INVASIVE SPECIES AND COMPENSATORY

WETLAND MITIGATION SUCCESS

by

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ABSTRACT

Invasive species and compensatory wetland mitigation

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Polygonum cuspidatum, *Lythrum salicaria,* and *Phalaris arundinacea* are invasive plant species that pose significant threats to the legal and functional success of compensatory mitigation sites because they have the ability to form dense monostands. Many compensatory mitigation wetlands fail to meet permit requirements because they exceed the 10% standard for aerial coverage of invasive species, but may still be providing functional replacement as required under the No-Net-Loss policy.

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Chapter 1

Introduction

In the last 200 years over 50% of the original wetlands in the United States were destroyed (Dahl 1991; Mitsch and Gosselink 2000). As the public's understanding of wetland functions and values increased, a variety of policies and laws were passed to protect wetlands (Mitsch and Gosselink 2000). In 1972 Congress passed the Federal Water Pollution Control Act, later amended as the Clean Water Act, which regulates the placement of dredge and fill materials in the Waters of the United States, including wetlands under Section 404. Compensatory mitigation is a significant part of the Section 404 permitting process (Kruczynski 1990), not because it is required under the Clean Water Act, but because the issuance of a Section 404 permit triggers mitigation requirements under the National Environmental Policy Act (Berry and Dennison 1993).

Many compensatory mitigation wetlands are not considered successful because they fail the 10% aerial coverage standard for invasive species. Invasive plants reduce biodiversity in wetlands (Wilcove, et al. 1998). Invasive plants that reproduce vegetatively and form monocultures are the most threatening to native plant communities (Pysek 1997; Kercher, et al. 2005). When species composition shifts in compensatory wetland mitigation some of the functions expected to be replaced by the mitigation wetlands may be lost, which is in direct conflict with the concept of "no-net-1oss" of wetlands and wetland functions. The 10% aerial coverage standard may not be appropriate for all invasive species and all compensatory mitigation sites because reference wetlands may also exceed the 10% aerial coverage standard.

Instruments of Corps Regulation

The Department of the Army has been involved in regulating certain activities in the nation's waters since 1890. Initially the mission of the U.S. Army Corps of Engineers (Corps) Regulatory program was to protect and maintain the navigability of the nation's waters. **In** the late 1960's, the Corps' regulatory saw a dramatic change with the addition of a second focus as a product of several new laws and court decisions. Today, the Corps' mission is still evolving as public needs and policy change, and as case law and new statutory mandates add to the complex character of the program's authority.

The legislative genesis of the regulatory program are the Rivers and Harbors Acts of 1890 (superseded) and 1899, which establish permit requirements to prevent unauthorized obstruction or alteration of any navigable water of the United States (33 U.S.c. 401, et seq.). The most regularly exercised authority is contained in Section 10 (33 U.S.c. 403) which covers "construction, excavation, or deposition of materials in, over, or under such waters, or any work which would affect the course, location, condition, or capacity of those waters."

In 1972, amendments to the Federal Water Pollution Control Act added Section 404 authority (33 U.S.c. 1344) to the Corps' regulatory program. Under section 404 of the Clean Water Act, the Chief of Engineers, acting for the Secretary of the Army, is authorized to issue permits for the discharge of dredged or fill material into waters of the

United States, including navigable waters and wetlands, at specified disposal sites after giving notice of the proposed discharge and opportunity for the public to comment at hearings.

In the 1975 decision of *Natural Resources Defense Council v. Riverside Bayview Homes, Inc.,* included wetlands in the definition of waters of the United States as defined by the Federal Water Pollution Control Act, prior to the decision the Corps only regulated Section 404 dredge and fill activities in navigable waters. In 1977 the Federal Water Pollution Control Act was amended and given the name Clean Water Act, and the Act's latest amendments in 1987 have changed criminal and civil penalties and added an administrative penalty provision.

The selection of placement sites for dredge or fill material is done in accordance with Section 404(b)(1) guidelines developed by the U.S. Environmental Protection Agency (40 CFR Part 230). Navigable waters of the United States are those waters that are subject to the ebb and flow of the tide, and/or are presently used, or have been used in the past, or may be susceptible to use to transport interstate or foreign commerce. In waters affected by the tide, the landward limit of the navigable water of the United States is the mean high water; in non-tidal waters the landward limit of navigable waters is the ordinary high water. In non-tidal waters where adjacent wetlands are present, the Clean Water Act extends jurisdiction to the limits of the adjacent wetlands, as defined by the 1987 Corps of Engineers Wetland Delineation Manual (1987 Manual).

The Corps issues two types of Department of the Army permits, Standard and General permits, with two permits in each category. The basic vehicle for authorization used by the Corps is the standard individual permit. Standard permits include standard individual permits and letters of permission, which both take into account the public interest when processing a permit decision. General permits are issued on a nationwide and regional basis to authorize categories of activities that are substantially similar in nature and cause only a minimal individual or cumulative adverse effects on the aquatic environment. There are currently 44 nationwide permits issued, with several either expired or revoked on a regional basis. A summary of Corps permit decisions is provided in Table 1.

Table 1: Summary of Corps permit decisions, by fiscal year

Permit Type	Fiscal Year				
	1999	2000	2001	2002	2003
individual permit	4,168	3,883	4,159	4,023	4,035
letters of permission	2,687	2,560	3,066	3,258	3,040
nationwide permit	44,913	41,385	37,088	35,768	35,317
regional general permit	38,595	40,702	38,759	38,125	43,486
denials	221	180	171	128	299
Totals	90,584	88,710	83,243	81,302	86,177
Source: Engineers 2006					

Wetlands

Wetlands have been dewatered and filled for farming, development, mosquito control, and numerous other activities throughout the nation's history (Toxicology 2001). Wetlands provide numerous ecosystem functions that are also lost when wetlands are filled. These include water quality improvement, flood storage, ground water recharge, shoreline stabilization, and habitat functions. **In** 1988 at the National Wetland Policy Forum the concept of "no-net-Ioss" was introduced by the Conservation Foundation, and was subsequently endorsed by the administration of President George H. W. Bush, and articulated in the 1990 Memorandum of Agreement (MOA) between the U.S. Environmental Protection Agency and the Corps. The Memorandum of Agreement set up a mitigation sequence that recognized that wetlands provided important ecosystem functions that were important to the goals of the Clean Water Act, which are to "restore and maintain the chemical, physical, and biological integrity of the Nation's waters" (33 U.S.c. 1251(a)). The U.S. Supreme Court has found that the larger goal of the Clean Water Act is the improvement of water quality, and that wetlands adjacent to navigable waters "playa key role in protecting and enhancing water quality... [and] serve significant natural biological functions" (States 2002). Working with the Memorandum of Agreement as a framework, the Corps stated that the goal of compensatory mitigation was to replace the affected aquatic resource functions that will be lost or impaired by the project, or to maintain or improve the overall aquatic environment (Corps 2006).

Mitigation may include avoiding, minimizing, rectifying, reducing, or compensating for resources that will be lost due to the construction of the proposed project. The Corps regulations state that losses will be avoided to the extent practicable, but impacts for many projects are unavoidable. 33 CFR 320.4(r) Provides general guidance about mitigation that is required for the Corps regulatory program; that mitigation will be directly related to the impacts of the proposed project, appropriate in scope and degree of impacts, and that the mitigation can be reasonably enforced. Mitigation is an important concern when evaluating Department of the Army permit applications because mitigation can be required to ensure that the project is adequately compensating for the impacted aquatic resources (40 CFR Part 230.

Table 2: Wetland impacts authorized by Corps permit and wetland compensatory mitigation required

Fiscal Year	Wetland impacts permitted (acres)	Wetland compensatory mitigation required (acres)
1999	21,556	46,433
2000	18,900	44,757
2001	24,089	43,832
2002	24,651	57,821
2003	21,413	43,550
Source: Engineers 2006		

After an application for a Department of the Army permit has been considered and either found to be consistent with activities already authorized under general permits, or not found to be contrary to the public interest and otherwise compliant with Corps regulations, a permit is issued contingent on appropriate approved mitigation being constructed if the impact threshold is over 0.1 acres.

The 1990 MOA stipulates that the Corps consider the functional values lost when determining compensatory mitigation requirements for a section 404 permit, and states that in-kind compensatory mitigation is generally preferable to out-of-kind mitigation. Therefore, under the section 404 permit review process, the Corps must attempt to achieve replacement of the impacted or lost wetland functions and values.

Ecological performance standards are used to establish that the approved compensatory mitigation is developing in the desired aquatic habitat and providing the expected ecological functions. To ensure the success of the approved compensatory mitigation, District Engineers may impose administrative and adaptive management requirements including: as-built surveys, performance bonds, real estate instruments for protection of mitigation sites, and long term management funding. These administrative requirements are intended to guarantee that the compensatory mitigation site is constructed as approved by the District Engineer, and that the mitigation is maintained and protected from future development. Adaptive management requirements may include modifications to management and maintenance of compensatory mitigation sites based on monitoring of ecological performance standards.

All compensatory mitigation is usually held to some type of performance standards (Engineers 2006). These standards are normally based on aquatic community structure and aquatic resource function as they relate to the criteria in the 1987 Manual for wetland hydrology, soils, and vegetation.

Species Invasions in Wetlands

In some cases, permit compliance is determined by survival of specific plantings. Plant species introductions from other eco-regions have put a strain on native plant communities in North America, and invasive species in wetlands can pose definite challenges for compensatory mitigation being able to meet aquatic community structure performance standards (Kennedy, et al. 2002). Mitchell and Gopal found that "there is some validity to the concept that disturbed ecosystems are the most susceptible to alien invasions," (Mitchell and GopaI1991).

Wetland mitigation sites are very susceptible to species invasions because they are typically devoid of vegetation, have multiple gaps within plant canopies, and have eutrophic water supplies (Toxicology 2001). Hobbs and Huenneke found that the habitat being colonized by an invasive species will be more invadable if there are gaps in the canopy or minor soil disturbances available for seedling colonization, conditions which are typical during and after construction at wetland mitigation sites (Hobbs and Huenneke 1992). Plants that invade compensatory mitigation wetlands are usually species with high seedling production and germination rates, and have the ability to spread vegetatively (Toxicology 2001).

The most frequently encountered invasive species in the Pacific Northwest is reed canarygrass *(Phalaris arundinacea),* a species that is difficult to eliminate because it spreads by both seeds and rhizomes, and creates monocultures that crowd out lower growing native plants (Merigliano and Lesica 1998). There is also great potential in the

Pacific Northwest for purple loosestrife *(Lythrum salicaria)*, a tall emergent hydrophyte, to creep onto mitigation sites, and this species has already shifted the species composition of many wetlands in the mid-west, and is causing alarm among wildlife managers (Stucky 1980; Balogh and Bookhout 1989). But the invasive species with the most invasion potential for compensatory mitigation sites is Japanese knotweed *(Polygonum cuspidatum)*, which poses many hazards to successful stream and riparian restoration projects, and which has already begun to appear in the Pacific Northwest, some in patches as large as half an acre (SolI and Morgan Undated).

Chapter 2: Japanese Knotweed *(Polygonum cuspidatum)*

History

Polygonum cuspidatum was first described in 1777 by Houttuyn as *Reynoutria japonica* and as *Polygonum cuspidatum* by Siebold in 1846 (Beerling, et al. 1994). It was not until the early part of the 1900's that these were discovered to be the same plant (Beerling, et al. 1994). The plant is referred to as *Polygonum cuspidatum* by Asian and American authors and as *Fallopia japonica* by European authors.

Polygonum cuspidatum is native to eastern Asia. It was introduced to the United Kingdom in 1825 as an ornamental (Townsend 1997), and in the late 1800's to North America as an ornamental and fodder plant, but rhizomes are reported to be toxic to some animals (Patterson 1976; Conolly 1977; Beerling, et al. 1994). Today in Japan it is used to hide garbage dumps and shield other unsightly areas as well as to stabilize seashore areas vulnerable to wave erosion (Locandro 1978; Jennings and Fawcett 1980). *Polygonum cuspidatum* is edible, newly emerged shoots can be used in salads, older shoots can be stripped and prepared in a manner similar to rhubarb, and are said to have an almond flavor (Kiple 2000; Doll and Doll 2002). Hu chang, the dried roots and stem of *P. cuspidatum*, are used by traditional Chinese medicine practitioners to treat high cholesterol and various other conditions (Huang 1999). *Polygonum cuspidatum* has also been used as a laxative (Lewis and Elvin-Lewis 1977). Roots contain a phytochemical called resveratrol, which his also found in red wine, that may shield against cancer and cardiovascular disease by acting as an antioxidant, antimutagen, and anti-inflammatory agent (Kimura and Okuda 2001). Aqueous extracts of *P. cuspidatum* have been found to inhibit the formation of new blood vessels in vitro (Wang, et al. 2004). During World War II, leaves of *P. cuspidatum* were used as a substitute for tobacco (Beerling, et al. 1994).

In Japan, *P. cuspidatum* is a pioneer species in the primary succession of volcanically disturbed slopes and is a colonizer in secondary succession of hilly or high mountain ecosystems on sites with direct sun exposure (Kanai 1983; Hirose and Tateno 1984).

European authors have found evidence that clones may persist on a single site for over 100 years (Pyek, et al. 2001). In North America, *P. cuspidatum* has been observed from Nova Scotia to North Carolina, and is widely distributed in the Midwest and in the coastal regions of Washington and Oregon, where it spreads along river banks, as well as wetlands, along roads and fence lines, and in other disturbed areas (Muenscher 1955; Pauly 1986).

Physical Description

Polygonum cuspidatum is an herbaceous perennial which can grow to a height of 10 feet. It is dioecious and reproduces by seed, but can also reproduce vegetatively by large rhizomes, which can be 20 feet or longer. Well established plants develop a central taproot. The hollow stems are simple and glabrous (non-haired) with thin membranous sheaths that extend from the erect base.

The leaves of *P. cuspidatum* are broad and ovate, truncating to cuneate at the leaf base, cuspidate at leaf apex, 5-15 cm long, 5-12 cm wide, with petioles 1-3 cm long. Flowers are greenish-white, 2-3 mm long, compactly arranged in axillary panicles. Male flowers have branched panicles on upright racemes with the distal end of the raceme in the highest position; individual panicles usually point up; 8-10 stamens with longitudinally dehiscing anthers. Female flowers are decumbent with the proximal end in the highest position; 3 styles, fruiting 6-10 mm long calyx. Both male and female flowers possess rudimentary organs of the other sex. Trigonous achenes are shiny black-brown and 3-4 mm long (Fernald 1950).

Reproduction

The primary mode of reproduction in the United States is through rhizomes which can be 15-20 meters long and which are dispersed when fragments of rhizomes are transported by water or more commonly when disturbed soil containing rhizomes is placed as fill; shoots commonly emerge in April and growth rates can exceed 8 cm per day (Locandro 1973; Conolly 1977; Locandro 1978). The capacity of *P. cuspidatum* rhizomes to generate viable shoots is affected by the source of rhizome fragments, size and depth in soil (Locandro 1973). *Polygonum cuspidatum* has been observed regenerating from internodal tissue (Locandro, 1973), and rhizomes fragments buried 1 meter deep can produce viable plants and have been observed growing up from two inches of impervious surface (Pridham and Bing 1975; Locandro 1978).

In Europe *P. cuspidatum* has been observed growing on soils with pH values ranging from 3.0 to 8.0 (Grime, et al. 1988). **In** its native Japan, *P. cuspidatum* it has been observed growing on sulphurous soils near volcanic fumaroles at pHs below 4.0 (Yoshioka 1974).

Seeds

Polygonum cuspidatum is both dioecious and gynodioecious, and has been observed to be subdioecious in New England, with male and female flowers on separate plants with males that sometimes set seed (Beerling, et al. 1994; Forman and Kesseli *2003). Polygonum cuspidatum* flowers from August to September in North America (Fernald 1950; Conolly 1977), and the main method of seed dispersal in North America is wind (Maruta 1976). Hirose and Tateno found high levels of seed production but low seedling survival in their 1984 study on Mt. Fuji, but noted that once a seedling had past the three-leaf stage, the seedling typically survived (Hirose and Tateno 1984). Wild plants in Asia are characteristically found in early successional environments, and first and second year seedlings can be found growing next to adult plants (Maruta 1981; Schnitzler and Muller 1998).

In Europe, seedling establishment in the wild has been noted, but several documented cases have turned out to be hybrids between *P. cuspidatum* and *P. sachalinense* (giant knotweed) (Bailey, et al. 1995). New genetic research has concluded that virtually all non-hybrid *P. cuspidatum* (referred to as *Fallopia japonica var. japonica)* in the United Kingdom are female, implying clonal growth (Hollingsworth and Bailey 2000).

Outside of Asia, "seeds do not appear to be a significant mode of reproduction" (Seiger 1995). Seiger found that 50% to 63% of seeds collected in Washington, D.C. germinated on a filter paper medium after a two year dormancy, while only 10% of seeds with no dormancy period germinated (Seiger 1995). Seiger also noted that the seeds collected appeared to be hybrids with *Polygonum aubertii*, and did not observed any seedling establishment in the field (Seiger 1995).

Forman, et al. found high viability of *P. cuspidatum* seeds from Massachusetts under various environmental treatments; seeds from the same parent were able to germinate in the fall almost immediately after seed set, or enter a dormancy period and germinate in spring (Forman and Kesseli 2003). Seeds in the Forman, et al. study remained viable in winter conditions whether attached to the parental plant, covered with soil, or exposed on the soil surface (Forman and Kesseli 2003). Forman, et al. did find that seedlings that germinated underneath well-established stands of *P. cuspidatum* were typically not able to survive because the canopy of the existing stands blocked most sunlight (Forman and Kesseli 2003). However, the Forman, et al. seeds dispersed into areas with open canopies did survive, and do not inevitably die at early stages as reported by Locandro and Seiger; who focused on *P. cuspidatum* areas which were already heavily infested and competition prevented any plant from growing, particularly seedlings (Locandro 1973; Seiger 1995; Forman and Kesseli 2003).

Locandro found in his 5 year study of *P. cuspidatum* in New Jersey that female plants often bore empty achenes, fertile males were rare, and plants that did germinate seldom developed past the three-leaf stage and did not survive beyond mid-summer (Locandro 1973). Forman, et al. reported drastically different findings in Massachusetts, in both greenhouse and field observations female plants isolated from males had ovaries that aborted with no remnant seed detected (Forman and Kesseli 2003). Additionally, Foreman, et al. found at least one fertile male plant within pollinating distance of each female plant, suggesting that isolation of female stands is not as dramatic as reported by Locandro (Locandro 1973; Forman and Kesseli 2003).

When *P. cuspidatum* invades riparian sites, simplification of forest structure can lead to decreases in small mammal habitat, and change nutrient cycling, prevent recruitment of large woody debris, disrupt the aquatic food webs of salmoinds, block fish passage, and simplify normally complex salmonid habitat (Potash 2001).

Chapter 3: Purple Loosestrife *(Lythrum salicaria)*

History

Purple loosestrife *(Lythrum salicaria)* is an emergent aquatic native to Europe and Asia that was first described by Turner in 1548. *Lythrum salicaria* was transported to the United States in the early 1800's in soil used as ballast for ships, livestock bedding, and as a garden plant (Hulten 1971). *Lythrum salicaria* became established so quickly in the costal eastern wetlands of North America that in the first edition of *A Flora of North America*, Torrey and Gray described *L. salicaria* as "probably native" (Voegtlin 1998). Torrey noted in 1877 that *L. salicaria* was "well established on the Northern R[ail] R[oad] of New Jersey, near Granton" (Torrey 1877). In 1879 *L. salicaria* was reported in abundance along the high water mark of the Hudson River, and in meadows on adjacent creeks (Rudkin 1879).

The colonization of *L. salicaria* into glacial marshes of the Midwest by 1900 is correlated with development in these wetlands habitats; construction of eastern canals, and marine traffic and trade extending into the Great Lakes region (Skinner, et al. 1994). In the 1940s disturbed areas in which *L. salicaria* colonizes increased significantly as construction commenced of the first series of interstate highways, and as more acreage was irrigated under the Federal Reclamation Act (Kuusvouri 1960).

By the late 1940's, *L. salicaria* was established in marine areas in the Pacific Northwest, and by 1985 Alaska and Montana were the only non-Southwestern states that had not reported *L. salicaria* (Thompson, et al. 1987). In the more recent studies *L. salicaria* was found to be invading new wetland sites by migrating down ditch and culvert drainage systems (Wilcox 1989). There has also been interest by bee keepers in using *L. salicaria* as a honey plant dating as far back as 1944, which could have contributed to the species spread into the west (Thompson, et al. 1987). More recently *L. salicaria* seeds have been found to be contaminating seed samples obtained from commercial suppliers of wildlife cover and prairie restoration plants, undoubtedly contributing to some current invasions (Thompson, et al. 1987).

Physical Description

Lythrum salicaria is a broad-leafed emergent aquatic perennial that can reach up to 8 feet in height. Leaves are lanceolate, terminating to cordate, usually opposite, but may also be alternate, or in whorls of three and four. An individual plant may have 30-50 angular annual stems that turn woody with age, rootstocks persist through winter (Skinner, et al. 1994). Some uncertainty exists in the literature as to whether *L. salicaria* rootstocks can send out rhizomes. Skinner, et al. said that *L. salicaria* may be considered a clonal plant with the rootstock acting as genet, and annual shoots as ramets (Skinner, et al. 1994). Ohwi described *L. salicaria* in Japan as "rhizomatous;" (Ohwi 1965) but this observation has been criticized as possibly referring to adventitious buds sprouting from lateral roots (Thompson, et al. 1987). Pearsall described *L. salicaria* as having "have tough rhizomes capable of penetrating the interstices of the hard substratum" (Pearsall 1918), while other British authors of the time only mention creeping rootstalks (Morse and Palmer 1925). In a more recent study the U.S. Fish and Wildlife Service reported adventitious buds on the buried stems, but no evidence of spread by rhizome activity after a thorough examination of plants throughout the United States and Canada (Thompson, et al. 1987).

Lythrum salicaria plants bloom July to September and seed set begins by late July. Spiked flowers are reddish-purple, six-petaled, showy, and have 5 sepals fused at the base into a tube. The bisexual complete and perfect flowers are insect-pollinated; self-pollination is achievable, but cross-pollination prevails (Thompson, et al. 1987). Each flower has 8-10 stamens of 3 distinct lengths, and three distinct style lengths, one of three pistil lengths and two different sets of stamen lengths in each flower. Shorter styles may be hidden within the calyx (Stout 1925). The trimorphic nature of *L. salicaria* flowers attracted the attention of Charles Darwin, who published two separate articles on the subject. Darwin noted that the three forms of flowers coexisted in wild populations in nearly equal frequencies (Darwin 1865). Kuusvouri found a high frequency of mid length style morphs in crowded stands of *L. salicaria* in Finland where vegetative reproduction was common (Kuusvouri 1960), but Halkka and Halkka while investigating 16 isolated populations of *L. salicaria* on small islands in the Gulf of Finland and reported that findings similar to Darwin's; the three style morphs occurred in nearly 1:1:1 frequencies (Halkka and Halkka 1974). The majority of the literature reported nearly equal frequencies of all style morphs, supporting the assertion that sexual reproduction is the driving factor in establishment and spread of *L. salicaria* (Shamsi and Whitehead 1974; Thompson, et al. 1987).

Seeds

The U.S. Fish and Wildlife Service reported the mean number of seeds produced by a single plant to be 2,700,000, with 30 stems per plant, 1000 capsules per stem, and 90 seeds per capsule (Thompson, et al. 1987). The main mode of seedling dispersal is floating seeds, which have been noted to sink upon contact with water, but rise after germination (Ridley 1930), but seeds have also been observed being dispersed by wind (Shamsi and Whitehead 1974), and by animals (Torrey 1931). Nilsson and Nilsson suggest that *L. salicaria* seeds can be dispersed by wind over snow and ice (Nilsson and Nilsson 1978). Thompson, et al. disputes this conclusion by noting that while seeds are light enough to carried by wind, observed densities of seedlings fall off dramatically within the first 10 m from the parent plant (Thompson, et al. 1987).

Shamsi and Whitehead found that 80% of seeds stored at 3°_4°C for three years were able to germinate, while 90% of freshly collected seeds were able to germinate (Shamsi and Whitehead 1974). Most *L. salicaria* seeds emerge by day 17, but low reserve of stored energy by the seeds suggests that floating germinated seeds would not survive beyond a few weeks (Shamsi and Whitehead 1974). Mitchell found that 75% of *L. salicaria* seeds exposed to diffuse light successfully germinated, versus 6% of seeds that were exposed to constant darkness (Mitchell 1926). Seeds need between temperatures between 15 and 20°C to germinate, and can germinate successfully on substrates with pHs between 4.0 and 9.1 (Shamsi and Whitehead 1974; Thompson, et al. 1987). Seeds or propagules that survive the winter must establish themselves in moist soil in late spring or early summer; summer-germinated seedlings which do not produce more than four pairs of leaves do not survive the following winter (Thompson, et al. 1987).

Seedlings of *L. salicaria* were found to be most affected by nitrogen deficiency, as compared to phosphorus and potassium deficiency (Shamsi and Whitehead 1977). Edwards, et al. noted that fecundity was similar among native *L. salicaria* populations in Europe exposed to low and medium nutrient treatments, but was significantly higher at both treatment levels amongst North American populations (Edwards, et al. 1998).

Effects of monostands on habitats

Many authors have also noted that invasions of *L. salicaria* often lead to changes in arthropod, mammal and avian fauna that rely on native plants, shelter, or breeding and nesting areas (Thompson, et al. 1987; Mal, et al. 1992; Kiviat 1996). This is because *L. salicaria* has the ability to radically shift species composition within wetlands.

Fernald recognized the potential for invasion in 1940 when he penned that ". . . the formerly unique and endemic flora of the estuary is being rapidly obliterated by . . . the purple loosestrife ... without mercy for the insignificant endemics ... " (Fernald 1940). In 1978, Coddington and Field suggested that competition between *L. salicaria* and Long's bulrush *(Scirpus longii)* may be partly to blame far the endangered status of S. *longii* in Massachusetts (Coddington and Field 1978). Rawinski expressed similar concern about a rare inland population of dwarf spike rush *(Eleocharis parvula)* in inland New Yark (Rawinski 1982).

Rawinski and Malecki conducted a three year study and found a negative correlation when comparing stem densities of *L. salicaria* and *Typha* (Rawinski and Malecki 1984). In 1994 the presence of *L. salicaria* in flood, control and infertile treatment plots caused an average 60% reduction in biomass of neighboring species (Keddy, et al. 1994). A study of 12 Minnesota wetlands in 1995 showed that increased *L. salicaria* biomass was associated with a decrease in *Typha* biomass (Emery and Perry 1995).

Weiher, et al. found that wetland microcosms inoculated with the seeds of 20 wetland plant species came to be dominated by *L. salicaria* after 5 years; most dicots had vanished from the wetland microcosms and despite 24 different treatments Weiher and colleagues concluded that initial planting conditions could not predict longer term trends in species competition in the wetland microcosms (Weiher, et al. 1996). Another 1996 study by Twolan-Strutt and Keddy found that *L. salicaria* was less sensitive to competition than *Carex crinita* by comparing biomass levels of roots and shoots in competition treatments (Twolan-Strutt and Keddy 1996).

Mal, et al. found in a four year study with differing initial density treatments of *L. salicaria* and *Typha angustifolia* that *L. salicaria* exceeded *T. angustifolia* in ramet production after year one, exceeded stem proportion in all treatments after year four (Mal, et al. 1997). Weihe and Neely had similar results when comparing *L. salicaria* and *Typha latifolia* in varying light treatments; *L. salicaria* was able to produce more above- and belowground biomass in all treatments (Weiher and Neely 1997). After herbicide treatments (Gabor, et al. 1996) and cutting (Wilcox, et al. 1988) to remove *L. salicaria*, densities of graminoids have increased. Farnsworth and Ellis recognized that interspecies competition was strong with *L. salicaria;* increasing *L. salicaria* biomass was linked with declining biomass of other species, but questioned the frequency of true monostands of *L. salicaria* (Farnsworth and Ellis 2001).

Brown studied the impact of *L. salicaria* on the native species *L. alatum,* and found that pollinator visitation and subsequent seed set was lower in *L. alatum* with the presence of *L. salicaria* (Brown 1999). Fickbohm and Zhu found that *L. salicaria* transpired about twice as much water as *Typha* species, and concluded that monostands of *L. salicaria* had the ability to cause changes in organic matter distribution, nitrogen cycling, and water quality of freshwater wetlands (Fickbohm and Zhu 2006).

Chapter 4: Reed Canarygrass *(Phalaris arundinacea)*

History

Reed Canarygrass *(Phalaris arundinacea)* is a rhizomatic perennial that may be native to North American costal areas, but that has spread under human influence (Anderson 1961). It is reported to be native to Japan, Eurasia, North America, and South Africa (Tsvelev 1983). It may be called 'canarygrass' either because the genus was first described in the Canary Islands or because a close relative, *P. canariensis* is the source of canary seed (Pojar and MacKinnon 1994). The first mention ofreed canarygrass was in Hesselgren's 1749 thesis studying the preferred feeding species of livestock (Stannard and Crowder 2001). European cultivation of reed canarygrass was documented in England as early as 1824, and in Germany in 1850 (Schoth 1938).

Herbarium collections in the Pacific Northwest from the mid-1880's found this species well represented, and the samples were frequently collected from remote locations, indicating that *P. arundinacea* is native to North America (Merigliano and Lesica 1998). Turner documented oral history that indicates Halq'emeylem and most likely other Salish groups used stems for decorating baskets prior to European contact (Turner 1992). But others speculate that freshwater wetland systems in Western Washington prior to European contact were dominated by *Thuja plicata, Picea sitchensis*, and *Tsuga heterophylla*, all of which would have created unfavorable understory conditions for *P. arundinacea* (Antieau 1998), because species richness drops as intense competition for available light increases during the stem exclusion stage of succession (Houston, et al. 1996). However, many native American groups are noted to have utilized burning of certain areas which would have maintained emergent communities instead of forested wetland communities (Pyne 2001). Some native American groups were even noted to burn wetlands for blueberry production (Adamson 1926).

Agricultural Use

In the 1830's *P. arundinacea* was used in livestock grazing trials on the Atlantic Coast, most likely using local germplasm, and by the 1850's native reed canarygrass stands were commonly used as grazing areas for livestock (Stannard and Crowder 2001). As popularity of *P. arundinacea* as a grazing grass skyrocketed, European companies began to export seed to North America, but most of the reed canarygrass currently growing in the Pacific Northwest can be attributed to commercial seed production of local germplasm in Coquille Valley of Oregon beginning in 1885 (Schoth 1938). The species was noted to produce 30% more hay than similar grasses (Wilkins and Hughes 1932). Marten warned that *P. arundinacea* contains alkaloid compounds that at high concentrations make it indigestible or toxic (Marten 1973).

Several authors have also noted that *P. arundinacea* agricultural stands do not typically develop without multiple seedlings (Wasser 1982; Antieau 1998). This may be due to the fact that *P. arundinacea* does not develop tillers until five to seven weeks after germination (Comes, et al. 1981). *Phalaris arundinacea's* delay in sending up tillers may limit it's initial completive ability against more rapidly tillering plants such as *Agrostis alba* and *Festuca rubra*, but after tillering it gains a distinct competitive advantage (U.S. Department of Agriculture 1996).

Use of *P. arundinacea* in the Pacific Northwest began in the late 1890's. *Phalaris arundinacea* was used as a "breaking in" crop after logging (Wheeler 1950). In the late 1970's interest was sparked again in reed canarygrass as a wastewater management species. Zeiders reported that "reed canarygrass is the most popular species for irrigation with wastewater from municipal and industrial sources as a pollution control measure" (Zeiders 1976). *Phalaris arundinacea* has also been utilized to stabilize shorelines and prevent gully erosion (Baltensperger and Kalton 1958; Figiel, et al. 1995). More recently, using *P. arundinacea* as a bio-fuel has been explored in Scandinavia (Katterer, et al. 1998).

Physical Description

Phalaris arundinacea is a hollow stemmed, sod forming perennial, clums 0.7 to 2 meters tall, with scaly long rhizomes. Leaves are slightly hairy, flat, 5-15mm wide, 7 to 41cm long, with 4 to 10 mm membranous ligules, usually frayed and turned down. Compact reddish panicle that changes to straw color as seeds mature; up to 25cm; up to 3 lanceolate spikelets per raceme, usually containing three florets, two of which infertile and reduced. Slightly hairy glumes 4-5mm, and shiny lemma 4mm (infertile florets have 1mm lemmas).

Rhizomes arising from a single plant grow radially outwardly until a terminal bud develops a shoot (Evans and Ely 1941). Comes, et al found that although *P. arundinacea* develops thick rhizomes, they are relatively shallow-rooted; 88% of new shoots originate from the upper *Scm* of soil, and 100% originate from upper 20cm of soil (Comes, et al. 1981). Whole plants dislodged during a disturbance event are able establish mono-stands at new sites by re-rooting in disturbed soils (Hovin, et al. 1973). 74% of new shoots arise from rhizomes, the remainder from auxiliary buds on basal nodes (Casler and Hovin 1980). The rhizomes of *P. arundinacea* are extremely tolerant of anoxiant conditions (Brandle 1983).

Seeds

Seeds of *P. arundinacea* provide a means for long distance dispersal, exchange of genetic information, and the impetus for multiple genotypes; multiple genotypes ensure that at least some genotypes will thrive and multiply in harsh environments (Morrison and Molofsky 1999). Seeds are naked, up to 3mm and germinate immediately after ripening on long clumps, which ripen from top to bottom, allowing for a prolonged period of dispersal. Seeds have no known dormancy requirements, and often dominate seedbanks within wetlands (Apfelbaum and Sams 1987). In 97% of greenhouse grown *P. arundinacea* seeds germinated immediately after harvest, while seeds stored in moist sand germinated after a year of fluctuating temperatures (Comes, et al. 1981).

Seedlings are initially very sensitive to interspecies competition, and frequently sprout in ephemeral ponds in spring (Morrison and Molofsky 1998). The ponded water creates anaerobic conditions that force the sprouts to rely on carbohydrate reserves stored in rhizomes (Hovin, et al. 1973). If water persists on the developing stand for extended periods of time, the reducing environment will deprive the roots oxygen, killing the stand, and in some cases removing oxygen from the roots (Stannard and Crowder 2001). Individual leaves of *P. arundinacea* grow from nodes along the clum, and become disadvantaged as the plant grows taller shielding lower leaves from light (Stannard and Crowder 2001). Large mono-stands are capable of producing up to 9 tons/acre of biomass, but this type of growth requires a tremendous amount of nutrients (Stannard and Crowder 2001).

Nutrient Enrichment

Nutrient influx into wetlands can be associated with either agricultural or residential runoff (Kercher, et al. 2005). Increases in runoff events can cause standing water in wetlands, causing a decline in species that are not flood tolerant, making much more light available for *P. arundinacea* (Kercher, et al. 2005). As the standing water is released from the wetland, many of the nutrients and sediments that the runoff contained increase growth in *P. arundinacea*. Kercher, et al. found that wetland mesocosms subjected to flooding, sedimentation, and nutrification became monostands of *P. arundinacea* after two growing seasons, with up to 50% of native species dying in the first 6 weeks due to prolonged flooding (Kercher, et al. 2005).

Phalaris arundinacea has shown significantly increased growth associated with nutrient enrichment of soil or water supply (Stannard and Crowder 2001). Ho found increased stem density and increased nitrogen and phosphorus levels in *P. arundinacea* that had been subjected to nutrient enrichment (Ho 1979). Both Green and Galatowitsch, and Maurer and Zedler showed an increase in biomass with high nutrient treatments, and Wetzel and van der Valk found a 73% increase in biomass of high nutrient treatments over low nutrient treatments of *P. arundinacea* (Wetzel and van der Valk 1998; Green and Galatowitsch 2001; Maurer and Zedler 2002). Maurer and Zedler did not find a significant relationship between nutrient treatments and emergence or survival of *P. arundinacea* communities, but did find that young ramets with nutrient treatments readily invaded shaded areas drawing nutrients from parental clones (Maurer and Zedler 2002).

Invasive Characteristics

Wetlands that experience invasion by *P. arundinacea* often have drastic declines in native species within several years Spuhler found that wetlands with *P. arundinacea* had 25-33% less species than neighboring sedge meadows, and at two sites *P. arundinacea* had formed monotype stands (Spuhler 1994). Likewise, Werner showed that wetlands invaded by *P. arundinacea* had 9 to 11 less species per square meter than nearby wet prairie communities. Native herbaceous species that begin growing in late spring can be dramatically affected by large monoculture stands of *P. arundinacea*, which deprives them of light (Stannard and Crowder 2001). On the other hand Lindig-Cisneros and Zedler found that sites that support dense native plant canopies because of ideal site conditions can inhibit *P. arundinacea* invasion from seeds, even when the native plant communities are not diverse(Lindig-Cisneros and Zedler 2002). Lindig-Cisneros and Zedler's results also showed that among native plant canopies that experienced a disturbance, canopies with higher diversity (more species) had greater resistance to invasion by *P. arundinacea* seeds than native monotype canopies (Lindig-Cisneros and Zedler 2002). Due to low tissue density *P. arundinacea* stems are twice as high as similar grasses when grown alone, even thought biomass allocation is similar; but when grown with native grasses *P. arundinacea* is able to change morphology and increase its total shoot length to biomass ratio by 50% (Miller and Zedler 2003).

Changes in hydrologic regime

The ability of *P. arundinacea* to invade sites that have experienced a change in hydrologic regime has been well documented. Apfelbaum, et al. considered *P. arundinacea* a forceful invader in disturbed habitats where the substrate was favorable (Apfelbaum and Sams 1987). Odland conducted vegetation transects on a reservoir that was subject to a permanent drawdown and observed that *P. arundinacea* gradually invaded not only the newly exposed substrate, but also the wetlands previously adjacent to the reservoir (Odland 2002). Barnes reported an expansion of *P. arundinacea* on small river islands following lower summer flows that exposed more river substrate (Barnes 1999). Good, et al. and Lech found increased germination and growth of *P. arundinacea* after a hydrologic drawdown (Good, et al. 1978; Lech 1996). Barnes noted that riparian areas may be particularly susceptible to invasion for two different reasons: because sedimentation caused by flooding regularly makes available new sites for *P. arundinacea* to establish; and because human activities along rivers can change the hydrologic regime and alter rates of erosion and sedimentation, and disturb existing vegetation (Barnes 1999). Similarly, Comes, et al. found that *P. arundinacea* is a ready invader at sites disturbed with chemical or mechanical control treatments which open up canopies (Comes, et al. 1981).

Phalaris arundinacea has been found to be productive under varying moisture levels (Morrison and Molofsky 1998), including flooding (Figiel, et al. 1995) and drought (Sheaffer, et al. 1992). Miller and Zedler found that in flood conditions *P. arundinacea* allocated more biomass above ground (Miller and Zedler 2003). Rubio and Lavado suggest that this may be a mechanism to decrease biomass and oxygen demand of the root systems, or to increase the ratio of root length to root biomass to aid in nutrient uptake ability (Rubio and Lavado 1999). In droughty conditions *P. arundinacea* reduces leaf surface area, heavily controls stomatal transpiration, and produces smaller cells with thicker cell walls which retain more water than larger thin walled cells (Frank, et al. 1996).

Chapter 5: Discussion

Performance of Compensatory Mitigation Wetlands

Roberts summarized the process of permitting wetland impacts and requiring compensatory mitigation by stating that "wetland trading is a loser's game"(Roberts 1993). Indeed, by simply evaluating compliance under the no-net-Ioss concept, wetland losses continue to happen because of the failure of compensatory wetland mitigation sites to be constructed. The U.S. EPA estimates that the United States is still loosing 70,000-90,000 acres per year, which does not take into account the losses from compensatory mitigation wetlands that are poorly designed or managed and therefore have reduced functional value that does not adequately compensate for the aquatic resources that were impacted (Lee and Chapman 2001).

Some types of herbaceous wetlands, such as freshwater emergent marshes and wet meadows, have been successfully restored or created for compensatory mitigation (Lindau and Rossner 1981; Niswander and Mitsch 1995; Wilson and Mitsch 1996; Brown and Veneman 1998). Wet prairies and sedge meadows have met with limited success (Galatowitsch and Valk 1996; Ashworth 1997). Shrub swamps and forested wetlands are the most difficult to create or restore for compensatory mitigation because of the time required to establish mature woody plants (Niswander and Mitsch 1995; Brown and Veneman 1998; King 2000).

Most studies suggest that there is much room for improvement in the construction of compensatory mitigation wetlands. Maguire found that 50% of mitigation sites in Virginia were considered successful using area, vegetative cover, and achievement of permit conditions to calculate mitigation success (Maguire 1985). Maguire noted that many mitigation efforts were not considered to be successful because they had not even been built (Maguire 1985). The U.S. Environmental Protection Agency found similar results (Reimold and Cobler 1985). Glubiak, et al. and Quammen both suggested the need for better management of compensatory mitigation wetlands (Glubiak, et al. 1986; Quammen 1986).

There is currently a disagreement among researchers as to the success of recent mitigation efforts. Harvey and Josselyn believed that compensatory mitigation was working in the mid 1980's, while Race suggested that many mitigation projects were frail and could easily fail (Harvey and Josselyn 1986; Race 1986). Kusler and Groman questioned the granting of permits to projects that were not water dependent, and when alternative sites were available, and Golet took a firmer stance and asserted that damage to wetlands should not be permitted unless there is absolutely no alternative (Golet 1986; Kusler and Groman 1986).

Recent research has shown that many mitigation sites are not constructed. Erwin (1991) found that in southern Florida only half of the 430 ha of wetlands required as mitigation had been constructed, and 60% (24 of 40) of projects were found to be incomplete or failures (Figure 1). Kentula et al. found similar results for mitigation sites in the Pacific Northwest; and found a net loss in area of 43% for Oregon and 26% for Washington (Kentula, et al. 1992).

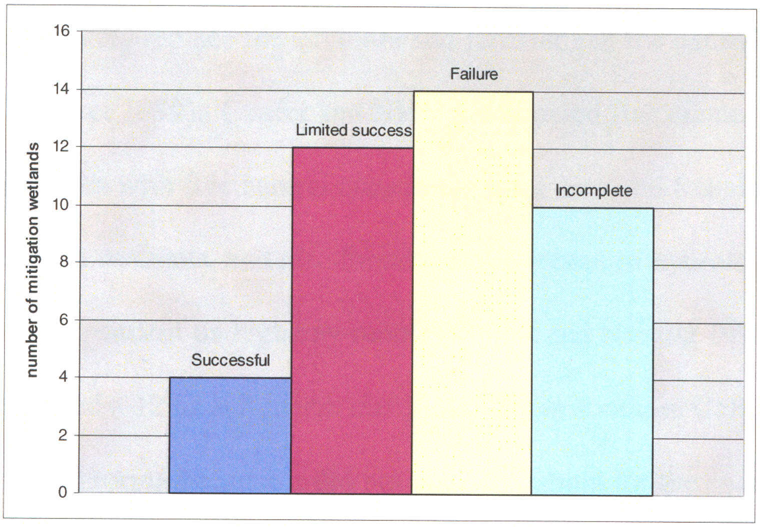

Figure 1: Evaluation of 40 wetland mitigation projects in south Florida (Erwin 1991).

For mitigation sites that are constructed, many are considered legally successful following construction, and no monitoring of any type takes place. Holland and Kentula found that only 31.5% of more than 300 issued permits in California from 1971 to 1987 required any monitoring of the mitigation even though 1260 ha of compensatory mitigation was required for the issuance of the permits (Holland and Kentula 1992). Sifneos et al. found that only 8% of Section 404 activities in Louisiana, Alabama, and Mississippi were compensated for in some way, and that only 10% of the compensatory mitigation wetlands were monitored in any way (Sifneos, et al. 1992).

Ecological success of compensatory mitigation wetlands

Many researchers suggest that compensatory mitigation wetlands do not adequately replace the structure and functions of the natural wetlands that are lost (Streever 1999). Confer and Niering compared five created compensatory mitigation wetlands with five natural ones in the same area and found more open water area in the created wetlands, and also found that the source of hydrology for the created wetlands was dependent on highway runoff (Confer and Niering 1992). Zedler and Malakoff declared a 12 ha salt marsh mitigation site in southern California a failure after ten years of monitoring because it did not provide habitat for the endangered light-footed clapper rail *(Rallus longirostris levipes)* (Zedler 1996; Malakoff 1998). Wilson and Mitsch concluded that while 4 of the 5 compensatory mitigation wetlands they studied were successful ecologically, they were not always considered successful legally (in compliance with permit conditions) (Wilson and Mitsch 1996). While Svengsouk and Mitsch found that a 6 ha compensatory mitigation wetland in Ohio was successful, a similar site in lllinois was not (Mitsch and Flanagan 1997; Svengsouk and Mitsch 1997).

Invasive species pose numerous challenges for compensatory mitigation. Many compensatory mitigation wetland sites are determined to be compliant once the vegetative community becomes established (Council 2001). While there are many things that could be used to monitor success of mitigation wetlands, vegetation is considered the easiest indicator of progress to observe (Mitsch and Gosselink 2000). In most cases, the wetland indicators of hydrology and soils are rarely used to evaluate compliance with permit conditions.

In 1998 Gallihugh studied the success of wetland mitigation sites in the Chicago region by evaluating 61 permits issued from 1990 to 1994 with a combined impact of 288.7 acres (Gallihugh 1998). The total mitigation proposed under the 61 permits was 354.9 acres, but 72.5 acres of mitigation was never established, a net loss of 6.3 acres. Of projects with constructed mitigation, 54 of 61 permits had special conditions related to mitigation, but only 2 of these permits were deemed to be in full compliance with all special conditions. Furthermore, 128 mitigation sites were required by the issuance of the 61 permits; 22 of those sites were established with correct plant communities as proposed, 28 mitigation sites had established wetlands, but with the wrong plant communities, and the remaining 78 sites either had excessive open water or insufficient hydrology to support the correct plant communities. The Chicago study also found that 64% of wetland mitigation sites had less than 20% of the plants that had initially been planted actually establish. The overall success rate for a given native plant to establish in an area where it was planted was 12.4%; only 10% of native plants had a success rate above 50%, and 68% of native plants had success rates of 0%. As illustrated in Figure 2, *Lythrum salicaria* was found at 48 of the 128 mitigation sites, and had a total aerial coverage of more than 20% at 6 sites, while *Phalaris arundinacea* was found at 81 sites, and had a total aerial coverage of more than 20% at 12 sites.

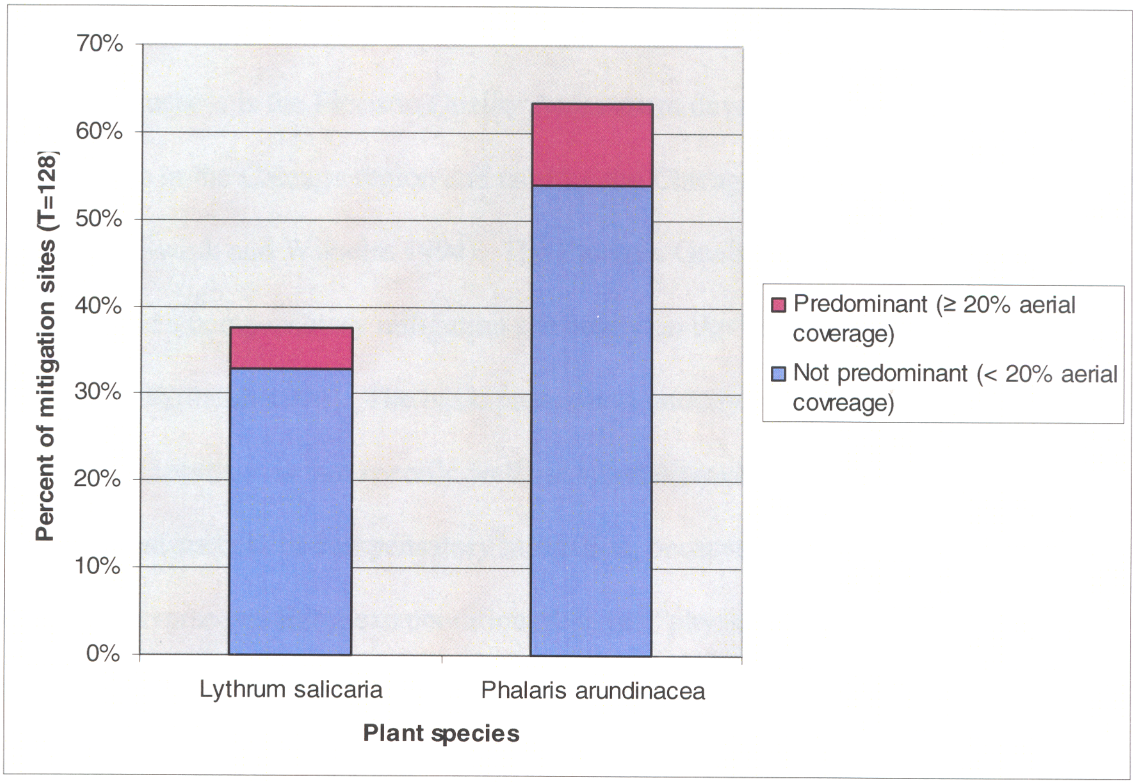

Figure 2: Occurrence and predominance of invasive plant species (Gallihugh 1998).

Atkinson et al., indicate that that vegetation should be the main criterion to measure the short-term success of compensatory mitigation wetlands (Atkinson, et al. 1993), but Reinartz and Warne warn that use of vegetative standards to measure success of compensatory mitigation wetlands may not correspond with the wetland's functional success (Reinhartz and Warne 1993). Furthermore, although permit requirements often suggest the need to consider area and function, structural characteristics (usually the amount of vegetation cover) may be used as a the criterion to judge whether functional replacement is achieved (Kentula, et al. 1992).

One example of a singular criterion used to judge compensatory wetland mitigation success is the Floristic Quality Assessment developed by Swink and Wilhelm for wetlands in the Chicago region and used by the Chicago District, U.S. Army Corps of Engineers (Swink and Wilhelm 1994). The Floristic Quality Assessment essentially characterizes a compensatory mitigation site solely on the vegetative community present (Swink and Wilhelm 1994). The basic postulation underlying the Floristic Quality Assessment approach is that specific wetland vegetation variables can be used to indicate the functional success of compensatory mitigation, because if a vegetative community displays a diverse pre-European condition, then the "physical, biological, and biochemical functions that support the vegetation must be present" (Council 2001). However, low plant diversity is not always characteristic of substandard hydro-geological and geo-chemical conditions in wetlands, and higher plant diversity is not automatically a de facto indicator of wetland functions (Council 1995).

Conclusion

The different types of floristic assemblages required for successful compensatory mitigation often require widespread plantings and persistent management to maintain the species composition desired (Council 2001). *Polygonum cuspidatum, Lythrum salicaria,* and *Phalaris arundinacea* all pose significant threats to the legal success of compensatory mitigation sites, because their aggressive growth often exceeds aerial coverage standards that are made part of permit requirements. Often eradicating one of these species from a site will cause disturbances which could allow new colonization by yet another invasive species (Ecology 2004). Invasive species also present significant challenges to the functional success of compensatory mitigation by changing species composition, which can alter the functions that mitigation site was designed to replace.

The Corps use of the 10% aerial coverage standard for invasive species was an attempt to implement a reasonable standard that could be easily measured, but the scientific literature does not support the use of the 10% standard (Tong 2006). Maintaining an aerial coverage of less than 10% for invasive species for the duration of the monitoring period of a compensatory mitigation wetland does not ensure that a mitigation site will not be invaded after the monitoring period ends, or that invasive species already present take over a site in the future.

While the intent of the 10% standard for invasive species aerial coverage was to prevent the establishment of monostands of invasive species from out competing native species, thereby compromising and degrading wetland functions, many sites failed to comply with the 10% standard and therefore permit conditions because of a high occurrence of *P. arundinacea* (Celedonia 2002; Ecology 2004; Ecology, et al. 2006). Because *P. arundinacea* does provide water quality, food web support, and habitat functions (Terzi 2006), coverage standards for *P. arundinacea* should be set as to not exceed aerial coverage at the impact site, or to not exceed the aerial coverage of nearby wetlands, which ever is lower. Implementing this new standard will not result in a net-loss of wetland functions, and will allow wetland mitigation to function in a manner that is similar to nearby natural wetlands.

Polygonum cuspidatum and *L. salicaria* both form dense stands that are spreading aggressively in many wetlands, displacing many native species(Mitsch and Gosselink 2000; Doll and Doll 2002). *Polygonum cuspidatum* also prevents woody species from establishing on stream banks which disrupts aquatic food webs for salmonids (Potash 2001). For *P. cuspidatum* a zero tolerance policy should be adopted because of the aggressiveness of recent invasions into wetlands, while the 10% aerial coverage standard may be appropriate for *L. salicaria* because while it does invade wetland habitats, the existence of true monostands of *L. salicaria* that exclude native species is questionable (Keddy, et al. 1994; Farnsworth and Ellis 2001).

Polygonum cuspidatum, Lythrum salicaria, and *Phalaris arundinacea,* are invasive species that form monostands which can undermine both the legal and ecological success of compensatory mitigation wetlands by failing to replace the aquatic resource functions that were impacted by the issuance of a Department of the Army permit. Compensatory mitigation sites that exceed the 10% aerial coverage standard for invasive species are not considered legally successful by the Corps because of permit conditions specifying specific allowable invasive species coverage ratios. Compensatory mitigation wetlands may also not be considered ecologically successful if the mitigation site fails to provide the same wetland functions and values as the impacted site. A diverse assemblage of native plants is always preferable at mitigation sites to ensure the legal and functional ecologic success of compensatory mitigation wetlands.

Policy recommendations

- Adopt a zero tolerance policy for *Polygonum cuspidatum.*

- Make case by case determinations for aerial coverage standards of *Lythrum salicaria* based on percent cover of natural wetlands close to the mitigation site (not to exceed 10%).

- Set *Phalaris arundinacea* cover standards to match the impact site.

- Design aerial coverage standards for all compensatory mitigation wetlands on a case by case basis and take into account the impact site, the mitigation site, and the functions lost to be replaced by mitigation wetlands.

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