from broadly oblong obovate to suborbicular, 2 12 cm long and 1.5 to 8 cm wide. Leaf margins are crenate serrate and leaf base cuneate to obtuse, tip acute to rounded. Petioles are 1 3 cm long.

Inflorescences are axillary cymes, usually containing 3 7 flowers. However inflorescences are sometimes terminal in male plants. Flowers are small, greenish yellow, and usually become unisexual by abortion or reduction of male or female parts, thus the plants are usually dioecious (Brizicky 1964). Occasional vines develop both unisexual and perfect flowers and are then termed polygamo dioecious (Gleason and Cronquist 1991). Another reported variation is occasional monoecious plants, i.e. with both male and female flowers on the same vine (Hou 1955).

The flowers have 5 sepals and 5 petals. Male flowers contain 5 stamens which are about as long as the petals and inserted at the edge of a cup shaped disk around a vestigial pistil. Female flowers have vestigial stamens, a 3 lobed stigma, columnar style and well a developed superior ovary, sometimes embedded in the disk (Gleason and Cronquist 1991).

The fruit are globose, loculicidal capsules, 6 to 8 mm in diameter, which change in color from green to bright yellow as they mature. The capsules are three valved with each valve (locule) containing one or two brown seeds completely enclosed in a fleshy red aril. Upon ripening, the yellow outer covering splits open to reveal the red aril, thus presenting a brightly bicolored “dispersal flag”.

This species can be reliably distinguished from the native Celastrus scandens only by the location of female flowers and fruit. In C. orbiculatus they are borne in clusters of 3 7 in the axils of leaves. Celastrus orbiculatus fruit are never arranged in terminal clusters. In contrast the flowers and fruit of C. scandens are borne in terminal panicles which may contain numerous flowers or fruits. A second, less reliable, difference is the yellow color of the outer fruit covering in C. orbiculatus vs. the orange color of C. scandens outer fruit cover. The color of the inner aril is red in both species. Identification by leaf shape or size, or by male inflorescence type is not reliable. Illustrations showing the differences between the two species can be found in Gleason (1952) and McNab and Meeker (1987).

The primary taxonomic reference for this section is Hou (1955).

Diagnostic Characteristics:
Celastrus orbiculatus is an invasive, non-native woody vine. It is particularly troublesome in natural areas in coastal Connecticut and New York state and in the southern Appalachians but may be found growing wild from Maine to Louisiana and the southeastern Great Plains. It may severely damage desirable plants by strangling and/or overtopping them and may blanket entire stands. Upland meadows, thickets, young forests, and beaches are most vulnerable to Asian bittersweet invasion and dominance.

The species may be distinguished from its native congener C. scandens by the location of its fruit - C. orbiculatus has small clusters in the leaf axils while C. scandens has clusters at its branch tips. The two

species may be capable of hybridizing and since the native is relatively rare it is possible that its distinct genetic identity is threatened.

Little research has been conducted on C. orbiculatus control but low-growing populations have been successfully treated by cutting and applying triclopyr herbicide to the regrowth about a month later. Larger vines may be cut and the stump treated immediately with triclopyr herbicide. Unfortunately, Asian bittersweet is frequently cultivated and its fruits are gathered for decorative use, which will make preventing further spread and reinfestation all the more difficult. For this reason it is of the utmost importance that land managers, naturalists, botanists, students, horticulturists, gardeners, retailers, etc. learn to distinguish between the native and the introduced bittersweet vine. The seriousness of the problem must be communicated to those in areas were C. orbiculatus either has not yet reached or is not well established.

II. STEWARDSHIP SUMMARY

III. NATURAL HISTORY

Range:
Celastrus orbiculatus is native to temperate east Asia, including central and northern Japan, Korea, and China north of the Yangtze River.

The exact date of Celastrus orbiculatus introduction to eastern North America is obscure, but appears to have been before 1879 (Patterson 1974). Patterson (1974) stated that C. orbiculatus has become "naturalized" in 21 of the 33 states in which it is cultivated. By the early 1970's it was naturalized north to central Maine, through New England, New York, Ohio and west to Iowa, south to Louisiana and Georgia. It was considered weedy in all of New England and most of the Atlantic Coast States by 1971. It is especially troublesome in the southern appalachians and is considered epidemic in the vicinity of Asheville, NC (Langdon 1993). Patterson (1974) also found it cultivated in the three Pacific Coast states, but it has not been reported "naturalized" there. It is reasonable to assume the vine has expanded its North American range in the twenty years since Patterson completed his research.

The western most citation of naturalized populations found was for the "southeastern Great Plains" (Great Plains Flora Assoc. 1986). In southern Illinois, it is sometimes found in woodlands (Mohlenbrock and Voigt 1974), while the authors of "Plants of the Chicago Region" call it "an aggressive weed which gives every indication of being a future problem in the area" (Swink and Wilhelm 1979). C. orbiculatus is also "sometimes found in semi natural situations, as in woods" in southern Ontario (Soper and Heimburger 1985).

Habitat:
Celastrus orbiculatus habitat on its native continent of Asia is said to be lowland slopes or thickets at altitudes from 100 to 1,400 m. The vine is widely distributed in northern and central Japan and Korea. In China it is found primarily in provinces north of the Yangtze River (Hou 1955).

Its North American habitat preferences are wide but seem to be exclusively terrestrial. It is variously described as occupying open woods and thickets (Gleason and Cronquist 1991), roadsides, fence rows, and thickets (Fernald 1970), alluvial woods, roadsides and thickets (Radford et al. 1968).

Reproduction:
Flowers bloom in late May to early June in Connecticut. Fruit ripens in September and remains on the vine through much of the winter. Brizicky (1964) notes that hymenopterous insects, especially bees, are its main pollinators, but Wyman (1950) also found wind pollination to be effective. Wyman also states C. orbiculatus and C. scandens can pollinate each other, and White and Bowden (1947) created a fertile hybrid through a controlled breeding program. No naturally occurring hybrid plants have been reported in the

literature. However, Dreyer et al, (1987) reported two distinct sizes of pollen grains on certain individuals, tentatively identified as C. orbiculatus, growing in close proximity to C. scandens. They speculated that these plants may be hybrids.

Fruit dispersal is generally thought to be by birds and small mammals. In an unpublished undergraduate study in Connecticut, removal of fruit from seven species of woody plants by birds was observed during fall and winter (Wheeler 1987). C. orbiculatus was considered an important winter food, and was comparable in lipid and sugar content to the fruit of other species, but was not taken at all by animals in the fall. Black capped Chickadees, Northern Mockingbirds, European Starlings and Blue Jays all fed on C. orbiculatus during the winter months.

Humans are also important dispersal agents. Fruiting stems are collected for dried flower arrangements, and are soon disposed of on compost and brush piles. The vine is highly attractive, easy to grow and propagate, and available at many nurseries, where it is often mislabeled as C. scandens. It was, and still may be, planted extensively in highway landscaping and for "conservation" plantings for wildlife food and cover, and erosion control, both as itself or mistakenly for C. scandens.

Seed germination is generally high in C. orbiculatus, particularly when compared to C. scandens. Patterson (1974) conducted a wide variety of germination tests with seed from 4 eastern US states and found between 30 and 95% germination. He also noted that the highest germination rates occurred at low light intensities. Dreyer et al. (1987) confirmed the ability to germinate at low light levels and reported germination from 59 to 82%. Also in Connecticut, Clement et al. (1991) found C. orbiculatus produced 4.2 viable seed per fruit compared to 3.2 in C. scandens. Mean germination rates for C. orbiculatus were 70% compared to 20 % for C. scandens.

In field experiments Clement et al (1991) found that C. orbiculatus photosynthetic rates increased with increasing light intensity. In contrast, C. scandens photosynthetic rates at the same sites, tended to reach a plateau beyond which additional light had no significant effect. The ability of C. orbiculatus to acclimate to a variety of irradiance levels may be one of the factors which has allowed it to spread rapidly.

C. orbiculatus rootsuckers prolifically, especially after the main vine is damaged or cut. Rootsuckering is a common occurrence and results in large clones or patches which often spread from one or a few original plants which originated as seedlings.

Patterson (1974) noted the scarcity of other plants under dense canopies of C. orbiculatus, but could not attribute this to soil moisture, soil nutrients, precipitation interception or temperature changes. However, shading by the C. orbiculatus canopy was considered a potentially significant factor.

IV. CONDITION

V. MANAGEMENT/MONITORING

Management Requirements:
Celastrus orbiculatus poses a serious threat to individual plants and plant communities due to its high reproductive rate, long range dispersal, ability to rootsucker, and rapid growth rates. Individual plants can be severely damaged and even killed by the aggressive growth habits of this vine. Tree and shrub stems are weakened and killed by the twining and climbing growth which twists around and eventually constricts solute flow (as shown by Lutz, 1943 for C. scandens). Trees with girdled stems and large amounts of vine biomass in their canopies are more susceptible to damage by wind, snow and ice storms (Siccama, et al. 1976, Langdon 1993). All types of plants, and even entire plant communities, can be over topped and shaded by the vine's rapid vegetative growth. Nearly pure stands of this vine are not uncommon in affected areas. Upland meadows, thickets and young forests, both natural and managed, appear to be most vulner-

able to C. orbiculatus dominance.

Langdon (1993) notes that many of the rarest plants in the southeastern U.S. require a natural disturbance regimen of a certain quality and frequency. Because many of these processes have been altered some of these species are now relegated to roadway and utility corridors which provide exactly the sort of habitat most often invaded and dominated by C. orbiculatus. Langdon (1993) also points out that the region's old growth forests such as cove hardwood stands lose 1-2% of their canopy each year which may provide C. orbiculatus with opportunities to invade. Fortunately, it has not yet been found in virgin forests in the Smokies.

Beaches are also open to invasion. In Connecticut, TNC is managing a C. orbiculatus infestation in sand dunes adjacent to a Piping Plover nesting area on Long Island Sound. The managers are concerned that the vines will either spread into actual nesting areas or alter the dynamics of dune formation and erosion (Lapin 1992). In either case, they could interfere with the reproduction of a bird officially listed as a Threatened Species by the State of Connecticut. Very vigorous patches of C. orbiculatus have also been observed growing in pure sand in coastal Rhode Island (Dreyer, pers. obser.).

It is considered of particular concern to forestry programs in some parts of the southern U.S. (McNab and Meeker 1987). The problems throughout the East are most noticeable along roadsides where vegetation is blanketed by bittersweet in a way reminiscent of Kudzu infested areas of the Southeastern U.S.

Given the fact that hybrids with C. scandens are clearly possible, and that C. scandens appears to be less common than in the recent past (Dreyer et al. 1987, Mehrhoff, 1986) the potential for introgressive hybridization, resulting in the loss of C. scandens genetic identity, is possible. Connecticut has recently listed C. scandens as a Species of Special Concern, i.e. one for which more information on distribution and abundance is needed. In Great Smoky Mountains National Park C. scandens is restricted to circumneutral soils and is considered a non-reproducing rare plant (Langdon 1993).

Recovery of natural areas highly infested with C. orbiculatus is unpredictable. Previous natural vegetation structure and function are often severely altered, although remnants of the flora may persist. Removal methods often further disrupt remnants of previous plant communities. A number of workers report that even with complete removal and rootkill of C. orbiculatus, substantial seedling regeneration occurs in following years, probably due to a persistent soil seed bank. Langdon (1993) stresses that individual clones are difficult to kill. For example, one 5m x 5m clone treated with triclopyr in 1986 has produced 50+ sprouts each year since. The sprouts are hand-pulled but often break and resprout later. Fortunately, the six years of work at this site appear to have nearly exhausted the seed bank. In cases where all nearby seed sources cannot be eliminated, however, reinfestation is a continual possibility.

Due to the ease of and apparent interest in cultivating C. orbiculatus in the eastern half of the U.S. and elsewhere, a very large geographic area is potentially threatened. The spread of the vine from coastal States westward should be monitored. In addition, the status of C. scandens populations, especially in east coast States, should be evaluated.

Langdon (1993) strongly suggests that natural areas be scouted for infestations about 2 weeks after the autumn foliage peak. In the Smokies this falls around November 10. By this time other native deciduous plants drop almost all of their leaves while C. orbiculatus leaves turn lemon- to golden-yellow making the plants easy to identify even at a distance or from a vehicle at moderate speeds. Since individual plants are usually exclusively male or female and the seeds may disperse a kilometer or more these types of searches are essential to locate individuals that have recently become established in previously uninfested areas. If carried out conscientiously, such searches will enable managers to implement control programs that contain infestations to particular areas and prevent them from spreading.

The following individuals have experience monitoring and controlling Celastrus orbiculatus:

Glenn Dreyer
The Connecticut College Arboretum
270 Mohegan Avenue
New London, CT 06320

Lise Hanners
Devil's Den Preserve
P.O. Box 1162
Weston, CT 06883
203 226 4991

Keith Langdon
Great Smokey Mt. National Park
107 Park Headquarters Road
Gatlinburg, Tenn. 37738
615 436 1218

W. Henry McNab
USDA Forest Service
SE Forest Experiment Station
Asheville, NC 28806

David Patterson
USDA Ag. Research Service
Botany Department
Duke University
Durham, NC 27706

Paula Piehl
Potomac State College
West Virginia University
Keyser, West Virginia 26726

Effective biological and additional effective chemical/mechanical control methods are needed. Education of nursery growers, retailers and the gardening public is also needed to reduce the demand for and the dissemination of the vine and its fruit.

HERBICIDES:

A successful control technique was developed by Dreyer (1988) for dense, low patches of C. orbiculatus where herbicide use is appropriate. Vegetation in the entire area is cut to the ground early in the growing season and allowed to resurge. Approximately one month later, foliar applications of an herbicide containing triclopyr (Garlon 4, a triclopyr ester, or Garlon 3A, a triclopyr salt) mixed at 1% to 2% in water and applied by backpack sprayer result in essentially 100% rootkill of C. orbiculatus. No off target damage or root uptake by adjacent plants has been noted in over four years of using this technique. The same study found foliar applications of glyphosate (Roundup, Rodeo) and amitrole (Amitrol, Weedazol) were both ineffective in rootkilling C. orbiculatus.

Another advantage to using triclopyr instead of glyphosate is that it does not kill monocots. Thus grasses, sedges, liliaceous plants, etc., will not be killed and will remain to prevent soils from being completely exposed. These remaining plants often dominate sprayed sites a year after treatment. Triclopyr is also the

active ingredient, in relatively dilute form, in the Ortho product Brush B Gone which, unlike Garlon, is not a restricted use chemical.

Hutchison (1992) reports foliar applications of a 2,4 D and triclopyr mixture (Crossbow) to C. orbiculatus will effectively "reduce the population" when applied in mid to late October.

In locations where large vines climb high into trees, cutting and treating the vine stump surface with a triclopyr-containing herbicide is a logical procedure. The vine stems hanging in the trees will decompose and fall within two to three years. Hutchison (1992) recommends cut surface treatment with "100% Roundup" (presumably undiluted with water) applied at the time of the last killing frost, but he included no data concerning the effectiveness of this technique.

MOWING:
Regular, weekly mowing will exclude C. orbiculatus. However, less frequent mowing, eg. 2-3 mowings per year, stimulates rootsuckering (Dreyer, pers. obs.).

CONTACTS WITH ADDITIONAL INFORMATION ON CONTROL:
In addition to the Connecticut College Arboretum program described above, the following organizations have some type of C. orbiculatus control programs or experience:

Connecticut DOT
Contact: James Stotler
Conn. DOT
24 Wolcott Hill Rd., Drawer A
Wethersfield, CT 06109)

Great Smokey Mountains National Park (Contact is Keith Langdon, see MONITORING PROJECTS section above for address)

Max Hutchison
Cache River Wetlands Project
The Nature Conservancy
Route 1, Box 53E
Ullin, Illinois, 62992
618 634 2524

Southeastern Forest Experiment Station (contact is W. Henry McNab, see MONITORING PROJECTS section above for address)

VI. RESEARCH

Management Research Programs:
Researchers at the Connecticut College Arboretum and the College's Botany and Zoology Departments (Dreyer, Clement, Wheeler, etc.) have pursued various aspects of the problem. No projects are currently underway. Future research will probably continue to examine the comparative species biology of C. scandens vs. C. orbiculatus.

No other active research programs are known.

Management Research Needs:
Research in species biology is needed in the following areas: pollination ecology; extent of natural hybridization with C. scandens; mechanisms of seed dispersal; annual vegetative growth rates; mechanisms of

rootsucker induction; possible allelopathic effects on other species; seed bank dynamics.

Research is also needed to define the current range of C. orbiculatus and to monitor subsequent spread.

Work on biological control methods is apparently completely lacking. Langdon (1993) located an ornamental planting of C. orbiculatus in north Georgia that was losing vigor due to an infestation of Euonymus scale (Unaspi euomyi) and suggests this lead should be followed. The little published on chemical and mechanical control indicates further work in these areas would also be fruitful.

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IX. DOCUMENT PREPARATION & MAINTENANCE

ACKNOWLEDGEMENTS
Keith Langdon of Great Smoky Mountains National Park contributed important information on the biology and control of C. orbiculatus in the southern Appalachians. Beth Lapin of TNC's Connecticut Field Office provided valuable editorial comments.

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Contributing Author(s): Glenn D. Dreyer, edited by: John M. Randall.

Autumn Olive

ELEMENT STEWARDSHIP ABSTRACT
for Elaeagnus umbellata
Autumn Olive
To the User:

Element Stewardship Abstracts (ESAs) are prepared to provide The Nature Conservancy's Stewardship staff and other land managers with current management-related information on those species and communities that are most important to protect, or most important to control. The abstracts organize and summarize data from numerous sources including literature and researchers and managers actively working with the species or community.

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Authors of this Abstract:
N. Sather, Nancy Eckardt

THE NATURE CONSERVANCY
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The Nature Conservancy
Element Stewardship Abstract
For Elaeagnus umbellata

I. IDENTIFIERS

Common Name: AUTUMN OLIVE Global Rank: G?

General Description:
Elaeagnus umbellata is a shrub or small tree with alternate, petioled leaves in small lateral clusters on twigs of the current year.

II. STEWARDSHIP SUMMARY

Elaeagnus umbellata is planted in some states for wildlife cover. It invades disturbed areas adjacent to the plantings where encroachment can be rapid due to the high production of seeds, high germination rate, and hardiness of the plants. It also resprouts quickly after burning or cutting. Repeating cutting or burning may prevent spread, but may need to be conducted for many years, as resprouting will occur. Herbicides offer more effective control, and glyphosate is commonly painted on stumps after cutting in a 10-20% dilution in late August or September. Foliar sprays of glyphosate and dicamba may be effective but will damage other vegetation under the olive. Basal applications of triclopyr alone or in combination with 2,4-D applied in March (dormant season) will also provide effective control.

III. NATURAL HISTORY

Range:
Elaeagnus umbellata is native to China, Korea and Japan and was introduced to the United States for cultivation in 1830 (Rehder 1940). It occurs from Maine to New Jersey and Pennsylvania (Fernald 1950) and west to Wisconsin, Illinois, and Missouri (Holtz 1981).

Habitat:
Elaeagnus umbellata grows well on a variety of soils including sandy, loamy, and somewhat clayey textures with a pH range of 4.8-6.5 (Holtz 1981). It apparently does not grow as well on very wet or dry sites (Allan and Steiner 1965), but Sharp (1977) described it as having excellent tolerance to drought. It does very well on infertile soils because its root nodules house nitrogen-fixing actinomycetes (Sternberg 1982). Mature trees tolerate light shade, but produce more fruits in full sun, and seedlings may be shade intolerant (Holtz 1981, Nestleroad et al. 1984).

Ecology:
Elaeagnus umbellata is one of the earlier shrubs to break dormancy, putting out foliage in mid-March in southern Illinois and advancing north with the season about 100 miles per week (Sternberg 1982). It grows rapidly, producing fruits in 3-5 years. Anthesis occurs after first leaves are out from May to June. Flowers are fragrant and pollinated by a variety of insects (Holtz 1981). The drupes are silvery with brown scales when immature, ripening to a speckled red in September-October. Most fruits are eaten by birds or fall to the ground by early winter (Sternberg 1982). E. umbellata produces a large amount of seed, each tree producing 2-8 lbs. of seed per year and the number of seeds per lb. ranging from 20,000-54,000. The seeds are widely distributed by birds and have a high rate of germination (Holtz 1981). Cold stratification is required to break embryo dormancy (Holtz 1981). The effect of stratification by passing through a bird's digestive tract has apparently not been reported.

Impacts:
Elaeagnus umbellata has the potential of becoming one of the most troublesome adventive shrubs in the central and eastern United States (Sternberg 1982). It exhibits prolific fruiting, rapid growth, is widely disseminated by birds, and can easily adapt to many sites. It is vigorous and competitive against native species, and resprouts after cutting (Nestleroad et al. 1984). Due to its nitrogen-fixing capabilities, it has the capacity to adversely affect the nitrogen cycle of native communities that may depend on infertile soils. E. umbel-

lata is just beginning to be recognized as a potentially serious problem exotic. Seeds are still distributed for wildlife plantings in some states such as Missouri, although the state conservation department is working to stop distribution (Kurz pers. comm.).

IV. CONDITION

V. MANAGEMENT/MONITORING

Management Requirements:
Elaeagnus umbellata seems to be a problem only in locations where small stands or rows were planted, usually within the last 10-20 years, and have begun to spread into adjacent fields or natural areas. It apparently can become troublesome where it occurs on or next to prairies with infrequent prescribed burns because it resprouts quickly after fire damage or cutting.

Since burning and cutting stimulate resprouting, herbicide treatment may be necessary to eradicate large patches. One method of application is to cut the plant off at the main stem and paint the herbicide on the stump. Glyphosate is effective and commonly used. Kurz (pers. comm.) and Nyboer (pers. comm.) recommended a 10-20% dilution for painting on stumps. Foliar applications may be adequate for small patches; the recommended dilution of glyphosate in this case is a 1-2% solution. Kurz (pers. comm.) stated that the best time for herbicide application is in late August or September when the plant is actively translocating materials to the roots.

Kuhns (1986) reported that March dormant season basal applications (stem injections) of triclopyr alone or in combination with 2,4-D provided excellent control of autumn olive at very low concentrations (down to 1% triclopyr in diesel oil). The lowest concentrations of triclopyr and all treatments with the 2,4- D/triclopyr combinations provided slower kills than higher concentrations of triclopyr alone, but only one of the treatment plants were expected to survive (Kuhns 1986). Foliar applications of 2,4-D, triclopyr or metsulfuron methyl in late May or June at recommended rates did not provide adequate control, and even plants that were severely injured recovered the following year. Dicamba applied in late June at 4 lbs/gal (2 qts/100 gal/acre) with a surfactant provided 90% total kill and severely retarded the growth of surviving stems the following year (Kuhns 1986). Glyphosate was not included in this study.

Ohlenbusch and Ritty (1979) reported excellent results for the control of russian olive (E. angustifolia) in Kansas using a variety of herbicides and treatments. Applications were made on June 14 and results evaluated in late August. Foliar applications of dicamba, picloram, and glyphosate, all in a 90% water/10% diesel oil carrier, resulted in total root kill. However, glyphosate in both 1% and 2% solutions damaged herbaceous plants under the trees so extensively that foliar application of this chemical is not recommended.

Basal application of triclopyr, all mixed in diesel oil and applied June 14, also resulted in 100% control. A second study by the same authors indicated that diesel oil alone also provides highly effective basal control of E. angustifolia (Ohlenbusch and Ritty 1979).

Management Programs:
Elaeagnus umbellata is not a problem on many preserve lands. It occurs on some state managed natural ares in Illinois and Missouri where management has implemented control programs consisting of herbicide application. Contact: Don Kurz, Natural History Section, Missouri Conservation Dept., P.O. 180, Jefferson City, MO 65102 and Randy Nyboer, Illinois Dept. of Conservation, 2612 Locust St., Sterling, IL 61081.

Monitoring Requirements:
Occurrences, especially those adjacent to or on natural areas, should be monitored to help substantiate the need for classification of autumn olive as a noxious weed where necessary. Planted stands near natural areas should be monitored to check their expansion.

VI. RESEARCH

Management Research Needs:
Elaeagnus umbellata is not a priority species for research. There are some indications that its abundance may be increasing, both by continued planting and by seed dispersal from naturalized populations (see Nestleroad et al. 1984 and Sternberg 1982) but little data is available on population dynamics within its range. Questions for consideration include: how well does E. umbellata compete with and displace native vegetation? What is the affect on growth and reproduction of repeated burning over several years?

VII. ADDITIONAL TOPICS

VIII. INFORMATION SOURCES

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IX. DOCUMENT PREPARATION & MAINTENANCE

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Contributing Author(s): N. Sather, Nancy Eckardt
Herbicide information partly updated: TunyaLee Martin, 8/2001

Winter Creeper

Euonymus fortunei (Turcs.) Hand.-Mazz.
Staff-tree family (Celastraceae)
NATIVE RANGE
China
DESCRIPTION
Climbing euonymus, also known as wintercreeper, Emerald'n Gold, and Gaiety, is an evergreen, clinging vine. It can form a dense groundcover or shrub to 3 feet in height, or climb 40-70 foot high vertical surfaces with the aid of aerial roots. Dark green, shiny, egg-shaped leaves, from 1-2 ½ inches long, with toothed margins and silvery veins, occur in pairs along the stems. Stems are narrow, minutely warty, and have abundant rootlets or trailing roots. Clusters of inconspicuous green-white flowers are produced on a long stalk from June to July and are followed in the autumn by pinkish to red capsules that split open to expose seeds adorned with a fleshy orange seed coat, or aril.
ECOLOGICAL THREAT
Traits that make climbing euonymus a desirable ornamental plant, such as its rapid growth, evergreen nature and tolerance of harsh conditions, also make euonymus a threat to natural areas. Climbing euonymus can outcompete native vegetation by depleting soil moisture and nutrients, blocking sunlight, and by forming a dense vegetative mat that impedes the growth of seedlings of native species. Vines on trees continue climbing and can eventually overtop them, covering the leaves and preventing photosynthesis.
DISTRIBUTION IN THE UNITED STATES
Climbing euonymus is currently scattered throughout the eastern U.S. in populated areas.
HABITAT IN THE UNITED STATES
Climbing euonymus tolerates a variety of environmental conditions, including poor soils, full sun to dense shade, and a wide pH range. It does not do well in heavy wet soils. Natural forest openings resulting from wind throw, insect defoliation or fire are vulnerable to invasion and provide conditions for satellite populations of climbing euonymus to get started.
BACKGROUND
Climbing euonymus was introduced into the U.S. in 1907 as an ornamental ground cover.
BIOLOGY & SPREAD
Climbing euonymus spreads vegetatively with the help of lateral shoots produced along its long main branches and by new plants that emerge from rootlets also produced along the stem at short intervals. Vines climb rocks, trees, and other supporting structures. Flowers formed in the summer produce mature fruits by fall that are equipped with fleshy edible structures (arils) that are fed on by birds and other wildlife which disperse it. Climbing euonymus also escapes from neglected gardens and is carried by by water, to undisturbed forest and riparian areas.
MANAGEMENT OPTIONS
A variety of mechanical and chemical methods are available for management of climbing euonymus. Grubbing, a rather labor intensive method, is effective for small populations or environmentally sensitive areas where herbicides cannot be used. Using a pulaksi or similar digging tool, remove the entire plant, inlcluding all roots and runners. Juvenile plants can be hand-pulled when the soil is moist and root systems are small. Any portions of the root system remaining may resprout. All plant parts including stem fragments and mature fruits should be bagged and disposed of in a trash dumpster to prevent reestablishment.
Chemical
Cut stem treatment, using systemic herbicides applied to freshly cut stems, is effective in areas where vines are well established on or around non-target plants, or where they have grown into tree canopies or other vertical surfaces. Cut the stem as close to the ground as possible and immediately apply a 25% solution of glyphosate (e.g., Roundup®) or triclopyr (e.g., Garlon®) and water to the cut stem. This procedure is effective at temperatures as low as 40° F. Subsequent foliar application of these herbicides may be required. Cutting without the application of herbicides is generally not recommended because this will lead to root sprouting.
Foliar application

Foliar applications of herbicide can be used to control large populations. It may be necessary to precede foliar sprays with cut stem treatments to reduce the risk of damage to non-target plants. Apply a 2% solution of glyphosate or triclopyr and water plus a 0.5% non-ionic surfactant to thoroughly wet all foliage but not so heavily that it drips off leaves where it may affect desirable plants. Glyphosate is a non-selective systemic (i.e., travels through the plant vessels) herbicide that may kill even partially sprayed plants. Triclopyr is selective to broad leaf species and is a better choice if desirable native grasses are present. Ambient air temperature should be above 65° F.

USE PESTICIDES WISELY: Always read the entire pesticide label carefully, follow all mixing and application instructions and wear all recommended personal protective gear and clothing. Contact your state department of agriculture for any additional pesticide use requirements, restrictions or recommendations.

NOTICE: mention of pesticide products on this page does not constitute endorsement of any material.

CONTACTS

For more information on the management of climbing euonymus, please contact:

• Kris Johnson, Great Smoky Mountains National Park, Gatlinburg, TN

SUGGESTED ALTERNATIVE PLANTS

There are a variety of native creeping or climbing vines that make good alternatives for climbing euonymus. Some examples from the eastern U.S. include trumpet creeper (Campsis radicans), Dutchman's pipe (Aristolochia macrophylla), crossvine (Bignonia capreolata), trumpet honeysuckle (Lonicera sempervirens), American bittersweet (Celastrus scandens), and American wisteria (Wisteria frutescens), our only native wisteria*.

*NOTE: When purchasing or planting wisteria, make certain it is the native American wisteria (Wisteria frutescens) and not exotic Chinese wisteria (Wisteria sinensis) or Japanese wisteria (Wisteria floribunda), both of which are aggressive exotic invaders of natural areas and are difficult to control.

OTHER LINKS

• http://www.invasive.org/search/action.cfm?q=Euonymus%20fortunei

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Material obtained from:

Plant Conservation Alliance®s Alien Plant Working Group

English Ivy

Weed Notes: Hedera helix L.

TunyaLee Morisawa

Wildland Weeds Management and Research
http://tncweeds.ucdavis.edu
30 June 1999

Introduction:

Hedera helix L. is commonly called English ivy. H. helix belongs to the family Araliaceae (ginseng) and is a native of Europe. Brought to North America by colonial settlers, H. helix has become naturalized in the US. English ivy is cultivated in Europe and North America in gardens, landscapes and as house plants. This plant grows easily in many types of soil and in sun or shade. English ivy is fairly drought tolerant once it is established. Leaves are alternate and simple with the juvenile leaves 3-5 lobed and adult leaves ovate to rhombic. Mature plants will bear greenish-white flowers. The fruit is berry-like and black.

Poultices for cuts and sores are made from the leaves. However, an allergic reaction can occur in sensitive people. Secondary compounds within the leaves may be natural product pesticides for insects and mollusks.

English ivy outcompetes both grasses, herbs and trees, often reducing animal feeding habitats. In warm areas, H. helix can grow throughout the year and probably outcompetes native vegetation that is dormant during the winter. In Australia, English ivy is found in disturbed areas of the forest. Seeds are often spread into these areas by birds or other animals. H. helix is a pest in Europe but only in disturbed habitats.

Cultural Control:

Cutting is successful with persistence but does not always kill the plant. However, the use of cutting and then applying a herbicide may provide better control (see Chemical control section).

Using a shovel to remove plants provided immediate control with little regrowth. Weeding plants by hand or with pliers successfully allowed regeneration of most native species in Australia. Do not leave the pulled plants on the ground; they can continue to grow. If removal of the plants is not possible, place the pulled plants on a wooden platform to dry and decompose.

Immediately control English ivy that is growing up trees by cutting the vine at waist height, loosening the vine around the limbs and removing the roots. If the root can not be removed by hand, strip the bark and notch the exposed section of the vine. Paint on an undiluted herbicide such as glyphosate. If English ivy is growing on tree-ferns, take care that all pieces of the ivy are removed. The growth of H. helix can be sustained by the fibrous nature of the trunk.

Chemical Control:

A wax layer on the leaves often prevents herbicides, especially hydrophilic compounds such as glyphosate, from permeating the leaves.

In container pots, two applications, one month apart, of 2,4-D (Weedar 64) applied at 1.1 kg/ha (1.0 lb/A) provided control of English ivy. Two applications of glyphosate (Roundup) applied at 4.5 kg/ha (4.0 lb/A) effectively inhibited regrowth and provided some control. Regrowth but reduced shoot weight was

observed with one treatment of 2,4-D and glyphosate at the rates stated above. The same observation was noted for one or two applications of glyphosate applied at a lower rate of 2.2 kg/ha (2.0 lb/A). Regrowth occurred with plants sprayed with one or two applications of Dicamba (Banvel) or triclopyr (Garlon) at the rate of 0.6 kg/ha (0.5 lb/A).

In another study, an application of glyphosate (25% solution) provided good control. Cutting (using a nylon cord weedeater to cut to the stem surface just before treatment) followed by a 25% solution of glyphosate also provided control of English ivy. Excellent control of H. helix that had been cut and then sprayed was achieved with a 2% solution of 2,4-D. A lower rate of glyphosate (2% solution) and cutting provided only slight control. Glyphosate only (2% solution) did not control English ivy. The herbicide triclopyr or mowing provided no control. Control evaluations were made 1 year post-treatment.

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Honeysuckle Vine

ELEMENT STEWARDSHIP ABSTRACT
for Lonicera japonica
Japanese Honeysuckle
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The Nature Conservancy
Element Stewardship Abstract
For Lonicera japonica

SCIENTIFIC NAME (GNAME)
Lonicera japonica Thunb.

The common name "Japanese honeysuckle" refers to the species Lonicera japonica Thunb. including the more aggressive cultivar Lonicera japonica var. halliana, also known as Hall's honeysuckle and the less common Lonicera japonica var. chinensis (P.W. Wats.) Baker. The original Latin name of the species was Nintooa japonica (Gleason and Cronquist 1963), but the species has been referred to as Lonicera japonica since at least 1889 (Wood and Willis 1889).

The genus name Lonicera refers to German naturalist Adam Lonitzer (1528-1586), the species epithet japonica to Japan, and the variety name halliana to Dr. George Hall, who introduced the variety to the United States in 1862 (Coombes 1991).

COMMON NAME

JAPANESE HONEYSUCKLE is the name most commonly used to refer to Lonicera japonica and its varieties, L. japonica var. halliana (Hall's Japanese honeysuckle) and L. japonica var chinensis. Hall's Japanese Honeysuckle is more common and aggressive than the species. In old floras Lonicera japonica was occasionally referred to as "woodbine" (Lounsbury 1899) and "Chinese honeysuckle" (Wood and Willis 1889; probably L. japonica var. chinensis).

DESCRIPTION (DIAGNOSTIC CHARACTERISTICS)

Lonicera japonica is a perennial trailing or climbing woody vine of the honeysuckle family (Caprifoliaceae) that spreads by seeds, underground rhizomes, and aboveground runners (USDA 1971). It has opposite leaves that are ovate, entire (young leaves often lobed), 4-8 cm long, with a short petiole, and variable pubescence. In the southern part of the range the leaves are evergreen, while in more northern locales the leaves are semi-evergreen and fall off in midwinter (Fernald 1970). Young stems are reddish brown to light brown, usually pubescent, and about 3 mm in diameter. Older stems are glabrous, hollow, with brownish bark that peels in long strips. The woody stems are usually 2-3 m long, (less often to 10 m). Lonicera japonica creates dense tangled thickets by a combination of stem branching, nodal rooting, and vegetative spread from rhizomes.

Lonicera japonica (including the varieties) is easily distinguished from native honeysuckle vines by its upper leaves and by its berries. The uppermost pairs of leaves of Lonicera japonica are distinctly separate, while those of native honeysuckle vines are connate, or fused to form a single leaf through which the stem grows. Lonicera japonica has black berries, in contrast to the red to orange berries of native honeysuckle vines. The fruits are produced September through November. Each contains 2-3 ovate to oblong seeds that are 2-3 mm long, dark-brown to black, ridged on one side and flat to concave on the other.

The fragrant white (fading to yellow) flowers of Lonicera japonica are borne in pairs on solitary, axillary peduncles 5-10 mm long, supported by leaflike bracts. The species has white flowers tinged with pink and purple. Individual flowers are tubular, with a fused two-lipped corolla 3-4(-5) cm long, pubescent on the outside. Flowers are produced late April through July, and sometimes through October. Lonicera japonica var. halliana may be distinguished from the species by its pure white flowers (fading to yellow; Dirr 1983) and more vigorous growth. Lonicera japonica var. chinensis has purple, essentially glabrous leaves, red flowers, and a more limited range than the species, occurring north to New Jersey and Pennsylvania (Fernald 1970), with an outlier in southern Illinois (Mohlenbrock 1986).

This description was derived from Gleason and Cronquist (1991) and Fernald (1970). Excellent illustrations of Lonicera japonica are contained in USDA (1971).

STEWARDSHIP SUMMARY

Lonicera japonica invades fields, forest edges and openings, disturbed woods, and floodplains, in eastern North America, where it spreads rapidly and outcompetes native vegetation by vigorous above- and below-ground competition. Once established, the vine may literally engulf small trees and shrubs, which collapse under the weight, and few plants survive beneath the dense canopy. It has also escaped cultivation at scattered locations in California and in Hawaii where it has the potential to become a severe pest in mesic and wet forest areas.

Lonicera japonica has few natural enemies in North America and is difficult to control once established. Thus, the best and most effective control method is to prevent its establishment by surveying a site for its presence regularly and immediately destroying every plant located. Unfortunately Lonicera japonica is difficult to locate when small and without careful attention may go unnoticed until it is well established.

Because Japanese honeysuckle is so difficult to control once established, an appropriate control program goal is 100% kill of all plants in the target area. Removing above-ground stems by cutting pulling or burning will temporarily weaken, but not kill, Lonicera japonica as it will resprout from subterranean buds and roots, and from cut branchlets.

In northern states, Lonicera japonica retains some leaves through all or most of the winter (semi-evergreen or evergreen), when most native plants have dropped their leaves. This provides a windows of opportunity from mid-autumn through early spring when it is easier to spot and treat with herbicides, fire or other methods without damaging native species. The most effective treatment is a foliar application of glyphosate herbicide (trade names Roundup, Rodeo or Accord; 1.5 v/v), applied after native vegetation is dormant and when temperatures are near and preferably above freezing. Applications within 2 days of the first killing frost are more effective than applications later in the winter. Lonicera japonica is less susceptible to herbicides after the first hard frost (-4oC). Combining fire and herbicides may prove to be more effective than either method by itself if late autumn or winter burns are used to reduce Japanese honeysuckle biomass and all resprouts are then treated with a foliar application of glyphosate about a month after they emerge. Prescribed burns may also be used to help prevent spread of Japanese honeysuckle because seedlings and young plants are most susceptible to fires. Soil disturbance should be avoided in infested areas to minimize germination of seed in the seedbank.

IMPACTS (THREATS POSED BY THIS SPECIES)

Lonicera japonica damages natural communities it invades by outcompeting native vegetation for both light (shoot competition [Thomas 1980, Bruner 1967]) and below-ground resources (root competition [Dillenburg et al. 1993a, 1993b, Whigham 1984]), and by changing forest structure (Sasek and Strain 1990, 1991). Lonicera japonica grows very rapidly, sending out numerous runners that give rise to still more runners. The vines overtop adjacent vegetation by twining about, and completely covering, small trees and shrubs. Dense Japanese honeysuckle growth can topple trees and shrubs due to its weight alone (Williams 1994, McLemore 1981). As Lonicera japonica becomes established in forest openings it forms a dense blanket that excludes most shrubs and herbs (Oosting 1956). Few tree seedlings can penetrate the mat and those that do are often quickly overgrown and bent down by the vine, and consequently die (Slezak 1976, Thomas 1980). Forests invaded by Lonicera japonica gradually lose their natural structure as canopy openings are invaded, and understory herbs shrubs and replacement trees suppressed and killed by thick mats of honeysuckle. This results in a simplified, increasingly open understory. Lonicera japonica, in turn, becomes even more vigorous with the increased light (Thomas 1980). These openings also promote further invasion by other non-native species including aggressive vines like kudzu (Pueraria lobata) and English ivy (Hedera helix) (Miller 1985; Thomas 1980).

Shading under the extensive and rapid aerial growth of Lonicera japonica poses the most obvious threat to

native species. However, Dillenburg et al. (1993a, 1993b) demonstrated that in the early stages of invasion, below-ground competition by Lonicera japonica reduced tree growth, particularly leaf size and expansion rate, significantly and more than above-ground competition. After two growing seasons, Lonicera japonica root competition significantly reduced growth of young sweetgum trees (Liquidambar styraciflua) and greatly exceeded root competition from the native vine Parthenocissus quinquefolia (Dillenburg et al. 1993b). The combined effects of above- and below-ground competition can suppress growth or result in direct mortality of trees and seedlings (Whigham 1984). Bruner (1967) documented that after five years of co-occurrence, 33% of yellow-poplar seedlings were dead, 22% were overwhelmed, and 45% were heavily draped with Lonicera japonica that germinated from seed in the first year.

Lonicera japonica has an additional competitive edge as it grows during part or all of the winter, when many native species are dormant (Carter and Teramura 1988a). This evergreen or semi-evergreen character allows Lonicera japonica to photosynthesize at winter temperatures and light levels. The shade it casts during early spring may inhibit ephemeral herbs that complete their life cycle in the six weeks prior to deciduous tree leaf-out.

Alteration of forest understory and overstory structure by Lonicera japonica may lead to a decline or alteration in songbird populations (Nyboer 1990). However, no studies have been conducted on interactions between Lonicera japonica and native animals, with the exception of white-tailed deer (Odocoileus virginianus) which favors Lonicera japonica leaves as food (Handley 1945, Harlow and Hooper 1971). In fact, wildlife managers in some states actively promoted growth of this aggressive vine to provide winter forage for deer (Dyess et al. 1994; Segelquist and Rogers 1975, Stransky 1984). Japanese honeysuckle foliage is most digestible and nutritious in spring, but it is still relatively high in nutritional value in winter (Blair et al 1983) when other food sources are less available to deer (Dyess et al. 1994). Seeds and leaves are eaten by cottontail rabbits, as well as birds (Dyess et al. 1994), and the tangled thickets provide cover for birds and small mammals.

Lonicera japonica is a severe threat in the southeastern and eastern states (Florida to Texas, north to Kansas, Missouri, central Illinois and New York), and a severe potential threat in northern states outside the current (1995) range. On the northern edge of the range, Lonicera japonica flower production is inhibited by winter temperatures (Swink and Wilhelm 1994), and the vine is thus a moderate threat. For example, in Illinois, Lonicera japonica is not a serious pest in the colder, northern third of the state, but is increasingly common in the central part of the state (Nyboer 1990). Lonicera japonica continues to spread gradually northward (Wagner 1986), possibly due to increasing cold tolerance, or to warm winters, or to other factors.

As of 1995 Lonicera japonica northern range was limited by winter temperatures, and its western range by drought-induced stress at the seedling stage (Sasek and Strain 1990). If atmospheric CO2 concentrations increase as predicted, resulting in a 3oC increase in average and minimum winter temperatures, the northern range of Lonicera japonica is predicted to shift up to 400 km north (Sasek and Strain 1990). Further westward expansion may be limited by decreased summer precipitation, although Lonicera japonica has improved water use efficiency and increased drought tolerance at higher CO2 levels (Sasek and Strain 1990). Lonicera japonica is also predicted to become a more serious competitor of native trees at higher CO2 levels, as it experiences much greater growth rates at higher CO2 levels than do native woody erect species (Sasek and Strain 1991).

Virginia and Illinois have produced honeysuckle control circulars (Williams 1994, Nyboer 1990). Minnesota ranks the species as a severe potential threat (MN DNR 1991).

GLOBAL RANGE

Lonicera japonica is native to east Asia, including Japan and Korea (Gleason and Cronquist 1991, Lee et al. 1990). From this native range it has spread to Hong Kong (Thrower 1976), England (Clapham et al.

1962), Wales (Martin 1982), Portugal (De Baceler et al. 1987), Corsica (Jeanmonod and Burdet 1992), Hawaii (Wagner et al. 1989), Brazil, (Bove 1993), Argentina (Bonaventura et al. 1991), possibly the Ukraine (Panova 1986), and the continental United States, primarily by way of horticultural introductions.

The species was introduced into the U.S. in 1806 on Long Island, NY (Leatherman 1955), and the similar but more aggressive variety halliana was introduced to the country in 1862 in Flushing, N.Y. As with many invasive species, Japanese honeysuckle initially had a very gradual rate of spread, primarily to the south and east. Lonicera japonica was not included in Chapman's Flora of the Southern States (1884; in Hardt 1986) but in 1889 Wood and Willis included the variety chinensis in their flora of the eastern United States and a decade later Britton and Brown (1898) reported that the species ranged from New York and Pennsylvania to North Carolina and West Virginia. In 1899 Lonicera japonica was described in a wildflower book as the most widely planted of the honeysuckles (Lounsbury 1899). Lonicera japonica was reported from Florida in 1903, and from Texas in 1918 (Hardt 1986). By 1912, it had "escaped from cultivation", and ranged from Connecticut to Florida (Atkinson 1912), and within a few years was identified as an invasive problem species from the Gulf of Mexico to Massachusetts, creating "a network of tangled cords that covers the ground wherever this ruthless invader gets a foot hold" (Andrews 1919).

Lonicera japonica now occurs throughout the eastern half of the United States, south of a line extending from Massachusetts west to Lake Michigan, Illinois, and Missouri, and then southwest through Texas to Mexico, an area encompassing 26 states (USDA 1971, Leatherman 1955). The northern range limit coincides with maximum 30-year winter temperatures of -25oC (Sasek and Strain 1990). The area of greatest infestation is in the center of this range, where annual precipitation averages 100-120 cm, and 30 year low temperatures are -8oC to -15oC (Sasek and Strain 1990). Lonicera japonica's range is limited to the north by severe winter temperatures, and to the west by insufficient precipitation and prolonged droughts which limit seedling establishment (Sasek and Strain 1990). At the northern edge of the range, plants have reduced growth due to a shorter growing season, and produce few or no flowers (Swink and Wilhelm 1994). Lonicera japonica continues to spread northward, however, possibly due to increasing cold tolerance or warmer winters (Wagner 1986). It may spread up to 400 km north if global temperature increases 3oC (Sasek and Strain 1990).

Japanese honeysuckle sporadically escapes from cultivation in California where it is present in scattered locations, primarily below 1000 m elevation (Hickman 1993). It has also escaped cultivation in scattered locations in the Hawaiian islands, particularly in mesic to wet forest in Kokee State Park on Kauai and near Volcano on the island of Hawaii (Wagner et al. 1990). It apparently does not produce seed at most locations in Hawaii and will likely become a much more serious pest there if fertile strains develop. Unfortunately, most plants in an escaped population in Manoa Valley on Oahu reportedly set seed (Wagner et al. 1990). A recent report from Kauai also indicates the Japanese honeysuckle population there may be spreading and has potential to become a severe pest in the Kokee area (Flynn, personal communication).

HABITAT

Lonicera japonica is native to east Asia. In Korea, Lonicera japonica is part of the understory in later successional forests dominated Carpinus cordata, Fraxinus rhynchophylla and Cornus controversa (Lee et al. 1990).

In North America, Lonicera japonica primarily occurs in disturbed habitat, including successional fields, roadsides, forest edges, and fencerows (Williams 1994). It is common in dry-mesic to wet-mesic upland forest, floodplain forest, and southern pine stands, and particularly common in forest openings created by disturbance, such as treefall, logging, or disease. Lonicera japonica continues to be planted for landscape purposes in gardens and along highways.

Lonicera japonica grows most vigorously in full sun and on rich soil, but is shade and drought tolerant and

therefore able to grow in a wide variety of habitats (Leatherman 1955). It develops high frequency and cover in young forests while densely shaded, mature forests support fewer, and smaller, colonies (Robertson et al. 1994). Lonicera japonica usually invades disturbed communities and rarely colonizes deeply shaded, mature forests unless canopy openings are created by human disturbances or natural processes (disease, wind throw, drought, etc.) (Slezak 1976; Thomas 1980). In Virginia Lonicera japonica quickly invaded a former forest site destroyed by avalanche (Hull and Scott 1982), and it grew vigorously in a forest opening in Arkansas (McLemore 1981). This species can persist in low numbers in relatively undisturbed forest and then "break out" following disturbances that open the canopy, e.g.; windthrow, ice storm, disease, scouring flood, or drought. Once established, Lonicera japonica's dense canopy inhibits establishment of later successional species (Myster and Pickett 1992). Lonicera japonica rarely invades deeply shaded, mature forests unless the canopy is somehow opened (Robertson et al. 1994).

In Pennsylvania, Lonicera japonica is a major component of the third stage of succession in old fields, increasing after fields have been abandoned for four years (Keever 1989). In New Jersey Lonicera japonica invaded an oldfield 13 years after abandonment, and was present for at least 18 years (Myster and Pickett 1992). In Virginia Lonicera japonica, is most abundant in the piedmont and coastal plant forests (Williams 1994). In Illinois Lonicera japonica grows where overstory canopy provides filtered light, especially oak forests, cedar glades, and barrens, and along the banks of streams where the natural break in canopy creates a light opening (Nyboer 1990). Plants then spread into adjacent shaded forest. Lonicera japonica has been found on Michigan sand dunes (Wagner 1986), and persists near abandoned homesites in the Chicago region (Swink and Wilhelm 1994). In Indiana, Lonicera japonica is abundant in urban forest preserves, but is absent from woodlots isolated by agricultural fields and distant from urban areas (Brothers and Springarn 1992).

BIOLOGY-ECOLOGY

Lonicera japonica is a strong competitor due to wide seed dispersal, rapid growth rate, extended growing season, ability to capture resources both above- and below-ground, wide habitat adaptability, and lack of natural enemies. Some of these factors have received considerable study, while others have been given little or no attention.

Lonicera japonica blooms most prolifically in full sun (Leatherman 1955), and decreases flowering activity as light decreases; in 8% of full light no flowers are produced (Blair 1982, Robertson et al. 1994). The blooming period extends from April to December in Georgia (Andrews 1919), late May to October in Kentucky (Sather, personal communication), May to June in Illinois (Mohlenbrock 1986), and June in Michigan. Flowers open a few hours before sunset, and remain open for approximately three days (Roberts 1979). In Wales, the majority of flowers are pollinated the day after opening by bumblebees (Bombus lucorum and B. pascuorum). Other bee species may be potential pollinators, as nectar is available to species with tongues > 4 mm long (Roberts 1979). Flowers remain open at night, indicating the possibility for moth pollination (Roberts 1979). In the United States Lonicera japonica is probably pollinated by a variety of insects, due to its extended blooming season and wide geographical range.

Relatively few studies have documented seed production, seed viability, germination requirements, or seedling establishment.

The inconspicuous black berries contain 2-3 seeds (USDA 1971). Fruit production is much higher in full sun than in shade (average 222 vs. 11 g seeds per plant, respectively) in Texas (Halls 1977). Fruit production decreases as soil nitrogen increases (Segelquist and Rogers 1975). Seed viability is highly variable. Leatherman (1955) determined that 85% of seed were viable, and obtained 63% germination. Haywood (1994) attempted to study long-term seed viability, but seed was unsound when collected. This variation is typical of the Lonicera genus, which is characterized by having variable seedcoat dormancy, embryo dormancy, and/or no dormancy both within and among species (Hartmann and Kester 1968). Bruner (1967)

reported rapid growth from seed in South Carolina, and Carter and Teramura (1988b) stated that Lonicera japonica reproduces abundantly from seed. Berries are consumed by a number of birds including robin, turkey, quail, bluebird, and goldfinch (Martin et al. 1951, Jackson and Cooper 1974), which then disseminate the seeds (Nyboer 1990).

Rate of growth from the seedling stage is not known; most researchers and nurseries propagate Lonicera japonica from stem cuttings, particularly the var. halliana, which forms roots "wherever the canes touch moist ground" (Hartmann and Kester 1968). Leatherman (1955) suggested that seedlings likely photosynthesize shortly after germination, due to the low food reserves in each seed. Seedlings are known to establish in shaded understories, which implies that light may not be necessary for seed germination. Seedling growth is apparently slow for the first two years (Little and Somes 1967). Lonicera japonica is drought sensitive, particularly at the seedling stage (Sasek and Strain 1990). Biomass appears to decline with summer drought (Faulkner et al. 1989).

Once established, Lonicera japonica is capable of extremely vigorous growth. In a moist bottomland forest vines overtopped a 4.5 m tree in one year (Bruner 1967), although growth rates of 1.5 m/year may be more typical (Leatherman 1955). Bell et al. (1988) recorded a maximum shoot elongation of 4.6 mm/day in Maryland. This rapid growth rate allows Lonicera japonica to outcompete native trees; In one year, Lonicera japonica overtopped three-year old sweetgum (Liquidambar styraciflua) trees (Dillenburg et al. 1993a). Lonicera japonica vines spread both vertically and horizontally (Williams 1994).

Individual vines have numerous long vegetative runners; the combined length of lateral and sublateral runners from one sprout in one year exceeded 15 m (Little 1961). Vines in high light have been recorded with > 7 runners, each over 60 cm long (Slezak 1976). The runners develop roots at nodes in contact with soil, and thus form dense mats. If the above ground parts are severed, each new root system develops into a separate, but genetically identical, plant. The root system has been recorded at up to 3 m across and 1 m deep (Leatherman 1955). Roots are highly competitive with native species (Carter and Teramura 1988a, 1988b).

Lonicera japonica's climbing architecture is adapted to early successional forest (Carter and Teramura 1988a), which typically has small diameter trees and a dense understory. The vines twine about vegetation in closely spaced spirals, thus creating a strong support structure that permits them to remain upright after the host tree is killed. Individual shoots may be very long, but due to the numerous spirals, a vine's height above the ground may not be great. Japanese honeysuckle vines typically climb stems <15 cm diameter (Andrews 1919). Larger stems are rarely used as hosts, as Lonicera japonica cannot climb wide boles unless small branches or other vines are present to provide support (Andrews 1919).

Longevity of individual plants has not been measured. As Lonicera japonica reproduces vegetatively, life span of individual stems or roots is not a measure of genet longevity.

Lonicera japonica is adapted to growing in 25-100% of full light, and grows vigorously in full sun. Stem density is greatest in full light, and decreases with increasing shade: In Pennsylvania, Robertson et al. (1994) recorded mean stem densities of 25.4/m2 in an oldfield, 15/m2 in a thicket, 13.6/m2 in a woodland, and 8.6/m2 and 8.1/m2 in riparian forest and upland mature forest, respectively. Stem density was similarly high in both oak and maple associations (Robertson et al. 1994). In Washington D.C. Lonicera japonica produced good growth at 47% of full sun (Thomas 1980). In this location winter light measurements in closed forest range from 49% to 86% of full light. Lonicera japonica is able to persist in deciduous forest at low summer light intensities, and put on growth in winter, or when canopy gaps occur.

Lonicera japonica tolerates low light conditions, and may spread vegetatively, but rarely produces flowers or fruits under low light (25% of full light; Robertson et al. 1994). Honeysuckle plants are severely stressed in low light, and lose substantial amounts of aboveground biomass after long periods of low light: Blair

(1982) reported that leaf biomass declined 94% after two years at very low light (8% of full sunlight), and plants suffered stem dieback and leaf loss, but did not die. Leatherman (1955) similarly reported that half of her experimental cuttings survived at 10% of full light, and the majority survived at 25% of full light. Once established, Lonicera japonica can persist at low light levels with little or even negative growth, and respond to winter sun and canopy openings with more vigorous growth (Carter and Teramura 1988a). Interestingly, as a twining vine Lonicera japonica is less physiologically adapted to low light levels than native tendril climbing vines, such as Parthenocissus quinquefolia (Carter and Teramura 1988a), which can rapidly climb up supporting trees to reach higher light levels.

Lonicera japonica has a long photosynthetic season due to its evergreen nature and its ability to grow in cold temperatures. Lonicera japonica shoots grow until the first frost, apparently because they are able to lignify rapidly, which gives them greater cold-hardiness than more tender species (Panova 1986). In southern locales Lonicera japonica retains its old leaves over winter (Schierenbeck and Marshall 1993) permitting year-round photosynthesis. In these areas, Lonicera japonica leaves are physiologically active during the winter and can grow when minimum predawn air temperatures are at or above -3oC. At these temperatures, net photosynthetic rates on warm winter days are comparable to those in summer (Carter and Teramura 1988b). The presence of old leaves during the period of new-leaf formation (January - March), combined with the higher photosynthetic rates in new leaves, increases total carbon gain and thereby growth rate and invasiveness (Schierenbeck and Marshall 1993). Shoots produce an early burst of growth in spring, before native species leaf out (Dillenburg et al. 1993a).

In the northern states Lonicera japonica retains its leaves until late December or January (semi-evergreen), while native trees lose their leaves in October. The vines continue to photosynthesize for several months after overstory trees have dropped their leaves, which allows them to maintain presence in low light communities (Robertson et al. 1994, Carter and Teramura 1988a). In Maryland, Lonicera japonica is physiologically active for 9 weeks after native deciduous vines have gone dormant (Parthenocissus quinquefolia and Vitis vulpina) (Bell et al. 1988). In spring Lonicera japonica begins growth some two months earlier than native species, from the period when temperatures are above freezing, until deciduous trees produce new leaves (Hardt 1986). Thomas (1980) calculated that in the Washington D.C. area there are an average of 52 days/year between first and last frost when temperature and light conditions in closed canopy forests are adequate for Lonicera japonica photosynthesis.

Lonicera japonica leaves are unaffected by minimum temperatures of -0.6oC, and continue to function, at lower efficiency, until temperatures drop below -3.0oC (Carter and Teramura 1988b). The relatively high rate of leaf gas exchange in autumn, winter, and spring indicates that carbon gain during this period may contribute substantially to Lonicera japonica's rapid growth rate. Although Lonicera japonica leaves photosynthesize in winter, the lowered activity reduces effectiveness of foliar herbicides applied after the first frost (Regehr and Frey 1988). In Tennessee, defoliation occurred at -26oC, but plants were not apparently killed (Faulkner et al. 1989).

Lonicera japonica is spread primarily by birds, which consume the fruits and pass the seeds, carrying them from landscape plantings to natural areas and disseminating them in forest openings and disturbance zones. Once established, Lonicera japonica can develop a large seedbank that germinates when the soil is disturbed. This attribute led to a dramatic increase in southern states in the 1950's, when timber companies promoted intensive site preparations (discing, burning, bush-hogging) to facilitate tree regeneration after clearcutting (Prine and Starr 1971). Honeysuckle grew so rapidly from both seedbank and top-killed plants that tree seedlings were outcompeted (Prine and Starr 1971). Consequently, forest companies have conducted much of the research to identify herbicides that control Lonicera japonica (Edwards and Gonzalez 1986, McLemore 1981).

Originally introduced as a landscape plant, Lonicera japonica is still considered a desirable species by some landscapers, highway designers, and wildlife managers. Wildlife managers promote increased growth of

Lonicera japonica to provide winter forage, particularly for deer (Dyess et al. 1994). Landscape architects plant Lonicera japonica for its fragrant flowers and rapid growth (Georges et al. 1993, Nam and Kwack 1992, Bradshaw 1991), and highway designers use the plant for erosion control and bank stabilization (Stadtherr 1982, Hardt 1986).

In China Lonicera japonica is a valued medicinal herb that contains anti-complementary polysaccharides (Shin et al. 1992). Polyphenolic compounds isolated from Lonicera japonica inhibit human platelet activation and provide protection from cellular injury, and thus help maintain human vascular homeostasis (Chang and Hsu 1992). Aden I, a mixture of Lonicera japonica flower buds and parts of other plants, has both antibiotic and antiviral effects, comparable to results produced by standard antibiotics (Houghton et al. 1993). Leaves and flowers are used in the therapy of chicken pox (Luo 1989), and may be used as a food additive to increase productivity of broiler chickens in Korea (Cho 1992).

CONTROL

Prevention/Legislation

In Illinois, the sale and distribution of Lonicera japonica is prohibited under the Illinois Exotic Weed Act (1988).

Biological control

The only technique that could control Lonicera japonica on a regional scale is biological control, but as of 1997 no formal program had been established. Interestingly, in China, a biocontrol program using Scleroderdermus spp. was established to protect Lonicera japonica from the cerambycid Xylotrechus grayi (Tian et al. 1986). Lonicera japonica is utilized by some insects in its native habitat and the U.S. In Sichuan, China, Lonicera japonica growing near cottonfields is an early spring host for aphids that feed on crops later in the growing season (Li and Wen 1988). In North Carolina, the two-spotted spider mite (Tetranychus urticae), an agricultural pest in corn and peanut fields, overwinters on Lonicera japonica growing on field margins (Margolies and Kennedy 1985). Lonicera japonica is also a suitable host for the cicadellid cotton pest (Empoasca biguttula) in Hunan, China (Chen et al. 1987), and may be a host for tobacco leaf curl virus, which was detected in the horticultural variety Lonicera japonica var. aureo-reticulata (Macintosh et al. 1992). The vine is susceptible to honeysuckle latent virus (Brunt et al. 1980), and to tobacco leaf curl bigeminivirus (TLCV) transmitted by whiteflies (MacIntosh et al. 1992).

Burning

Fire removes above-ground vegetation, and reduces new growth, but does not kill most Lonicera japonica roots, and surviving roots produce new sprouts that return to pre-burn levels of cover within a few years (Oosting and Livingstone 1964). A single spring fire reduced Japanese honeysuckle cover 50% in Illinois (Nyboer 1990). Two sequential fires topkilled Lonicera japonica, reducing crown volume (m3/ha) by 80%, but new growth from root sprouts maintained Lonicera japonica as a dominant groundcover species in North Carolina (Barden and Matthews 1980). In Virginia burning is used to reduce abundance of Lonicera japonica, and inhibit spread for 1-2 growing seasons (Williams 1994). Prescribed burning significantly reduced Lonicera japonica biomass in Tennessee, by 93% when burned in October, and by 59% when burned January - March (Faulkner et al. 1989). Top-killed honeysuckle resprouted in spring (March - April), apparently from roots or runners just below the unburned litter layer. In this situation, follow-up application of 2% glyphosate in spring, 2 - 6 months after burning, appeared to control honeysuckle better on unburned than burned plots, possibly because tall herbaceous vegetation that grew up after the fire on the burned plots intercepted the herbicide before it could reach the shorter honeysuckle resprouts (Faulkner et al. 1989). In Texas, burning in February removed all above ground foliage, but did not kill plants (Stransky 1984). However, burned plants produced fewer and shorter runners than unburned plants, and fire therefore reduced total vegetative growth (Stransky 1984).

Combining fire and herbicides may prove to be more effective than either method by itself if late autumn

or winter burns are used to reduce Japanese honeysuckle biomass when most native species are dormant and all resprouts are then treated with a foliar application of glyphosate about a month after they emerge (Johnson, personal communication). Prescribed burns may also be used to help prevent spread of Japanese honeysuckle because seedlings and young plants are most susceptible to fires (Richter, personal communication) .

Chemical

The evergreen and semi-evergreen nature of Lonicera japonica allows application of herbicides when many native species are dormant. Timing of application is critical to effectiveness; in general, applying herbicide shortly after the first killing frost, and before the first hard frost (ca. -4.0oC) is most effective. Herbicide effectiveness can be reduced in areas where large stones or fallen logs protect root crowns from soil-active herbicides (Miller 1985) or where overtopping vegetation intercepts foliar herbicides (Faulkner et al. 1989). Many herbicides produce a short-term reduction in foliar coverage, but do not kill the plant and buds left undamaged by the herbicide can produce new growth that often exceeds growth from untreated plants within a year (Prine and Starr 1971). A foliar application of 1.5% glyphosate shortly after the first frost appears to be the most effective treatment. Treated plants should be re-examined at the end of the second growing season, as plants can recover from herbicide application (McLemore 1981).

GLYPHOSATE (brand names include: Roundup, Rodeo, Accord)

- October applications of 0.75% and 1.5% v/v glyphosate killed 99% of treated Lonicera japonica within six months in Delaware, and few plants resprouted within 30 months of treatment (Regehr and Frey 1988). The two application rates were equally effective. The same experiment conducted in December resulted in 68% mortality at the lower concentration, and 86% mortality at the higher concentration, and regrowth from buds was much greater than in plants treated in October. The authors concluded that timing of application was critical; applying glyphosate within 2 days of the first frost resulted in very high mortality. After the first frost, higher concentrations of glyphosate were needed to achieve somewhat lower mortality. Defoliation after glyphosate treatment was very slow; only 5-15% of leaves were gone one month after treatment, although 78-90% of stems were dead.
- A mid-August application of 2.2 kg/ha glyphosate controlled 83% of actively growing Lonicera japonica in North Carolina; control was reduced under drought conditions (Younce and Skroch 1989). Glyphosate (2 lb active ingredient/gal) at 1 to 1.5 gallons/acre controlled “most” Lonicera japonica in Alabama (Miller 1985).
- In Arkansas, a 6.72 kg active ingredient/ha application resulted in 85% control after one growing season, and 80% control after two growing seasons (McLemore 1981). Lower application rates were less effective two years after treatment.
- Effectiveness of glyphosate increased linearly with increasing herbicide concentration (0.48-4.8% w/w), but no concentration gave complete control with one application; repeated treatment with 4.8% glyphosate produced complete shoot necrosis in only 50% of plants (Ahrens and Pill 1985).
- Efficacy of glyphosate was not increased by addition of surfactants (Younce and Skroch 1989, Regehr and Frey 1988).

DICHLORPROP + 2,4-D

- Dichlorprop mixed with 2,4-D at 3.6 grams active ingredient/liter (1.5% v/v) resulted in 94% mortality when applied within two days of the first frost in October, but only 46% mortality when applied in December. Thirty months after treatment, 14% of stems sprayed in October resprouted, and 75% of stems sprayed in December produced new growth (Regehr and Frey 1988).

2,4-D + PICLORAM (brand names include: Tordon)

- Picloram is a restricted use soil-active herbicide that is prohibited in California, as it is relatively persistent and subject to leaching.
- Tordon 101 (4:1 2,4-D amine + picloram, at 1 to 2 gal/acre) “reduced existing honeysuckle to a few surviving crowns” (Miller 1985). Tordon 10K at 50 lb/acre had similar effectiveness (Miller 1985).

• Tordon 101 at 10 gal acre reduced foliage by 72.5% one year after treatment; a second application of Tordon 101 reduced foliage by a total of 90% one year after re treatment (Prine and Starr 1971)
• A foliar spray of Tordon 101 at 2.8-8.4 kg/ha gave 84-94% control in a pine stand (McLemore 1982), similar to control provided by amitrole at 2.24 and 4.48 kg/ha. (McLemore 1982).

TEBUTHIURON (brand names include: Spike)
• Spike 80W (80% tebuthiuron) and Spike 20p (20% tebuthiuron) provided very effective control when applied at 4-5 lbs active ingredient/acre, "resulting in essentially bare plots with yellowing sprigs of vegetation" (Miller 1985).

DICAMBA (brand names include: Banvel, Brushkiller)
• Banvel 720 (2 lb 2,4-D and 1 lb dicamba) was very effective when applied at 4 gal/acre, but had only partial effectiveness at 3 gallons/acre (Miller 1985).
• Lower rates of Dicamba, as in Brushkiller 4-41 and 10-51, resulted in limited or no mortality (Miller 1985). In fact, Lonicera japonica growth was stimulated by application of Brushkiller 10-51 (Miller 1985).

SULFOMETURON (brand names include: Oust)
• A February application of sulfometuron methyl in South Carolina at .25 lb/acre active ingredient, applied when vegetation is less than 30-45 cm high, is recommended for control of Lonicera japonica in loblolly pine stands (Michael 1985).
• In Georgia, Lonicera japonica was not controlled by a late application of Sulfometuron applied at 3 oz/acre (Withrow et al. 1983)
• Lonicera japonica was almost completely killed (99% mortality) by a May application of 2 oz metsulfuron-methyl + 0.25% surfactant in central Georgia (Edwards and Gonzalez 1986)

INEFFECTIVE
• In Illinois, herbicides that are not used by the Department of Conservation due to ineffectiveness or environmental persistence are: picloram; amitrole; aminotriazolel atrazine; dicamba; dicamba + 2,4-D; 2,4-D; DPX 5648; fenac; fenuron; simazine; and triclopyr (brand names for triclopyr include Garlon 3A, Garlon 4 and Brush-B-Gone) (Nyboer 1990).
• Hexazinone at 2.24 and 6.72 kg Active ingredient/ha was ineffective (McLemore 1981), as was application at 1 or 2 lb active ingredient/acre (Michael 1985). Hexazinone pellets at 8 lb active ingredient/acre reduced Lonicera japonica cover from 100% to 25% cover after three years, while a 2 lb/acre rate resulted in a decrease in cover from 100% to 52% over the same time period (Michael 1984).
• Oryzalin is apparently ineffective, as it is recommended for use in controlling weeds that threaten Lonicera japonica planted as a groundcover (Bowman 1983)
• Brushkiller 10-51 at 1.5 gal/acre "encouraged" growth of Lonicera japonica (Miller 1985). Brushkiller 170 resulted in a 45% decrease in foliar cover one year after June treatment (Prine and Starr 1971).
• June application of 2,4-D (4 lb active ingredient/acre at 10 gal/acre) increased foliar growth of Lonicera japonica by 48% one year after treatment (control plants increased by 0.9%) (Prine and Starr 1971).
• June application of Banvel resulted in increased foliar growth one year after treatment (Prine and Starr 1971).
• Triclopyr in both ester and salt formulations (3 and 4lb/gal, respectively) and as an ester combined with 2,4-D (1 and 2lb/gal respectively) failed to control Lonicera japonica one year after treatment (Dreyer 1988). However, in Illinois the latter formulation is reputedly effective (Nyboer 1990).

Mowing, Discing and Pulling
Removing the above-ground portion of Lonicera japonica reduces current-year growth but does not kill the plant, and generally stimulates dense regrowth. Cut material can take root and should therefore be removed from the site (not practical with most infestations).

Mowing is an ineffective control method, stimulating growth and encouraging formation of dense, albeit shorter, mats. Plants mowed in February formed a dense, 20 cm tall mat within two months, growing from cut stems and rooting from severed runners; by the following November (21 months later) mowed plants were 60 cm high (Stransky 1984). Twice-yearly mowing in Virginia slowed vegetative spread but increased stem density (Williams 1994).

Bush-hogging is an ineffective control, as Lonicera japonica re-invades within one growing season (McLemore 1985).

Discing is apparently an effective control method: McLemore (1985) reported that "control of the honeysuckle was still effective after two years". Discing depth was not indicated. Discing is a highly destructive procedure that destroys native groundlayer species, and may stimulate Lonicera japonica seed bank germination.

Hand-pulling is a time-consuming procedure with limited effectiveness, as the entire plant (roots and shoots) must be removed. Pulling may be a practical method to remove small patches of seedlings.

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Honeysuckle Shrub

ELEMENT STEWARDSHIP ABSTRACT
for
Lonicera maackii (Rupr.) Maxim (Amur honeysuckle)
Lonicera morrowii A. Gray (Morrow's honeysuckle)
Lonicera tatarica L. (Tatarian honeysuckle)
Lonicera x bella Zabel (Bell's honeysuckle)
The Bush honeysuckles
To the User:

Element Stewardship Abstracts (ESAs) are prepared to provide The Nature Conservancy's Stewardship staff and other land managers with current management related information on species and communities that are most important to protect or control. The abstracts organize and summarize data from many sources including literature and from researchers and managers actively working with the species or community.

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SCIENTIFIC AND COMMON NAMES

Lonicera maackii (Rupr.) Maxim	Amur honeysuckle
Lonicera morrowii A. Gray	Morrow's honeysuckle
Lonicera tatarica L.	Tatarian honeysuckle
Lonicera x bella Zabel	Bell's honeysuckle

DESCRIPTION AND DIAGNOSTIC CHARACTERISTICS

Exotic bush honeysuckles are upright, multi-stemmed, oppositely branched, deciduous shrubs that range in height from 2 m to 6 m. The opposite leaves are simple and entire, and paired, axillary flowers are showy with white, pink, or yellow corollas. The fruits of Lonicera spp. are red, or rarely yellow, fleshy berries (Gleason and Cronquist 1991).

In flower, exotic bush honeysuckles can be distinguished from all native bush honeysuckles except swamp fly-honeysuckle (L. oblongifolia) by their hirsute (hairy) styles. In fruit, the red or rarely yellow berries of the exotics separate them from the blue- or black-berried natives waterberry (L. caerulea) and bearberry honeysuckle (L. involucrata) (Gleason and Cronquist 1991). The exotic bush honeysuckles also generally leaf-out earlier and retain their leaves longer than the native shrub honeysuckles (Trisel and Gorchov 1994).

Within the exotic bush honeysuckles, L. maackii alone has acuminate, lightly pubescent leaves (Luken and Thieret 1995) that range in size from 3.5 to 8.5 cm long (Gleason and Cronquist 1991) and peduncles generally shorter than 6 mm (Pringle 1973). Its flowers are white to pink, fading to yellow, 15-20 mm long. Its berries are red or with an orange cast. Height ranges to 6 m (Luken and Thieret 1995).

In North America, there has been considerable confusion regarding the correct identification of L. morrowii, L. tatarica, and L. x bella, their hybrid. The literature contains a number of references to plants called by the name of one of the parents, but described as having characters more like those of the hybrid. L. x bella. The hybrid therefore, may be more common than the literature would indicate (Barnes 1974, Wyman 1977), and accurate field identification may be similarly problematic.

The two parent species of L. x bella, however, are dissimilar. L. morrowii has leaves that are elliptic to oblong gray-green, soft-pubescent beneath, and are 3-6 cm long. Its flowers are pubescent, white fading to yellow, 1.5-2 cm long, on densely hairy peduncles 5-15 mm long. The fruits are red. The height ranges to 2 m (Gleason and Cronquist 1991, Rehder 1940, Wyman 1977). L. tatarica has leaves that are ovate to oblong, glabrous, and are 3-6 cm long. Its flowers are glabrous, white to pink, 1.5-2 cm long, on peduncles 15-25 mm long. The fruits are red or rarely yellow. Height ranges to 3 m (Gleason and Cronquist 1991).

L. x bella has intermediate characteristics. The leaves are slightly hairy beneath. Flowers are pink fading to yellow, on sparsely hairy peduncles 5-15 mm. long. Fruits are red or rarely yellow. Height ranges to 6 m (Gleason and Cronquist 1991).

STEWARDSHIP SUMMARY

The exotic bush honeysuckles are increasingly common throughout much of the eastern and mid-western United States and south-central Canada where they have contributed to reduced richness and cover of native herb communities and to reduced tree regeneration in early to mid-successional forests. Although disturbance of some kind usually precedes invasion, the exotic bush honeysuckles are adapted to a wide variety of habitats. Reproduction is almost entirely by seed. Seed production and short-term seed viability are consistently high, and seeds are readily dispersed by birds and, perhaps, small mammals. The group is relatively

free of known significant diseases and insect or other predators. Mechanical controls include grubbing or pulling seedlings and mature shrubs, and repeated clipping of shrubs. Effective mechanical management requires a commitment to repeated treatments for a period of three to five years. Winter clipping should be avoided as it encourages vigorous re-sprouting. Repeated annual prescribed burns during the growing season will top-kill shrubs and inhibit new shoot production. Because exotic bush honeysuckles readily resprout, it may be necessary to re-burn every year or every other year for several years. Most managers report that treatment with herbicides is necessary to control the exotic bush honeysuckles. Water-soluble formulations of glyphosate (brand names Roundup, and for use near waterbodies, Rodeo), a non-selective herbicide, and formulations of triclopyr (brand names Garlon, Pathfinder, and others), a selective herbicide for broad-leaved plants, have been used as foliar sprays or cut stump sprays and paints with varying degrees of success. Both glyphosate and triclopyr should be applied to the foliage late in the growing season, and to cut-stumps from late summer through the dormant season. The flush of seedlings that sometimes follows herbicide treatments must also be controlled.

RANGE

L. maackii is native to central and northeastern China, Manchuria, Korea and, less commonly, Japan. It was introduced to Europe beginning in 1887 and to North America at the Dominion Arboretum in Ottawa, Canada in 1896 and the New York Botanical Garden in 1898 (Luken and Thieret 1995). It is now naturalized in twenty-four states of the eastern and central United States and in Ontario, Canada (Trisel and Gorchov 1994). Reported occurrences of L. maackii in North America include: Arkansas, Delaware, District of Columbia, Georgia, Illinois, Indiana, Kansas, Kentucky, Maryland, Massachusetts, Michigan, Mississippi, Missouri, New Jersey, North Dakota, Ohio, Ontario, Pennsylvania, South Carolina, Tennessee, Texas, Virginia, West Virginia, and Wisconsin.

L. morrowii is native to Japan and was introduced to North America circa 1875 (Rehder 1940). It is now common in southeastern and south-central Canada, and in most northeastern and mid-Atlantic states and in some midwestern states. Reported occurrences of L. morrowii in North America include: Arkansas, Colorado, Connecticut, District of Columbia, Illinois, Iowa, Kentucky, Maine, Maryland, Massachusetts, Michigan, Minnesota, Missouri, New Brunswick, New Hampshire, New Jersey, New York, North Carolina, Ohio, Ontario, Pennsylvania, Quebec, Rhode Island, Saskatchewan, Tennessee, Vermont, Virginia, West Virginia, Wisconsin, and Wyoming.

L. tatarica is native to western and central Russia (Barnes 1974) and was introduced to the United States as early as 1752 (Rehder 1940). It is common in southeastern and south-central Canada, and in most northeastern and mid-Atlantic states and in some midwestern and western states. Reported occurrences of L. tatarica in North America include: Alberta, California, Colorado, Connecticut, Delaware, District of Columbia, Illinois, Indiana, Iowa, Kansas, Kentucky, Maine, Manitoba, Maryland, Massachusetts, Michigan, Minnesota, Montana, Nebraska, New Brunswick, New Hampshire, New Jersey, New York, North Dakota, Nova Scotia, Ohio, Ontario, Pennsylvania, Quebec, Rhode Island, Saskatchewan, South Dakota, Utah, Vermont, Virginia, West Virginia, Wisconsin, and Wyoming.

L. tatarica and L. morrowii hybridize to form L. x bella, which is widely naturalized from Alberta, Canada, east to Maine, south to northern North Carolina, west to Missouri, and north through Kansas, eastern Nebraska, and the Dakotas (Barnes 1974). Reported occurrences of L. x bella in North America include: Connecticut, Illinois, Indiana, Kentucky, Maine, Maryland, Massachusetts, Michigan, Minnesota, New Brunswick, New Hampshire, New Jersey, New York, North Carolina, Ohio, Ontario, Pennsylvania, Quebec, Rhode Island, Saskatchewan, South Carolina, Vermont, Virginia, Wisconsin, and Wyoming.

HABITAT

In its native habitat, L. maackii is found in mixed forests in association with oaks, elms and other hard-

woods, and with softwoods such as fir, spruce, and hemlock; in floodplain forests; and in scrub communities. It is often found in calcareous soil (Luken and Thieret 1995; Luken et al. 1995a). In Japan, L. morrowii is often associated with mesic sites and acidic soils (Barnes 1974). In Eurasia, L. tatarica occurs in dry, relatively cool semi-desert locations (Barnes 1974). The group as a whole favors disturbed sites and forest edges or openings (Barnes 1974; Luken et al. 1995b), but in a New England study of the L. tatarica-L. morrowii-L. x bella group, it also invaded the interior of intact forests (Woods 1993).

In North America, L. maackii is often found in urban forests or in forests with histories of fragmentation, grazing, or woodcutting (Luken and Thieret 1995), and in semi-shaded fencerows, weedy thickets, and brushy groves (Cochrane 1995). It is especially aggressive on calcareous soils (Cochrane 1995; Luken and Goessling 1995). In North America, L. morrowii, L. tatarica, and their hybrid L. x bella occupy a wide range of sites. They are most often found on forest edges and in forest interiors but are also found in lacustrine (lakeside) and riparian habitats and in a variety of waste places such as abandoned agricultural land and road and railroad rights-of-way (Barnes 1974; Woods 1993). They grow in soils ranging from poorly to well drained and non-calcareous to limey and tolerate low nutrient availability (Barnes 1974; Woods 1993). A 1998 survey of The Nature Conservancy's land managers documented L. maackii and/or L. tatarica in a number of habitats and communities, including mesic woodlands, old-growth northern hardwood forests, oak woodlands, floodplain forests, maritime forests, shale barrens, shrub fens, maritime shrublands, dry prairie, grasslands, and serpentine grasslands (Randall and Meyers-Rice, unpublished).

BIOLOGY AND ECOLOGY

Flowering and Fruiting

Reproduction of the bush honeysuckles is almost entirely by seed (Converse 1985), although greenwood and hardwood cuttings have been used extensively in their commercial propagation (Wyman 1977). L. maackii and L. tatarica (and perhaps the others as well) consistently produce abundant annual seed crops (Schopmeyer 1974). L. maackii may first fruit when as young as three to five years (Luken and Thieret 1995). L. maackii seeds ripen September through November; L. morrowii and L. tatarica seeds, June through August (Schopmeyer 1974).

Seed Germination

Horticultural recommendations for germination of bush honeysuckle seeds call for a three-month stratification at 40° degrees F. (Wyman1977), but in a greenhouse experiment, L. maackii seeds collected in November began to germinate in just 18 days and continued to germinate three months from planting. Light promoted but was not necessary for germination; germination rates at the end of 88 days ranged from 50% in dark to 80% in light (Luken and Goessling 1995).

Seed Dispersal

Seeds of bush honeysuckles are dispersed by birds and perhaps by small mammals. L. tatarica and L. maackii fruits, which persist on the plants into the middle of the winter, are often consumed by a variety of birds (Ingold 1983; White 1992). Bird dispersal contributes to germination success by increasing the likelihood that seed will be dropped in lighted tree fall gaps and other openings rather than in shaded settings (Hoppes 1988). Bird dispersal also increases germination success where allelopathy is present, as has been suggested for some Lonicera species (Converse 1985).

Phenology

In a study conducted in three northeastern forest stands, L. tatarica was the earliest deciduous plant to leaf-out, with leaf break beginning two weeks earlier than for co-occurring trees. It also retained its leaves longer

than any other woody plant (Woods 1993). In Wisconsin, L. x bella showed a wide adaptability to different light regimes (Barnes 1975). Although L. maackii (and perhaps the others) are not unusually shade tolerant (Luken et. al. 1995b), their unusually long photosynthetic period may help explain their competitiveness. The competitive success of L. x bella may also be due in part to hybrid vigor (Barnes 1974).

ECONOMIC USES

Lonicera spp. have been widely used in ornamental plantings. L. tatarica has also been used in mine reclamation (Wade 1985).

IMPACTS AND THREATS POSED BY EXOTIC LONICERA SPP.

Forest regeneration following disturbance can be severely impeded by these species. The group is widely considered an aggressive, highly successful weedy complex (Barnes 1974; Luken and Thieret 1996; Woods 1993 and others). In a survey of Ohio forests, tree seedling density, tree seedling species richness, and herb cover were all inversely related to L. maackii cover, and tree regeneration appeared to have been inhibited (Hutchinson 1997). In a study in New England, the L. tatarica-L. morrowii-L. x bella complex reduced the richness and cover of herb communities and the establishment of new seedlings. Seedlings that predate L. tatarica establishment were more tolerant of its presence. Annual herbs were entirely suppressed. (Woods 1993).

The L. tatarica-L. morrowii-L. x bella complex is an aggressive invader of lower elevation forests throughout the northeastern United States (Woods 1993). L. maackii becomes a dominant shrub in a large variety of plant communities growing on calcareous soils (Luken and Goessling 1995) and is increasingly common in a variety of disturbed, early to mid-successional forests throughout the eastern and central United States and south-central Canada (Luken and Thieret 1996). In a study conducted in the Oxford, Ohio area, L. maackii moved outward from its urban point of origin at a rate of from 0.1 to 0.5 km/year. Agricultural land can act as a barrier to L. maackii spread while greater forest cover and connectivity facilitates its extent, presumably due to the relationship between cover type and dispersal by birds (Hutchinson 1998).

Schmidt and Whelan (1999) studied nest predation on American robins (Turdus migratorius) and wood thrushes (Hylocichla mustelina) for 6 years in a 200 ha woodland fragment near Chicago. They found that robin nests in Lonicera maackii and another non-native, invasive shrub, Rhamnus cathartica, experienced higher predation rates than nests in similar native shrubs (Crataegus, Viburnum) and in native trees. Part of this difference was due to nests in L. maackii being built closer to the ground. The authors speculate that absence of thorns on the exotics and a branch structure that facilitates movement of predators like raccoons may also help explain the difference. Robin use of Lonicera increased sharply during the 6-year study and the authors suggest this may be due to the exotic shrub's early leaf-out. If so, higher predation rates early in the season may also help explain the difference between nest success in exotic and native plants. Predation on wood thrush nests in native and exotic plants was not significantly different. High proportions of thrush nests were in L. maackii and as use of L. maackii by robins increased, predation rates on thrushes increased. The authors caution that these results are specific to a single site and to the two bird species followed and that it is not known whether they will be applicable to other sites or species. But they note that if higher nest predation rates are found in exotic shrubs elsewhere, restoring native shrubs would serve several conservation goals simultaneously.

The exotic bush honeysuckles may provide an important source of winter food for birds in areas where it is abundant and few other shrubs survive (Whelan and Dilger 1992). In a study of New Jersey frugivores, L. tatarica was one of two introduced species used most by birds during the winter sampling period, after high-quality native fruit sources had been exhausted (White 1992). Despite their low fat content and extreme bitterness, L. maackii fruits, which persist into the middle of the winter in Ohio, are also consumed by a variety of birds (Ingold 1983). In a study undertaken in southwestern Ohio, deer mice (Peromyscus manicu-

latus) were found to be the major small mammal consumers of L. maackii fruits (Williams et al. 1992).

The bush honeysuckles have been promoted for decades by the U.S. Department of Agriculture and by commercial nurseries for their wildlife, shelterbelt, and ornamental value. Many state and private nurseries still sell them, although less widely than previously (Luken and Thieret 1996). Commercial sources continue to introduce Lonicera spp. to areas not already colonized, but most future invasions will originate in naturalized populations of the shrubs. Only cropland and large, closed canopy forests, where bush honeysuckles remain relegated to the edge (Luken and Thieret 1996), will be secure from invasion (Barnes 1974; Hutchinson and Vankat 1998).

MANAGEMENT

Potential for Restoration of Invaded Sites

Lonicera spp. annually produce large numbers of viable seed that are readily dispersed by birds and germinate at high rates in a wide range of conditions. The different species are extremely adaptable and have successfully invaded a wide range of habitats and communities in North America. Manual and mechanical, environmental/cultural, and chemical methods are all useful to varying degrees in controlling Lonicera spp. The use of prescribed fire may be effective in some cases where the density of Lonicera spp. is low and sufficient fuels are available. Restoration potential is likely to be lowest where Lonicera spp. occur in high densities and there is a high likelihood of continued dispersal of seeds into the restoration area. Lonicera spp. have a high degree of reproductive vigor, a wide range of adaptability, and few pests and predators in North America. The potential for large-scale restoration of unmanaged natural areas or wildlands infested with Lonicera spp. is probably low. Restoration potential for managed natural areas or wildlands infested Lonicera spp. is probably moderate. If attacked during the early stages of colonization, the potential for successful management is high.

Biological Controls

There are no known biological controls of Lonicera spp. The aphid Hyadaphis tatariacae (Aizenberg) is an obligate feeder on the tips and shoots of L. tatarica and perhaps L. morrowii and L. x bella . The resulting “witches brooming” may somewhat reduce flowering (Voetglin and Stoetzel 1988) and therefore fruiting. Hyadaphis is present throughout the northern U.S. and southern Canada, as are a number of native ladybug beetles that prey on it. (Nyboer 1992).

Mechanical Control

Mechanical controls include grubbing or pulling seedlings and mature shrubs, and repeated clipping of shrubs. Effective mechanical management requires a commitment to cut or pull plants at least once a year for a period of three to five years (Virginia Natural Heritage Program, no date). Grubbing or pulling by hand (using a Weed Wrench or a similar tool) is appropriate for small populations or where herbicides cannot be used. Any portions of the root system not removed can resprout (Tennessee Exotic Pest Plant Council 1997). Because open soil can support rapid re-invasion, managers must monitor their efforts at least once per year and repeat control measures as needed (Nyboer 1992). Mature L. maackii shrubs growing in shaded forest settings can be eradicated by clipping once a year, during the growing season, until control is achieved (Luken and Mattimiro 1991). Other bush honeysuckles growing in more open settings can be managed by clipping twice yearly, once in early spring and again in late summer or early autumn. Winter clipping should be avoided as it encourages vigorous re-sprouting (Virginia Natural Heritage Program, no date). Mature honeysuckle wood is tough and easily dulls power-tool blades (Nyboer 1992).

Prescribed Burning

Repeated annual prescribed burns during the growing season will top-kill shrubs and inhibit new shoot

production. Because exotic bush honeysuckles readily resprout, it may be necessary to re-burn every year or every other year for several years to achieve good control (Nyboer 1992).

Herbicides

Most managers report that treatment with herbicides is necessary for the control of L. maackii populations growing in full sun and may be necessary for all large bush honeysuckle populations. Formulations of glyphosate (brand names Roundup, and for use near waterbodies, Rodeo) and formulations of triclopyr (brand names Garlon, Pathfinder, and others), have been used as foliar sprays or cut stump sprays and paints with varying degrees of success (Nyboer 1992). Glyphosate is a non-selective herbicide which kills both grasses and broad-leaved plants while triclopyr is a selective herbicide that kills broad-leaved plants but does little or no harm to grasses. A survey of The Nature Conservancy land managers undertaken in 1998 found that most used glyphosate, and used it as a cut stump treatment, to control L. maackii and/or L. tatarica (Randall and Meyers-Rice, unpublished). For cut stump treatments, 20-25% solutions of glyphosate or triclopyr can be applied to the outer ring (phloem) of the cut stem. 2% solutions of glyphosate or triclopyr can be used for foliar treatments. Both glyphosate and triclopyr should be applied to the foliage late in the growing season, and to the cut stumps from late summer through the dormant season (Tennessee Exotic Pest Plant Council, 1997; Virginia Natural Heritage Program, no date). The subsequent flush of seedlings following all herbicide treatments must also be controlled (Luken and Mattimiro 1991).

EXAMPLES OF LONICERA SPP. CONTROL ON TNC PRESERVES

Exotic bush honeysuckles have been reported from TNC preserves in Kentucky, Indiana, Wisconsin, Vermont, Maryland, Massachusetts, Minnesota, Ohio, and in New York. Garth Fuller reported that prescribed fires are useful in controlling Lonicera spp. in Minnesota, but only if the fires are hot and repeated at regular intervals. Steve Richter of Wisconsin similarly reported that fire is useful in controlling the growth of small seedlings.

All preserves reported that pulling is effective for smaller plants, but is labor intensive. Cutting the shrubs was also effective, especially if the herbicide glyphosate (tradename RoundUp and others) is applied to the cut-stump immediately after cutting. Margaret Shea of the Kentucky preserves and Ross Lebold of Ohio reported positive results using the cut-stump method. David Banks of Indiana, however, reported that results from cutting were poor without an herbicide application. Garth Fuller (of Minnesota) added that the cut-stump technique using glyphosate or triclopyr (tradename Garlon and others) was effective if applied in fall, but not effective if applied in spring.

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MONITORING

Control efforts must be repeated and monitored for three to five years following the initial treatment (Virginia Natural Heritage Program, no date). In natural areas management, monitoring programs will likely combine assessing changes in abundance of exotic bush honeysuckles with changes in abundance of species or changes in community attributes that are the targets of management. Such programs should be designed to meet certain minimum confidence levels (for example, a 90% confidence level that changes of 20% and more will be accurately captured). And such programs should have explicit objectives that can be measured and that are meaningful from both a biological and management standpoint. These objectives may vary depending on the abundance of exotic bush honeysuckles and other invasives. For instance, in a forest that currently has 40% cover of exotic bush honeysuckles an appropriate management objective might be to reduce honeysuckle cover to 20%; In a forest that currently has just 10% honeysuckle cover, an appropriate management objective might be to prevent honeysuckle cover from increasing by more than 10% (total honeysuckle cover 20%). In addition, increasing the regeneration and abundance of native species may be an important objective. Monitoring the status of other conservation targets, such as invertebrates dependent on specific nectar sources, may be more important than tracking invasive plant species abundance. In general, the objectives of monitoring should track those of management.

In terms of effort (number of plots established and monitored), transects or long, linear plots are generally more effective in providing sufficient statistical power to determine change than square or broadly rectangular or otherwise regularly shaped quadrats. Analyses of plant species composition and abundance can be simplified by (1) collecting data on abundance of dominant species; (2) collecting data on all species and pooling data on less abundant species; and (3) pooling data on species by placing them in guilds (invasive grasses, invasive legumes, native grasses, etc.).

While generally a research technique, measuring change, or lack thereof, in control (unmanaged) areas can be an effective way of assuring that changes seen in treated areas are actually the result of management actions and not due to other factors. In forest communities that are in early successional stages or recently disturbed, declines in abundance of the exotic bush honeysuckles may occur over time without management.

RESEARCH NEEDS

Among the research topics suggested in the literature are: 1) What are the mechanisms of exotic honeysuckle invasion and spread in a variety of fragmented forest landscapes? (Brothers 1992); 2) What is the light environment of disturbed forests and bush honeysuckle's corresponding tolerance limits for critical life history events? (Luken et al. 1995b); 3) What can comparisons between invasives and native congeners (or other less prolific non-native congeners) tell us about those traits that are most likely responsible for successful invasion and establishment versus those that are merely coincident with them? (Schierenbeck

et al. 1994); and 4) What are the effects of bush honeysuckle invasions on herb layer species? (Hutchinson and Vankat 1997).

White tailed deer are a major influence on the composition of forest communities in the midwest and northeast . No information exists on the relationship between high deer populations and the abundance of the exotic bush honeysuckles. To what extent selective browsing by deer is important to the abundance of the exotic bush honeysuckle as a forest invader, is an important area of research.

Other research questions might include: 1) Which if any biocontrols are effective in the native ranges of the taxa?; 2) What role do logging and other forestry practices play in the successful spread of these taxa?; 3) How could forestry operations be carried out to prevent invasion by exotic bush honeysuckles?; 4) Which species replace honeysuckle when control succeeds?; 5) What is the effectiveness of prescribed burning on reducing or eliminating these species and encouraging regeneration of native species in forest types that are fire influenced?; and 6) What are the latitudinal limits of the species? Have they been reached in North America?

Work is also needed on more efficient control methods, especially where cutting is used. Standard tools such as weed whackers, brush hogs and other equipment are not designed for cutting this species or the kinds of habitat where work is needed.

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Chinese Privet

ELEMENT STEWARDSHIP ABSTRACT
for Ligustrum spp.
Privet
To the User:

Element Stewardship Abstracts (ESAs) are prepared to provide The Nature Conservancy's Stewardship staff and other land managers with current management related information on species and communities that are most important to protect or control. The abstracts organize and summarize data from many sources including literature and from researchers and managers actively working with the species or community.

We hope, by providing this abstract free of charge, to encourage users to contribute their information to the abstract. This sharing of information will benefit all land managers by ensuring the availability of an abstract that contains up-to-date information on management techniques and knowledgeable contacts. Contributors of information will be acknowledged within the abstract.

For ease of update and retrievability, the abstracts are stored on computer at The Nature Conservancy. Anyone with comments, questions, or information on current or past monitoring, research, or management programs for the species described in this abstract is encouraged to contact The Nature Conservancy's Wildland Invasive Species Program.

This abstract is a compilation of available information and is not an endorsement of particular practices or products.

Please do not remove this cover statement from the attached abstract.

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SCIENTIFIC AND COMMON NAMES

Ligustrum amurense (Carr.): Amur privet
Ligustrum japonicum (Thun.): Japanese privet
Ligustrum lucidum (Ait.f.): Tree privet; glossy privet
Ligustrum obtusifolium (Sieb. and Zucc.): Blunt-leafed privet; border privet
Ligustrum ovalifolium (Hassk.): California privet; waxy-leaved privet
Ligustrum quihoui (Carr.): Wax-leaf privet
Ligustrum sinense (Lour.): Chinese privet
Ligustrum vulgare (L.): European privet, common privet

DESCRIPTION AND DIAGNOSTIC CHARACTERISTICS

Ligustrum spp. are deciduous, semi-evergreen, or evergreen shrubs and small trees in the Oleaceae (olive family). There are approximately 50 Ligustrum species that are native to Europe, North Africa, and Asia. Ligustrum spp. have been cultivated and developed into several horticultural varieties, and were introduced to North America as a common hedge in landscaping. Ligustrum spp. can easily escape cultivation to invade adjacent areas and can form dense monospecific thickets.

Some Ligustrum spp. can grow to 5 m tall and have a stem diameter of 2.5-25 cm. Ligustrum spp. bark is whitish-tan to gray in color and smooth in texture. Slender twigs are straight, rounded or four-angled below the nodes, and gray-green in color. Winter buds are ovoid with two outer scales. Terminal buds are present. Leaves are elliptic to ovate in shape, oppositely arranged on slender twigs, often leathery and thick. Flowers have both male and female parts, and the corollas are white. The calyx is small, obconic or campanulate, and 4-toothed. Each flower has petals that are fused into a tube below with four separate lobes above. Flowers are borne on small panicles terminating the main axis and on short lateral branches. Bloom time is usually June-July. The fruit is a subglobose or ovoid drupe containing 1-4 seeds. Fruit clusters generally ripen during September and October and persist through the winter. Mature specimens can produce hundreds of fruit (Rehder 1977).

L. amurense grows to 5 m. Leaves are elliptic to oblong or oblong-ovate, 2.5-6 cm long, acute or obtuse, rounded or broad-cuneate at base; ciliolate, sometimes lustrous above, and smooth except on the midrib below. Petioles are 2-4 mm long, pubescent. Panicles are 3-5 cm long and pubescent. The calyx is glabrous or slightly pubescent. The corollas (from base of tube to tip of lobe) are 7-9 mm long, with the tube far longer than the lobes.

L. japonicum generally grows to 3 m, rarely to 6 m. Leaves are broad-ovate to ovate-oblong, 4-10 cm long, obtusely short-acuminate or acute to obtuse, rounded at the base with reddish margins and midrib and with 4-5 pairs of indistinct veins. Petioles are 6-12 mm long. Panicles are 6-15 cm long. Flowers are short-stalked with the corolla tube longer than the calyx. Stamens are slightly longer than the corolla lobes.

L. lucidum grows as a large shrub or medium-sized tree, to 10 m high, with spreading branches. Leaves are ovate to ovate-lanceolate, 8-12 cm long, acuminate or acute, usually broad-cuneate with 6-8 veins, usually distinct above and beneath. Petioles are 1-2 cm long. Panicles are 12-20 cm long and nearly as wide. Flowers are subsessile. The corolla tube is as long as the calyx. Stamens are as long as the corolla lobes. Fruits are oblong, 1 cm long, bluish or purplish-black.

L. obtusifolium grows to 3 m with spreading or arching branches. Leaves are elliptic to oblong or oblong-obovate, 2-6 cm long. Leaves are acute or obtuse, cuneate or broad-cuneate, glabrous above, pubescent below (or occasionally only on midrib). Petioles are 1-4 mm long, pubescent. Panicles are 2-3.5 cm long, nodding. Corollas are 8-10 mm long with anthers nearly as long as the corolla lobes. Fruits are subglobose, black and slightly bloomy (glaucous).

L. ovalifolium grows to 5 m. Leaves are elliptic-ovate to elliptic-oblong, 3-6 cm long, acute, broad-cuneate, dark lustrous green above, yellowish green below. Petioles are 3-4 mm long. Flowers are creamy-white with an unpleasant scent, subsessile in panicles 5-10 cm long. Corollas are 8 mm long with anthers as long as lobes. Fruits are 5-7 mm across, black.

L. quihoui grows to 2 m with spreading, rigid branches. Leaves are elliptic to elliptic-oblong or obovate to obovate-oblong, 2-5 cm long, obtuse, sometimes emarginate, glabrous, subcoreaceous. Petioles are 1-3 mm long, puberulous. Flowers are sessile, in small clusters on long spikes collected into 10-20 cm long panicles. Corolla tubes are as long as the lobes with anthers exceeding the lobes. Flowers appear in late summer.

L. sinense is a shrub or small tree to 7 m. Leaves are elliptic to elliptic-oblong, 3-7 cm long, acuminate, acute to obtuse, dull green above, pubescent on the midrib below. Petioles are 6-15 mm long. Flowers are small, distinctly stalked, on panicles 10-16 cm long. Fruits are dull black.

L. vulgare grows to 5 m with spreading branches. Leaves are oblong-ovate to lanceoloate, 3-6 cm long, obtuse to acute, glabrous. Petioles are 3-10 mm long. Flowers are pediceled in dense puberulous panicles, 3-6 cm long. Anthers exceed the corolla tube. Fruits are subglobose or ovoid, 6-8 mm long, black and lustrous.

STEWARDSHIP SUMMARY

Several Ligustrum species have become common invaders of cultivated landscapes, disturbed areas and wildlands throughout the U.S. L. amurense is found in many eastern and some south-central states. L. japonicum is found in the Southeast and in Puerto Rico. L. lucidum is present from Maryland south and west to Texas. L. sinense and L. obtusifolium are found throughout the eastern and central U.S. L. ovalifolium is common in California and in parts of the central and eastern U.S. L. quihoui is seen in the southeast. L. vulgare is widely naturalized throughout much of the U.S. and southern Canada.

Ligustrum spp. may invade natural areas such as floodplain forests and woodlands. They may displace shrubs in regenerating communities and remain persistent in these areas. Ligustrum spp. can form dense thickets that outcompete many kinds of native vegetation.

In North America, Ligustrum spp. are seen along roadsides, in old fields and in other disturbed habitats and in a variety of undisturbed natural areas, including bogs, wetlands, floodplains, old fields, calcareous glades and barrens, and mesic hardwood forests.

Ligustrum spp. control methods include mowing and cutting, seedling removal, herbicide application, and burning. Mowing and cutting are appropriate for small initial populations or environmentally sensitive areas where herbicides cannot be used. Stems should be cut at least once per growing season as close to ground level as possible. Repeated mowing or cutting will control the spread of Ligustrum spp., but may not eradicate it. Ligustrum spp. can also be effectively controlled by manual removal of young seedlings. Herbicide control measures include foliar spraying in late autumn or early spring with glyphosate, triclopyr, or metsulfuron; cut stump applications using glyphosate or triclopyr; and basal bark applications of triclopyr. Some reports indicate that burning top-kills L. vulgare and L. sinense and, if repeated, can eliminate them over time.

RANGE

Ligustrum spp. are native to east Asia, Europe and North Africa: Ligustrum amurense is native to north China; L. japonicum to Korea and Japan; L. lucidum to China, Korea and Japan; L. obtusifolium to Japan; L. ovalifolium to Japan; L. sinense to China; and L. vulgare to the Mediterranean region.

Reported occurrences of the different Ligustrum spp. in North America include:

L. amurense: Arkansas, Kentucky, Maine, Maryland, Massachusetts, New Jersey, New York, North Carolina, Pennsylvania, South Carolina, Tennessee, Texas, and Virginia.

L. japonicum: Alabama, Florida, Georgia, Louisiana, Maryland, Mississippi, North Carolina, South Carolina, Tennessee, Texas, Virginia, and Puerto Rico.

L. lucidum: Alabama, Florida, Georgia, Louisiana, Maryland, Mississippi, North Carolina, and Texas.

L. obtusifolium: Connecticut, District of Columbia, Illinois, Indiana, Kentucky, Maryland, Massachusetts, Michigan, Missouri, New Hampshire, New Jersey, New York, North Carolina, Ohio, Pennsylvania, Rhode Island, Tennessee, Utah, Vermont, and Virginia.

L. ovalifolium: California, Connecticut, Delaware, District of Columbia, Florida, Kentucky, Louisiana, Maryland, Massachusetts, Michigan, Missouri, New Jersey, North Carolina, Ontario, Pennsylvania, Texas, Vermont, Virginia, and Puerto Rico.

L. quihoui: Louisiana, North Carolina, Texas, and Virginia.

L. sinense: Alabama, Arkansas, Connecticut, Florida, Georgia, Iowa, Kentucky, Louisiana, Maryland, Massachusetts, Mississippi, Missouri, New Jersey, North Carolina, Oklahoma, Rhode Island, South Carolina, Tennessee, Texas, and Virginia.

L. vulgare has the broadest range of the invasive Ligustrum species established in North America. It has been documented in: Alabama, Arkansas, British Columbia, Connecticut, Delaware, District of Columbia, Florida, Georgia, Great Smoky Mountain National Park, Illinois, Indiana, Kentucky, Louisiana, Maine, Maryland, Massachusetts, Michigan, Missouri, New Hampshire, New Jersey, New York, Newfoundland Island (Newfoundland), North Carolina, Nova Scotia, Ohio, Ontario, Pennsylvania, Rhode Island, South Carolina, Tennessee, Texas, Utah, Vermont, Virginia, West Virginia, and Wisconsin.

IMPACTS AND THREATS POSED BY EXOTIC PRIVETS

Ligustrum spp. can form dense thickets that outcompete native vegetation. The privets can invade natural areas such as floodplain forests, woodlands, and disturbed agricultural fields. They generally expand along fence-rows, windbreaks and roadsides (Haragan 1996). In New Zealand, L. sinense may displace the shrub layer and marginal shrubs of alluvial forests, and remain persistent in these areas. L. lucidum replaces mid-canopy trees in forests and may completely dominate an area of forest or forest fragments if not controlled (New Zealand Weeds Web Site 1999). L. japonicum and L. sinense invade woodlands in the eastern and southeastern U.S. (Faulkner et al. 1989; Stone 1997). Forest gaps can also become invaded since birds often disperse Ligustrum seeds.

HABITAT

In North America, Ligustrum spp. often grow along roadsides, in old fields and in other disturbed habitats and in a variety of undisturbed natural areas. Examples of Ligustrum invasions include:

1) L. obtusifolium was found invading an old field succession site in Illinois. The field had an average of more than 6,082 plants per ha (2.5 acres) (Tennessee Exotic Pest Plants Council 1996).

2) L. sinense has been reported in bogs, an oak-hickory-pine forest, a longleaf pine-turkey oak forest, and mesic hardwood forests in Alabama. In Arkansas, L. sinense has been reported in virtually all non-xeric

habitats. In Georgia, L. sinense has been reported in floodplain/wetland habitats, and in North Carolina, in woodland edges (Randall and Meyers-Rice. unpublished.).

3) L. vulgare has been recorded in bottomlands and mesic and riparian forests in Arkansas. In Ohio, L. vulgare is found in old fields, primary woodlands, and closed canopy forests. In Tennessee, the species has been recorded in calcareous glades and barrens and in deciduous cove forests (Randall and Meyers-Rice. unpublished.).

4) In New Zealand, L. sinense is found in alluvial forest remnants, waste places, shrublands, and open stream systems, particularly in coastal areas. L sinense is widespread and common, especially near towns. It is a common farm hedging plant. L. lucidum is found in forests (lowland and coastal), forest fragments, shrublands, along roadsides, in farm hedges, wastelands, and domestic gardens (New Zealand Weeds Web Site).

ECOLOGY AND BIOLOGY

Ligustrum spp. are perennial shrubs that grow readily from seed or from root and stump sprouts. They can escape from cultivation when the fruits are consumed by wildlife, particularly birds, which often excrete the seeds unharmed at distant locations where they may germinate and become established. Germination rates have been variously reported as low as 5%-27% (Tennessee Exotic Plants Council 1996) and as high as 77% (Schopmeyer 1974). Unlike most woody species, experimental defoliation did not result in reduced percentages of flowers producing fruits, decreased seed number, or decreased seed quality (Obeso and Grubb 1993).

Ligustrum spp. leaves are high in phenolic compounds that defend against herbivores, especially insects. These work by inhibiting digestive enzymes and proteins (Konno et al. 1998). Despite this, L. sinense has been identified as an important forage plant for deer in the southeastern U.S. (Stromayer et al. 1998).

L. vulgare grows well in high light, low nutrient soils, but will tolerate lower light levels if nutrients are increased (Grubb et al. 1996).

MANAGEMENT

Potential for Restoration of Invaded Sites

In North America, Ligustrum spp. have no important pests or predators. The various species are widespread and occasionally locally abundant. Manual and mechanical, environmental/cultural, and chemical methods are all useful in varying degrees in controlling Ligustrum spp. Fire management may be useful in some cases where the density of Ligustrum spp. is low and sufficient fuels available. Restoration potential is likely to be lowest where Ligustrum spp. occur in high densities and there is a high likelihood of continued dispersal of seeds into the restoration area. Ligustrum spp. have a high degree of reproductive vigor, a wide range of adaptability, and, in its present settings, few pests and predators. Ligustrum spp. produce large numbers of viable seed that are readily dispersed by birds and germinate at high rates in a wide range of conditions.

The potential for large-scale restoration of unmanaged natural areas or wildlands infested with Ligustrum spp. is probably low. Restoration potential for managed natural areas or wildlands infested Ligustrum spp. is probably moderate. If attacked during the early stages of colonization, the potential for successful management is high.

Mechanical Controls

Mowing and cutting are appropriate for small populations or environmentally sensitive areas where herbicides cannot be used. Stems should be cut at least once per growing season as close to ground level as

possible. Repeated mowing or cutting will control the spread of Ligustrum spp., but may not eradicate it (Tennessee Exotic Pest Plants Council 1996). Managers of The Nature Conservancy preserves in Ohio reported eradication of L. vulgare after two cutting treatments (Randall and Meyers-Rice, unpublished).
Ligustrum spp. can be effectively controlled by the manual removal of young seedlings. Plants should be pulled as soon as they are large enough to grasp but before they produce seeds. Seedlings are best pulled after a rain when the soil is loose. Larger stems (up to 6 cm in diameter) can be removed using a weed wrench or similar uprooting tools. The entire root must be removed since broken fragments may resprout (Tennessee Exotic Pest Plants Council 1996).

Biological Controls

Ligustrum spp. have no known biological controls, although a few pathogens are known to attack them in North America. Cercospora adusta, C. lilacis, and Pseudocercospora lugustri are fungal leaf spots that affect L. vulgare and L. amurense. Nectriella pironi creates galls on L. sinense, L. lucidum and L. quihoui. Pseudomas syringae impacts members of the olive family including L. amurense. Agrobacterium tumefaciens, Ganoderma lucidum and Glomerella cingulata affect L. vulgare (Sinclair et al. 1987).

Herbicides

Foliar Spray Method: This method may be effective for large thickets of Ligustrum spp. where risk to non-target species is minimal. Air temperatures should be above 17°C to ensure that herbicides are absorbed. The ideal time to treat is while plants are in leaf in late autumn or early spring but when many native species are dormant.

Glyphosate (brand name Roundup and others): A number of concentrations have been used successfully. The Tennessee Exotic Pest Plants Council (1996) suggests a 2% solution of glyphosate and water plus a 0.5% non-ionic surfactant to thoroughly wet all leaves. The New Zealand Weeds Web Site (1999) recommends, for a handgun sprayer, 1 liter Roundup and 100 mls of a surfactant per 100 liters of water (1% solution); for a backpack sprayer, the recommendation is 100 ml Roundup and 20 mls of a surfactant per 10 liters of water. (Roundup is a non-selective herbicide.)

Triclopyr (brand name Garlon, Pathfinder II and others): The Tennessee Exotic Pest Plants Council (1996) suggests a 2% solution of triclopyr and water plus a 0.5% non-ionic surfactant, sprayed to thoroughly wet all leaves. Use a low pressure and coarse spray pattern to reduce spray-drift damage to non-target species. (Triclopyr is a selective herbicide for broadleaf species only.)

Metsulfuron (brand name Escort and others): The New Zealand Weeds Web Site (1999) recommends, for a handgun sprayer, 35 g metsulfuron and 100 mls of a surfactant per 100 liters of water; for a backpack sprayer, the recommendation is 5 g metsulfuron and 10 mls of a surfactant per 10 liters of water. Metsulfuron methyl was identified as the most cost-effective herbicide in an experimental treatment comparing metsulfuron methyl, triclopyr ester and 2,4-D (Madden and Swarbrick 1990). (Metsulfuron is a selective herbicide active upon broadleaf and some annual grass species.)

Cut Stump Method: This control method should be considered when treating individual shrubs or where the presence of desirable species precludes foliar application. The Tennessee Exotic Pest Plants Council (1996) recommends this treatment only as long as the ground is not frozen, but other researchers have found it effective on Rhamnus spp. in frozen ground (Reinartz 1997). Immediately after cutting stems at or near ground level, apply a 25% solution of glyphosate and water or triclopyr and water to the cut stump, being careful to cover the entire surface (Tennessee Exotic Pest Plants Council 1996). Effectiveness of the herbicide is increased if holes are cut in the top of the freshly felled stump, to hold the herbicide in for better absorption by plant (New Zealand Weeds Web Site 1999).

Basal Bark Method: Apply a mixture of 25% triclopyr and 75% horticultural oil to the basal parts of the shrub to a height of 30-38 cm (12-15 in) from the ground. Thorough wetting is necessary for good control; spray until run-off is noticeable at the ground line. Like the cut stump application, this method may be effective throughout the year, if Ligustrum spp. responds similarly to Rhamnus spp. (Reinartz 1997). In New Zealand, researchers have killed standing Ligustrum trees by drilling downward-sloping 20 mm wide holes 5 cm into the trunk at no greater than 5 cm spacing around the trunk, and filling the holes with a stump paint-herbicide mix (New Zealand Weeds Web Site 1999).

Prescribed Burning

Faulkner et al. (1989) reported that in experimental trials of prescribed burning, there was no significant difference in the abundance of L. sinense in burned vs. unburned plots. Ligustrum litter has a low flammability and fires did not carry well in these treatments.

The Nature Conservancy land managers in Alabama reported that burning top-kills L. vulgare and L. sinense and eliminates them over time, and that burning is effective at controlling L. sinense if done annually with low fuel moisture and high Keetch-Byram Drought Index (Randall and Meyers-Rice. unpublished).

EXAMPLES OF LIGUSTRUM SPP. CONTROL ON TNC PRESERVES

Ligustrum spp. have been reported as problems weeds on TNC preserves in Alabama, Arkansas, Louisiana, Georgia, Florida, Mississippi, Tennessee, North Carolina, and in Ohio.

In Alabama and in Florida, Carlen Emanuel and Greg Seamon, respectively, reported that annual burning was effective in controlling L. sinense. Furthermore, cutting is also effective if done when conditions are dry. George Ramseur Jr. in Mississippi found that a combination of pulling and burning provided good control of L. sinense.

Richard Martin reports that L. sinense is one of the worst weeds on Louisiana preserves, and has found that the application of Garlon 4 (triclopyr) has produced excellent control results, but RoundUp (glyphosate) did not provide good results. In North Carolina, however, Robert Merriam found that RoundUp was useful in controlling large infested areas of L. sinense. Additionally, cutting was very effective if coupled with the use of Arsenal (imazapyr) on cut stumps. Rates of herbicide application should follow those recommended by the manufacturer. Rates that have been applied successfully for control of Ligustrum are described above.

In Arkansas, Scott Simon reports that burning only top-kills L. vulgare and L. sinense, but will eventually eliminate the plants over time if burns are repeated. Burning is not effective however, in moist bottomland areas.

L. vulgare was successfully controlled in central Ohio preserves. Ross Lebold reported that the cut-stump method, using RoundUp (glyphosate) was effective, and that repeated cutting also seemed effective. In Tennessee, L. vulgare was partially controlled by cutting, and Gabby Call reports that the use of goats to control privet works well. The goats however, must able to reach and destroy adult privet plants.

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MONITORING

In natural areas management, monitoring programs will likely follow changes in abundance of Ligustrum spp. AND changes in abundance of desirable native species or changes in community attributes that are the targets of management. Such programs should have explicit objectives that can be measured and that are meaningful from both a biological and management standpoint. These objectives may vary depending on the abundance of Ligustrum spp. and other invasive plants. For instance, the objective of managing a forest with 40% cover of Ligustrum spp. may be to reduce Ligustrum cover to 20%. On the other hand, on a site with 10% an appropriate management objective might be to prevent an increase of more than 10% of total cover (20% total). In addition, increasing regeneration of native species may be an important objective. Monitoring the status of other conservation targets such as invertebrates dependent on specific nectar sources or plant species that are conservation targets may be more important than tracking invasive plant species abundance. In general, the objectives of monitoring should track those of management.

In terms of effort (number of plots established and monitored), transects or long linear plots are more effective in providing the statistical power to necessary to detect changes than square, broadly rectangular, circular or other regularly shaped quadrats. Analyses of plant species composition and abundance can be simplified by (1) collecting data on abundance of dominant species; (2) collecting data on all species and pooling data on less abundant species; and (3) pooling data on species by placing them in guilds (e.g. invasive grasses, invasive legumes, native grasses, etc.).

While generally a research technique, measuring change, or lack thereof, in control (unmanaged) areas can be an effective way of assuring that changes observed in treated areas actually result from the treatmentand not from other factors such as fire, rainfall, etc. In forest communities that are in early successional stages or recently disturbed, declines in abundance of the Ligustrum spp. may occur with time without management.

RESEARCH

Additional research is needed on more efficient control methods, especially where cutting is used. Standard tools such as weed whackers, brush hogs, and other equipment are not designed for cutting this species or for use in the kinds of habitat which Ligustrum species often invade.

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Japanese Stilt Grass

ELEMENT STEWARDSHIP ABSTRACT
for Microstegium vimineum
Japanese stilt grass, Nepalese browntop, Chinese packing grass

To the User:

Element Stewardship Abstracts (ESAs) are prepared to provide The Nature Conservancy's Stewardship staff and other land managers with current management related information on species and communities that are most important to protect or control. The abstracts organize and summarize data from many sources including literature and from researchers and managers actively working with the species or community.

We hope, by providing this abstract free of charge, to encourage users to contribute their information to the abstract. This sharing of information will benefit all land managers by ensuring the availability of an abstract that contains up-to-date information on management techniques and knowledgeable contacts. Contributors of information will be acknowledged within the abstract.

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Abstract Written: 8/00

THE NATURE CONSERVANCY

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SCIENTIFIC NAME

Microstegium vimineum (Trin.) A. Camus

SYNONYMS

Andropogon vimineus Trin.
Eulalia viminea (Trin.) Kuntze
Microstegium imberbe (Nees ex Steud.) Tzvelev
Microstegium willdenovianum Nees ex Lindl.
Pollinia imberbis Nees ex Steud.
Pollinia viminea (Trin.) Merr.
Pollinia willdenoviana (Nees ex Lindl.) Benth.

COMMON NAMES

Japanese stilt grass, Nepalese browntop, Chinese packing grass

DESCRIPTION AND DIAGNOSTIC CHARACTERISTICS

Microstegium vimineum is a shade tolerant, annual C4 grass (family Poaceae). It is a straggling or decumbent plant, usually 6-10 dm in height, and the reclining stems can grow up to 1.0 m (40 in) long. Its culms (stems) are typically branched, rooting at the lower nodes, and the nodes and internodes are smooth and hairless. The lanceolate leaf blades are 5-8 cm long and 2-15 mm wide, sparsely pubescent on both surfaces, and distinctly tapered at both ends. The ligules are membranous, usually ciliate, and are 0.5-2.0 mm long (Radford et al. 1968).

The terminal or axillary inflorescence is a raceme, 2-7 cm long, with an elongate peduncle and an angled disarticulating rachis. The hirsute fertile spikelets are deciduous, and occur in pairs, with one spikelet sessile and the other pedicellate. The glumes are equal in length (4.5-5.0 mm) and awnless. The first glume is flat and 2-3 veined. The second glume is keeled and 3-veined. There are two lemmas per spikelet, with the lower one sterile and the upper, fertile one awnless or often with a slender awn 4-8 mm. Both cleistogamous (flowers closed at pollination) and chasmogamous (flowers open) conditions have been reported for M. vimineum in Japan, with the axillary flowers all being cleistogamous (Tanaka 1975, in Barden 1987).

The fruit or caryopsis (grain) of M. vimineum is yellowish to reddish, and ellipsoid (2.8-3.0 mm) in shape. Fruiting occurs in September and October in North America (Radford et al. 1968; Hitchcock 1971; Gleason & Cronquist 1991).

M. vimineum can be distinguished from other grasses by its thin, pale green, tapered leaf blades, and by its multiple spikelets that may be either terminal or arising from leaf axils. The alternate leaves have a silvery stripe of reflective hairs down the middle of the upper leaf surface. In the fall, identification becomes somewhat easier after the plant develops a slight purplish tinge (LaFleur 1996; Swearingen 2000).

While M. vimineum is an annual, there has been some confusion regarding whether M. vimineum also occurs as a rhizomatous, perennial (Ehrenfeld 1999; Mehrhoff 2000). According to Mehrhoff (2000), this confusion resulted when specimens of a native perennial, Leersia virginica, were incorrectly identified as M. vimineum. The annual M. vimineum can be distinguished L. virginica (which it frequently grows alongside) by the former's ciliate leaf sheath collars and paired spikelets (versus L. virginica's glabrous or

pubescent leaf sheaths and 1-flowered spikelets).

STEWARDSHIP SUMMARY

M. vimineum is an annual C4 grass native to Asia from India and Japan. It possesses characteristics typical of many invasive species: it grows quickly, fruits within a single season, produces abundant seed, and easily invades habitats that have been disturbed by natural (e.g., flood scouring) and anthropogenic (e.g., mowing, tilling) sources. M. vimineum was first discovered in the United States in 1919 (Fairbrothers & Gray 1972), and has since spread rapidly to all states east of the Mississippi, and south of and including Connecticut. M. vimineum is locally abundant, able to displace native wetland and forest understory vegetation with its dense, expanding monospecific patches. It is usually found under moderate to dense shade in moist conditions, but it does not persist in areas with periodic standing water, nor in full sunlight (Barden 1987, 1991). Once established, the removal of M. vimineum requires major eradication and restoration efforts (Bruce et al. 1995).

Manual or mechanical techniques may be the best method for controlling M. vimineum, since it is a shallowly-rooted annual. Hand pulling, however, is extremely labor-intensive, is feasible only for small infestations, and will need to be repeated and continued at least seven years to exhaust the seed supply in the seed bank (Virginia Native Plant Society 2000). Mowing or burning early in the season does not control the plant as the plants resprout and new seeds germinate. Following these treatments, plants can still set seed by the end of the season. Mowing may be an effective control method if carried out in late summer, when the plants are in peak bloom but before seed is produced (J. Ehrenfeld, pers. comm.). For extensive infestations, where mechanical methods are not practical, systemic herbicides such as imazameth (tradename Plateau) or glyphosate (tradename RoundUp, or Rodeo in wetland sites), or grass-specific herbicides like sethoxydim (tradenames Vantage or Poast) may be effective (Johnson 1997; Swearingen 2000). No biological controls are currently available for this plant.

RANGE

M. vimineum was introduced to North America from Asia, where it is native to India, Nepal, China, and Japan. It was first identified in the United States in 1919 in Tennessee, and by 1960 had spread (probably by hay and soil) to Ohio and Pennsylvania, and all Atlantic coastal states from Florida to New Jersey. It was widely used as a packing material for porcelain from China, and this was likely the means of its introduction into the U.S. M. vimineum occupies riparian habitats, lawns, woodland thickets, damp fields, and roadside ditches. Reported occurrences of M. vimineum in North America currently include: Alabama, Arkansas, Connecticut, Delaware, Florida, Georgia, Illinois, Indiana, Kentucky, Louisiana, Maryland, Mississippi, New Jersey, New York, North Carolina, Ohio, Pennsylvania, South Carolina, Tennessee, Texas, Virginia, West Virginia, and Puerto Rico (USDA-NRCS 1999).

IMPACTS AND THREATS POSED BY MICROSTEGIUM VIMINEUM

M. vimineum is capable of invading wildland areas and swiftly replacing natural communities with nearly monospecific stands. It is generally slow to invade undisturbed areas, but rapidly fills disturbed areas such as flood-scoured stream sides and sewer line rights-of-way that are mowed once a year. An individual plants of M. vimineum can produce up to 1000 seeds, and the seeds remain viable in the soil for three to five years. Once established, M. vimineum is able to crowd out native herbaceous vegetation in wetlands and forests within three to five years (Hunt 1992; Barden 1987).

M. vimineum is a C4 plant, and C4 plants are typically adapted to high temperatures and high light regimes. However, unlike most C4 plants, M. vimineum is adapted to low light levels and is able to grow and produce seed in only 5% full sunlight (Winter et al. 1982). Additionally, M. vimineum may be responsible

for altering natural soil conditions, creating an inhospitable environment for many native species. Kourtev et al. (1998) reported that in areas that have been invaded by M. vimineum, both litter and organic soil horizons were thinner than in uninvaded areas, and that the pH of soils in invaded sites was significantly higher than in uninvaded sites. There is no indication that M. vimineum produces allelopathic chemicals (Woods 1989).

Established populations of M. vimineum usurp quality nesting habitat from quail and other wildlife. In addition, it creates excellent habitat for rats, especially cotton rats (Sigmodon spp.), that often prey on the nests of native bobwhite quail and attract other predators as well (A. Houston, pers. comm.).

HABITAT

In North America, M. vimineum occurs in a variety of disturbed sites. It thrives in along mesic roadsides, ditches, woodland borders, floodplains, and streamsides (Fairbrothers & Gray1972; Hunt & Zaremba 1992). It can also be found in mesic upland sites, and is almost always found in moderate to dense shade (Redman 1995). It does not survive, however, in areas with periodic standing water, nor in areas with full sunlight.

BIOLOGY AND ECOLOGY

Light, Moisture, and Temperature

M. vimineum possesses characteristics typical of many invasive species: it grows quickly, fruits within a single season, produces abundant seed, and easily invades naturally (e.g., flood scouring) and artificially (e.g., mowing, tilling) disturbed habitats. Once established, the removal of M. vimineum requires major eradication and restoration efforts (Bruce et al. 1995).

M. vimineum is unusual in that although it is a C4 plant, it is adapted to low light conditions (Winter et al. 1982; Barden 1991). It can grow and produce seeds at as little as 5% full sunlight, but maximum growth and seed production occurs at 25-50% full sunlight (Winter et al. 1982; Horton & Neufeld 1998).

Most sites invaded by M. vimineum in the United States, have acidic soils (pH 5.8 to 4.8), but some populations are on soils derived from limestone or marble with surficial soil that is neutral or only slightly acidic in reaction. Soils on which M. vimineum occurs are typically average in levels of potassium and phosphorus, and high in nitrogen (Redman 1995). The overall acidity of the soils, however, may limit nutrient availability. Soils are usually moist, and are often well-drained silty loams, sandy loams, or loams. Clay was not a significant component of the upper soil horizons in any of the soils invaded by M. vimineum at sites studied by Hunt & Zaremba (1992).

No information was found regarding the optimal growing temperatures or the temperature limits of this species. The coldest winter temperature at which invasive populations of M. vimineum occur is approximately -21° to -23° C (Redman 1995).

Seed Dispersal

M. vimineum fruits and seeds disperse by water, animals, and by humans. (It was originally introduced as packing material or for basket-weaving.) The floating fruits of M. vimineum can disperse throughout an entire wetland or alluvial floodplain during high-water events (Woods 1989; Mehrhoff 2000). Even though M. vimineum does not exhibit any special adaptations for seed/fruit dispersal such as hooks or barbs, its seeds are small and often adhere to animal fur or clothing. Further, the fruits have been observed being transported on automobiles (Mehrhoff 2000).

M. vimineum relies entirely on its seed bank for its annual recruitment. Seeds of M. vimineum may need a period of stratification (cool temperatures and high moisture) before they will germinate (Woods 1989). M. vimineum seeds stored in the soil may remain viable as long as five years (Barden 1991). M. vimineum seeds may have low germination rates (Woods 1989), but many seeds are produced by each plant. Seeds of M. vimineum are also able to survive submersion in water for periods of up to 10 weeks. Barden (1991) reports that seeds can germinate while under water, but the plants do not grow. If standing water is removed, more seeds will germinate shortly afterwards.

ECONOMIC USES

In the early 1900s, M. vimineum was used extensively as a packing material for porcelain, especially fine China porcelain, which may have contributed to its invasion into the United States. Culms of this grass have also been used for basket weaving. M. vimineum has not been documented as being intentionally planted as an ornamental, for erosion control, or for forage.

MANAGEMENT

Potential for Restoration of Invaded Sites

Manual and mechanical, environmental/cultural, and chemical methods are all useful to varying degrees in controlling M. vimineum. Prescribed burns have not been successful in controlling this species so far, but fall burns may have the potential for partial control. M. vimineum produces a large number of viable seed that can remain in the soil seed bank for seven years or more. If controlled during the early stages of invasion, the potential for successful management is high. The potential for large-scale restoration of wildlands where M. vimineum has become established is probably moderate.

Manual and Mechanical Control

Hand pulling of M. vimineum is the preferred method of removal as it is highly specific and provides minimal impact (except trampling and soil disturbance) to the surrounding environment. Hand pulling is an effective method of control if it is thorough and timed correctly. It is, however, labor-intensive and time-consuming. Pulling late in the season (September-early November) before seed production reduces the unintentional spread of the current year's seeds. Pulling early in the season (before July), however, allows germination of new plants from the seed bank which will mature during the remaining season and produce seeds. In the northeast, August and late September are good times to pull plants by hand (LaFleur 1996). Yearly weeding is necessary because new plants can appear as a result of seed banking or re-infestation from new seed being dispersed into the area (G. Edinger, letter to J. Randall).

Mowing using a weed whacker (or a weed-eater) is an effective control method if carried out in late summer just before seeds are produced. Mowing at any other time is not useful as the plants have the ability to re-sprout and can produce seed heads in the axils of their lower leaves (Woods 1989; Barden 1991). Mowing can also be useful in reducing the amount of litter and plant biomass prior to herbicide application, making the herbicide more effective.

Grazing

Grazing is not a control option for M. vimineum since cattle, deer, and even goats avoid feeding on it (A. Houston, pers. comm.; Barden 1991).

Flooding

Flooding for more than three months, or intermittent flooding during the growing season, may be an effective control method for mature plants of M. vimineum. The seeds of M. vimineum, however, can survive periods of inundation of at least ten weeks (Barden 1991).

Prescribed Burning

Spring burns are ineffective at controlling M. vimineum because a new cohort of seeds will germinate soon after the burn. Burns in the late fall, however, may be useful in controlling this species (Barden 1991). Burning is also useful in reducing the amount of litter and plant biomass prior to herbicide applications.

Herbicides

For large infestations of M. vimineum, the use of herbicides may be the only viable option for good control. A series of control experiments using herbicides was carried out at the Ames Plantation (University of Tennessee), and the researchers reported that it is relatively easy to kill M. vimineum, but that managing for a desirable plant community is difficult. They found that the herbicide imazameth (tradename Plateau) was the herbicide of choice for controlling M. vimineum. This is because imazameth (applied at a rate of 6 ounces per acre) kills M. vimineum, but allows the development of (a.k.a., does not kill) the desirable native sedges, ragweeds, and legumes.

The grass-specific herbicide fluazifop-p (tradename Fusilade) applied at the rate of 1.2 liters per hectare (1 pint per acre) also controlled M. vimineum, but left a less desirable plant community. Glyphosate (tradename RoundUp) was also tested, but resulted in a complete kill of all plants, which could potentially lead to possible re-invasion by M. vimineum or other undesirable species. Barden (1991) also found glyphosate useful in killing M. vimineum. Formulations of glyphosate registered for use aquatic systems (Rodeo), has been effective for M. vimineum control in wetlands. Woods (1989) in Tennessee found that the grass-specific herbicide sethoxydim (tradenames Poast, Vantage), applied during late summer at rates of 1 pint per acre, also provided excellent (more than 95%) control of M. vimineum and released dicots from competition without injuring them. Pre-emergent herbicides such as diphenamid (tradename Enide) and benefin (tradename Balan) have also demonstrated excellent control of M. viminium seedlings under conditions of good herbicide-to-soil contact (Woods 1989), but do not encourage the germination of native species.

Allan Houston (pers. comm.) reports that if there is a heavy build-up of litter (dead plant material) in M. vimineum stands, burning the debris may first be necessary to get adequate herbicide coverage. He suggests applying herbicide when the plants reach a height of 5-10 centimeters (2-4 inches).

Biological Control

No biological controls are currently available for M. vimineum.

EXAMPLES OF MICROSTEGIUM VIMINEUM MANAGEMENT ON TNC PRESERVES

According to TNC's 1998 Weed Survey, M. vimineum has been reported from TNC preserves in New Jersey, North Carolina, Virginia, Georgia, Alabama, Arkansas, Maryland, and in Connecticut. Several preserves reported M. vimineum is one of their worst weed problems, but only a few had begun active control measures.

In Maryland, Donnelle Keech reported that burning is not effective in controlling M. vimineum. In North Carolina, Robert Merriam reported hand pulling was effective. Elizabeth Farnsworth in Connecticut, how-

ever, indicated that hand pulling may be effective in small populations, but seems futile for large populations since it is difficult to eliminate the seed sources. She added that it is important to attack small infestations as soon as possible, and to attack them vigorously!

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MONITORING

The distribution of M. vimineum should be monitored annually or biannually where there is a threat to protected species. Following all control treatments, further control efforts and monitoring is needed for at least seven years due to the viability of seeds in the seedbank or re-invasion from nearby propagule sources (Barden 1991).

Since M. vimineum usually occurs in dense, nearly monospecific stands, permanent line intercepts (or transects) across population borders are an easy technique for periodic monitoring of changes in M. vimineum distribution. Where it is less abundant, visual estimates of percent cover changes in permanent plots, or photographic documentation, carried out at the same (phenologic) time each year, may be for monitoring change over time. Additionally, new invasions of M. vimineum should be identified as soon as possible, since small populations are the easiest to eradicate.

Research Needs

The following research topics need attention: 1) What are the impacts of M. vimineum on native communities? 2) What are the mechanisms of M. vimineum invasion in a variety of landscapes? 3) Is biological control by inoculation with fungal pathogens a possible control technique? 4) Which species replace M. vimineum when control succeeds? And 5) What is the most effective method (for each specific area) of M. vimineum control, and how can this method encourage the regeneration of native species?

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Bamboo

Southeast Exotic Pest Plant Council Invasive Plant Manual
Common Name: Golden Bamboo, Fishpole Bamboo
Scientific Name: Phyllostachys aurea Carr. ex A.& C.

There have been over 750 bamboo plant introductions into the United States. Of these, twenty-four species and eleven cultivars have been in the Phyllostachys genus. Golden bamboo was introduced in Alabama in 1882. In China, these plants grow in deciduous and coniferous forests. It is used for paper pulp, handicrafts and as a food source in many countries throughout Asia. The name Phyllostachys comes from phyllon, meaning leaf, and stachys, meaning spike.

Height: Golden bamboo culms can reach a height of 8 to 10 meters. The basal internodes of this species are inflated, a distinguishing characteristic.

Leaves: The leaves are lanceolate; 1.5 dm long and 1 to 2 cm wide. The edges of the leaves may be rough or smooth without lobes.

Flowers: Golden bamboo flowers infrequently and may not flower for several decades. Spikelets are solitary and 8 to 12 flowered.

Life History
P. aurea is a monopodial bamboo and primarily spreads by rhizomes. Shoots develop in the spring with initition primarily controlled by temperature. The culms grow from side shoots at alternate nodes of the rhizome. Flowering is infrequent and in many cases will preclude the death of the plant.

Origin and Distribution
Golden bamboo is native to China but has been cultivated in Japan for centuries. It was introduced to the United States in 1882 in Alabama. Since that time it has spread or been introduced to the Southeastern U.S. from Maryland to Florida, Louisiana to Arkansas and Oregon.

Similar Species
One possible look-alike to P. aurea is Arundinaria gigantea or Cane, which is native throughout the Southeastern United States. These two species can be distinguished by P. aurea having one side of the stem flattened. The cross section of A. gigantea is more or less round.

Habitat
Golden bamboo thrives in full sun in all but the hottest climates where it requires some shade. It will grow in sparsely wooded secondary forests. Vigorous growth and spread is seen in moist, deep loamy soils. In habitats less than ideal, P. aurea will continue to grow and spread although at a diminished rate.

Management Recommendations

Mechanical Control
Cutting/Mowing: This method can be used on small infestations or where herbicides cannot be used. Cut plants as close to the ground as possible. Repeat several times throughout the growing season as plants resprout. Monitoring and re-treatment will be necessary for several growing seasons until the energy reserves in the rhizomes are exhausted.

Herbicidal Control
Foliar Spray Method: This method should be considered for large areas of bamboo where risk to non-target species is minimal. Air temperature should be above 65°F to ensure absorption of herbicides.

Glyphosate: Apply a 2% solution of glyphosate and water plus a 0.5% non-ionic surfactant to thoroughly wet all leaves. Use a low pressure and coarse spray pattern to reduce spray drift damage to non-target species. Glyphosate is a non-selective systemic herbicide that may kill non-target, partially sprayed plants.

Cut Stump Method: This control method should be considered when treating individual trees or where the presence of desirable species precludes foliar application. Stump treatments can be used as long as the ground is not frozen.

Glyphosate: Horizontally cut stems at or near ground level. Immediately apply a 25% solution of glyphosate and water to the cut stump, covering the outer 20% of the stump.

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JAPANESE KNOTWEED

ELEMENT STEWARDSHIP ABSTRACT
for Polygonum cuspidatum
Japanese knotweed, Mexican bamboo
To the User:

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The Nature Conservancy
Element Stewardship Abstract
For Polygonum cuspidatum Sieb. & Zucc.
(Reynoutria japonica Houtt.)
(Fallopia baldschuanica)

I. IDENTIFIERS

Common Name: Japanese knotweed, Mexican bamboo

General Description:
Polygonum cuspidatum was classified as Reynoutria japonica by Houttuyn in 1777 and as Polygonum cuspidatum by Siebold in 1846. It was not until the early part of the 20th century that these were discovered to be the same plant (Bailey, 1990). The plant is generally referred to as Polygonum cuspidatum by Japanese and American authors and as Reynoutria japonica in the United Kingdom and Europe. Recent evidence suggests that this plant should be reclassified as Fallopia japonica (Bailey, 1990).

The following is based on descriptions by Fernald (1950), Muenscher (1955), Locandro (1973,1978), and Ohwi (1965).

Polygonum cuspidatum is an herbaceous perennial which forms large clumps 1-3 meters high. It is fully dioecious and can reproduce by seed and by large rhizomes which may reach a length of 5-6 meters. The stout stems are hollow and bamboo-like, extend from an erect base and are simple or little branched and glabrous with thinly membranous sheaths. Leaves are broadly ovate, truncate to cuneate at base, abruptly cuspidate, 5-15 cm long, 5-12 cm broad, with petioles 1-3 cm long. Greenish white flowers 2.5-3 mm long, densely arranged in axillary panicles; 3 styles; 8-10 stamens with longitudinally dehiscing anthers. Fruiting calyx wing-angled, 6-10 mm long. Achenes shiny black-brown, 3-4 mm long, acutely trigonous. Male flowers have branched panicles on upright racemes with the distal end of the raceme in the highest position; individual panicles generally point up. Female flowers are drooping or decumbent with the distal end in the lowest position; individual panicles are not oriented in a particular direction. Both male and female flowers possess vestigial organs of the other sex.

Polygonum cuspidatum closely resembles Polygonum sachalinense (Reynoutria sachalinensis), an exotic species native to northern Japan and the Sakhalin Islands (Ohwi, 1965). P. sachalinense can be distinguished primarily by its larger size, greenish flowers and cordate leaves which gradually taper to the tip (Fernald, 1950). Polygonum cuspidatum is known to hybridize with Polygonum sachalinense and with Fallopia baldschuanica (Bailey, 1985,1988). Hybrids between Polygonum cuspidatum and Polygonum sachalinense have been frequently mistaken for Polygonum cuspidatum in the U.K. (Bailey, 1990).

II. STEWARDSHIP SUMMARY

Polygonum cuspidatum is widely distributed in much of the eastern U.S. In western Pennsylvania it already occupies hundreds of acres of wetlands, streambanks and hillsides and has spread along the banks of the Allegheny and Ohio Rivers and dominates the edges of many of the islands in these rivers (Wiegman, pers.comm.).It is present on at least two sites belonging to the Pennsylvania Chapter of The Nature Conservancy (Long Pond in the Poconos and Bristol Marsh, an urban preserve near Philadelphia)and has become a problem on creeks in suburban Philadelphia (Broaddus, pers. comm.). It is also a serious problem in Rock Creek Park, a national park in Washington D.C. In the U.K., it is considered a major weed and a threat to conservation, and it is legally prohibited to introduce Japanese knotweed into the wild (Beerling, 1990; Nature Conservancy Council, 1989; Palmer, 1990). Its early emergence and great height combine to shade out other vegetation and prohibit regeneration of other species (Sukopp and Sukopp, 1988). Thus it reduces species diversity and damages wildlife habitat (Palmer, 1990; Scott and Mars, 1984; Wiegman, pers. comm.). It does not appear to be a threat in undisturbed forest and other low light areas, but it is likely that, if unchecked, it will continue to expand its range in open habitats. Once Polygonum cuspidatum has established it forms large, almost pure stands which are extremely persistent and difficult to eradicate.

III. NATURAL HISTORY

Habitat:
Polygonum cuspidatum is native to eastern Asia. It was introduced from Japan to the United Kingdom as an ornamental in 1825, and from there to North America in the late nineteenth century (Conolly, 1977; Patterson, 1976; Pridham and Bing, 1975). In Japan, Polygonum cuspidatum is widely distributed and is usually found in sunny places on hills and high mountains (Kanai, 1983; Ohwi, 1965). It is a dominant pioneeer in the primary succession of volcanic slopes and is frequently a colonizer in secondary succession (Hirose, 1984).

In the U.K., Polygonum cuspidatum has spread extensively, occurring in virtually every 10 km square (Nature Conservancy Council, 1989). Stands range in size from individual plants to clumps more than 500 square meters (Palmer, 1990). Polygonum cuspidatum has also become naturalized in much of central Europe (Sukopp and Sukopp, 1988). In North America, it is widely found in the eastern U.S. and has been observed as far north as Nova Scotia and New Foundland, as far south as North Carolina, in much of the midwest and in the coastal areas of Washington and Oregon (Locandro, 1978; Patterson, 1976; Pauly, 1986). Large stands have been noted in western Pennsylvania, in particular along the banks of the Ohio and Allegheny Rivers and on the islands in these rivers (Wiegman, pers. comm.). Polygonum cuspidatum spreads primarily along river banks, but also grows in wetlands, waste places, along roadways, and in other disturbed areas (Beerling, 1990; Conolly, 1977; Muenscher, 1950).

Polygonum cuspidatum can thrive in a wide variety of habitats. In Japan, it can grow on volcanic soils high in sulfur and having a pH less than 4 (Conolly, 1977). In the U.S., it has been observed growing in a variety of soil types, including silt, loam, and sand, and in soils with pH ranging from 4.5 to 7.4 (Locandro, 1973). Analyses of soils from 17 stands in Wales showed no correlation between stand size and vigor and soil chracteristics. The stands studied grew in soils with a broad range of pH, organic matter and nutrients (Palmer, 1990). In Japan, Polygonum cuspidatum growth is slow, but steady in nutrient poor sites, and rapid in nutrient rich sites (Hirose, 1984). In areas where Polygonum cuspidatum has been introduced, it is found primarily in moist, unshaded habitats. Distribution maps from the U.K. show that it is generally associated with regions of high precipitation (Conolly, 1977). Locandro (1973) reports it growing on xeric as well as hydric sites in the U.S. Its distribution appears to be limited by light. It is found primarily in open sites, and its growth and abundance are depressed in shady sites (Beerling, 1991; Seiger, unpublished data).

Polygonum cuspidatum flowers from July to October in Japan (Ohwi, 1965) and in August and September in the U.K. and North America (Conolly, 1977; Fernald, 1950; Muenscher, 1955). It is pollinated by bees and other insects (Bailey, 1990; Locandro, 1978). Seeds appear about two weeks after flowering (personal observation) and are wind dispersed (Maruta, 1976). In Japan, reproduction in Polygonum cuspidatum is characterized by high seed production and low seedling survival, but plants have a very high probability of survival once established (Hirose and Tateno, 1984). However, in the U.S., U.K, and Europe, seeds do not appear to be a significant mode of reproduction. In a study of the reproductive ecology of Polygonum cuspidatum populations in New Jersey, Locandro (1973) found viable pollen, but noted that fertile males were rare. Seedling germination was observed in the presence of males, but no seedling survival was recorded during five years of observation. In the absence of males, females produced empty achenes. In the U.K., fruit set is very rare. This was originally attributed to the rarity of male fertile flowers (Conolly, 1977). It has since been found that there are no male fertile forms of Polygonum cuspidatum in England and that the pollen source is actually a hybrid between Polygonum cuspidatum and Polygonum sachalinense (Bailey, 1990). Polygonum cuspidatum also hybridizes with Fallopia baldschuanica (Bailey, 1985,1988,1990). In the U.S., hybrids morphologically similar to those between Polygonum cuspidatum and Fallopia baldschuanica have been grown from seeds collected in the field, but seedling establishment has not been observed in the wild (Seiger, unpublished data).

The primary mode of reproduction in the U.S., U.K. and Europe is through extensive rhizomes which can reach 15-20 meters in length (Conolly, 1977; Locandro, 1973). Dispersal can occur naturally when rhizome fragments are washed downstream by the current and deposited on banks or, as more commonly occurs,

when soil is transported by humans as fill dirt (Conolly, 1977; Locandro, 1978). Rhizomes can regenerate from small fragments, and have even been observed to regenerate from internode tissue (Locandro, 1973). Rhizomes can regenerate when buried up to 1 meter deep and have been observed growing through two inches of asphalt (Locandro, 1978; Pridham and Bing, 1975). The ability of rhizomes to generate shoots was found to be affected by the source of rhizome fragments, fragment size and depth planted, the optimal depth being just below the surface (Locandro, 1973).

Polygonum cuspidatum requires high light environments and effectively competes for light in such environments by emerging early in the spring and using its extensive rhizomatous reserves to quickly attain a height of 2-3 meters. Shoots generally begin to emerge in April and growth rates exceeding 8 cm per day have been recorded (Locandro, 1973). In addition, its deep root system gives it an advantage in foraging for nutrients and water, and contributes to soil stabilization on disturbed sites (Hirose and Tateno, 1984; Nakamura, 1984). Hirose and Tateno (1984) found that organic nitrogen levels on Mt. Fuji increased following colonization by Polygonum cuspidatum on bare volcanic desert and concluded that Polygonum cuspidatum contributes to the development of the ecosystem, in part, by acting as a nutrient reservoir.

Polygonum cuspidatum is found on open sites and does not appear to be able to invade forest understory due to its high light requirements (Beerling, 1991). Studies of the very closely related Polygonum sachalinense indicated that Polygonum sachalinense plants grown in low light did not have higher photosynthetic rates than plants grown in high light, and thus would not be expected to adapt to sites with low light intensity (Patterson, Longstreth and Peet, 1977). Transplant studies of Polygonum cuspidatum in closed understory sites showed poor survival and growth compared to open bank sites, confirming that it was environmental factors and not limitations on dispersal which exclude Polygonum cuspidatum from understory sites (Seiger and Merchant, 1991). Follow-up studies in the greenhouse showed that Polygonum cuspidatum grown under light levels comparable to those foundin the understory had significantly less rhizomatous reserves at the end of the season than did plants grown under full sunlight (Seiger, unpublished data).

Polygonum cuspidatum occurs in much of the temperate U.S. Though not yet a major weed in the U.S., it is spreading, particularly in the eastern states. Dispersal is limited to areas where rhizome fragments from existing stands are washed downstream or soil containing rhizomes is transported by humans. Once established, it forms large, monospecific stands which displace virtually all other vegetation. Establishment can be prevented by monitoring for the introduction of Polygonum cuspidatum and manually removing the entire plant. Small stands may be controlled by repeated cutting, which may need to be supplemented by revegetation once growth of Polygonum cuspidatum has been reduced. At present, the only method to control large stands is with repeated application of herbicides. Complete eradication may not be possible.

IV. CONDITION

V. MANAGEMENT/MONITORING

Management Requirements:
Current control methods (mechanical, herbicidal) require continued treatment to prevent reestablishment of Polygonum cuspidatum. It may be feasible to reintroduce competitors as an alternative to continued treatment. There is a need for more research on whether native species might serve effectively as competitors and methods of reintroduction. Only very preliminary work has been done towards developing a biological control for Polygonum cuspidatum and much research remains to be done (see below).

The International Institute for Biological Control in conjunction with the National Agricultural Research center in Japan has produced a partial list of insect herbivores in Japan which are associated with Polygonum cuspidatum. A number of pathogens have been collected from Japan by CAB International and are held at the International Mycological Institute (Fowler and Schroeder, 1990). Plans are under way to begin a study of Polygonum cuspidatum in its native habitat (Fowler, pers. comm.). Contact:

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Recent studies in Rock Creek park, Washington, D.C. indicate potential for control by mechanical means combined with revegetation (Seiger, unpubished data). Field tests will be conducted in 1992. Contact:

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In areas where Polygonum cuspidatum has not yet become established, the focus of management should be to prevent establishment by monitoring areas for introductions of Polygonum cuspidatum and eradicating newly established stands before they can become established.

Manual control consists of digging out the rhizomes or cutting the stalks. Digging is extremely labor intensive and tends to spread the rhizome fragments and promote disturbance and is not recommended (Palmer, 1990). Cutting, on the other hand, may be quite effective in eliminating Polygonum cuspidatum. It has been observed that Polygonum cuspidatum does not establish where grazing pressure is high (Beerling, 1990; Palmer, 1990). In a review of control methods, Palmer (1990) noted that eradication is not complete with cutting alone, but has been nearly achieved in some cases and should be feasible with persistence. A number of authors claim that cutting is ineffective (Pauly, 1986; Pridham and Bing, 1975; Orchowski, 1991). These conclusions are based on the observation that shoots are regenerated following cutting. However, a greenhouse study found that cutting stems results in a significant reduction of rhizomatous reserves. The same study also found that cutting was equally effective at any time during the growing season prior to the beginng of senescence (Seiger and Merchant, 1990). A study of the effects of repeated cutting showed that at least three cuts are needed in a growing season to offset rhizome production (Seiger, unpublished data). Manual control can be labour intensive, but where populations are small and isolated, may be the best option. No research has been done to test the effectiveness of burning. It may act similarly to cutting by removing above ground material.

Shading, particularly in conjunction with cutting, may be another useful means of control on smaller stands. Studies showing that P. cuspidatum requires high light environments suggest that covering stands with black plastic or shade cloth may reduce growth. Pridham and Bing (1975) state that applying several layers of black polyethylene film covered by asphalt, blocks or stones to a leveled soil surface may provide some control. However, they also note that P. cuspidatum is able to emerge through asphalt. If shade cloth (or plastic) is to be applied without cutting, then, to prevent P. cuspidatum from emerging through the covering, shade cloth should be placed over shoots after the plants have reached their full height or placed well above newly emerging shoots, or raised as plants grow.

A number of biocidal chemicals have been found to be effective against Polygonum cuspidatum. Most of these are undesirable for use in conservation areas because they are nonselective, may be persistent in the soil and/or are not safe for use near water. One frequently used way to minimize the effects of non-selective herbicides on non-target species is to paint herbicides directly onto the target plants (Broaddus, pers. comm.). In the case of P. cuspidatum, this would probably require prior cutting for easier access if herbi-

cides are to be applied after the plants have reached their full height. Herbicides appear to be more effective when combined with cutting (Scott and Mars, 1984; Orchowski, 1991).
Glyphosate [N-(phophonomethyl)glycine] has been found to be very effective against Polygonum cuspidatum (Ahrens, 1975; Beerling, 1990; Pauly, 1986). Glyphosate is a nonselective systemic herbicide with a short residual life (Ahrens, 1975; Lynn, Rogers and Graham,1979). Application is more effective in the fall when leaves are translocating to rhizomes (Lynn, et al, 1979). The British Nature Conservancy Council (1989) recommends applying 2.0 kg/ha in August with a prior cut in late spring or early summer. Glyphosate is available from Monsanto under the trade names RoundupTM and RodeoTM. Only Rodeo has been approved for use near water (Bender, 1988). Glyphosate has been used with limited success on some nature reserves in the U.K. (Palmer, 1990). Repeated applications over several years may be necessary (Beerling, 1990; Palmer, 1990; Pauly, 1986).

The Nature Conservancy Council (1989) also recommends picloram to be applied at a rate of 2.6 kg/ha in the spring. Picloram is a selective herbicide which is persistent in the soil. It must not be used near water, thus excluding its use in many of the areas where Polygonum cuspidatum is a problem (Gritten, 1990; Scott and Mars, 1984). Dicamba (3,6-dichloro-o-anisic acid) has also been found to be effective against Polygonum cuspidatum, but is persistent in the soil and nonselective (Pridham and Bing, 1975). A number of other herbicides have been tested against Polygonum cuspidatum, both alone and in combination with other herbicides (Orchowski, 1991; Scott and Mars, 1984). Herbicide may have to be used on stands that have been allowed to attain a large size. However, their use is not recommended in nature reserves because of their undesirable effects on other biota and the need for repeated applications to maintain control of Polygonum cuspidatum.

Regardless of whether control is manual or chemical, as long as some rhizomes remain in the soil Polygonum cuspidatum returns once management is relaxed (Beerling, 1990; Nature Conservancy Council, 1989; Palmer, 1990). It has been suggested that the reintroduction of effective competition might be possible (Eaton, 1986).

Research has only recently begun on biological control. The herbivores and pathogens of Polygonum cuspidatum in Wales have been examined for their potential as control agents (Fowler and Schroeder,1990). A program is underway at the International Institute of Biological Control to identify biological control agents (Fowler, pers. comm.). The genetic uniformity of Polygonum cuspidatum makes it a good candidate for biological control (Bailey, 1990). Biological control may be necessary where Polygonum cuspidatum has taken over vast areas as it has done in the U.K., but it may be years before a successful control agent can be found.

The following individuals are familiar with Polygonum cuspidatum and its control in natural areas:

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VI. RESEARCH

Management Research Programs:
It is extremely difficult, if not impossible to eradicate large established stands of Polygonum cuspidatum. However, establishment can be prevented fairly easily by removing Polygonum cuspidatum before it becomes firmly entrenched. Areas known to be near established stands of Polygonum cuspidatum, particularly those downstream from such stands, should be monitored for the introduction of Polygonum cuspidatum.

The following individuals have direct experience monitoriing Polygonum cuspidatum:
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VII. ADDITIONAL TOPICS

VIII. INFORMATION SOURCES

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KUDZU

FACT SHEET: KUDZU

Kudzu

Pueraria montana var. lobata (Willd.) Maesen & S. Almeida

Pea family (Fabaceae)

NATIVE RANGE

Asia

DESCRIPTION

Kudzu ia a climbing, semi-woody, perennial vine in the pea family. Deciduous leaves are alternate and compound, with three broad leaflets up to 4 inches across. Leaflets may be entire or deeply 2-3 lobed with hairy margins. Individual flowers, about ½ inch long, are purple, highly fragrant and borne in long hanging clusters. Flowering occurs in late summer and is soon followed by production of brown, hairy, flattened, seed pods, each of which contains three to ten hard seeds.

ECOLOGICAL THREAT

Kudzu kills or degrades other plants by smothering them under a solid blanket of leaves, by girdling woody stems and tree trunks, and by breaking branches or uprooting entire trees and shrubs through the sheer force of its weight. Once established, Kudzu plants grow rapidly, extending as much as 60 feet per season at a rate of about one foot per day. This vigorous vine may extend 32-100 feet in length, with stems ½-4 inches in diameter. Kudzu roots are fleshy, with massive tap roots 7 inches or more in diameter, 6 feet or more in length, and weighing as much as 400 pounds. As many as thirty vines may grow from a single root crown.

DISTRIBUTION IN THE UNITED STATES

Kudzu is common throughout most of the southeastern U.S. and has been found as far north as Pennsylvania.

HABITAT IN THE UNITED STATES

Kudzu grows well under a wide range of conditions and in most soil types. Preferred habitats are forest edges, abandoned fields, roadsides, and disturbed areas, where sunlight is abundant. Kudzu grows best where winters

are mild, summer temperatures are above 80 degrees Farenheit, and annual rainfall is 40 inches or more.

BIOLOGY & SPREAD

The spread of kudzu in the U.S. is currently limited to vegetative expansion by runners and rhizomes and by vines that root at the nodes to form new plants. Kudzu also spreads somewhat through seeds, which are contained in pods, and which mature in the fall. However, only one or two viable seeds are produced per cluster of pods and these hard-coated seeds may not germinate for several years.

BACKGROUND

Kudzu was introduced into the U.S. in 1876 at the Philadelphia Centennial Exposition, where it was promoted as a forage crop and an ornamental plant. From 1935 to the mid-1950s, farmers in the south were encouraged to plant kudzu to reduce soil erosion, and Franklin D. Roosevelt's Civilian Conservation Corps planted it widely for many years. Kudzu was recognized as a pest weed by the U.S. Department of Agriculture and, in 1953, was removed from its list of permissible cover plants.

MANAGEMENT OPTIONS

For successful long term control of kudzu, the extensive root system must be destroyed. Any remaining root crowns can lead to reinfestation of an area. Mechanical methods involve cutting vines just above ground level and destroying all cut material. Close mowing every month for two growing seasons or repeated cultivation may be effective. Cut kudzu can be fed to livestock, burned or enclosed in plastic bags and sent to a landfill. If

conducted in the spring, cutting must be repeated as regrowth appears to exhaust the plant's stored carbohydrate reserves. Late season cutting should be followed up with immediate application of a systemic herbicide (e.g., glyphosate) to cut stems, to encourage transport of the herbicide into the root system. Repeated applications of several soil-active herbicides have been used effectively on large infestations in forestry situations.

Biological Efforts are being organized by the U.S. Forest Service to begin a search for biological control

agents for kudzu.
USE PESTICIDES WISELY: Always read the entire pesticide label carefully, follow all mixing and application instructions and wear all
recommended personal protective gear and clothing. Contact your state department of agriculture for any additional pesticide use requirements, restrictions or recommendations.
NOTICE: mention of pesticide products on this page does not constitute endorsement of any material.
CONTACTS
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SUGGESTED ALTERNATIVE PLANTS
Native vines such as trumpet creeper (Campsis radicans), pipevine (Aristolochia macrophylla), passionflower (Passifloralutea), trumpet honeysuckle (Lonicera sempervirens), and native bittersweet (Celastrus scandens) have attractive flowers and fruits, provide food for wildlife and make excellent substitutes for kudzu. These plants should be used in landscaping and for land restoration where they are known to occur as natives.
OTHER LINKS
• http://www.invasive.org/search/action.cfm?q=Pueraria%20montana
• http://www.lib.uconn.edu/webapps/ipane/browsing.cfm?descriptionid=23
• http://www.hear.org/starr/hiplants/images/thumbnails/html/pueraria_montana_var_lobata.htm
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Department of Conservation and Recreation, Richmond, Virginia.

PRINCESSTREE

Weed Notes: Paulownia tomentosa
Princess tree, empress tree, royal paulownia
The Nature Conservancy
Wildland Invasive Species Team

Summary: Paulownia tomentosa can be controlled most effectively using an integrated management approach. Cutting or girdling trees with power or manual saws are effective at preventing seed production, but repeated cutting or a herbicide treatment is necessary following cutting since Paulownia readily resprouts.

For descriptions of each of the following methods or for herbicide information, see TNC's Weed Control Methods Handbook at: http://tncweeds.ucdavis.edu/handbook.html

Manual & Mechanical Control
Hand pulling: Young seedlings of Paulownia can be successfully controlled by manual removal. Pulling is easiest following a rain event, as the soil becomes loose. It is important to remove all root fragments as Paulownia can resprout from root fragments.

Cutting and Girdling: Cutting of Paulownia trees is most effective at the onset of flowering. Cutting at ground-level can prevent seed production for that year, but cutting alone must be repeated for several years to successfully kill the tree. Girdling results in top-kill of that stem, but may also induce increased resprouts. Cutting and girdling followed immediately by a herbicide application (cut-stump, hack-and-squirt) has good rates of efficacy.

Chemical Control
Paulownia can successfully be controlled by herbicides. The most common herbicides used to control Paulownia include glyphosate (tradenames RoundUp®, Rodeo®) and triclopyr (Garlon 3A® or Garlon 4®).

Foliar Spray: Paulownia seedlings can be controlled by using a foliar spray of either glyphosate (2% solution with 0.5% nonionic surfactant) or triclopyr (2% solution with 0.5% nonionic surfactant). These spray herbicides should be applied directly to the leaves and sprayed-to-wet.

Cut-Stump: Following cutting of the trees at ground-level, glyphosate (25% solution) or triclopyr (50% solution) should be directly applied to the stump. The cut-stump method can be used at all times of the year, as long as the ground is not frozen.

Girdling or Hack-and-Squirt: Immediately following girdling (approximately 15 cm above the ground and the cut should be well into or below the cambium layer) or hacking, directly apply glyphosate (25% solution) or triclopyr (50% solution) into the cut area.

Basal Bark: Apply a mixture of 25% triclopyr and 75% horticultural oil to the basal parts of the tree to a height of 30-38 cm (12-15 in) from the ground. Thorough wetting is necessary for good control, and spray until run-off is noticeable at the ground line. Basal bark applications should not be applied when the ground is frozen.

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The Nature Conservancy's Wildland Invasive Species Team
Written: December 2002

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Multiflora Rose

ELEMENT STEWARDSHIP ABSTRACT
for Rosa multiflora
Rambler Rose, Multiflowered Rose

To the User:

Element Stewardship Abstracts (ESAs) are prepared to provide The Nature Conservancy's Stewardship staff and other land managers with current management-related information on those species and communities that are most important to protect, or most important to control. The abstracts organize and summarize data from numerous sources including literature and researchers and managers actively working with the species or community.

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Authors of this Abstract:
Nancy Eckardt

THE NATURE CONSERVANCY
1815 North Lynn Street, Arlington, Virginia 22209 (703) 841 5300

I. IDENTIFIERS

Common Name: RAMBLER ROSE Global Rank: G?

General Description:
Rosa multiflora is a perennial shrub with compound leaves and white to pinkish white flowers.

Diagnostic Characteristics:
Rosa multiflora may be told from R. setigera, which it resembles, by a more trailing or arching habit, mostly 7 or 9 leaflets, 2-4 cm long, abundant, mostly white flowers in a pyramidal inflorescence, a glabrous style, and smaller fruit (Fernald 1950).

II. STEWARDSHIP SUMMARY

Rosa multiflora is a concern on several preserve lands, notably in New Jersey and Indiana. It is considered a serious problem on old fields and agricultural land in many southeastern states. Monitoring should be conducted on preserves where it presents a potential problem, followed by active management if necessary. The most effective means of eradication seem to be cutting followed by herbicide application. Glyphosate is commonly used and can be effectively applied in a 1% V/V solution, or 0.5% V/V solution if a surfactant is added, applied directly to the plants, cut branches, or stumps. Spring applications should show increasing control over the season with complete residual control the following spring. Repeat applications may be necessary in subsequent years to prevent recurrences.

III. NATURAL HISTORY

Range:
Rosa multiflora is a common pasture weed in the northeastern and midwestern United States. It was originally introduced to the East Coast from Japan in 1886 as an understock for ornamental roses (Wyman 1949). It is no longer used among horticulturalists and is not available from nurseries (Doudrick 1987).

The present range of multiflora rose is throughout the U.S., with the exception of the Rocky Mountains, the Southeastern Coastal Plains, and the Nevada and California desert areas, although the plant does less well in the northern tier of states (Fawcett 1980).

Habitat:
Rosa multiflora grows best on deep, fertile, well-drained but moist uplands or bottomlands, but is capable of enduring a wide range of edaphic and environmental conditions (Wyman 1949, Steavenson 1946). Steavenson (1946) reported successful plantings even on the eroded clay pans of central Missouri and southern Illinois. Schery (1977) reported that multiflora rose endures shade or sun and damp or dry environments, but does not grow well in standing water.

Reproduction:
Rosa multiflora reproduces by seeds and by rooting at the tips of its drooping canes (Albaugh et al. 1977). Flowering begins in May, and the fruits develop in mid to late summer. The rose hips do not split apart to release the seed, but dry gradually to form a leathery capsule too dense to be wind-carried. The fruits are highly sought after by birds, especially the Cedar waxwing and American Robin (Scott 1965, Albaugh 1977, Barbour and Meade 1980). Birds are responsible for spreading the seeds, and as Schery (1977) noted, rose seedlings are often found under bird perch sites. Wyman (1940) observed better germination of seeds after scarification by passing through the digestive tract of birds. Uneaten rose hips remain on the plant until the following spring (Fawcett 1980) and the seeds remain viable for a number of years (Wyman 1949).

The seeds germinate readily following deposition in the soil. Steavenson (1946) recommended cold strati-

fication from Feb. 1 to April to people planting multiflora rose. Seedlings appear within 60 days at soil temperatures above freezing (Steavenson 1946). Seedlings are generally inconspicuous the first one or two years due to their low growth habit (Schery 1977).

Impacts:
In the 1930's, the U.S. Soil Conservation Service advocated the use of multiflora rose for soil erosion projects and as a "living fence" to confine livestock (Albaugh et al. 1977). Experimental plantings were conducted in Missouri and Illinois (Steavenson 1946), and as recently as the late 1960's state conservation departments in many states were giving away rooted cuttings to property owners (Schery 1977). Hedges of multiflora rose have also been used as a crash barrier and to reduce headlight glare in the medians of highways (Schery 1977, Hipkins et al. 1980). The plant is extremely prolific, however, and successfully invades pastures and other unplowed lands, crowding out existing vegetation and creating dense, impenetrable thickets. In some areas entire pastures have been taken over (Barbour and Meade 1980, Doudrick 1987). Cattle are often reluctant to enter fields dominated by multiflora rose (Fawcett 1980), and it has also been shown that rose hedges lower the crop yields on adjacent fields by competing for nutrients (Labisky and Anderson 1965).

IV. CONDITION

V. MANAGEMENT/MONITORING

Preserve Selection & Design Considerations:
Active control of multiflora rose is necessary mainly on agricultural land when it threatens to dominate pastures. It may also require management on preserve lands if found in old recovering pastures, as it can crowd out desirable grasses and other species.

Management Requirements:
Mechanical Control: Repeated mowing will control the spread of multiflora rose, particularly where the grass cover is dense (Scott 1965, Fawcett 1980). Fawcett (1980) stated that mowing several times a year would prevent multiflora rose seedlings from becoming established. At the Woodborne Santurary in Pennsylvania, annual mowing in July helped control the spread of multiflora rose, but did not eradicate it (Stone 1982). Mowing can be difficult due to terrain, when the hedges become established in wooded and brushy pastures. It is also difficult, if not impossible, to mow when the individual clumps reach their mature size, which may exceed 10 ft. in height by 20 ft. in diameter (Doudrick 1987).

Hand cutting of established clumps is difficult and time consuming. Fawcett (1980) recommended use of a bulldozer to knock down large rose clumps but cautioned that further control would be necessary due to resprouting and because seeds will be spread and germinate readily on the disturbed soil. At Woodborne, a large hedge cutter was used to top cut ten foot high rose clumps. Following this, annual mowing has prevented the re-establishment of large clumps and kept the field open (Stone 1982, Davison 1987).

Burning: Burning has not, apparently, been tried for multiflora rose. However, it has been tested in southeastern Texas as a management practice for Mccartney rose (Rosa bracteata), another exotic pasture species in the southern U.S. Gordon and Scifres (1977) tested head fires at 2 to 3 month intervals starting in February, 1975. Fire intensity and fuel components varied seasonally; however, regardless of the date of the burning, topkill of Mccartney rose was greater than 90%. Regrowth was initiated within two weeks after burns, again, regardless of the date of the burning. The average cane elongation was about 4 cm per month and canopy cover replacement averaged 10 to 15% per month following burning. Burning in winter effectively reduced the rose canopy for short-term gains in brush control, and allowed native grasses to take advantage of the entire spring growing period. There were higher herbage yields following winter burns than other seasons. Scifres (1982) believes that multiflora rose response to burning would be similar.

Prescribed burning in combination with herbicides has also been evaluated for Mccartney rose in southeast Texas. Scifres (1975) found that mechanical methods such as raking and stacking were effective for initial removal of mature, dense and ungrazed stands of the rose, allowing access for subsequent treatment. Prescribed burning removes the debris that remained after spraying and should reduce live Mccartney rose top growth by 75%. Periodic burning or respraying is probably necessary to prevent re-invasion of the rose (Scifres 1975).

Biological Control: The European Rose chalicid, Megastigmus aculeatus Swederus (Hymenoptera:Torymida), and rose rossette disease are potential biological control agents for multiflora rose.

M. aculeatus is a phytophagous wasp. The life cycle and distribution in North America has been summarized by Milliron (1949) and Balduf (1959). The adults are minute, weak flyers of limited lifespan. In May and June the long terebras of the female ovipositor pierces the still soft achene and deposits one egg in the soft, pulpy seed. The larvae subsequently develop during June and July, and consume the entire contents of the seed. After full growth in mid to late summer, the larvae undergo a long diapause and overwinter inside the now seedless achene. Pupation occurs in late April to June, and the adult emerges from the rose hip in early summer to renew the cycle. Populations are heavily female in number, suggesting that the majority of reproduction is parthenogenetic (Milliron 1949, Balduf 1959).

It is important to note that M. aculeatus adults are limited fliers, and do not appear to disseminate even locally through their own powers of flight (Balduf 1959). Their spread is dependent upon the use of rose seed, which explains the presence of these insects in nurseries on the East coast, where imported rose seed was used to start root stocks. Subsequent plantings, however, were done vegetatively, far from the nurseries where the plants were grown. It is possible that some of the large-scale plantings of multiflora rose throughout the Midwest are isolated from their chalicid limiting agent (Scott 19865). If true, this suggests that local reintroductions of M. aculeatus could be an effective control method for multiflora rose.

The rose rosette disease is another potential biocontrol agent for R. multiflora. Characteristic symptoms of the disease include abnormal floral development, a “witches broom” effect, and reddening of leaves and shoots (Doudrick et al. 1986). It was originally reported on wild native roses in the northwestern United States and Canada (Thomas and Scott 1953), and first showed up on multiflora rose at a Nebraska nursery in 1964 (Doudrick 1987). By the 1980’s, rose rosette was widespread on multiflora rose in Kansas and Missouri (Crowe 1963), and the rose industry became concerned about the spread of the disease to ornamental roses. It is apparently spreading eastward and was first reported east of the Mississippi in southern Indiana and northern Kentucky in 1987 (Hindal et al. 1987). The disease is lethal to all roses, and Doudrick (1987) and Hindal (1987) reported the occurrence of entire fields in Missouri dominated by multiflora rose where 80-90% of the plants were dead or dying. However, the causal agent of the disease is unknown, and it is considered unsafe for use in a control management program for multiflora rose because of the potential threat to ornamental roses. Doudrick (1987) believed that the disease may have reached equilibrium status in Missouri, and that multiflora rose may begin to “bounce back” (i.e., most of the non-resistant genomes of R. multiflora have been attacked, leaving the more resistant ones). The natural spread of the disease may eliminate the need for active control of multiflora rose in some areas.

Chemical control: Plant growth regulators have been used to control multiflora rose in southwestern Virginia where it has been used as a safety barrier along highways. Of the four regulators tested in Spring 1977, chlorflurenol, maleic hydrazine, and MBR- 18337 effectively prevented fruit set and subsequent spread. The fourth regulator, gyloxime, did not give adequate control although it caused some fruit abscission after fruit set (Hipkins et al. 1980).

Various herbicides have been tested and found effective for control of multiflora rose. It is important to note that multiflora rose has the typical regenerative power of members of the rose family (Scott 1965), and control programs must be monitored and followed up if necessary by repeated herbicide application or used

in conjunction with other control methods such as mowing or burning.

Glyphosate is effective against multiflora rose in a 1-2% V/V solution (Ahrens 1977, Lynn et al. 1979, Barbour and Meade 1980, Albaugh et al. 1977, Sherrick and Holt 1977, Fawcett et al. 1977). Although Reed and Fitzgerald (1979) reported glyphosate to be relatively ineffective, giving 25-75% stem kill over one season after a spring application, they did not follow-up their results to check for residual control the following year. Lynn et al. (1979) reported that a spring glyphosate treatment on R. multiflora showed increasing control over the growing season to complete control by the following spring. Treatments in the fall showed no results until the following spring, when effective control was realized (Lynn et al. 1979). Ahrens (1977) reported almost complete control of multiflora rose by the end of the second growing season after a late June application of either 1.5 or 3.0 lb/100 gal glyphosate, and noted that grasses growing underneath the roses were unaffected indicating that the spray on the rose overstory did not penetrate to the ground. Albaugh et al. (1977) found that the rate of application of glyphosate could be reduced to a 0.5% V/V solution for effective control with the addition of a surfactant.

2,4-D, and picloram also give excellent control of multiflora rose (Sherrick and Holt 1977, Fawcett et al. 1977, Reed and Fitzgerald 1979). Sherrick and Holt (1977) reported 2,4-D plus picloram, or picloram alone were also effective (all as foliar sprays). Picloram was found to be relatively ineffective as a soil application. Reed and Fitzgerald (1979) also found erratic results using picloram in pellet form (soil application), with stem kills ranging from 25-100% over one growing season (they did not look for the effects of residual control the following spring, however). Barbour and Meade (1980) reported picloram pellets to be effective, studied over a three- year period, at 2,4, or 5 lb/A.

Other foliar sprays found to be effective against multiflora rose include dicamba (Sherrick and Holt 1977, Fawcett et al. 1979), triclopyr (Sherrick and Holt 1977, Reed and Fitzgerald 1970) and fosamine (Kmetz 1978, Ahrens 1979). Fosamine controls only woody species and is non-volatile, and may be suitable in situations where there is concern to protect herbaceous species (Fawcett 1982). Pelleted and granular treatments found adequate include tebuthiuron (Lynn et al. 1978, Link et al. 1981) while dicamba was not found adequate (Sherrick and Holt 1977, Fawcett et al. 1977, Ahrens 1977, Barbour and Meade 1980).

Management Programs:
Multiflora rose has been declared a noxious weed in many states, including Kansas, Iowa, Missouri, Ohio, Pennsylvania, and West Virgina. It is mainly a threat to agricultural land, but has been reported to be a concern on at least two TNC preserves: the Spinn Prairie in Indiana and the Eldora Nature Preserve in New Jersey.

On the Spinn Prairie it occurs in small patches and monitoring may be necessary to determine if active control is necessary (Heitlinger 1987, McGrath 1987). At the Eldora Nature Preserve it is reportedly taking over old fields and there is concern about loss of habitat for some native moth species that feed on grasses in these areas (Davison 1987). No monitoring or management of multiflora rose has taken place at Eldora, but active control measures are considered necessary (Davison 1987).

Contact: Stewardship Director, The Nature Conservancy, Pennsylvania Field Office, 1218 Chestnut St., Suite 807, Philadelphia, PA 19107. (215) 925-1065.

Denny McGrath, Assistant Director, The Nature Conservancy, Indiana Field Office, 4200 N. Michigan Road, Indianapolis, IN 46208. (317) 923-7547.

A number of states where multiflora rose is a problem on agricultural land have cost share eradication programs whereby landowners can be reimbursed for a portion of the costs to control the plant on their property. These programs may also be available for preserve areas.

Contact: Iowa. Secretary of Agriculture, Iowa Dept. of Ag. and Land Stewardship, Wallace State Office Bldg, Des Moines, IA 50319.

Ohio. Larry Vance (614) 265-6610. Larry Summers (614) 265-6684. Ohio DNR. Div. of Soil & water Conservation, Fountain Square Bldg. E-2, Columbus, OH 43224.

Monitoring Requirements:
Monitoring should be conducted on preserve land where multiflora rose presents a potential management concern to determine changes in area occupied and density; also to track changes where control measures are being implemented.

Populations can be monitored with aerial photography and field measurements of abundance and density. Continuous monitoring over a period of several years may be necessary to check for the spread of small clumps and/or recurrences after implementation of control measures.

Monitoring Programs:
Heitlinger (1987) suggested monitoring of multiflora rose through the use of line intercept transects at the Spinn Prairie in Indiana to track its density and contraction/expansion. Contact: Denny McGrath, Ass't. Director, Indiana Field Office, The Nature Conservancy, 4200 N. Michigan Road, Indianapolis, IN 46208 (317) 923-7547.

VI. RESEARCH

Management Research Programs:
Research is currently being conducted at West Virginia University on rose rosette as a control for multiflora rose. Contact:

Dr. Dale Hindal, Division of Plant & Soil Sciences, Dept. of Plant Pathology and Agricultural Microbiology, 401 Brooks Hall, West Virginia University, Morgantown, WV 26506. (304) 293-3911.

Dr. James Amrine, Dept. of Entomolgy, West Virginia University, Morgantown, WV 26506. (304) 293-6023.

Management Research Needs:
Further research is needed in the area of biological control for multiflora rose. Both the phytophagous wasp Megastimus aculeatus and the Rose Rosette disease are potential biological control agents (see Management Procedures), but also represent a potential threat to ornamental roses.

In the case of M. aculeatus, the degree of host specificity is not fully understood. Milliron (1949) recognized two varieties of the wasp: a "light form" (M. aculeatus aculeatus) and a "dark form" (M. aculeatus nigroflavus). Milliron believed the dark form to be host specific on multiflora rose. However, Balduf (1959) recovered M. aculeatus nigroflavus from Rosa eglanteria and R. virginiana, both native roses. No further research has been conducted on M. aculeatus host specificity. A more promising control agent is the rose rosette disease. However, research to determine the causal agent of the disease has met with little success. Transmission of the disease is accomplished by an eriophyid mite, Phyllocoptes fructiphilus (Amrine et al. 1987). Symptoms of rose rosette, such as the witches broom and reddening of leaves, suggest a mycoplasma- like organism (MLO) as the causal agent, but the mite mouthpart (a sucking tube) is too small to suck up an MLO and also does not penetrate the phloem where an MLO would be found (Doudrick 1987). Other characteristics suggest a viral causal agent, but Doudrick et al. (1987) were unable to find anything resembling viral particles associated with diseased plants. Until more is known about the cause of rose rosette, it probably will not be employed in management programs for control of multiflora rose due to the threat to ornamental roses.

Other questions that may aid management of multiflora rose if carefully researched include the following. What are the germination requirements of multiflora rose and under what conditions are seeds least likely to germinate? How persistent is the rose in recovering grasslands that are no longer grazed? What are the effects of fire on seed viability and vegetative reproduction? How effective is fire in conjunction with herbicides or other control methods?

VII. ADDITIONAL TOPICS

VIII. INFORMATION SOURCES

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IX. DOCUMENT PREPARATION & MAINTENANCE

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HYDRILLA

ELEMENT STEWARDSHIP ABSTRACT
for Hydrilla verticillata (L.F.) Royle
Hydrilla
To the User:

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THE NATURE CONSERVANCY

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SCIENTIFIC NAME
Hydrilla verticillata (L.F.) Royle
Synonyms: Elodea verticillata (L.f.) F. Muell, Hydrilla lithuanica (Andrz. ex Besser) Dandy, Serpicula verticillata L.f.
COMMON NAME
Hydrilla

DESCRIPTION AND DIAGNOSTIC CHARACTERISTICS

H. verticillata is a submerged aquatic perennial plant that is highly variable in appearance, depending upon the conditions under which it is growing (Verkleij et al. 1983; Pieterse et al. 1985). H. verticillata generally is rooted to the bottom, although sometimes fragments will break loose and survive in a free-floating state. Erect stems can be quite long when the plant grows in deep water. Branching is usually sparse until the plant grows to near the water surface, where branching becomes profuse. Many horizontal aboveground stems (stolons) and underground stems (rhizomes) are also produced. Leaves are lanceolate, usually toothed, and arranged in bottlebrush-like whorls. Reproduction can occur sexually (through the production of seeds) or vegetatively. Vegetative reproduction in H. verticillata occurs by fragmentation of the stem, or by the production of above-ground and below-ground (subterranean) turions. The above-ground turions are sometimes referred to as axillary turions, as they are formed in the leaf axils, and the subterranean turions are often incorrectly referred to as tubers.

H. verticillata occurs in both monoecious (both male and female flowers on the same plant) and dioecious (male and female flowers on different plants) forms. Dioecious forms occur primarily in the southeastern U.S., California, Texas, Poland, Malaysia, Indonesia and Panama, while monoecious H. verticillata has been found in the mid-Atlantic U.S. (on the Potomac River near Washington, D.C.), India, and Indonesia. Isoenzyme comparisons of the Potomac River and Florida types are genetically distinct (McFarland and Barko 1990). Evidence indicates, however, that spatially disjunct and genetically distinct populations should be partially fertile and sexually compatible, with respect to chromosome pairing (Langeland et al. 1992). There is little evidence of self-incompatibility (Cook and Luond 1982).

Female flowers consist of three whitish sepals and three translucent petals. Flowers are 10-50 mm long and 4-8 mm wide, and are borne from a green spathe attached at the leaf axils. Flowers are clustered toward the tips of the stems, and float on the water surface. The stem tips from which female flowers arise are often very compact and have very short leaves. Female flowers are resistant to wetting and when returned to the water surface after submergence, will immediately float. A submerged female flower has been described as an inverted bell filled with a large bubble (Cook and Luond 1982). Male flowers have three whitish red or brown sepals 3 mm long and 2 mm wide, three whitish or reddish linear petals approximately 2 mm long, and three stamens. The male flowers are borne on a short stalk (Reed 1977), and as they approach maturity, are released and float to the surface, where they release pollen. Female flowers are wind-pollinated. Pollen that lands on the water surface is lost for reproductive purposes (Cook and Luond 1982). Thousands of free-floating male flowers have been observed in windrows on ponds (Langeland and Schiller 1983). In monoecious plants, male and female flowers are produced singly from the spathe.

Leaves are 2-4 mm wide, 6-20 mm long, sessile, and arranged in whorls of 3-8 (generally five leaves per whorl) (Cook and Luond 1982). The leaves have 11-39 sharp teeth per cm along the margins and often have either spines or glands on the underside of the midrib. The midrib is often red. The teeth are deciduous and leave behind elevated projections (Reed 1977). Adventitious roots are usually glossy white, although when growing in highly organic sediments, may take on the reddish brown color of the sediment. When exposed to light, the roots may have a greenish cast caused by the presence of chlorophyll (Langeland 1996).

The axillary turions of H. verticillata, also called winter buds, are condensed shoots of 12-15 internodes surrounded by fleshy leaves arranged in alternating whorls. They are oval to oblong in shape, 3-12 mm

long and 2-3.5 mm wide. Axillary turions are green in color and can be distinguished from vegetative buds by the absence of spines on the midrib of their leaves. They fall from the plant when they mature. Subterranean turions are formed terminally on rhizomes or stolons, and can be found up to 30 cm deep in the sediment. They are 5-10 mm long and are off-white to yellow, unless they take on darker colors from organic sediments (Langeland 1996). They are filled with reserve food material in the form of starch (Miller et al. 1992).

H. verticillata was originally confused with Elodea sp., a common native aquatic plant of the central and northern U.S. and parts of Canada. H. verticillata can be distinguished from Elodea by its sharply serrated leaf margins (usually visible to the naked eye), red veins, spinous midrib and scabrous texture, and its anthers that open explosively. However, leaf morphology, the variation in the number of leaves per node, and variation in hydrilla under different conditions, make misidentification common (Hench et al. 1994).

PEST WEED STATUS

H. verticillata is a federal noxious weed and an A-ranked noxious weed in California. It is considered a pest species throughout its introduced range.

STEWARDSHIP SUMMARY

H. verticillata is native to the warmer areas of Asia, and was first discovered in the United States in 1960. It possesses specialized growth habits, physiological characteristics, and reproductive strategies that allow for rapid growth and expansion in freshwater environments. H. verticillata has spread rapidly through portions of the United States and has become a serious weed in aquatic systems, causing substantial economic hardships, interference with water uses, displacement of native aquatic plant communities, and other adverse impacts to freshwater habitats. Management methods currently include mechanical removal and drawdowns, herbicides, and biological controls.

Mechanical removal is a primary method of H. verticillata removal, but because it is so costly, it is only used in proximity to domestic water supply intakes, in rapidly flowing water, or when immediate removal is necessary. Water drawdown can be effective if done while subterranean turions are developing in the fall and prior to regrowth in the spring, but drawdowns are restricted to water bodies with water control structures, and where they will not interfere with other primary water uses. Several herbicides have also been used to control hydrilla. Most effective have been the contact poisons copper sulfate (brand name Komeen and others) and endothal (brand name Aquathol and others), and the systemic herbicides fluridone and bensulfuron methyl. Contact poisons are especially toxic to other plants and/or animals. Several organisms have shown promise as biocontrol agents against hydrilla. Sterile, triploid Chinese grass carp are available and legal by permit in some states in the U.S. They are useful in small ponds or lakes and canal systems where the fish can be retained within the water body and where the removal of all vegetation is acceptable. Two weevils, two leaf-mining flies, and one aquatic moth, have also been purposely or accidentally introduced to control hydrilla.

RANGE

H. verticillata is native to the warmer regions of Asia (Cook and Luond 1982), and now occurs in Asia, Australia, New Zealand, the Pacific Islands, Africa, Europe, South America, and North America (Cook and Luond. 1982; Langeland 1996). In North America, states which have recorded occurrences of H. verticillata include: Alabama, Arizona, California, Delaware, District of Columbia, Florida, Georgia, Louisiana, Maryland, Mississippi, North Carolina, South Carolina, Tennessee, Texas, Virginia, and Washington. H. verticillata was recently reported in Connecticut (USDA, NRCS 1999).

IMPACTS AND THREATS POSED BY HYDRILLA

Because of unique biological and physiological characteristics, and an aggressive growth habit, H. verticillata has established itself in a wide range of aquatic habitats. Once established, hydrilla can replace native aquatic vegetation and affect fish populations. H. verticillata can have long-term impacts on native systems by reducing seed production of native species, resulting in a decline in the native species proportion of the seed bank (De Wintono 1996). H. verticillata can also displace native aquatic plants such as pondweeds (Potamogeton sp.) and eelgrass (Vallisneria americana Michaux). Hydrilla can elongate 2-3 cm per day as it approaches the water surface. Near the water surface, it branches profusely, with 50% of the standing crop occurring in the upper 0.5 m of the water column (Haller and Sutton 1975). By producing such a large mat of vegetation on the water surface, H. verticillata intercepts sunlight to the exclusion of other submersed plants. Both inorganic and organic sediment levels increase with increasing hydrilla abundance (Joyce et al. 1992). Such increases may themselves have effects on aquatic community composition. In natural systems, as well as in channels and rivers providing urban and agricultural water supplies, hydrilla slows the movement of water. Supply, drainage and irrigation uses are hampered. Recreational uses (e.g., boating) are also degraded. H. verticillata can, therefore, have a serious impact on the functioning of natural systems, cultural water uses, and economic uses of water supplies.

HABITAT

H. verticillata has a large ecological amplitude and is found in a variety of aquatic habitats, but rarely in swift-flowing water. Water quality is rarely limiting. It is able to grow under a wide range of water chemistry conditions, and is found in oligotrophic (low nutrients) to eutrophic (high nutrients) lakes (Langeland 1996). H. verticillata can grow in water up to about 7% the salinity of seawater (Haller et al. 1974) or higher (Steward and Van 1987), and tolerates a wide range of pH values, but prefers a pH of 7.0 (Steward 1991). H. verticillata grows very rapidly from rootstocks, stem fragments, and from turions. Only one node or whorl of leaves is necessary for growth. H. verticillata is normally associated with low amounts of sulfate, chloride, nitrate and iron, and in deep sediments (Cook and Luond 1982). It is adapted to low light levels (Van et al. 1976).

ECOLOGY AND BIOLOGY

Light, temperature and nutrients

Hydrilla has low rates of dark respiration and photorespiration, and a high rate of photosynthesis (Cook and Luond 1982). It can therefore, begin photosynthesis at lower light levels (early in AM, later in PM, and seasonal) than other aquatic plants. Its low light requirements (1% of full sunlight or less) also allow H. verticillata to colonize deeper waters than other aquatic plants (Langeland 1996). In clear waters, hydrilla can grow from more than 10 meters deep (Hall and Vandiver 1991). H. verticillata most commonly occurs in around three meters deep, but has been found growing from depths of 15 meters.

Submersed plants are often subjected to more constraints from photosynthesis, in comparison to terrestrial plants. Carbon dioxide has a low diffusion rate in water, so the efficient use of bicarbonate ion as a dissolved inorganic carbon source is an important competitive characteristic for existence in aquatic environments. H. verticillata can use free carbon dioxide from surrounding water when it is available, and can easily switch to bicarbonate utilization when conditions favors its use, such as when waters have a high pH or a high carbonate concentration (Salvucci and Bowes 1983). These conditions occur in highly productive waters during warm water and high photosynthesis conditions. Under these conditions, H. verticillata can also switch to C4-like carbon metabolism, characterized by low photorespiration and inorganic carbon fixed into malate and aspartate (Holaday and Bowes 1980).

Monoecious and Dioecious Biotypes

Comparisons of monoecious and dioecious biotypes indicate that the monoecious type can germinate at lower temperatures and produce the maximum number of subterranean tubers under short day lengths. This indicates that the monoecious type may be better adapted to cooler climates (Steward and Van 1987).

Allelopathy

H. verticillata does not produce allelopathic chemicals. It is also not affected by allelopathic exudations from native aquatic species such as Ceratophyllum demersum, Potamogeton nodosus, or Vallisneria americana (Jones 1995).

Reproduction

H. verticillata reproduces by both vegetative and sexual methods. Vegetatively, H. verticillata reproduces by stem fragmentation and by the production of turions (axillary and subterranean). In general, most hydrilla plant fragments (more than 50%) can resprout and grow into new plants (Langeland 1996).

Turions are vegetative propagules. Formation of turions begins in September and increases from October to November, decreases in December and January, and increases again in the late spring. Turion production is significantly higher in floating plants (75.6 turions/kg) than in rooted plants (28.6 turions/kg). Production of turions decreases with increasing plant density of H. verticillata (Miller et al. 1992).

Subterranean turions are considered the more important of the two methods of turion reproduction. One single subterranean turion has been shown to produce over 6000 new turions per m2 (Sutton et al. 1992), per year, while only 2,803 axillary turions can be produced per m2 (Thullen 1990). Subterranean turions can remain viable for several days out of water (Basiouny et al.1978), and for over four years in undisturbed sediments (Van and Steward 1990). They can also survive ingestion and regurgitation by waterfowl (Joyce et al. 1980), and herbicide applications (Haller et al. 1990).

Subterranean turion and biomass production are closely correlated with temperature. In laboratory experiments, both are inhibited at 12° to 16° C. Biomass production increased linearly with increasing temperature to 28° C and reached a maximum at 28° to 32° C. Subterranean tuber production was highest at 24° C. Both subterranean tuber and biomass production were higher on inorganic than organic sediments. Most subterranean tubers are formed during the short days of spring and autumn. Higher temperatures during simulated short day lengths increased the number of subterranean tubers; higher temperatures during simulated long day-lengths increased subterranean tuber mass. Amendments of N had no significant effect (McFarland and Barko 1990).

Seed production is probably of minor importance to H. verticillata reproduction, compared to its successful vegetative reproduction. Seed production and viability is low compared to many other weedy species. Survivability of seeds after ingestion is unknown. The importance of seed dispersal, therefore, after ingestion by birds or other species, is also unknown (Langeland 1996).

ECONOMIC AND OTHER USES

H. verticillata causes major negative impacts on water use. In drainage canals, it greatly reduces flow and can cause flooding and damage to canal banks and structures. In irrigation canals, it impedes flow and clogs the intakes of pumps used for conveying irrigation water. Hydrilla has interrupted water flow patterns in utility cooling reservoirs. It can severely interfere with the navigation of both recreational and commercial craft. Furthermore, H. verticillata interferes with boating by fisherman and water-skiers, and interferes with swimming. Limitations on water use can reduce real estate values and tourism.

Some sport fishermen consider H. verticillata to benefit largemouth bass habitat (Tucker 1987), although research results differ (Estes et al. 1990; Porak et al. 1990; Colle and Shireman 1980; Canfield and Hoyer 1992). Because H. verticillata displaces native plant communities, it may adversely impact sportfish populations.

H. verticillata may have some positive economic consequences. It is eaten by waterfowl, and maintaining hydrilla populations is sometimes advocated by waterfowl scientists because it increases feeding habitat for ducks (Johnson and Montalbano 1984; Esler 1989). Highly transparent water is often desired by the public, and submersed aquatic macrophytes such as H. verticillata tend to increase water clarity (Canfield et al. 1984) probably by lowering suspended sediment loads, and removing nutrients on which phytoplankton populations depend.

MANAGEMENT

Potential for Restoration of Invaded Sites

H. verticillata has an unusually high level of reproductive vigor, and is also highly adaptable to different habitats. There are currently no known important native insect pests of H. verticillata. In wildlands, the application of herbicides can kill other aquatic plants and animals. Unless and until biological controls become biologically and economically feasible, the potential for large-scale restoration of wildlands infested with H. verticillata is probably very low.

Management methods currently include mechanical removal and drawdowns (controlled water drainage), herbicides, and the use of some biological controls.

Specialized machines are used for mechanically removing H. verticillata. This is not a widespread practice because of the high cost involved (often over $1000 per acre) and because of logistical constraints. Up to six harvests may be required annually due to the rapid growth rate of H. verticillata (McGehee 1979). Mechanical removal is used for H. verticillata management only in areas that are in close proximity to domestic water supply intakes, in rapidly flowing water, or when immediate removal is necessary. The high cost of harvesting H. verticillata, and its low nutrient value, greatly restrict hydrilla's value as a forage plant (Easley & Shirley 1974; Bagnall 1978).

Drawdowns can be an effective mode of hydrilla control if the drawdown is performed while subterranean turions are developing in the fall, and prior to regrowth in the spring (Haller et al. 1976). Drawdowns for aquatic plant management are restricted to only those lakes or ponds that have water control structures, and have hydrologic characteristics that permit water levels to be controlled. Additionally, the drawdown must not negatively impact other primary water uses, such as domestic or irrigation supplies, navigation, or hydrologic power. Even in drained lakes and ponds, subterranean turions may remain dormant and viable in organic substrates (Haller and Shireman 1983).

Several herbicides have been used to control hydrilla. Most effective have been the contact poisons copper sulfate (brand name Komeen and others) and endothal (brand name Aquathol and others), and the systemic herbicides fluridone and bensulfuron methyl. For both contact and systemic herbicides, concentration in the water column and exposure time are key variables determining effectiveness. Copper sulfate and endothal are non-selective herbicides, and copper sulfate is highly toxic to fish. Fluridone has been used to control H. verticillata in Lake Okeechobee in Florida with minimal to no long-term impact on native aquatic plants (Langeland 1996). Application rates vary according to a number of factors, including water depth, water chemistry, whether the water is still or moving, and the size of the infestation. Getsinger & Netherland (1997) report that the following formulations have been effective: for endothal, 2.0 mg ae/L for 48 hours or 3.0-5.0 mg ae/L for 24 hours; for fluridone, 15-30 ug/L for 20-40 days (minimum of 4 ug/L); and for bensulfuron methyl, 25 ug/L and higher for in excess of 42 days. The use of plant growth regulators

such as fluridone and bensulfuron methyl is relatively recent, and is intended to reduce, but not to necessarily eliminate, H. verticillata. Less vigorous remnant plants may perform useful functions such as providing oxygen, stabilizing sediment loads, and creating habitat (Lembi and Chand-Goyal 1994).

Acetic acid in concentrations of 9-26 mmol/L (which is less concentrated than commercial vinegar) for 24 hours reduced growth by 50% in laboratory studies (Spencer and Ksander 1995). The use of compounds from native aquatic plant species with allelopathic properties has not been shown to be an effective control for H. verticillata (Jones 1995).

Grass carp or white amur (Ctenopharyngodon idella Val.) is a biological control agent that effectively controls H. verticillata (Van Dyke et al.1984). Possession of this fish species, however, is illegal in many states because of the potential environmental damage that could result if escaped fish were to establish breeding populations. Sterile, triploid grass carp are available and legal by permit in some states in the U.S. Not all triploid grass carp are sterile, however, and every individual released needs to be genetically checked. Grass carp is recommended for small ponds or lakes and canal systems where the fish can be retained within the water body and where the removal of all vegetation is acceptable. There is no adequate method of recapture. Since Chinese grass carp prefer food other than H. verticillata, a reduction in the overall abundance of native aquatic plants, and the potential reduction in food and habitat for invertebrates, other fish, and waterfowl, are to be expected. Stocking rates for partial control have not been established.

Worldwide surveys for natural H. verticillata enemies were begun in 1981 in a cooperative study undertaken by the University of Florida-IFAS, United States Department of Agriculture, and U.S. Army Corps of Engineers. Snails consume large amounts of H. verticillata when they are present in high densities in enclosed experimental areas, but not in natural settings. Plant pathogens effective against H. verticillata under experimental conditions have been ineffective in the field (Charudattan and Lin 1974; Charudattan and McKinney 1978). Several pathogens have been identified from Asia, but their effectiveness has yet to be tested (Shearer 1997).

Over 40 species of insects have been found that feed on H. verticillata. Several are being evaluated as potential H. verticillata biocontrols in the United States. Other insects from Australia are also under consideration (Balciunas et al.1996). Bagous affinis Hustache is a weevil that was discovered in Pakistan and India. Adults lay eggs on rotting wood and other organic matter. After hatching, the larvae burrows through the sediment until it encounters a H. verticillata subterranean turion, which it then feeds on and destroys (Buckingham and Bennett 1994). This insect is useful only where there are periodic lake drawdowns or intermittently wet and dry shorelines. Another Bagous species has been released in the U.S. but has not become established. Hydrellia pakistanae Deonier is a leaf-mining fly that is very promising as a H. verticillata biosuppressant (Buckingham et al.1989). H. pakistanae is established in Florida, but its impact on H. verticillata has yet to be determined. H. balciunasi, released in 1989, has had limited establishment, apparently due to several factors including competition with other biological agents, parasitism by native wasps, genetic differences in H. verticillata types, and possible inbreeding depression (Grodowitz et al. 1997). An aquatic moth, Parapoynx diminutalis Snellen, was accidentally introduced into the United States (Del Fosse et al. 1976). The larvae of this moth can frequently be found feeding in large numbers on H. verticillata, though usually not until late in the growing season. Large areas may be defoliated but viable stems remain and the plant remains a problem.

MANAGEMENT PROGRAMS

Management programs are extensive where H. verticillata impacts recreational and economic resources. In the USA, California and Florida have spent millions of dollars on control efforts.

MONITORING

In general, the objectives of monitoring should track those of management. Abundance (cover) is often measured following control applications. Most areas subject to management for H. verticillata are of economic or recreational importance (e.g., recreational lakes, canals, and irrigation systems). In those areas, the goal is removal of H. verticillata to a degree sufficient to permit continued economic or recreational uses. Monitoring is generally designed to monitor changes in abundance of H. verticillata alone. In natural areas management, monitoring programs will likely combine changes in abundance of H. verticillata with changes in abundance of species or changes in community attributes that are the targets of management. Such programs should have explicit objectives that can be measured and that are meaningful from both a biological and management standpoint. These objectives may vary depending on the abundance of H. verticillata and other invasive aquatic plants.

In terms of effort (number of plots established and monitored), transects, or long, linear plots are more effective in providing sufficient statistical power to determine change, than square or broadly rectangular or otherwise regularly shaped plots. Such techniques, however, are difficult to apply in aquatic situations. While generally a research technique, measuring change, or lack thereof, in control (unmanaged) areas can be an effective way of assuring that changes are actually resulting from management and not from other factors.

Extensive monitoring programs exist where management programs have been implemented, especially in California and in Florida. In Florida, monitoring the presence/absence and abundance of H. verticillata has been coupled with that of other species. Monitoring the abundance of subterranean turions has also been undertaken to determine long-term effects of management (Sutton 1996).

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RESEARCH

A great deal is known about the biology of this species. Further work is needed on the effectiveness of replanting native species to slow reinvasion by H. verticillata following control (Madsen 1997).

Extensive research on the biology and management of this species has been done and is ongoing. Several journals, such as the Journal of Aquatic Plant Management, regularly carry articles on management of this species. A computer model of growth and reproduction under various site conditions has been developed (Best and Boyd 1996). However, there is little research on management and restoration of natural areas where native species are targets of conservation. Further work is needed on the application of biocontrol agents and of the potential impacts of such agents on native species and natural ecosystems. Work is needed on the use of plant growth regulators, such as flurprimidol, which reduce plant growth but do not cause mortality (Lembi and Chand-Goyal 1994). In addition, work is needed on integrated management strategies that combine herbicide use with biological control and restoration techniques (Madsen 1997).

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