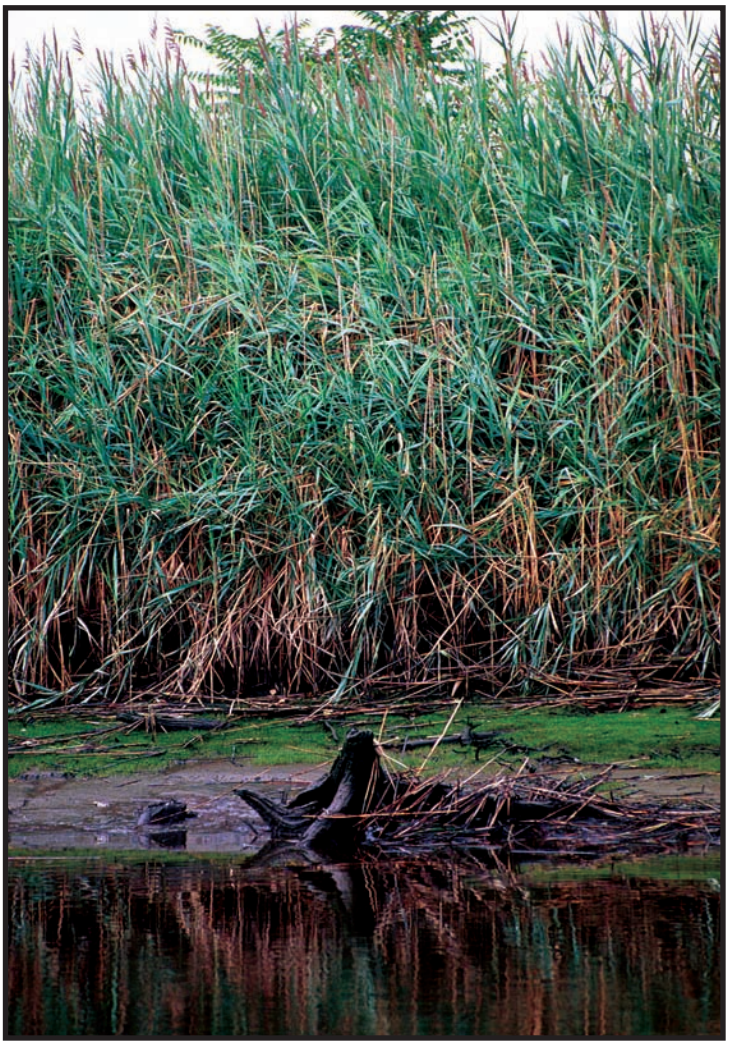
VI. Invasive and Exotic Species



A stand of common reed (*Phragmites australis*), Haplotype M.

VI. Invasive and Exotic Species

During the last century, a nonnative form of common reed (Haplotype M shown along the bottom) that thrives in disturbed wetlands has spread throughout the Meadowlands and much of North America. Common reed is tolerant of certain contaminants and can stabilize sediments in heavily contaminated wetlands. However, Haplotype M grows into vast stands that choke out other plant life and reduce animal diversity and production. Also, Haplotype M is difficult to eradicate and is re-invading restoration sites. Thus, when and where to eradicate common reed must be evaluated on a site-specific basis for enhancing and restoring the Meadowlands.

Common reed is not the only invasive species that is widespread in the Meadowlands: new exotic species arrive in the New York-New Jersey Harbor on an almost daily basis. Most exotic species do not survive for long, but those that do compete with or prey upon native life forms. Purple loosestrife (*Lythrum salicaria*, upper right), Japanese knotweed (*Polygonum cuspidatum*, upper left), and tree-of heaven (*Ailanthus altissima* middle insert) are spreading at many sites. Several invasive mollusks and other animals have also spread throughout the Meadowlands' waterways, though their impacts are unknown. Because invasive species, including disease-causing agents, may change biological communities and ecosystem processes, restoring the Meadowlands will not be successful unless monitoring and control programs are developed in the Meadowlands and the entire New York-New Jersey Harbor.

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Extensive monoculture stands of common reed such as that shown here can stabilize eroding areas of contaminated sediments. However, such stands provide minimal habitats for native fish and wildlife populations.

VI. INVASIVE AND EXOTIC SPECIES

A. INTRODUCTION

One of the first features of the Meadowlands that a person will notice while driving by on the New Jersey Turnpike, visiting Giants Stadium, or flying into Newark Airport is the vast sea of the giant, grass-like reed known as *Phragmites*. The form of *Phragmites* that dominates the Meadowlands is an invasive species that is not native to the United States. The species has spread over the Meadowlands in recent history to form a monoculture that has degraded the habitat suitability for many native plant and animal species and reduced the overall biological diversity of the Meadowlands. This invasive plant has been the primary subject and target of restoration activities in the Meadowlands.

At the 1994 national meeting of the Ecological Society of America, it was proclaimed that the earth was entering a new geologic/biologic period, "the Homogocene Age," in which natural borders are lacking and the planet's biota is increasingly homogenized with the potential for dramatic and far-reaching changes on the earth. Since that pronouncement, scientists in diverse disciplines (*e.g.*, epidemiology, agriculture) and organizations with broad scopes (*e.g.*, United Nations) are becoming increasingly aware of and concerned about the impacts of invasive species on the biology of natural systems and on the people of the earth (Myers and Knoll, 2000; United Nations Earthwatch, 2003).

Invasive species have diverse and well-documented impacts on native species, communities, and ecosystems (Drake *et al.*, 1989). For example, introduction of a single invasive species can have a wide variety of effects including:

(1) the displacement of one or more native species (*e.g.*, purple loosestrife displacing native wetland plant species; Gaudet and Keddy, 1988);

(2) reduced abundance of wildlife species dependent on displaced native species (*e.g.*, native butterflies that pollinate or obtain nectar from introduced plant species may have larvae that are entirely dependent on a single native plant species; Kearns *et al*., 1998);

(3) altered habitats (*e.g.*, purple loosestrife overgrows grassy nesting habitat of some shorebirds and waterfowl; Faanes, 1979); and

(4) disrupted community and ecosystem ecology (*e.g.*, European earthworms [*e.g.*, *Lumbricus terrestris*] altering fern, decomposer, and forest communities; Frelich and Holdsworth, 2002; Groffman *et al*., 2004).

As a result of their diverse actions on species, communities, and ecosystems, invasive species may directly or indirectly result in considerable economic and other costs that far exceed the costs of treatment or control (Simberloff, 1996; Connelly, 2003; Pimentel, 2004). For example, 80 introduced fishes are causing about $3 billion dollars in losses annually to commercial and sport fisheries in the northeastern U.S. alone. European starlings (*Sturnus vulgaris*) are

responsible for nearly $1 billion annually in crop losses nationwide. Compounding the costs of eradicating or managing invasive species are research and educational costs.

Recognizing the threats to native flora and fauna in the United States, President Clinton issued an Executive Order in February 1999 (E.O. 13112) establishing an Invasive Species Council and directing federal agencies to identify, prevent, and mitigate the adverse environmental effects of invasive species. Despite local efforts to publicize invasive-species problems and educate the public (*e.g.*, the Service's 2004 Issue of *Field Notes* titled "Invasion! In Your Back Yard?"), the scope of the invasive-species problems in the Meadowlands and the surrounding urban area is not sufficiently recognized or publicized, and concerted efforts to address the problems to date are just beginning. In 2004, the Governor of New Jersey signed an executive order forming an Invasive Species Council at the State-level, charged with submitting an Invasive Species Management Plan for the State in 2006. The NJDEP has also issued a new policy directive prohibiting the planting of non-native species on State lands (Policy Directive 2004-02, New Jersey Department of Environmental Protection, 2004e). However, non-native species are often included in recommended planting lists for landscaping within the HMD, and nurseries within the HMD and surrounding areas still market a number of invasive species (*e.g.*, purple loosestrife). Such inconsistencies have helped create public indifference toward invasive species. Certain particularly noxious, recent invaders of wetlands in New Jersey and elsewhere in the northeastern U.S. (*e.g.*, giant hogweed [*Heracleum mantegazzianum*]) may contribute to a change in the public's perception of invasive species, but more comprehensive efforts, including education, are needed to address the many problems of invasive species in the NY-NJ Harbor area, including the Meadowlands.

While the number of invasive species occurring in the Meadowlands is not known, it has been estimated that 20,000 of the 50,000 exotic species introduced into the U.S. were first introduced into the northeastern United States (Pimentel, 2004). Many species now known nationwide were introduced first within the NY-NJ Harbor region. Some of the best-known examples include purple loosestrife (Torrey *et al.*, 1819), carp (*Cyprinus carpio*; DeKay, 1842); European starlings (Adeney, 2001), and disease-causing agents (West Nile virus [*flavivirus*]; Animal and Plant Health Inspection Service, 2004). Although some of the above species were purposely introduced, traditional colonial shipping (*e.g.*, dumping dry ballast material in marsh areas) and agricultural practices (*e.g.*, landscape clearing and burning) are thought to have promoted introduction and establishment of an untold number of exotic insect and plant species. Later shipping practices such as using water as ballast introduced the larvae of freshwater and marine invaders (*e.g.*, zebra mussel [*Dreissena polymorpha*], Great Lakes Science Center, 2000; Japanese shore crab [*Hemigrapsus sanguineus*], McDermott, 1991) and probably some infectious diseases (*e.g.*, cholera [*Vibrio cholera*]; DePaulo *et al*., 1992; Ruiz *et al.*, 2000). Considerable attention is now paid to transportation practices to address dispersal and transport of exotic species (*e.g.*, National Ballast Information Clearinghouse, which collects, analyzes, and interprets data on the ballast water management practices of commercial ships that operate in the waters of the United States; Smithsonian Environmental Research Center and the U.S. Coast Guard, 2005). Despite such efforts, disturbed urban systems adjacent to shipping and other transportation centers will remain at considerable risk from existing and novel invasive species.

Without effective biological or other environmental controls to limit their abundance as occurs with native species, several invasive species have spread throughout the Meadowlands and significantly diminished the habitat values of the area for many native species. The invasive species discussed below are among the most abundant invasive species in wetlands within the HMD; each of these presents significant, complex, and potentially costly challenges for restoration of the Meadowlands. Given the continuing status of the NY - NJ Harbor estuary as one of the world's busiest shipping centers, a comprehensive effort to assess, evaluate, control, and manage these and other invasive species (and identify and respond to "new" exotic species) is needed.

B. COMMON REED (*Phragmites australis*)

At the present time, common reed (Figure 57) is the most significant invasive species affecting the restoration of not only the Meadowlands but also marshes throughout much of eastern North America (Chesapeake Bay *Phragmites australis* Working Group, 2003). During the past century, common reed has become one of the most abundant and widely distributed plants throughout the world (Den Hartog *et al.*, 1989). In North America, the invasive form of common reed has become more abundant in areas where the native form has occurred for several thousand years and spread into areas where common reed did not previously occur (Chambers *et al.*, 1999; Orson, 1999). During this century, common reed has been replacing the dominant plant species (*e.g.*, *Spartina* species in saltmarshes and *Typha* species in freshwater marshes) along the Atlantic coast at rates of 1 to 6 percent per year (Chambers *et al*., 1999; Warren *et al.*, 2002). Common reed appears to have become well established throughout most of the Meadowlands by the early 1930s (Sipple, 1972) as a result of conditions favorable to the species. Salinities throughout most of the Meadowlands (0 to 15 ppt; New Jersey Meadowlands Commission, 2005c) are ideal for growth of common reed (Saltonstall, 2003a).

1. Haplotype M: Characteristics and Biology

Recent molecular research has established that a competitively superior, non-native lineage of *Phragmites australis*, referred to as Haplotype M, is responsible for the widespread invasion of this species throughout wetlands and adjacent lands in eastern North America, including the Hackensack Meadowlands (Saltonstall, 2002; 2003b). Haplotype M denotes a specific set of closely linked genes occurring in the invasive form, now recognized to differ from native Mid-Atlantic haplotypes (strains) in several characters including stem color, root growth, and rhizome growth (Saltonstall and the IAN Group, 2003; Meadows and Saltonstall, 2004; Saltonstall, 2004). The attachment of leaf sheaths is considered the most reliable character to distinguish Haplotype M from native Mid-Atlantic strains (Meadows and Saltonstall, 2004). Haplotype M is the only strain that has been reported recently in the Meadowlands, though native strains undoubtedly occurred there previously (Waksman *et al.*, 1943; Orson *et al*., 1987), and may still persist in some locales, especially in the HRW outside of the HMD.

Haplotype M was most likely brought to northeastern North America from Europe during the 19th Century. During most of that century, ships' ballast largely consisted of dry materials that were readily available in coastal areas (*e.g.*, coastal rocks and earthy debris). Common reed has

been documented as growing in the U.S. where ships' ballasts were regularly dumped or used to fill marshes for railroad and shipping facilities (Saltonstall, 2002). Recent genetic analysis of museum materials has indicated that Haplotype M was present at a number of sites but uncommon in the U.S. prior to 1910 (Saltonstall, 2002); however, by 1940 it was present in nearly all sampling locations. Presently, Haplotype M is the most abundant plant in the Meadowlands, where it forms dense monotypic stands (Saltonstall, 2002). The spread of Haplotype M in the Meadowlands and elsewhere is largely due to human activities, including historical disturbances, such as the clearing or burning of vegetation, and more recent stresses such as pollution, alteration of the natural hydrologic regime, dredging, filling, eutrophication, and decreased oxygen levels in water (Roman *et al.*, 1984; Marks *et al.*, 1994).

Haplotype M is a successful colonizer of disturbed aquatic and adjacent upland areas due to a unique combination of adaptive features. Unlike many other plants, its seeds germinate readily on oxygenized, disturbed soils (Witje and Gallagher, 1996). Its colonies expand by rhizomes in aquatic sediments and by rhizomes and stolons in sandy soils; growth of rhizomes may exceed

Figure 57. Monotypic stand of the invasive form of common reed (Haplotype M) growing along Berry's Creek Canal. The Oritani Marsh along Berry's Creek is one of many common-reed dominated marshes in the Hackensack Meadowlands.

10 m in a single growing season (Ailstock, 2000; Chesapeake Bay *Phragmites australis* Working Group, 2003). Haplotype M is an aggressive form that outcompetes other strains of common reed across a wide range of salinity and nutrient conditions (*i.e.*, low or high concentrations singly or in combination; Saltonstall, 2004). Haplotype M is more efficient in gas exchange and mineral uptake than other emergent marsh plants and is more tolerant of low oxygen levels and acidic sediments, characteristics that enable it to survive in habitat unsuitable for other emergent marsh plants (Bart and Hartman, 2000). Common reed also possesses several adaptations that inhibit the growth of other plant species: (a) it attains greater heights, thus shading other emergent marsh plants; (b) it grows more densely than most other emergent plant species, thus reducing available space for other species; and (c) its colonies generate dense thatch layers that inhibit germination and growth of other species (Haslam, 1971; 1972; Ailstock, 2000).

2. Effects on Wetlands

Invasion of marshes by Haplotype M is increasingly recognized as capable of affecting entire ecosystems; however, many of the changes are subtle and at some sites do not always represent an ecological disaster (*e.g.*, Weis and Weis, 2003b). For example, certain ecosystem functions provided by common reed and smooth cordgrass (*e.g.*, sediment trapping, litter decomposition) may be comparable, but common reed's tolerance of high levels of heavy metals enables the plant to grow in and stabilize contaminated substrates (*e.g.*, Weis and Weis, 2003b). However, even subtle differences between the invasive and native haplotypes may have substantial effects on certain features of biological communities and ecosystem functioning. Marshes dominated by common reed have somewhat different seasonal temperature cycles, freezing earlier in fall and thawing later in spring than other marshes (Saltonstall, 2003a); thus, invasion of marshes by common reed may cause changes in the seasonality of primary and secondary production of Atlantic coast marshes. Dense stands of common reed result in: (1) decreases in topographic relief (*i.e.*, reduced microhabitat availability), abundance of standing water habitats, soil salinity, light levels, and benthic algal production, and (2) increases in detritus loading, depth to the water table, and soil aeration (Weinstein and Balleto, 1999; Windham and Lathrop, 1999; Wainwright *et al.*, 2000). Common reed thus changes the structure of the marsh, which facilitates its continuing invasion. As it spreads throughout the marsh, common reed alters the hydrology of marshes and tidal creeks. For example, common reed-dominated marshes tend to accrete sediment and organic material more rapidly than other marshes, resulting in the filling of creeks and tidal channels and reduced water flow over the marsh surface (*e.g.*, Weinstein and Balleto, 1999). During this reed-induced transformation of the marsh, the accumulation of detritus and other debris may degrade the aquatic habitat for many species. Species affected by this transformation include habitat generalists (*i.e.*, species such as benthic invertebrates [Angradi *et al.*, 2001] and larval fishes [Able and Hagan, 2003] that may use many marsh structures, features, or vegetation as habitat, and thus range widely over the marsh surface) and habitat specialists (*i.e.,* species such as willets [*Catoptrophorus semipalmatus*] and seaside sparrow [*Ammodramus maritimus*] that require very specific features, structures or vegetation for habitat, and thus are more restricted in their distribution within a marsh; Benoit and Askins, 1999). Common reed also ties up nutrients in its biomass, including its litter; nutrients are not released as quickly by litter of common reed as by litter of other common native vegetation in North American marshes (*e.g.*, salt hay grass; Windham, 2001). This trait can provide beneficial effects on polluted sites by reducing the bioavailability of environmental contaminants (see

discussion below). Microbial growth on common reed litter may be slow in comparison to microbial growth on litter of other plants (Findlay *et al.*, 2002).

The aggressive and pervasive growth of common reed has reduced plant diversity within the Meadowlands; however, a detailed understanding of the effects of common reed on the Meadowlands is lacking. The factors affecting the growth of common reed and its re-invasion of sites from which it has been eradicated (Table 5) must be understood better to restore the Meadowlands successfully. This species is known to have reduced plant diversity in many freshwater and estuarine marshes (Fell *et al.*, 1998; Chambers *et al.*, 1999; Meyerson *et al.*, 1999; Windham and Lathrop, 1999) and shifted plant community composition to tall-grass species (Saltonstall, 2003a). Typical estuarine landscape components (*e.g.*, high marsh communities) appear to be disappearing from the Meadowlands and other marshes along the Atlantic coast, though invasion by common reed appears only partially responsible (Bertness *et al.*, 2002). Common reed is reinvading nearly every site in the Hackensack Meadowlands that has been "restored" by removal of common reed and replacement with cordgrass (T. Doss, pers. comm., 2004). The only exception is the Marsh Resources Inc., Mitigation Bank site, which is the most recent site in which common reed was replaced with other vegetation. Thus, it may only be a matter of time before common reed reinvades this site. Because common reed is widely distributed on uplands (*e.g.*, especially on ditches, landfills, brownfields), numerous stands of common reed that potentially serve as sources for reinvasion of wetlands restoration sites occur throughout the Meadowlands. Reinvasion of estuarine restoration sites has occurred in other Mid-Atlantic states (*e.g.*, Virginia; Havens *et al.*, 1997). Currently, State and federal agencies have limited jurisdiction and resources to address management and control of common reed in the HMD. For example, current federal permits typically require only 5 years of monitoring of mitigation sites; longer periods of monitoring and control (10 years or longer), as are being considered elsewhere (Chesapeake Bay *Phragmites australis* Working Group, 2003), would be beneficial in the Meadowlands.

Though used by some wildlife, dense stands of Haplotype M are recognized generally as reducing native species diversity and providing poor-quality habitat for most wildlife (Cross and Fleming, 1989). Fewer than half of the bird species reproducing in the Meadowlands will nest in common reed (*e.g.*, red-winged blackbird, marsh wren, swamp sparrow; Kane, 2001); however, additional species may forage or rest in stands of common reed on migration stopovers. The usage of common reed by birds in the Hackensack Meadowlands may be due to the high value of common reed as shelter (*i.e.*, a refuge from predation or harsh weather) or simply due to the lack of other suitable vegetation to provide birds with food and cover. Large monotypic stands of common reed generally contribute to declines in overall avian species richness and marsh usage in most other estuaries (Roman *et al.*, 1984; Marks *et al*., 1994; Benoit and Askins, 1999). Wildlife use of *Phragmites* edges, mixed stands, and patches is higher than use of large, dense stands of common reed (Kiviat *et al.*, 2002).

The majority of studies of fishes and invertebrates suggest that the nursery value of common reed marshes is inferior to that of marshes dominated by native plant species. Invertebrate community structure has been observed to change following invasion by common reed (*e.g.*, Talley and Levin, 2001). Studies in the Hackensack Meadowlands suggest the epifaunal community of smooth cordgrass is more diverse and abundant than the community of common

reed (Robertson and Weis, 2002). This may be due to the poor food quality of live leaves and litter of common reed for many common invertebrates (*e.g.*, gammarid amphipods, Agnew *et al.*, 2003). Common reed-dominated marshes in the Meadowlands and elsewhere in New Jersey (Able and Hagan, 2000; 2003; Raichel *et al.*, 2003) had substantially lower abundances of larval and juvenile mummichog than cordgrass-dominated marshes. Mummichog is widely recognized as one of the most important and pollution-tolerant fish species inhabiting the Meadowlands and other marshes. Able and Hagan's (2003) results suggest that *Phragmites* marshes produce less overall biomass of fish and wildlife than native estuarine marsh communities. Studies in other areas have made similar observations comparing use of cordgrass- and common reed-dominated marshes by larval and juvenile fishes, the most abundant life-history stages inhabiting estuaries (Fell *et al.*, 1998).

3. Interaction of Haplotype M with Environmental Contaminants

Long recognized for its tolerance of various pollutants (*e.g.*, heavy metals, hydrocarbons), common reed was at one time widely planted to limit adverse environmental effects in contaminated sites (Mulamoottil *et al.*, 1999) and to improve water quality in constructed wetlands and restoration sites in this and other countries (*e.g.*, Botch and Light, 1994; Brooks and Agate, 2001). Even today, one widely perceived benefit of common reed in the Meadowlands is its ability to stabilize wetlands and other areas (*e.g.*, landfills) that have highly contaminated sediments, which can be toxic to native plant species. Dense stands of common reed limit the spread of certain contaminants and reduce their adverse biological impacts upon ecosystems by:

(1) restricting sediment mobility (Leonard *et al.*, 2002);

(2) trapping new organic and inorganic matter, which may sequester additional contaminants (Rooth *et al.*, 2003);

(3) incorporating contaminants into various living tissues and its litter (*e.g.*, Ye *et al.*, 1997; Windham *et al.*, 2003); and

(4) facilitating bioremediation of contaminants (Tischer and Hüber, 2002).

Although common reed may serve well in "localizing" certain contaminants, common reed may also affect a wetland's capacity to serve as a "sink" or "source" of contaminants (Weis and Weis, 2004). For example, common reed appears to process and excrete certain contaminants (*e.g.*, lead) into the water at slower rates than cordgrass species (*e.g.*, Windham *et al.*, 2001a; 2001b). Common reed also sequesters more of those same contaminants below ground than other plant species (*e.g.*, Windham *et al.*, 2001a; 2001b). Finally, stem litter of common reed initially (during the first 2 years) breaks down and releases heavy metals (lead, copper, and zinc) more slowly but subsequently decomposes and releases metals at rates similar to decomposing litter of smooth cordgrass (*e.g.*, Windham *et al.*, 2004). Studies conducted in the Meadowlands and elsewhere suggest that restoration activities replacing common reed with other plant species potentially affect the distribution and abundance of contaminants. However, because of differing results, studies to date indicate the need to base decisions on site-specific information. The

effects of replacing common reed with cordgrass on contaminant fates (*e.g.*, their distribution and accumulation in the biota) in the Meadowlands are largely unknown. Thus, replacing common reed on heavily contaminated sites is not advisable until the availability, bioaccumulation, biomagnifications, and effects of contaminants are better understood.

4. Control

Control of common reed presents a considerable challenge to the overall restoration of the Hackensack Meadowlands, most of which provides ideal salinity (0-15 ppt) and nutrient (*e.g.*, high nitrates and ammonium; New Jersey Meadowlands Commission, 2005c) conditions for growth of Haplotype M. High salinity, sulfide, and prolonged flooding are known to restrict the invasion and spread of common reed (*e.g.*, Osgood *et al.*, 2002); as a result, excavation of tidal marshes to improve tidal flooding and increase salinities has been a common feature of most restoration projects in the Meadowlands to control proliferation of Haplotype M. However, common reed has a propensity to re-invade restoration sites from which it has been eradicated. Another consideration is that opening sites to increased tidal flow and exchange where contaminant remediation has been achieved may exacerbate contaminant loading.

Widely employed methods to eradicate and/or control common reed include chemical treatments, mechanical treatments, and combination (mechanical and chemical) treatments. The most efficient eradication method to date is aerial or broadcast spraying with a glyphosate-based herbicide; other methods of application of glyphosate (*e.g.*, wipe-on or injection) may be more effective but are labor-intensive and costly. Because glyphosate is a broad-spectrum herbicide (*i.e.*, kills other plant species) and is toxic to invertebrates, mechanical methods (disking, burning) are sometimes preferred for small sites or for preventing damage to other vegetation. Glyphosate has been used to eradicate common reed from most restoration sites in the Meadowlands (*e.g.*, Marsh Restoration Inc., Mitigation Bank; Louis Berger Group, Inc., 2000). Irrespective of the method used, repeated treatment is often required. Removal or eradication by herbicides is rarely 100 percent effective, nor is it long-term. Treated areas generally require additional applications. Thus, programs to monitor the treatment success and reinvasion of sites by common reed are recommended (Chesapeake Bay *Phragmites australis* Working Group, 2003), and plans for repeated treatments need to be taken into account.

The Service has extensive experience in restoring *Phragmites*-dominated wetlands in New Jersey. The Service's *Partners for Fish and Wildlife* Program has demonstrated on-the-ground success in restoring wetlands sites on a number of private lands invaded by common reed. From 1999 to the present, common reed occurred on and was eradicated from approximately 80 percent of the 5,500 wetland acres in New Jersey restored by this program; other invasive species (*e.g.*, purple loosestrife, see below) occurred on and were eradicated from the remaining acreage that was restored. Similarly, the Service's *Coastal* Program, which focuses exclusively on coastal watersheds, restored more than 2,100 acres of wetlands in the state on private and public lands from 1997 to 2004. The Service's experience in successfully eradicating common reed from other restoration sites in New Jersey will prove beneficial in future efforts to restore marshes in the Meadowlands, particularly on those sites where contamination has been remediated.

Research into biological control of common reed is underway in the United States and elsewhere and offers some promise for the future. Approximately 30 arthropods in North America are reported to attack common reed; however, the majority of these species appear to have been introduced also, at least some of them along with Haplotype M (Tewksbury *et al.*, 2002). Native herbivorous (plant-eating) insects of common reed include one gall midge (*Calamomyia phragmites*), and the larvae of certain skippers (*Ochlodes yuma, Poanes viator*) and dolichopodid flies (*Thrypticus* spp.); however, these native insects apparently target only native strains (haplotypes) of common reed (Tewksbury *et al.*, 2002). Thus, most research into biological control of common reed has focused on the exotic pests of common reed already introduced into the United States (Blossey *et al.*, 2003). Some of these pest species (*e.g.*, an eastern European gall midge, *Lasioptera hungarica*, and several gall flies in the genus *Lipara*) appear to feed solely on common reed. Also, more than 100 other pest species of common reed occur outside the United States (Tewksbury *et al.,* 2002). The potential of some of these species to provide biological control of Haplotype M is being evaluated in laboratory experiments conducted by Cornell University and the University of Rhode Island. If any of these species is effective in controlling Haplotype M, additional research will be needed to establish that these species do not feed on native haplotypes of common reed or on other native plant species (*e.g.*, other marsh grasses). Some non-native herbivore species of common reed appear to be spreading along waterways from the NY-NJ Harbor area (Blossey *et al.*, 2003). To date, it is not known if these non-native insects are capable of providing effective biological control of Haplotype M in the Meadowlands. Regardless of the control mechanisms and restoration activities that are implemented, common reed is likely to remain one of the major vegetative cover types in the Meadowlands for a long time to come. Some have questioned whether it is actually feasible or desirable to eradicate *Phragmites* from the Meadowlands entirely, but without its removal, restoration will never truly be achieved and the potential diversity of plant and animal communities will be diminished.

C. PURPLE LOOSESTRIFE (*Lythrum salicaria*)

Purple loosestrife (Figure 58) has occurred in New Jersey for nearly 200 years (Torrey *et al.*, 1819); during the past 10 to 20 years, it has invaded freshwater marshes, wet fields, and drainage ditches throughout much of the state. The spread of purple loosestrife throughout most states has been facilitated by its widespread use for landscaping and honey production (Thompson *et al.*, 1987). Despite its recognition as a problem invasive species, purple loosestrife remains widely available at nurseries throughout New Jersey, including some within the HRW. Because its numerous seeds are readily dispersed, established landscape plants serve as source populations for introduction of the species into new locales and reintroduction into wetlands from which it has been eradicated. In the HRW, purple loosestrife has established populations on several freshwater wetland sites (*e.g.*, Kearny Marsh, Teterboro Woods, Losen Slote, Harrier Meadows), public parks (*e.g.*, Laurel Hill Park, New Bridge Landing, Johnson Park), and many privately owned properties. The infestation of purple loosestrife is also conspicuous in the adjacent Passaic River basin, especially in the glacial Lake Passaic central wetlands.

Figure 58. Purple loosestrife growing in Teterboro Woods; a close-up of the flowers in the inset. Purple loosestrife flowers are attractive to butterflies and other pollinators; the plant's attractiveness has facilitated its use for ornamental and landscape plantings.

1. Biology and Effects on Wetlands

Like many invasive plant species, purple loosestrife exhibits tremendous seed production (more than one million per plant; Welling and Becker, 1990) and may form a monoculture under some growing conditions (Thompson *et al.*, 1987). Purple loosestrife is known to displace native wetlands plant species (*e.g.*, Gaudet and Keddy, 1988), including plant species that provide forage for waterfowl (*e.g.*, *Potamogeton* spp., *Utricularia vulgaris*, *Lemna minor*; *e.g.*, Haramis and Thompson, 1985). Stands of purple loosestrife may also alter hydrology and nutrient cycling (Emery and Perry, 1996; Dixon and Johnson, 1999) and the distribution and abundance of native invertebrates (*e.g.*, Gardner *et al.*, 2001). The species may adversely affect use of wetlands by certain wildlife (*e.g.*, waterfowl, Faanes, 1979; bog turtle, New Jersey Department of Environmental Protection, 2003). Purple loosestrife-dominated wetlands have been reported to support higher bird densities but lower bird diversity than other vegetative communities (Whitt *et al.*, 1999).

2. Control

Since 1997, the New Jersey Department of Agriculture, Beneficial Insect Rearing Laboratory has reared and released two beneficial insects (*Galerucella pusilla* and *Galerucella calmariensis*) that feed upon the leaves of purple loosestrife. These insects have been highly effective in controlling purple loosestrife on some loosestrife-dominated sites in New Jersey but have required repeated releases (of approximately 10,000 beetles per acre) for several years in order to generate sufficient beetle populations and densities to achieve control of loosestrife on other sites (Scudder *et al*., 2002). These beetles have been used successfully in the HMD to control purple loosestrife at the NJMC's Harrier Meadows marsh restoration site and in the Great Swamp NWR approximately 20 miles west of the Meadowlands.

Some researchers (*e.g.*, Hager and McCoy, 1998) are not convinced that the adverse impacts of purple loosestrife on plant communities and wildlife are sufficient to justify releasing non-native insects. Other researchers (*e.g.*, Galatowitsch *et al.*, 1999) have recognized the absence of herbivory as the major contributor to the explosive growth of purple loosestrife in North America. Although purple loosestrife may not always form dense, monotypic stands everywhere under all conditions (*e.g.*, in southeastern New York wetlands; Morrison, 2002), the potential for dense stands of purple loosestrife to alter wetland ecosystems and promote succession toward terrestrial habitats suggests that its abundance should be closely monitored to avoid extensive control efforts.

D. JAPANESE KNOTWEED (*Polygonum cuspidatum = Fallopia japonica)*

Japanese knotweed (Figure 59), a member of the buckwheat family (Polygonaceae), is believed to have been introduced as an ornamental plant to the United States through the United Kingdom's Royal Botanical Gardens, which began distributing the plant in 1855 (Mehrhoff *et al.*, 2003). In addition to its use as an ornamental plant (especially for landscape screening), Japanese knotweed was originally used in this and other countries for erosion control. It was

reported as naturalized in Atlantic Highlands, New Jersey and in several northeastern cities (*e.g.*, Philadelphia, Schenectady) in the mid 1890s. Information was being published on how to eradicate this plant in gardens by 1938 (Mehrhoff *et al.*, 2003). This species is now widespread in the United States in a variety of disturbed habitats (*e.g.*, abandoned fields, roadsides, utility rights-of-way) and thrives in riparian and wetland areas in at least 36 states (National Park Service, 2005). Japanese knotweed is widespread at a number of wetlands sites throughout the Meadowlands (*e.g.*, Kearny Marsh, Teterboro Airport, Harrier Meadow) and also occurs extensively throughout many other freshwater wetlands and riparian areas within the HRW (*e.g.*, Laurel Hill Park, Overpeck County Park, Teaneck Creek Conservancy, Schmidt's Woods).

Figure 59. A close-up of a thicket of Japanese knotweed, showing its leaves and flowers. Note the location of the leaves on alternating sides of each stem.

Photo courtesy of Rutgers University

1. Biology and Effects on Wetlands

Comparatively little information is available on the biology of this plant in the United States; however, information is available on this plant in Europe and especially in the United Kingdom, where Japanese knotweed has been banned from planting or growing outdoors since 1981 (Shaw and Seiger, 2002). Once established, populations are persistent and spread downstream. This species spreads quickly to form dense thickets that exclude native plant species, reduce native herbivorous insect populations, and provide little value as wildlife habitat (Holzner, 1982; Beerling and Dawah, 1993). Japanese knotweed survives floods and rapidly colonizes scoured shores and islands but does not provide year-round erosion control in riparian areas as do other native species because it dies back in winter (Child *et al.*, 1992). Slow rates of decomposition of its leaves and stalks relative to native riparian species (Seiger, 1997) result in considerable standing biomass that may create a fire hazard (Ahrens, 1975). In urban areas, Japanese knotweed has been reported to grow through sidewalks and building foundations (Japanese Knotweed Alliance, 2005).

2. Control

Much debated until recently, the common invasive form of Japanese knotweed is now identified as *Fallopia japonica*, a relative of other exotic species (*e.g.*, *Polygonum* and *Reynoutria* spp.; Shaw and Seiger, 2002). Japanese knotweed may hybridize with the related species (*e.g.*, hybrids with other *Polygonum* and *Fallopia* species are already known) and create additional invasive forms (Japanese Knotweed Alliance, 2005). Herbicide treatment on a watershed basis is effective but usually only when repeated for several years (Shaw and Seiger, 2002); research into biological control with insects and other agents (*e.g.*, fungi) are ongoing in the United Kingdom but have shown limited promise to date. Given the abundance of Japanese knotweed in the Meadowlands and its adverse impacts elsewhere, a management program to consider eradication and control measures for this species should be established immediately.

E. TREE-OF-HEAVEN (*Ailanthus altissima*)

A native of China, tree-of-heaven (Figure 60) was reportedly first brought to North America from England by William Hamilton, a Pennsylvania gardener, for ornamental purposes in 1784. Commercially distributed in the east as early as 1840, the species was spread throughout the U.S. and especially to the West Coast by Asians, who used the plant for medicinal purposes (Hu, 1979). Its tolerance to air pollution contributed to its being planted throughout some metropolitan areas in the United States, though it has also been used as an indicator species for ozone pollution (Gravano *et al.*, 2003). All parts of this plant exhibit a strong odor, likened to cat urine or rotting peanuts; this property has given rise to one local name, stinking sumac. Tree-of-heaven is common in a number of different habitats throughout the New York Bight, including abandoned fields and lots, roadsides, and coastal dunes. This species is possibly the most common tree of disturbed uplands and wetland/upland borders in the Meadowlands (Spendiff, 2004).

Figure 60. Tree-of-heaven. A tree in full flower in summer (upper photo) and a close-up of the upper and lower surface of the leaflets (lower photo). Small glandular notches occur on the underside of leaflets in summer.

Photos courtesy of the University of Georgia Bugwood Network

1. Biology and Effects on Wetlands

A member of the mostly tropical quassia family (the Simaroubaceae), tree-of-heaven grows rapidly, seeds prolifically, and forms dense thickets that displace native plant species. This species thrives on poor soils and is tolerant of many air pollutants, sulfide, and mercury (Kozlowski, 2002; Mehrhoff *et al.*, 2003). This species' ability to spread from metropolitan areas and displace native plants may also result in part through its production of allelopathic chemicals that have non-selective, post-emergent herbicidal properties similar to glyphosate or paraquat (De Feo *et al.*, 2003; Ferguson and Rathinasabapathi, 2003).

This species has not only been recognized for a variety of adverse impacts on native plants, but for other diverse impacts in urban areas (*e.g.*, root growth clogging sewer lines, growing into building foundations). Although the tree is reportedly unpalatable to native herbivores and only its seeds are consumed (Mehrhoff *et al.*, 2003; Howard, 2004), Spendiff (2004) reports it is variously used by some insects and other arthropods, and its wood is easily excavated by cavity-nesting birds. Eastern cottonwood (*Populus deltoides*), grey birch, native sumacs (*Rhus* spp.), and box elder (*Acer negundo*) are native species that have likely been displaced by tree-of-heaven. The species is prohibited or listed as a noxious invasive or weed in 43 states (U.S. Department of Agriculture, 2005); however, despite threats to native species, *Ailanthus* seeds are still readily available in nurseries and seed catalogs in the United States. Because of its tolerance of polluted air and barren soils, tree-of-heaven may also be of some benefit in highly urbanized landscapes where native tree species are unable to grow and thus provide some habitat requirements such as roosting or nesting sites for migratory birds.

2. Control

Although controlled with several herbicides that can be applied in different ways, this species is reported to re-sprout from root or stem fragments. Effective control thus requires careful and repeated herbicide treatments and girdling (National Park Service, 2003). Natural control methods appear limited: fungi that kill tree-of-heaven (*e.g.*, *Verticillium* spp., *Fusarium oxysporum*; National Park Service, 2003; Connecticut Agricultural Experiment Station, 2005) also kill many native tree species.

F. OTHER INVASIVE PLANTS, MARINE SPECIES, AND PATHOGENS

Many invasive species are a familiar component of the region's urban landscape and are no longer widely recognized as non-native. Thus, problems caused by invasive species have not been fully recognized and have been considered a lower priority for resolution than other issues throughout the NY-NJ Harbor estuary. For example, comprehensive sampling of aquatic and wetland habitats throughout the NY-NJ Harbor estuary for invasive species did not begin until 2003 (U.S. Environmental Protection Agency, 2003d). Thus, except for well-recognized and conspicuous species such as common reed that may dominate a landscape, the distribution and abundance of invasive plant and animal species in the Meadowlands and elsewhere in NY-NJ Harbor estuary are not well known. Available information suggests that invasive species,

including pathogens, are probably more abundant and widespread than is generally recognized, and their impacts are unknown.

There are approximately 2,200 species of plants in New Jersey, of which roughly 1,300 are believed to be non-native (Snyder and Kaufman, 2004), originating mostly from Europe or Asia. Most were introduced as ornamental plants or for specific purposes (*e.g.*, food, fiber, medicine). New Jersey has approximately the same number of non-native plant species as New York, California, Florida, Pennsylvania, and Massachusetts, five of the states among the highest in reported totals of non-native plant species (Williams and Meffe, 1998). The large number of non-native plant species in these states is due to their long history as commerce and transportation centers. Historically, the single most significant mechanism for introducing non-native plants (and animals) in the northeastern U.S. has been the dumping of foreign ballast soils and ballast waters at seaports (Snyder and Kaufman, 2004).

The problems presented by invasive plant species to restoring the Meadowlands are likely to increase. In addition to the four invasive plant species discussed in this report, the NJMC has begun efforts to control other invasive plant species, including mile-a-minute (*Polygonum perfoliatum*), Amur peppervine (*Ampelopsis brevipedunculata*), and hedge false bindweed (*Calystegia sepium*, also known as wild morning glory). A preliminary list of native and non-indigenous plants species occurring in the Hackensack Meadowlands and the surrounding watershed is being generated by the Service from unpublished sources (*e.g.*, field-trip reports by the Torrey Botanical Society [Brooklyn Botanic Garden, 2004b] and collection records of the Brooklyn Botanical Society) and field surveys to assist in determining the magnitude of the invasive plant problem in both the Meadowlands and the surrounding watershed.

Many invasive animals, especially terrestrial species, are accepted as common components of the urban landscape in the New York metropolitan area. Nonetheless, efforts should be made to identify all terrestrial invasive animal species and determine their impact upon native fish and wildlife populations, especially upon federally and State-listed species for which recovery plans are being implemented and for other species of concern. For example, the common street pigeon or rock dove (*Columba livia*, a European species) in urban areas may contribute to bioaccumulation of certain contaminants in peregrine falcon (*e.g.*, DeMent *et al.*, 1986), and its excrement is estimated to harbor over 50 diseases. Also, invasive bird species may differ in their susceptibility to recently-introduced diseases of wildlife and humans such as West Nile virus, and facilitate the spread of the virus among wildlife and humans (see below; *e.g.*, Male, 2003). Exotic earthworms, likely brought in ballast soils, are being increasingly recognized for their adverse impacts on North American forest ecosystems (*e.g.*, altered carbon, nitrogen, and phosphorus cycles, which stress certain trees; Groffman *et al.*, 2004). Thus, the impacts of exotic and invasive species that have been widely accepted as "naturalized" components of the landscape may have novel, poorly recognized impacts on native fish and wildlife (Rappole *et al.*, 2000; Male, 2003).

Approximately 300 aquatic animal and plant species have been identified as invasive in estuaries and coastal waters of the United States (*e.g.*, Ruiz *et al.*, 2000; Ray, 2005). Several invasive mollusks have been collected in the NJMC's fishery surveys (*e.g.*, dark false mussel [*Congeria leucophaeta*], Baltic macoma clam [*Macoma bathica*]; Kraus and Bragin, 1989; Bragin *et al.*,

2005); however, the total number of invasive species present in the Meadowlands and the HRE is presently unknown. Approximately 200 invasive species are known in Chesapeake Bay (Ruiz *et al.*, 2000) and San Francisco Bay (Cohen and Carleton, 1998). Clearly, additional effort must be made to monitor and assess invasive species in the Meadowlands and where these pose a significant ecological risk; methods for their control should be sought and implemented.

Studies in other areas (*e.g.*, Ray, 2005) indicate that some invasive species have the potential to alter the ecology of the Meadowlands and adversely affect native fish and wildlife populations. Ballast-water released from large ships has introduced non-native meroplanktonic (planktonic as larvae only) and holoplanktonic (planktonic as larvae and adults) species (Lavoie *et al.*, 1999). Invasive copepod species now dominate the zooplankton of many estuaries (Cordell and Morrison, 1996), and estuarine faunas are increasingly dominated or affected by invasive species (Ray, 2005). For example, introduction of the Amur River corbula clam (*Potamocorbula amurensis*) into San Francisco Bay displaced entire benthic assemblages (Carlton *et al.*, 1990) and significantly reduced phytoplankton abundance, which caused a "trophic cascade" affecting the biology of dependent species including zooplankton and fishes (Alpine and Cloern, 1992; Kimmerer *et al.*, 1994; Feyrer *et al.*, 2003). Even with recent development and implementation of ballast water management and programs to reduce the likelihood of introducing invasive species (U.S. Coast Guard, 2005), programs are needed to assess the status of existing exotic and invasive species in the Meadowlands, prioritize species for development of control programs, and monitor and target new exotic species before they spread. Successful eradications of established marine invaders in Australia (a mussel, *Mytilopsis* sp.; Bax *et al.*, 2002) and California (a polychaete worm, *Terebrasabella hetrouncintata*; Culver and Kuris, 1999) have been reported; however, in both instances, invaders were detected during their initial colonization.

Finally, no program addressing impacts of invasive species would be complete without some focus on invasive pathogens, that is, invasive species that cause disease. Some of the most familiar invasive pathogens that have already devastated native plant and animal communities and ecosystems throughout North America were introduced in the New York City area. For example, Chestnut blight (formerly *Endothia parasitica,* now known as *Cryphonectria parasitica*, *e.g.*, Gryzenhout *et al.*, 2005) was introduced into the New York City area shortly before 1900. The blight contributed substantially (along with another invasive fungus, *Phytophthora cinnamomi*, likely introduced in the southeastern United States, Zentmyer, 1988; Schlarbaum *et al.*, 2002) to essentially eliminating native American chestnut trees from more than 9 million acres in North America (American Chestnut Foundation, 2004), where over large regions of its range it had previously constituted over 50 percent of the forest tree biomass. West Nile virus (flavivirus), a causative agent of a type of meningitis and encephalitis in horses and humans, first appeared in the New York City area in 1999 (Lanciotti *et al.*, 1999). West Nile virus is now known to occur in more than 250 species of birds and 18 species of mammals in North America; there is considerable concern about its potential impacts on raptor populations and on other bird species as the virus is spread throughout the Americas (U.S. Geological Survey, 2005c). Other pathogens were first recognized in other port cities (*e.g.*, Dutch elm disease in Cleveland, Ohio; Schlarbaum *et al.*, 2002). Many invasive pathogens are often introduced along with other invasive species (*e.g.*, plants, insects); therefore, it is prudent and cost-effective to identify and intercept invasive species in port areas prior to infestations.

G. SUMMARY

As the movements of goods and people around the globe increase, the opportunities for invasive species to arrive and spread throughout the United States are similarly increasing (Carleton, 2001). Invasion of coastal ecosystems surrounding major urban seaports by exotic species is considered a national problem (National Invasive Species Council, 2001). Agencies participating in the restoration of the Meadowlands and elsewhere in the NY-NJ Harbor estuary have not addressed invasive species in a coordinated manner to date. Harbor-wide efforts must be developed to prevent current and future invasions of non-native species that may have considerable adverse impacts on indigenous fish and wildlife populations in the region. Development of comprehensive and coordinated programs to monitor, assess, and control invasive species are increasingly common in seaports and surrounding urban areas elsewhere in the United States (*e.g.*, Chesapeake Bay, San Francisco Bay). Such programs have done much to inform the public of the problems and costs associated with invasive species. Human activities are the primary cause of invasive-species introductions. Public education, blocking pathways of future introductions, on-the-ground control (including biological, mechanical, and chemical), working with local nurseries to supply native plant materials and seeds, and habitat remediation and restoration are necessary to stop the transport and introductions of exotic and invasive species in the Meadowlands and the surrounding NY-NJ Harbor estuary.