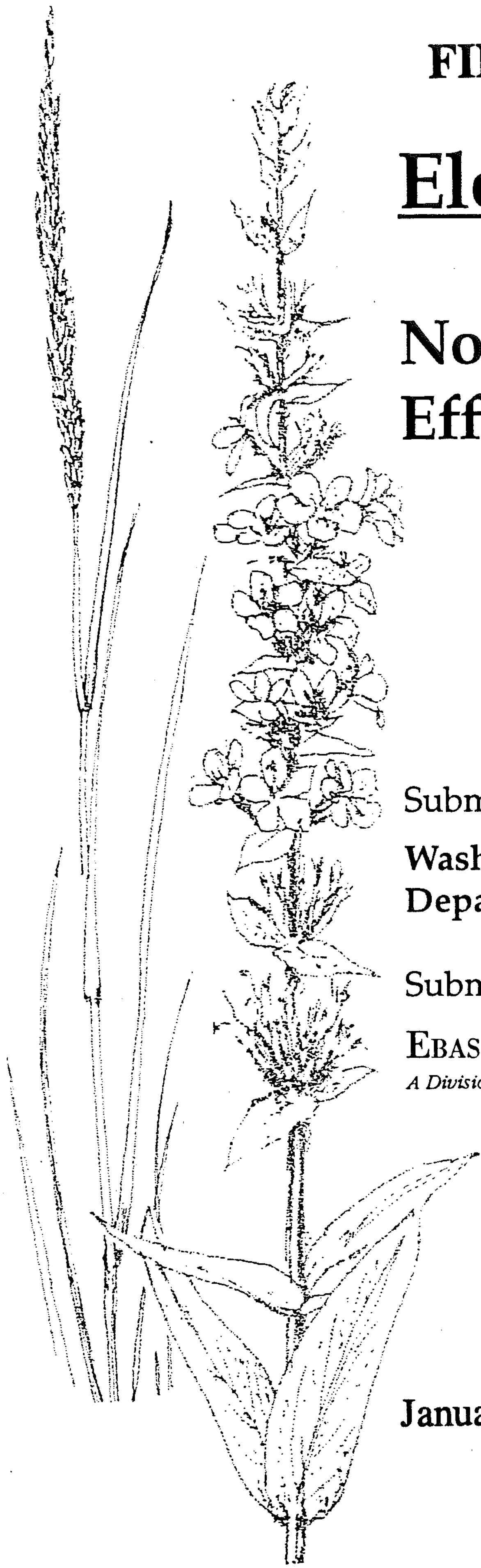

FINAL REPORT

Element C

No Action: Efficacy and Impacts

Submitted to

Washington State Department of Ecology

Submitted by

EBASCO ENVIRONMENTAL

A Division of Ebasco Services Incorporated

January 1993

FINAL REPORT

Element C: No Action: Efficacy and Impacts

Submitted to:

Washington State Department of Ecology

Submitted by:

Ebasco Environmental

Project Manager
Jay Brueggeman

Technical Lead
Mary Harenda
Stacy Giles Emory
Clay Antieau

January 1993

CONTENTS

TABLES

ABSTRACT

Infestations of noxious emergent plant species in Washington are raising ecological and economic concerns. The Washington State Departments of Agriculture, Ecology, Fisheries, Natural Resources, and Wildlife, and the Washington State Noxious Weed Control Board have proposed to develop and implement a management plan for these species. One management option is the no action alternative under which distributions and infestations of noxious species would be determined by natural processes. Potential efficacy of the no action alternative for controlling infestations and potential impacts on natural, agricultural, and built environments are described in this report.

The no action alternative will probably not result in regional stabilization or decrease of *Spartina alterniflora* and *S. anglica* distributions within the next several decades. Extensive *Spartina* infestations could significantly impact the physical, chemical, and biotic characteristics of estuarine environments. Potential impacts include colonization of large areas of tidal flats, displacement of native vegetation, and changes in available habitat for fish, wildlife, and shellfish.

It is unlikely that a no action management scenario would cause effective management of purple loosestrife infestations in Washington. Despite a lack of specific information on potential impacts, such an approach would appear to intensify significant adverse impacts to components of the natural and built environments such as biota (including rare flora and fauna), sedimentation rates, water flows and hydrologic regimes, agricultural irrigation, and economies of private and public sectors. These adverse impacts appear to outweigh the few apparent beneficial impacts of infestations, which relate primarily to perceived aesthetics and honey production.

Implementation of a no action alternative may not cause containment, stabilization, or dieback of Washington infestations of garden loosestrife (*Lysimachia vulgaris*), dotted loosestrife (*L. punctata*), giant hogweed (*Heracleum mantegazzianum*), and indigobush (*Amorpha fruticosa*). Despite a lack of information on the biology and ecology of these species, infestations elsewhere in the world suggest these species may adversely impact physical, chemical, and biotic characteristics of wetland and riverine environments in Washington. These adverse impacts would appear to become more severe in time under a no action scenario. Specific adverse impacts could include displacement of native plant and animal communities, increased threats to human health, increased erosional rates, and possible localized decreases in water quality or water quantity.

1.0 INTRODUCTION

1.1 PURPOSE

The Washington State Departments of Agriculture, Ecology, Fisheries, Natural Resources, and Wildlife, and the Washington State Noxious Weed Control Board, acting as co-lead agencies, have proposed to develop and implement a management plan for noxious emergent plant species in the State of Washington. Species of concern include three species of cordgrass or *Spartina* (*S. patens, S. alterniflora,* and *S. anglica*), purple loosestrife (*Lythrum salicaria* and *Lythrum virgatum*), garden loosestrife (*Lysimachia vulgaris*), dotted loosestrife (*L. punctata*), giant hogweed (*Heracleum mantegazzianum*), and indigobush (*Amorpha fruticosa*). Most of these species are included on the Washington State Noxious Weed List (Chapter 16-750 WAC) because they are considered detrimental to the agricultural, aquacultural, and natural environments of the state. The effort required to control a noxious species varies according to its current distribution, likelihood of spread to uninfested areas, and other factors (WSNWCB 1991). The lead agencies seek to determine which management alternative or combination of alternatives would provide the most effective management of noxious emergent plants with the least environmental impacts. The ultimate goal of this effort is to develop criteria and approaches for managing infestations of both existing noxious species and new invaders.

The lead agencies have determined that management of these noxious emergent plant species could have probable significant adverse impacts on the environment. Thus, an environmental impact statement (EIS) is required under RCW (Revised Code of Washington) 43.21C.030(2)(c). The lead agencies, through a public scoping process, have identified topics to be discussed in the EIS, including biology and ecology of problem species, management alternatives, efficacy and impacts of alternatives, and mitigation strategies. Ebasco Environmental was contracted by the nominal lead agency, the Washington State Department of Ecology, to assemble and synthesize available information on the topics of interest for probable inclusion into the EIS. This report provides information on the "no action" alternative for managing infestations of noxious emergent plants in Washington.

1.2 OBJECTIVES

The objectives of this report are to:

(1) evaluate, based on available information, the efficacy of a no action management scenario in controlling populations of *Spartina alterniflora, S. patens, S. anglica, Lythrum salicaria, L. virgatum, Lysimachia vulgaris, L. punctata, Heracleum mantegazzianum,* and *Amorpha fruticosa*; and

(2) describe potential impacts of the no action alternative on natural, agricultural, and built environments and their associated human uses.

Information sources for this report included published journal articles, published and unpublished studies, and communications with knowledgeable individuals. Information was obtained from both national and international sources.

2.0 DESCRIPTION OF ALTERNATIVE

Active management of noxious emergent plant infestations would not occur under the no action alternative. Distribution and spread of noxious species would be regulated by natural processes. Primary factors controlling distribution and spread of invasive species include environmental variables, plant genetics, and biotic interactions. Under this alternative, interested agencies would continue to monitor infestations of noxious weeds and participate in public outreach activities. Species would continue to be listed as noxious weeds, as required under state law. Public and private landowners who knowingly fail to control noxious weeds on their lands would be in noncompliance with state law (Chapter 16 - 750 WAC).

3.0 *SPARTINA*

This section addresses the efficacy and impacts of the no action alternative for managing infestations of *Spartina* species in Washington. Major positive and negative effects are summarized in Table 1.

3.1 EFFICACY OF NO ACTION ALTERNATIVE AS A MANAGEMENT TOOL

Spartina species are colonizing intertidal flats and, in some areas, are displacing salt marsh and eelgrass communities. Associated impacts, which are fully discussed in Section 3.2, may include accretion of fine sediments, changes in the topography and elevation of intertidal areas, and changes in available habitat for fish, wildlife, and shellfish.

The no action alternative could result in the control of noxious *Spartina* species in Washington if natural processes effectively stabilize (stop their spread) or decrease their distributions at the regional level. Vegetative spread of an individual colony (a group of *Spartina* stems that consists of one or more clones) is limited by the availability of suitable habitat and sometimes by competition with other high marsh species. Thus, individual colonies will stop spreading vegetatively once all suitable habitat at a site has been colonized. In some areas, such as south Willapa Bay, vegetative spread of *S. alterniflora* may stop only after large acreages of shallow, intertidal habitat have been colonized. However, stabilization of *Spartina* distribution throughout a water body also depends on whether populations are producing viable seeds or vegetative propagules. Successful seedling or propagule dispersal and establishment would further increase *Spartina* distribution within a water body, even if parent colonies stop expanding vegetatively. Although successful seedling establishment of *Spartina* has been shown to be variable in both natural and introduced populations (Hill 1984, Sayce 1988, Gray *et al.* 1990, Calloway and Josselyn

Table 1. Summary of major effects of the no action alternative on *Spartina* infestations in Washington intertidal waters.

Potential Positive Effects	Potential Negative Effects
• Increased cover availability for some fish, waterfowl, and wading birds	• Increased flooding of residential and commercial properties, agriculture/aquaculture facilities, and public works developments
• More nesting habitat for some birds	• Significant loss of habitat for shellfish, other invertebrates, and fish dependent upon intertidal flats
• Greater foraging habitat for some small mammals	• Reduction in feeding and roosting habitat for migratory and wintering waterfowl and shorebirds
• Beach stabilization	• Displacement of salt marsh and eelgrass communities
	• Reduction in nutrient inputs to estuarine waters
	• Continued spread of *Spartina* species
	• Loss of biodiversity

1992), one successful reproductive season could increase *Spartina* distribution within a water body and contribute to its spread to new locations.

The infestation of *S. patens* in Hood Canal at the mouth of the Dosewallips River is the only known locale of *S. patens* in Washington. Intensive efforts are being undertaken by the Washington State Department of Natural Resources to eradicate this population (J. Civille 1992, pers. comm.). Because of these control efforts, spread of the species to additional locations appears unlikely. If control efforts are abandoned without eradicating the population, *S. patens* will probably continue to expand vegetatively at the site. Seed viability of the population appears low (Frenkel and Kunze 1984), although viability has not been measured recently (L. Kunze 1992, pers. comm.). No information was found to indicate if potential exists for introduced populations of *S. patens* to experience rapid changes in seed viability or seed production. Potential for spread of *S. patens* to additional locations would increase if seed viability or production increased.

Natural stabilization of *S. alterniflora* and *S. anglica* distributions in Washington is unlikely to occur within the next several decades. *S. alterniflora* is widespread and spreading in Willapa and Padilla bays. In addition, *S. alterniflora* in Willapa Bay is producing viable seed that could disperse to other water bodies. Seeds or propagules from Willapa Bay have apparently been transported to Grays Harbor and the Copalis River estuary. *Spartina anglica* distribution in Puget Sound has also been steadily increasing. Habitat suitable for *S. alterniflora* and *S. anglica* abounds in intertidal areas throughout Washington. Thus, even if vegetative spread in certain areas slows or stops, spread to new locations by seed or propagule transport will probably continue to occur.

Spartina distributions could decrease if colonies experience dieback without subsequent recolonization. Dieback is a natural phenomenon and refers to recession or senescence of *Spartina* patches. It typically occurs in shallow depressions or "pans" within stands of the grass, or along the edges of stands. The most frequently cited causes of dieback are soil waterlogging and wave action. Soil waterlogging leads to formation of anaerobic, reducing soils and accumulation of significant amounts of free sulphide, beyond the range that normal growth of *Spartina* can be supported, resulting in stunting or death of plants (Goodman and Williams 1961, King *et al.* 1982, Delaune *et al.* 1983, Mendelssohn and McKee 1988).

Diebacks can be small, localized phenomena or may be extensive. Dieback along a stand edge is most likely related to the erosive force of wave action. Extent of dieback within a stand appears related to the degree of waterlogging within an area. Extent of waterlogged area, in turn, is determined by sediment type and drainage and sedimentation patterns within a stand. Dieback has been observed most frequently in *Spartina* stands on fine-particled substrates (silts and clays) (Goodman *et al.* 1959, ACOE 1992).

Although localized dieback has been reported in stands of *S. alterniflora* in Willapa (ACOE 1992) and Padilla (Riggs 1992) bays, the prospect of extensive dieback in Washington *S. alterniflora* populations is remote. The most extensive diebacks reported for *S. alterniflora* have occurred in Louisiana, apparently from the combined effects of sea-level rise and rapid land subsidence. Rates of vertical sediment accretion in Louisiana marshes are not keeping pace with water level rise (Wilsey *et al.* 1992). Since rapid land subsidence is not occurring in Washington estuaries, it is unlikely that scenarios observed in Gulf coast populations will occur in the state.

Dieback of mature *S. anglica* stands has been widespread in southern England since about 1960. Dieback of isolated clumps has also been reported. Intense wave action and strong tidal currents have been reported to prevent coalescence of clumps into closed stands and cause pans to form within clumps (University of Hull 1987). Recent research indicates that dieback in *S. anglica* may involve a strong genetic component. Documented low rates of allelic recombination and mutation contribute to genetic uniformity within stands that may cause increased susceptibility to environmental changes that promote dieback and to herbivore or pathogen infestations (Raybould *et al.* 1991). Because of the probable influence of genetic factors, *S. anglica* populations in Washington may also experience widespread dieback. Dieback of *S. anglica* stands in Port Susan Bay occurred in the 1960s, however, the species persists in Port Susan and continues to spread to other areas in Puget Sound (Kunze 1992, pers. comm.). The environmental conditions that caused extensive dieback in southern England required about a century to develop. During that period, *S. anglica* continued to spread throughout Britain (Charman 1990). Thus, natural dieback does not appear promising as a control method for *S. anglica* in the short term.

In addition, the fate of dieback areas is unclear. A return to pre-invasion conditions (i.e., tidal flats) would probably not occur if sediments accreted by a *Spartina* marsh remain in place. Recolonization of *S. anglica* dieback areas by native high marsh species has been observed in Britain (Scholten and Rozema 1990). Evidence of buried *S. alterniflora* culms below areas of living *Salicornia virginica* in Willapa Bay (ACOE 1992) suggests recolonization of dieback areas or displacement of *Spartina* by native species. Recolonization of dieback areas by *Spartina* is also possible, although Tubbs (1984) indicated this had not occurred for *S. anglica* in Britain by the mid-1980s. Once sediment in a *Spartina* marsh accretes to the level of the higher marsh areas, *Spartina* stands may be invaded and displaced by native high marsh species such as *S. virginica* and *Distichlis spicata*. Areas of *S. alterniflora* dieback in North Carolina were recolonized by native salt marsh plants at higher intertidal sites and by *S. alterniflora* at lower intertidal sites (Linthurst and Seneca 1980). Competition from native high marsh species may ultimately reduce the landward extent of *Spartina* infestations in some areas. However, the abundance of lower intertidal habitat in Washington estuaries suitable for *Spartina* colonization leave doubt that competition with native high marsh plants will substantially impact *Spartina* distribution.

Natural herbivore and pathogen infestations affect *Spartina* populations by impacting seed production and killing individual plants (Bertness and Ellison 1987; Strong 1990; Thompson 1991). However, none of the insect herbivores associated with *Spartina* populations in the eastern United States or California are present in Washington, and it is unknown if any local insect species could potentially control *Spartina* (Strong 1992, pers. comm.). Thus, natural control of *Spartina* in Washington by insect herbivores is improbable.

Some waterfowl and aquatic mammals feed on *S. alterniflora*, although not preferentially (Daiber 1974). Migrating waterfowl consume large quantities of *S. anglica* seeds in Britain (Ranwell 1967). Consumption or incidental transport of seeds and rhizomes by animals may contribute to *Spartina* spread. It is unlikely that grazing by wildlife will decrease *Spartina* distributions in Washington.

Lastly, the ergot fungus, *Claviceps purpurea*, has been shown to effectively prevent viable seed set in infested populations of *S. anglica* in Britain by infecting all the embryos in an inflorescence (Thompson 1991). This parasitic fungus has been observed growing on *S. alterniflora* in Willapa Bay and infestations may have lowered seed viability (Friebaum 1992, pers. comm.). Natural localized occurrences of this fungus may prevent seed dispersal from infested areas. However, the potential for extensive infestations of this or other fungal species in Washington *Spartina* populations is unknown.

3.2 IMPACTS

This section discusses documented and possible impacts to natural, agricultural, and built environments that may result from the no action alternative.

3.2.1 Natural Environment

For this report, the natural environment includes those areas that have not been directly modified by residential, commercial, or public works developments, or agricultural or aquacultural activities. Impacts to wildlife refuges, nature preserves, natural areas, etc., established to preserve ecosystems are considered in this section. Other "open space" areas managed for recreation or other human use, such as state, county, or city parks, are addressed in the Built Environment section. Implications for commercial fisheries from effects of noxious species in natural environments are discussed in this section. Impacts to agricultural or aquacultural species are described under the Agricultural/Aquaculture Environment section.

3.2.1.1 Sediment Composition and Dynamics

Impacts to the sediment dynamics of local waters from *Spartina* colonization include accretion of fine sediments within a *Spartina* marsh, concurrent reduction in sediment inputs to surrounding areas, and changes in sediment composition and topography of intertidal areas. The ability of *Spartina* stems and rhizomes to trap and accrete sediments from river and tidal flow is well-recognized. A developing *S. alterniflora* marsh in New England accreted sediment at 5.2 cm/yr (2 inches/yr) (Redfield 1972). An expanding colony of *S. anglica* in Britain accreted 8-10 cm (3.1-3.9 inches) of sediment per year (Ranwell 1964). Lee and Partridge (1983) noted rates of sediment deposition in New Zealand *S. anglica* between 3 and 12 mm/yr (0.1-0.5 inches/yr.). Accretion rates have not been measured for Washington *Spartina* infestations. Sediment core samples were taken in different aged *S. alterniflora* patches in Willapa Bay in 1990. However, limited funding has prevented adequate processing to determine sedimentation rates (Thom 1992, pers. comm.).

Changes in the pattern and rate of accretion occur as a marsh matures. Accretion is generally most rapid along the edges of a developing marsh. As the margins increase in elevation, the accretion rate will diminish, providing an opportunity for the interior of the marsh to "catch up" so that the entire surface ultimately becomes nearly level (Redfield 1972). Accretion rates in a *Spartina* marsh appear directly correlated with stem density and sediment supply, and inversely related to wind and wave action (Chung 1985).

Topographic changes in intertidal areas will likely occur from *Spartina* colonization. Sediment accretion increases the elevation of a growing *Spartina* marsh above surrounding tidal flats, although occasionally compaction may keep pace with accretion. In contrast to the formerly bare, gently sloping flats with shallow tidal channels, fully-developed *Spartina* marshes have steeply sloping seaward edges and deep, steep-sided tidal channels. The substrate of *Spartina* marshes also tends to consist of a higher proportion of softer, finer particles such as silts and clays, than intertidal flats dominated by either sand or mud.

Finally, extensive dieback of *Spartina* stands and subsequent erosion of accreted sediments by waves or tidal currents could cause large amounts of sediment to be redeposited in other areas. Erosion at the seaward edges of living, closed *Spartina* stands or individual clones has been documented in Britain (Ranwell 1964, Hubbard 1965), New Zealand (Lee and Partridge 1983), and Australia (Vanderzee 1992). Erosion of sediments from areas of *S. anglica* dieback has also been observed in England (University of Hull 1987). The amount of erosion in dieback areas would likely depend on the strength of waves or tidal currents in the area. Temporary or permanent burial of bottom-dwelling organisms and dispersal of viable root or rhizome fragments to uninfested areas could result from redeposition of sediments. Vanderzee (1992) suggested that dieback of *S.* x *townsendii/anglica* in Australia could result in the release of significant amounts of sediments in some areas.

3.2.1.2 Water Quality and Movement

The no action alternative may impact water quality and water circulation patterns. Although no studies have evaluated the effects of *Spartina* infestations on water quality and movement, it is conceivable that water quality in areas with extensive *Spartina* infestations may be impacted from decreased dissolved oxygen content caused by natural decomposition of *Spartina* litter, particularly during late summer and fall. Conversely, water quality in some areas may improve because of *Spartina* trapping suspended sediments from stream and tidal flows.

Water circulation patterns may also be impacted by *Spartina* presence. Changes in water circulation, resulting from *Spartina* infestations have been documented in Britain (Hubbard

1965), New Zealand (Franko *et al.* 1985), and Australia (Vanderzee 1992). Large infestations in or near the mouths of rivers may decrease flow velocities and cause increased flooding in river delta areas, particularly during periods of heavy precipitation or above normal tides. Effects to flows may be less pronounced during fall through spring when above-ground stems are absent. Increased flooding frequency in shoreline and river delta areas may also occur if sediment in *Spartina* marshes accretes to a level sufficient to block the retreat of tidal waters.

Some areas of *Spartina* may also be subject to decreased frequency of tidal inundation due to vertical accretion of sediments (Hubbard 1965, Franko *et al.* 1985, Vanderzee 1992). This situation has been observed at the Willapa Bay National Wildlife Refuge (Paveglio 1992, pers. comm.). In these areas, tidal exchange is limited to deeper channels. Decreased tidal inundation may increase surface water temperatures and either increase or decrease soil salinities (depending on freshwater inputs and rate of evapotranspiration) within a *Spartina* marsh (Franko *et al.* 1985). Increased water flows and velocities could also occur in tidal or stream channels if surrounding areas no longer transport tidal flows.

3.2.1.3 Biota

Possible impacts to the biotic communities of Washington bays and estuaries from the proliferation of *Spartina* are numerous and broad in scope. They include: loss of intertidal mud and sand flat habitats and eelgrass and macroalgae beds; displacement of native salt marsh plants; changes in available habitats for animals; disruptions in food webs; decreases in benthic algal production; and loss of critical habitat for birds, fish, shellfish, crustaceans, and other animals; and decreases in habitat diversity and plant and animal species diversity in infested areas.

Tidal Flats and Existing Vegetation

Spartina species are rapidly colonizing intertidal flats in Washington. For example, large acreages of formerly bare tidal flats have been colonized by *S. alterniflora* in Willapa Bay and by *S. anglica* in Port Susan Bay. Extensive losses of this habitat type are likely in some water bodies if growth of *Spartina* remains unchecked.

Growth of *Spartina* may also significantly impact existing plant communities in some areas. At some locations in Washington, *Spartina* species appear to have displaced native salt marsh flora, while in other areas, little direct competition has been observed between *Spartina* and native salt marsh plants. *Spartina patens* in Siuslaw Estuary, Oregon, successfully established in bare areas in the relatively "open" middle marsh community dominated by *Deschampsia cespitosa* and *Scirpus maritimus* and has expanded vegetatively, displacing native salt marsh plants in the process (Frenkel and Boss 1988). *Spartina patens* at Dosewallips State Park is displacing *S. virginica* and *D. cespitosa*. In addition, wracks of *Spartina* litter wash up onto native vegetation in the fall and winter, effectively covering it (J. Civille 1992, pers. comm.). Frenkel and Kunze (1984) also noted that *S. anglica* in Port

Susan Bay appeared to successfully compete in the native brackish marsh community dominated by *Triglochin maritimum*, *Scirpus americanus*, and *Scirpus maritimus*, and in the lower marsh community characterized by *Salicornia virginica* and *Distichlis spicata*. *Spartina anglica* in Britain has also displaced native low salt marsh communities (Doody 1990). Some mixing of *S. alterniflora* and native salt marsh plants has been observed at the edges of *Spartina* stands in Willapa Bay , however little direct competition has been observed (ACOE 1992). At Leadbetter Point in Willapa Bay, *S. alterniflora* initially established on the tidal flats and is now encroaching into the low salt marsh and into the tidal channels of the higher marsh areas (Kunze 1992, pers. comm.)..

Impacts to eelgrass and macroalgae beds from *Spartina* colonization are also of concern. Reports from New Zealand indicate that native eelgrass (*Zostera marina* and *Z. muelleri*) and macroalgae beds have been displaced by *S. anglica* (Franko *et al.* 1985). Corkhill (1984) notes an instance where the replacement of *Zostera* by *S. anglica* was apparently enhanced by increased siltation of eelgrass beds from construction of a nearby road. The replacement of *Z. marina* beds by *S. anglica* has been observed in British estuaries (Doody 1984, Hubbard 1965). *Zostera marina* populations worldwide abruptly began to decline beginning in the early 1930s and the species has since disappeared from a large part of its historical range. Several causes have been postulated for this worldwide "wasting disease" including a parasite and global climatic changes. Ranwell and Downing (1960, cited in Doody 1984) suggested that the decline of *Zostera* in British estuaries from wasting disease was further aggravated by the very rapid spread of *S. anglica.*

Replacement of eelgrass habitat by *S. alterniflora* has also been observed in some areas of Willapa Bay. Hundreds of acres on Willapa National Wildlife Refuge that once supported either *Z. marina* or *Z. japonica* have been colonized by *S. alterniflora* (Paveglio 1992, pers. comm.). Phillips (1974 cited in Phillips 1984) reports a general upper limit of native eelgrass (*Zostera marina*) growth as 1.8 m (6 feet) above mean lower low water (MLLW). The non-native *Zostera japonica* is usually found between 1.2 m (4 feet) and 2.4 m (8 feet) above MLLW (Harrison 1979 cited in Phillips 1984). *Spartina alterniflora* in Willapa Bay has been observed naturally growing between approximately 1.75 m (5.7 feet) and 2.75 m (9 feet) above MLLW (Sayce 1988). Both species of *Zostera* also occur in depressions at tidal elevations higher than their reported general upper limits. These higher eelgrass beds are most vulnerable to competition from *Spartina*. Displacement of eelgrass by *S. alterniflora* has not been observed to date in Padilla Bay (Riggs 1992, pers. comm.). However, wracks of dislodged eelgrass have covered and killed small patches of *Spartina* within a larger stand.

Displacement of eelgrass by *S. anglica* is also possible although no confirmed reports are available. *Spartina anglica* in Port Susan Bay has been reported to colonize areas within 50 cm (1.7 feet) above MLLW (Frenkel and Kunze 1984). *Spartina patens* will probably have minimal effects on eelgrass beds because it is typically confined to the upper and middle intertidal zones; its lower limit of colonization in Siuslaw Estuary is 1.8 m (6 feet) above MLLW (Frenkel and Boss 1988).

Once the sediment in a *Spartina* marsh accretes to the level of the higher marsh areas, *Spartina* stands may be replaced by species of the upper salt marsh. Such succession has occurred in *S. anglica* stands in Britain. However, the species that tend to displace *S. anglica* form plant communities that have much lower diversity than the original low salt marsh communities or mature high marsh communities (Doody 1990). Apparent replacement of *S. alterniflora* by native salt marsh plants has been observed in Willapa Bay (ACOE 1992) and Padilla Bay (Bulthuis 1992, pers. comm.).

Animals

Importance of Native Habitats

Colonization of tidal flats and displacement of native flora by *Spartina* may have significant implications for the fauna associated with these communities. Unvegetated tidal flats in Washington are primary habitats for clams, oysters and other shellfish species, and other benthic organisms, and are critical feeding areas for shorebirds and waterfowl, particularly during spring and fall migrations. Small mammals, such as shrews and moles, also utilize salt marsh areas. Fishes may also utilize tidal flats and salt marshes as feeding areas during incoming tides. Eelgrass beds are important seasonal and year-round habitat for many fishes (Phillips 1984). Permanent residents include tube snout (*Aulorhynchus flavidus*), English (*Parophrus vetulus*), rock (*Lepidopsetta bilineata*), and C-O sole (*Pleuronichthys coenosus*), and buffalo (*Enophrys bison*) and Pacific staghorn sculpin (*Leptocottus armatus*). Numerous fish species including Pacific herring (*Clupea harengus pallasi*) and top smelt (*Atherinops affinis*) spawn in eelgrass beds (Mayer 1989, Phillips 1984). Eelgrass is also important nursery habitat for salmonids (*Oncorhynchus* spp.), striped seaperch (*Embiotoca lateralis*), shiner perch (*Cymatogaster aggregata*), several species of sculpin, gunnel, sole, and other fishes. Oysters, clams, shrimp, crabs, polychaete worms, snails, and other invertebrates inhabit the various niches of an eelgrass community, including the water column, sediment, sediment surface, and roots and blades of eelgrass. In addition, shorebirds, waterfowl, and gulls eat a variety of foods from eelgrass beds, including eelgrass seeds and blades, epifauna, and herring and other fish and invertebrate eggs. The fishes that inhabit eelgrass beds are also prey for birds and other fishes.

Habitat Values of *Spartina* Marshes

Native *Spartina* marshes on the Atlantic and Gulf coasts provide habitat for numerous macroinvertebrates, including snails, clams, crabs, grass shrimp, amphipods, isopods, worms, and insects; and fishes, such as *Fundulus* spp., weak fish (*Cynoscion regalis*), striped mullet (*Mugil cephalus*), bay anchovy (*Anchoa mitchilli*), Atlantic silverside (*Menidia menidia*), and Atlantic menhaden (*Brevoortia tyrannus*) (LaSalle *et al.* 1991). Primary prey species of *Fundulus heteroclitus* captured in *Spartina* marshes included fiddler crab juveniles, leafhoppers, scale insects, and copepods. Hettler (1989) collected 20 families and 35 species of fish from a regularly flooded *S. alterniflora* marsh in North Carolina including juveniles and adults of eight resident species, and juveniles of 26 estuarine-dependent and five marine species. Rountree and Able (1992) collected 64 species of fish, 13 species of invertebrates, and the diamondback terrapin in tidal creeks within a New Jersey marsh dominated by *S. alterniflora* and *S. patens*. *Spartina* marshes are documented nursery areas for many fish and shellfish species (Landin 1990). Some species such as the fiddler crab *(Uca pugnex)* and the ribbed mussel *(Geukensia demissa)* appear to have mutually beneficial relationships with *S. alterniflora* in New England marshes (Bertness 1984, 1985).

Muskrats (*Ondatra zibethica*) will eat *Spartina* although they prefer brackish or freshwater plants. Some waterfowl species, such as the black duck, *Anas rubripes* (rare in Washington state) (Iten 1992, pers. comm.) will feed on *Spartina* and upon invertebrates in the marshes. Wading birds, such as herons and bitterns, will also utilize *Spartina* marshes (Daiber 1974). However, waterfowl and many of the fish species that inhabit native *Spartina* marshes are most commonly associated with tidal creeks, channels, or unvegetated areas within the marshes or marsh edges.

Although native *Spartina* marshes support a variety of animals, environmental conditions (e.g., tidal patterns) and species composition are very different between East coast and Washington estuaries. The animals that effectively utilize native *Spartina* marshes have adapted over time to cordgrass habitats. Because of the fundamental differences between East coast and Washington estuaries, it is not possible to directly associate the habitat values of native *Spartina* marshes with those in Washington. Unfortunately, little information exists on the value of *Spartina* marshes in Washington to fish and wildlife. Potential positive and negative impacts to animals from *Spartina* infestations, based on the results of studies and personal observations conducted in Washington and elsewhere in the world evaluating the habitat value of introduced *Spartina* marshes, are discussed in the following section.

<u>Impacts to Animals</u>

Detailed studies documenting the impacts of *Spartina* spread in Washington waters to fish and wildlife populations have not been undertaken and, in some instances, will be complex to conduct. Thus, actual consequences to fish and wildlife from *Spartina* invasions are difficult to predict. Some native fish and wildlife species may successfully exploit *Spartina* habitats. However, significant losses of tidal flat, eelgrass, and/or salt marsh habitats may negatively impact animal species that have narrow habitat requirements and are unable to utilize *Spartina* habitats.

Invertebrates: The dense root mat and thick layer of soft, fine sediments associated with *Spartina* marshes may preclude colonization by and the survival of shellfish species that commonly occur in tidal flats or eelgrass beds. Thus, populations of some species may decline in areas with extensive *Spartina* infestations. A preliminary investigation to assess the impacts of *S. alterniflora* to benthic macroinvertebrates in Willapa Bay showed, in some plots, decreases in numbers or absence of shellfish species (*Tapes semidecussata*, *Mya arenaria*, and *Macoma balthica*) in areas colonized by expanding clones of *S. alterniflora* compared to adjacent bare flats (Atkinson 1992). In addition, benthic samples taken in a mature *S. alterniflora* stand were devoid of the shellfish, amphipod, and polychaete species present in bare tidal flats and the younger *Spartina* stands. Although the results of this study are inconclusive because of small sample sizes, they do indicate potential exclusion of shellfish and other benthic macroinvertebrate species from *Spartina* marshes. A New Zealand study found low invertebrate diversity in *S.* x *townsendii/anglica* marshes compared to native habitats (Franko *et al.* 1985).

Mammals: Small mammals, such as shrews, voles, moles, and mice, will feed upon vegetation or invertebrates in salt marshes. They likely move from uplands into marsh areas at low tide to feed. In a trapping study conducted at the Willapa National Wildlife Refuge, large numbers of insectivorous vagrant shrews (*Sorex vagrans*) were captured in both a native and a lower, adjacent *S. alterniflora* marsh (Atkinson; Hidy 1992, pers. comm.). Additionally, Townsend voles (*Microtus townsendi*) have been reported to feed upon the shoots of *S. alterniflora* in Willapa Bay (Sayce 1992, pers. comm.). Thus, conversion of tidal flat to *Spartina* marsh may benefit some small mammal species by increasing the amount of available habitat. However, others may be negatively impacted by displacement of native salt marsh species.

The palatability of *Spartina* to large herbivores (elk and deer) has not been investigated. Herds of elk have been observed grazing on *S. alterniflora* in some areas of Willapa Bay (Fresh 1992, pers. comm.). However, in other areas, elk and deer have not been observed to heavily graze upon *S. alterniflora* in Willapa Bay. (Sayce 1992, pers. comm.). It is possible that elk and deer may no longer frequent areas where *Spartina* has replaced native salt marsh vegetation.

Birds: Conversion of large areas of tidal flats or eelgrass beds to *Spartina* marshes may also cause reductions in numbers of migratory and wintering shorebirds and waterfowl that can be supported in critical staging and wintering areas such as Willapa Bay, Padilla Bay, Skagit Bay, and Grays Harbor. Reductions in available habitat may lead to population declines in some species. *Spartina anglica* infestations have apparently impacted shorebird habitat in British estuaries. Numbers of dunlin (*Calidris alpina alpina*) overwintering in several British estuaries where *S. anglica* has spread over most of the intertidal flats where the birds feed, steadily decreased during a ten-year period (Goss-Custard and Moser 1988). Dunlin numbers declined at significantly higher rates in estuaries where *S. anglica* was present. Although other factors such as decreased survival and recruitment due to mortality during migration or on their Arctic breeding grounds may have been responsible or contributed to declines in dunlin numbers, the authors felt that the continuing spread of *S. anglica* may have been responsible. They hypothesized that *Spartina* affects bird numbers by decreasing available feeding habitat (the birds were not observed feeding in the *Spartina* marshes) and reducing feeding time within a tidal cycle, both of which would be expected to increase the rates of mortality and emigration to other areas. It also appeared that birds displaced from estuaries by *S. anglica* had been unable to reestablish in other estuaries within their overwintering range, perhaps because they lacked the behavioral flexibility to move. Furthermore, numbers of overwintering dunlin did not increase in estuaries where *S. anglica* had declined through natural dieback, indicating that areas where natural dieback has occurred may not be suitable for foraging.

The relationship among *S. anglica* spread, sediment and invertebrate composition, and shorebird use was examined at one estuary in Britain. Fine sediment and water content of the substrate increased with progression from open silt or sand flats to expanding clones to closed stands of *Spartina* (Millard and Evans 1984). Nutrient content of the substrate was highest and thickness of the oxygenated surface layer of the substrate was least for closed stands. The benthic invertebrate composition was also different between habitats. Worms, bivalves, and amphipods dominated the open flats while only gastropods occurred in high numbers in *Spartina* clumps and only shore crabs occurred at high densities in closed stands. Correspondingly, the diversity and total numbers of shorebirds feeding in *S. anglica* clumps and stands were dramatically lower than in tidal flats. It appeared that most species of shorebirds avoided areas of *Spartina*, however, low numbers of redshank (*Tringa totanus*), which tend to feed in loose flocks or as solitary individuals, were observed feeding only in patches of open mud in *Spartina* marshes. The tight flocking behavior of many species of shorebirds may preclude their use of *Spartina* marshes because the visual communication necessary to maintain flock structure is impaired. Moreover, the high density of *Spartina* stems is likely to prevent birds from landing easily and also to restrict their movement on the ground.

Some evidence also exists that local waterfowl populations may be negatively impacted by *Spartina*. Large flocks of migratory waterfowl and shorebirds that congregate in Willapa, Padilla, and Skagit bays, Grays Harbor, and other waters heavily feed on invertebrates in the tidal flats, and on eelgrass and its associated fauna. Smaller numbers of waterfowl also overwinter in Washington bays and estuaries. Willapa and Padilla bays are the two major overwintering areas for black brant (*Branta bernicula nigricans*) in Washington. Landin (1990) reported that geese and ducks on the Atlantic and Gulf coasts tend to use only salt marshes in which open water is interspersed with emergent marsh, and are rarely observed feeding in vegetated salt marshes. *Spartina alterniflora* and *S. patens* are the dominant salt marsh species on the Atlantic and Gulf coasts. Observations in Willapa Bay indicate that waterfowl no longer frequent areas that have converted from tidal flat to *S. alterniflora* stands (Hidy 1992, pers. comm.). Thus, conversion of large areas of tidal flats or eelgrass beds to continuous *Spartina* marsh will likely cause decreased availability of suitable waterfowl habitat. Possible barriers to bird utilization of *Spartina* stands include decreased accessibility to and mobility within stands and decreased diversity and abundance of prey inhabiting *Spartina* marshes. If behavioral controls or lack of habitat preclude emigration to other estuaries, populations could decline.

Elimination by *S. anglica* of the bulrush marsh species in Skagit and Port Susan Bays would have negative effects on wintering snow geese which utilize bulrush marsh species as their primary food resource. Dabbling ducks also utilize bulrush seed for a significant portion of their diet. Plants that compose the bulrush marshes of Skagit and Port Susan Bays include three-square bulrush (*Scirpus americanus*), hardstem bulrush (*Scirpus acutus*), arrowgrass

(*Triglochin maritimum*), and sedges (*Carex* spp.). These are preferred food sources for both greater and lesser snow geese as well as other waterfowl. If *Spartina* causes significant losses of these plants, snow geese would be forced to forage to a greater extent in adjacent agricultural fields where they would be vulnerable to hunting mortality. Loss of bulrush marshes would decrease the ability of these estuaries to support wintering waterfowl (Iten 1992, pers. comm.).

Results of a study monitoring waterfowl and shorebird use of native salt marsh and *S. alterniflora* stands in Padilla Bay will be available in February 1993 (Morris 1992, pers. comm.).

Although continuous stands of *Spartina* marsh appear unsuitable for waterfowl, impacts may be less deleterious when isolated *Spartina* clones are scattered throughout a tidal flat area. Such a situation may not impair flocking behavior and could provide the added benefit of increased cover. Similarly, sizable bare or open areas within large *Spartina* stands may provide adequate waterfowl habitat.

Herons and bitterns have been observed hunting in *S. alterniflora* stands in Willapa Bay (Sayce 1992, pers. comm.). Increased cover associated with *Spartina* marshes may benefit these birds. In addition, the physical structure of *Spartina* stands may benefit some songbirds. High densities of marsh wrens have been observed nesting in mature *S. alterniflora* stands in Willapa Bay.

Fish: Replacement of tidal flat, eelgrass beds, or native salt marsh with *Spartina* may have positive or negative effects on fish populations. Numerous fish species are associated with the surface and tidal channels of native *S. alterniflora* and *S. patens* salt marshes. Many Washington species could possibly adapt to *Spartina* habitats. The cover afforded by *Spartina* may be particularly beneficial to juvenile salmon, surfperch, and other species. However, this benefit may only apply for small clones and the edges of larger clones or stands. Additionally, retention of sediment in *Spartina* marshes could prevent further degradation of spawning habitats, for sturgeon and other species, that have apparently been impacted by siltation (Cheney *et al.* 1986).

Some fish species may not successfully utilize *Spartina*. For example, epibenthic predators such as sole and sturgeon may not adapt to the physical structure of *Spartina* marshes. In addition, *Spartina* marshes are probably not suitable spawning habitat for Pacific herring, smelt, or other species that typically spawn in eelgrass beds. The fine, soft sediments may smother eggs if they fall to the substrate, and the higher elevation of the marshes may decrease tidal inundation and cause dessication of eggs. In some areas, vertical accretion of sediments may prevent entry of tidal waters and fish in *Spartina* marshes. Stranding of fish that entered marshes at extremely high tides could also occur on the outgoing tide, leaving them susceptible to predation, temperature extremes, and low dissolved oxygen.

Small clones or the edges of larger *Spartina* stands may be suitable cover or feeding habitat for some species. However, the value of isolated clones to fish is temporary since clones eventually grow together into closed stands. A one-year study in Willapa Bay, the only study to date to investigate *Spartina* use by Washington fishes, found no significant differences (at a 90% level of confidence) between the total number of fish collected in *S. alterniflora* clones 3-6 meters (10-20 feet) in diameter and mudflat habitats (Allard 1991). Species composition was also similar between habitats; shiner surfperch (*Cymatogaster aggregata*), staghorn sculpin (*Leptocottus armatus*), surf smelt (*Hypomesus pretiosus pretiosus*), threespine stickleback (*Gasterosteus aculeatus*), northern anchovy (*Engraulis mordax*), and juvenile chum (*Oncorhynchus keta*) and chinook salmon (*Oncorhynchus tshawytscha*) were collected in both habitats. Total numbers of some of these species were higher in *Spartina* than in mudflat habitats. Several species were also collected in very low numbers in either habitat.

Although some fish may utilize *Spartina* as cover, they may make only limited use of its associated prey base. For example, feeding of juvenile salmon may be limited in *Spartina* because favored prey species, such as epibenthic filter-feeding harpactecoid copepods, may not survive in the fine, soft sediments of *Spartina* marshes.

Large losses of tidal flat or eelgrass habitats could negatively impact some native fish species that depend on these areas for feeding, spawning, or rearing habitats. In addition, prey items, particularly epibenthic fauna and their consumers, would likely be different in *Spartina* marshes than in tidal flat, eelgrass, or native salt marsh habitats. The effects of such shifts on fish populations and predator-prey relationships are unknown.

Lastly, *Spartina* stands located at river mouths may constrict access routes for migrating adult or juvenile salmon (*Oncorhynchus* spp.), steelhead (*Oncorhynchus mykiss*), or sea-run cutthroat (*Oncorhynchus clarki*). Unless access is completely blocked, however, migrations should not be impeded.

Commercial and Recreational Fisheries: As previously stated, detailed studies documenting the impacts of *Spartina* spread in Washington waters to fish and wildlife populations have not been undertaken. Thus, impacts of *Spartina* spread to fisheries are difficult to accurately predict.

Commercial fish and shellfish populations may be negatively impacted through conversion of large areas to *Spartina* causing reductions in available habitat, and associated alterations in trophic dynamics that may lead to decreases in prey availability. Commercial species that may be affected include several species of salmon and flatfish, sturgeon , and dungeness crab (*Cancer magister*). It is also possible that some species may be unaffected or benefit from *Spartina*. For example, juvenile salmon and crab may utilize *Spartina* marshes for food and cover, although eelgrass is their preferred habitat. Fishing could also be impacted by loss or impediment of navigable waters and fouling of nets and other gear by wracks of *Spartina* litter. *Spartina* infestations could cause significant economic hardships in the commercial fishing industry.

Threatened and Endangered Species

No information was found that documents known impacts of *Spartina* infestations on federally or state-listed threatened or endangered species in Washington. However, inferences can be made regarding the potential impacts of *Spartina* on threatened or endangered species.

Table 3 lists the threatened and endangered wetland plant species in Washington (WDOE 1991a, b). None of the habitats currently occupied by these plants in Washington are vulnerable to infestations by *Spartina* species. However, two state sensitive species, Alaska alkaligrass (*Puccinellia nutkaensis*), known from Puget Sound, and sharp fruited peppergrass (*Lepidium oxycarpum*), known from San Juan county, could potentially be impacted by *Spartina* (Gamon 1992, pers. comm.).

Populations of three state endangered animal species, brown pelican (*Pelecanus occidentalis*), Aleutian Canada goose (*Branta canadensis leucopareia*), and peregrine falcon (*Falco peregrinus*), and two state threatened species, bald eagle (*Haliaeetus leucocephalus*) and marbled murrelet (*Brachyramphus marmoratus*), could potentially be impacted by *Spartina* infestations (Kilbride 1992, pers. comm.). Populations of these bird species could be impacted by changes in availability of prey species or foraging habitats resulting from *Spartina* infestations.

Trophic Dynamics

Sources of organic matter for consumers (herbivores, detritivores, and predators) in Pacific Northwest estuaries include phytoplankton, benthic algae, seagrasses (e.g., *Zostera* spp.), seaweeds (benthic macroalgae), marsh plants, and allochthonous inputs from rivers and streams (Thom 1987). Detrital pathways are vital links in estuarine food webs. The pathway from detritus to epibenthic crustacea to fish, birds, and mammals is particularly important. Benthic macro- and microalgae may be the primary energy resource for detritus-based food webs in Pacific Northwest estuaries (Simenstad and Wissmar 1985 cited in Thom 1987).

The use of plant matter by animals is based on food preferences of the consumers. It appears that algal material is preferred over other plant matter by herbivores and detritivores because algal tissue is more easily assimilated (Pomeroy 1977; Simenstad and Wissmar 1985 cited in Thom 1987). Studies in southern California salt marshes found that *Spartina foliosa* and *Salicornia virginica* detritus were poor food sources for mussels (Zedler 1982). Detritus from flowering plants such as grasses and sedges requires microbial breakdown over an extended period (perhaps several months) to allow assimilation by detritivores. Detritus from flowering plants may be more important as an energy source when algae are not abundant, although data are lacking to substantiate this hypothesis (Thom 1987).

Large *Spartina* infestations could potentially alter the trophic dynamics of Washington estuarine systems by reducing nutrient inputs to estuarine waters, causing decreases in epibenthic or epiphytic algal production, changing habitat availability for animals, and causing major shifts in invertebrate prey bases. Native *Spartina* marshes are highly productive and may contribute a considerable amount of organic material and nutrients to the estuarine ecosystem (Marinucci 1982). *Spartina alterniflora* is also capable of accumulating, transforming, excreting, and transporting a variety of heavy metal compounds (Kraus 1989). However, *Spartina* marshes may also act as nutrient sinks, indicating that *Spartina* productivity does not always subsidize the high productivity of surrounding estuarine waters. Reductions in nutrient inputs to estuarine waters could occur if *Spartina* marshes that act as nutrient sinks replaced habitats that provided nutrients to surrounding waters through living and detrital matter.

In addition, epibenthic and epiphytic algae populations may be limited in *Spartina* stands due to the effects of shading, increased water temperatures, and increased frequency of desiccation. Kathleen Sayce (1992, pers. comm.) found lower numbers of live diatoms inside *S. alterniflora* clones in Willapa Bay than in the surrounding tidal flats. She also counted an order of magnitude fewer protozoans inside than outside clones. Changes in physical habitat availability for algae will probably cause shifts in the associated herbivores and detritivores that are primary prey for fish, such as juvenile salmonids. A study of outmigrating juvenile chum salmon in Hood Canal determined that eelgrass and its associated algae are the basis of the food web for this salmon species in the estuary (Simenstad and Wissmar 1985 cited in Thom 1987). These findings may differ for other Northwest estuaries and salmon species, however, this example illustrates the importance of epiphytic and epibenthic algae to estuarine ecosystems.

Changes in habitat availability for plants and animals and shifts in prey bases, particularly benthic and epibenthic fauna for higher level consumers, will probably occur in infested estuaries and may drastically alter estuarine food webs. Although the long-term effects of such changes are unknown, it is possible that competition among organisms for decreased resources and inability of organisms to adapt to the new physical conditions in *Spartina* stands may negatively impact populations of fish and wildlife in the state.

3.2.1.4 Areas Specifically Designated for Preservation

The impacts of *Spartina* infestations to areas that were specifically established to preserve and protect the natural environments and flora and fauna of the state, such as wildlife refuges, natural areas and reserves, are of particular concern. Continued spread of *S. alterniflora* and *S. anglica* is likely under the no action alternative. Negative impacts to some components of natural ecosytems will likely occur with continued spread. Such impacts may not be compatible with the purposes for which areas such as refuges or natural reserves were established.

3.2.1.5 Recreation and Aesthetics

Colonization of tidal flats, river deltas, and other areas by *Spartina* could be perceived to lower their aesthetic value. Conversely, *Spartina* marshes may be considered aesthetically pleasing. Recreational opportunities could also be affected by *Spartina* infestations. Loss of beach habitat, reduced access to water, beaches or streams, loss of navigation routes, and alterations to estuarine ecosystems may result from *Spartina* growth. Such changes could negatively impact recreational activities including sport fishing, waterfowl hunting, bird watching, recreational shellfish harvesting, boating, and hiking. Conversely, in some areas, *Spartina* colonization may increase access for waterfowl hunters. Reductions in recreational use in some areas may be viewed positively.

3.2.2 Agricultural/Aquacultural Environment

This section describes possible impacts of *Spartina* infestations to agricultural and aquacultural environments, species, and practices.

3.2.2.1 Sediment Composition and Dynamics

Spartina impacts to the sediment composition and dynamics of estuarine agricultural/aquacultural environments are the same as those described under the natural environments section. Increases in sediment accretion and changes in sediment composition and substrate topography of intertidal areas apparently occur in areas colonized by *Spartina*. The specific effects of these changes on aquaculture species and practices are discussed in Section 3.2.2.3.

3.2.2.2 Water Quality and Movement

The effects of *Spartina* infestations on the water quality and water circulation patterns of agricultural/aquacultural areas are similar to those described under the natural environments section. Decreases in dissolved oxygen content may seasonally occur from the natural decomposition of large quantities of *Spartina* litter. Suspended sediment content of water exiting *Spartina* marshes may be lower. Flooding frequency and extent may increase in shoreline and river delta areas. The specific effects of these changes on aquaculture species and practices are discussed below.

3.2.2.3 Affected Species and Practices

Aquaculture

Spartina species are colonizing intertidal habitats that support commercial aquaculture of oysters and clams. The most commonly cultured oyster species is the Pacific oyster (*Crassostrea gigas*), a native of Japan. This species is cultivated in Willapa Bay, Grays Harbor, and Puget Sound. The native Olympia oyster (*Ostrea lurida*) is also cultivated in southern Puget Sound. Limited culture of Eastern oysters (*Crassostrea virginica*), European flat oysters (*Ostrea edulis*), and several hybrid varieties of Pacific oyster may also occur in some areas. The most commonly cultured clam species are the manila or Japanese littleneck clam (*Tapes japonica*) and the Pacific or native littleneck clam (*Protothaca staminae*). Limited culture of butter clams (*Saxidomus giganteus*) also occurs (Cheney *et al.* 1986).

The primary methods for growing oysters are ground, longline, and rack culture. In ground culture, the most common method, whole or broken "mother" shells with attached immature oyster larvae or "seed" are spread onto the substrate and left to grow. Seed is obtained from hatcheries or from natural reproduction in local waters. Pacific oysters reach marketable size in two to four years. Oysters are harvested from the beds either by hand at low tide or mechanically at high tide. Ground culture is not well-suited for soft substrates because oysters may sink into soft bottoms or be smothered by depositions of silt or mud.

Longline and rack culture, however, are suitable for use in both firm and soft substrates since oysters are suspended off the bottom. In longline culture, mother shells with seed are attached to lines strung between upright poles and suspended 30-90 cm (1-3 feet) above the bottom. In rack culture, trays or mesh bags filled with oysters are placed in racks that are suspended off the bottom. Either seed attached to mother shell or free-swimming larvae can be cultured using racks. Rack culture is practiced less frequently than ground or longline culture.

Oysters grown using ground and longline methods are usually shucked or removed from the shell prior to sale. Total production costs for these culture methods are generally similar (Cheney *et al.* 1986). Costs to grow oysters using rack culture are higher than either ground or longline culture. However, the increased costs can be offset because oysters grown by rack culture may command a higher price due to their suitability for the half-shell trade.

Clams are commercially cultivated on self-propagating beds. Their ground culture requires a firm substrate dominated by sand and gravel. Clams burrow several inches into the substrate and typically filter their food from water circulating through the substrate (that is, they do not expose their siphon tubes to open water). A coarse-grained substrate is inherently stable and allows adequate water circulation within a burrow. *Spartina* species can colonize a broad range of substrates including sand and gravel and could potentially establish in clam beds.

It is uncertain how *Spartina* infestations will impact aquaculture activities. However, conversations with oyster growers in Willapa Bay (contacts are listed in references section) indicated that *Spartina* stands are probably not suitable habitat for aquaculture species currently grown by ground culture because fine sediments accreted in *Spartina* marshes could smother shellfish and increase elevation of bed areas above levels of regular tidal inundation. Dense root mats may also preclude burrowing by clams. Thus, conversion of tidal flats to *Spartina* marsh will result in loss of areas currently suitable for ground culture. In Willapa Bay, all of the oyster growing lands abandoned due to growth of *S. alterniflora* have been seed beds, which are generally high in the intertidal zone. Oyster grower Lee Weigardt estimates that 25 to 30 % of the Class IV oyster beds in Willapa Bay, which are used primarily as seed beds, have been taken out of production because of *Spartina* infestations. Although Class IV oyster lands are of poor quality to grow out adult oysters, they are valuable as sources of seed. The loss of seed beds would result in a greater dependence on larvae grown in hatcheries. This may not be desirable if the quality of hatchery seed is inferior to that harvested directly from Willapa Bay.

Bed areas not directly displaced by *Spartina* may be harmed from deposition of large amounts of *Spartina* litter. Conversely, the sediment trapping ability of *Spartina* may prevent siltation of downslope beds. Representatives of the aquaculture industry in Willapa Bay are also concerned that *Spartina* infestations may lead to massive changes in the availability of planktonic and detrital food sources for shellfish, and to alterations in water circulation and current patterns that may affect the quality and quantity of food carried to the beds. Finally, increased frequency and extent of flooding that may result from *Spartina* infestations could impact aquaculture facilities and equipment.

Livestock Pasturing

Spartina alterniflora, S. patens, and *S. anglica* are palatable to livestock such as cattle and sheep. Thus, increased availability of pasturage may result from conversion of tidal flats to *Spartina* marsh.

3.2.2.4 Economic Impacts

Spartina infestations may have significant economic impacts in regions where ground culture is the primary method of shellfish aquaculture. It is possible that oyster longline and rack culture could occur in some infested areas, since oysters are suspended off the bottom, if adequate tidal exchange occurs within the *Spartina* stand. Long-term impacts to aquacultural economies will depend on the feasibility of expansion in the use of longline, rack, and other culture methods compatible with *Spartina* presence. Infestations of *Spartina* may also present opportunities for expansion of livestock ranching in some areas, although water quality concerns may preclude the introduction of or pasturing of large numbers of livestock at certain sites. An additional concern is the consumption of ergot-infested *Spartina* by livestock. The alkaloids present in ergot cause abortion in cows and gangrene of the hooves and tails in cattle (Bold *et al.* 1980). These alkaloids can also be deadly to humans when consumed.

3.2.2.5 Sensitive Agricultural/Aquacultural Areas

Infestations of *Spartina* would likely be detrimental to the state oyster reserves. These areas were set aside early in the century to ensure continued existence of the Olympia oyster which was being overharvested. The reserves today also provide limited quantities of Pacific oyster seed for growers (Cheney *et al.* 1986). Thus, displacement of oyster beds in the reserves would likely decrease availability of local Pacific oyster seed.

3.2.3 Built Environment

This section describes possible impacts of noxious species to the built environment, including residential and commercial properties, public works developments, and parks.

3.2.3.1 Sediment Composition and Dynamics

Increased rates of sediment accretion and changes in sediment composition and substrate topography could occur in intertidal lands associated with private residences, commercial properties, parks, etc., from *Spartina* colonization.

3.2.3.2 Water Quality and Movement

Extensive *Spartina* infestations may cause increased frequency and extent of flooding in shoreline areas. Residences, resorts, parks, and other areas could potentially be impacted by floods.

Public water supplies located near *Spartina*-infested areas may also be impacted by increased flooding. Decreased water quality in surface water supplies or shallow wells may result from sediment loads in floodwaters or salt water intrusion from tidal waters. Additionally, water table levels in septic fields may rise during periods of flooding. Septic systems may be unusable at certain times of the year and leachates from septic systems could impact the quality of surrounding ground or surface waters.

3.2.3.3 Recreation, Aesthetic, and Cultural Resources

Conversion of tidal flats to *Spartina* marsh may be perceived positively or negatively. In addition, recreational opportunities on private or park lands may be altered. Beach areas may be lost and beach and water access may be diminished by *Spartina* growth. Opportunities for boating, bird-watching, recreational fish and shellfish harvesting, etc., may be impacted.

Areas of archaeological significance may also be affected by *Spartina* growth. Opportunities for scientific study and public enjoyment of such areas could be reduced. However, the long-term preservation of some areas could be enhanced from sediment deposition in *Spartina* marshes.

3.2.3.4 Economic Impacts

Property damage from flooding may occur in some areas. Costs will be incurred by property owners wishing to maintain ready access to beach and water. Costs will also be incurred by other parties undertaking control efforts. Property values may also be impacted due to the perceived aesthetic changes associated with *Spartina* growth. Reductions in recreational use of areas infested with *Spartina* may negatively impact the economies of some regions. Conversely, reductions in recreational use of some areas may be viewed positively.

3.3 ECONOMIC USES OF *SPARTINA*

Beneficial uses of *Spartina* may exist. Development of economically viable uses of *Spartina* species may directly contribute to their control and offset some of the economic impacts associated with infestations.

Several residents of Willapa Bay have been experimenting with production of paper from *S. alterniflora*. Initial results indicate that *Spartina* paper may be a quality product for the specialty paper trade (Wiegardt 1991, pers. comm.). *Spartina* paper products may also be suitable for use in a variety of crafts and as packing material, flooring, or room dividers.

4.0 PURPLE LOOSESTRIFE

This section addresses the efficacy and impacts of the no action alternative for managing infestations of *Lythrum salicaria*, and possibly *L. virgatum*, in Washington. Major positive and negative effects are summarized in Table 2.

4.1 EFFICACY OF NO ACTION ALTERNATIVE AS A MANAGEMENT TOOL

Purple loosestrife is infesting natural, disturbed, and constructed freshwater wetland communities in Washington. Associated impacts, which are discussed in Section 4.2, may include displacement of native species and the resulting loss of species diversity, reduction and degradation of available habitat for wildlife (primarily waterfowl, shorebirds, and fur-bearing mammals), decreases in irrigation canal and ditch channel capacities, degradation of hunting and fishing areas, and declines in the quality of wetland pastures and hay fields.

If natural processes effectively stabilize or decrease the distribution of purple loosestrife at the regional level, the no action management scenario could control this plant in Washington. Vegetative spread within individual infestations is limited by the availability of suitable

Table 2. Summary of major effects of the no action alternative for managing purple loosestrife infestations in Washington wetlands.

Potential Positive Effects	Potential Negative Effects
• Abundant source of pollen and nectar	• Decreased species diversity
• Aesthetic and horticultural values	• Decreased cover and forage for wetland wildlife—primarily waterfowl, shorebirds, and fur-bearing mammals
	• Decreased irrigation canal and ditch channel capacity and concomitant increase in operation and maintenance costs
	• Degradation of hunting and fishing opportunities and areas
	• Continued spread

habitat and, to some degree, by competition with existing wetland vegetation. It is therefore expected that the areal extent of individual infestations will stabilize once all suitable habitat has been exploited. Suitable purple loosestrife habitat includes alluvial floodplains, wetlands, irrigation wasteways and canals, roadside ditches, wet pastures, stream or river banks, and pond and lake shores (Stuckey 1980).

Stabilization of a purple loosestrife infestation throughout a local area also depends on the populations capacity to produce viable propagules. Purple loosestrife is effective at both sexual and vegetative modes of spread. Individuals of this species produce copious amounts of seed (production increases with age, size, and vigor of plant). The seed maintains viability of over 80% for at least 3 years (Parker and Burrill 1992) and germinates across a broad range of environmental conditions (Shamsi and Whitehead 1974). Vegetative dispersal mechanisms are also extremely effective in increasing the extent of a purple loosestrife infestation. Adventitious roots and shoots will sprout from clipped, trampled, or buried stems. These vegetative propagules are most effectively dispersed in riverine, lacustrine, or other open water systems (Heidorn and Anderson 1991). Wetlands with undisturbed, resident emergent vegetation are less susceptible to invasion because purple loosestrife seedling spread will be minimized due to competition for space and sunlight.

Apparent stabilization of a purple loosestrife infestation can be disrupted by either natural or human disturbance. Wetland habitats that have been disturbed or degraded by drought, draining, drawdowns, mud flat exposure, bulldozing, siltation, shore manipulation, dredging, and/or human, livestock and wildlife trampling often experience large population explosions (Henderson 1987). Moist areas exposed following drawdowns or other disturbances have been quickly colonized if purple loosestrife is a component of the existing vegetation. This population explosion is due to the presence of a large purple loosestrife seed bank. Abundant seedling densities [10,000 to 20,000 germinations/meter^2 (10.8 ft^2)] suppress the growth of native seedlings. Native vegetation is eventually displaced (Rawinski 1982). If an infestation is left unchecked, wetland eventually may convert to a monoculture of purple loosestrife.

Once established, infestations of purple loosestrife do not appear to be short-lived and the displacements of native species by purple loosestrife are seemingly permanent. In the northeast United States, monospecific stands of this plant have been self-maintaining and robust for more than 20 years (Thompson 1989). However, the longevity of purple loosestrife populations differs between North America and in its native distribution. In general, stands of purple loosestrife are more stable in North America than European purple loosestrife communities. In the British Isles and central Europe, purple loosestrife may invade disturbed areas and temporarily form dense monospecific stands. However, within a few years, purple loosestrife is only an occasional component in mixed-species stands in these areas (Shamsi and Whitehead 1974; Bodrogkozy and Horvath 1977, 1979).

Thompson *et al.* (1987) suggested three major factors to explain the relative stability of North American stands of *L. salicaria*: (1) the insect fauna associated with the plant in North America is missing several important species that decrease the vigor of European plants; (2) the selective foraging of muskrat (*Ondatra zibethica*) on cattail (*Typha* spp.) seems to facilitate the spread of purple loosestrife by reducing the cattail population (providing opportunity for purple loosestrife encroachment) and, incidental muskrat damage to purple loosestrife plants produces stem fragments that become vegetative propagules; and, (3) North American *L. salicaria* biotypes may be more adaptive and vigorous than European stock.

While local (intrapopulational) stabilization depends primarily on availability of suitable habitat, regional stabilization primarily depends on vectors for interpopulation dispersal. That is, the expansion of existing infestations would slow and eventually stabilize if current infestations did not have the opportunity or ability to migrate. There are several factors that, if left uncontrolled as proposed in the no action alternative, will continue to provide opportunity for dispersal of purple loosestrife to presently uninfested areas.

Areas most likely impacted by purple loosestrife expansion under the no action alternative are those areas interconnected to existing drainage channels (natural or artificial). The drainage channels provide opportunity for the dispersal of floating seed or vegetative propagules. Areas isolated from surrounding drainage channels are relatively secure from infestation (Thompson 1989). Additionally, the shape and continuity of a river or waterway determines its susceptibility to colonization. Streams with low gradients tend to have wider floodplains, meandering channels, and silt and sand deposits. These conditions offer abundant opportunity for floating seeds and propagules to settle and colonize. Water courses with high gradients are relatively less suitable for colonization. Streambank cover is also an important determinant with respect to vulnerability. Streams bordered by woody vegetation have well-shaded banks (low light situations) that preclude establishment of purple loosestrife seedlings (Thompson 1989).

Highway corridors are also susceptible to purple loosestrife infestation and represent a migration corridor (Wilcox 1989). Construction and maintenance of highways create disturbed sites that favor the establishment of purple loosestrife. *Lythrum salicaria* seedlings are tolerant of common roadside contaminants found in snowmelt. Isabelle *et al.* (1987) demonstrated that purple loosestrife was able to germinate even though snowmelt contained major contaminants such as sodium, chlorine, calcium, and lead and minor contaminants such as nickel, copper, zinc, iron, cadmium, oils and greases. Of the six wetland species tested, only one other species (*Typha latifolia*) was able to germinate under these conditions. Migration of purple loosestrife along the highway corridor is often along short distances. The spread of seed and vegetative propagules is limited by drainage patterns, topographic features, and physical barriers. Culverts also provide opportunity for downstream dispersal. Wilcox (1989) found that seeds may also be transported in wind currents created by trucks passing at high speed.

Surveys of native herbivores associated with *L. salicaria* in North America were conducted in the northeastern United States (Hight 1989) and in Washington (G. Piper 1992, pers. comm.). Purple loosestrife experiences a low level of herbivory by unspecialized phytophages. None of the arthropods collected on *L. salicaria* reduced populations of the plant or caused appreciable damage. Stabilization or control of purple loosestrife by the native arthropod fauna is, therefore, unlikely.

There are only a few reports of animals grazing or feeding on purple loosestrife. White-tailed deer (*Odocoileus virginianus*) were observed grazing on purple loosestrife in a wet meadow in New York (Rawinski and Malecki 1984), but the grazing had little impact on plant growth. Muskrats occasionally used dead loosestrife stalks as building material for dens but did not feed on it. As mentioned above, the selective foraging behavior of muskrats may even accelerate invasion by the plant.

In North America and Europe, literature studies and field surveys were conducted to investigate plant pathogens associated with purple loosestrife. The literature studies yielded little information and no important plant pathogens associated with this plant were collected during the field studies (Malecki *et al.* 1991). It is therefore unlikely that pathogens will contribute to the natural control of purple loosestrife.

The rate of spread of purple loosestrife between 1940 and 1980 has been estimated to be relatively slow—approximately 1,160 km^2/year (381 mi^2/yr) (Thompson 1989). Based on available information, the rate of spread will probably continue if no management efforts are undertaken.

4.2 IMPACTS

4.2.1 Natural Environment

4.2.1.1 Sediment Composition and Dynamics

Little information exists on the possible impacts to the sediment dynamics of waterways and wetlands. Sediment and detrital debris have been observed to build up around the roots of purple loosestrife (Bender 1988). In wetlands and areas with slow-moving water, this build-up enables purple loosestrife to gradually invade deeper water and to form dense stands that fill in open water areas. These stands shade out other emergents and eliminate floating vegetation and open water habitat. The no action alternative would probably result in increased sediment deposition in waterways and wetlands as infestations expand.

4.2.1.2 Water Quality and Movement

Impacts of purple loosestrife infestations on the water quality and groundwater recharge/discharge functions of wetlands have not been examined. Potentially, increased sedimentation associated with purple loosestrife infestations could decrease the water storage capacity of a wetland. Decreased water storage capacity could reduce the detention time contaminate-laden water remains in a wetland, thus reducing the water quality of the discharge. Such alterations in wetland hydrology and the potential increases in evapotranspiration caused by infestations may decrease the amount of groundwater recharge.

These impacts may ultimately affect the groundwater on which some public water supplies depend. Increased vegetation and sediment deposition could also potentially alter flows and currents in infested watercourses.

4.2.1.3 Biota

Possible impacts to the biotic communities of Washington wetlands, streams, and rivers from the invasion of purple loosestrife have not been well-studied. However, the loss of plant species diversity associated with purple loosestrife infestation will cause elimination of natural foods and cover essential to many wetland wildlife inhabitants (Malecki *et al.* 1991). Based on the few studies that have been conducted, the primary impact is habitat loss for birds and fur-bearing mammals. Wildlife impacts are closely related to the replacement of native plant communities. Additionally, purple loosestrife colonization of moist soil areas results in loss of open-water space, which may also have negative affects on wildlife.

<u>Birds</u>

In Voyageurs National Park, Minnesota, waterfowl avoided wetlands dominated by purple loosestrife and duckling productivity significantly decreased in infested wetlands (Benedict and Grim 1989). One of the primary impacts to waterfowl resulting from colonization by purple loosestrife is the loss of traditional food sources. Also, stands of purple loosestrife [which can be up to 3 meters (9.8 ft) high] are dense at the top but open at the base. The large root masses can be separated by 0.6 to 0.9 m (2 to 3 ft) of water, leaving no cover for surface-nesting ducks (Timmerman 199X).

Studies in a created impoundment in central New York (Thompson *et al.* 1987) documented the complete infestation within a thirteen-year period by purple loosestrife. Prior to the infestation, vegetation consisted of cattail and floating aquatics, and waterfowl (primarily wood ducks and mallards) were utilizing the area. Within thirteen years, purple loosestrife represented 90% of the vegetation biomass, having displaced the more desirable original mixture of native food and cover species. Canada geese foraging and cover areas were also negatively affected. Thompson *et al.* (1987) suggest that shifts from cattail to purple loosestrife decrease the carrying capacity of the habitat for waterbirds.

Other effects associated with the invasion of purple loosestrife include the loss of mudflats used as feeding areas by shorebirds and loss of nesting opportunities. Red-winged blackbirds, however, nested in purple loosestrife to a greater degree than in cattail (Rawinski and Malecki 1984). There was no evidence of any bird species feeding on purple loosestrife.

<u>Mammals</u>

In wetlands where purple loosestrife has displaced cattail, the habitat value for muskrat declines because cattail is their principal food and cover plant. Although the impact of purple loosestrife on North American muskrat populations has not been measured, Thompson *et al.* (1987) suggest several interactions are probable. As mentioned above, muskrat probably accelerate the invasion of purple loosestrife and subsequent dominance because they selectively forage cattail, a principal competitor. While cattail stems are heavily exploited by muskrats, purple loosestrife stems are only partially cut and not eaten. Each of these partially cut stems represents a potential vegetative propagule. The shift from cattail to purple loosestrife could cause a decrease in the carrying capacity of the habitat for muskrats. The importance of purple loosestrife as a forage plant for other animals has not been studied. White-tailed deer were observed grazing on purple loosestrife plants in a wet meadow (Rawinski and Malecki 1984). The importance of this species in their diet has not been documented.

Fish

No information is available on the impact of purple loosestrife infestations on fish populations.

Threatened and Endangered Species

Monospecific stands of purple loosestrife can jeopardize threatened and endangered native wetland plant and animal species. Thompson *et al.* (1987) provide examples from other parts of North America, including a threatened bulrush (*Scirpus longii*) in Massachusetts, dwarf spike rush (*Eleocharis parvula*) in New York, the bog turtle (*Clemmys muhlenbergi*) in northeastern United States, the black tern (*Chlidonias niger*) in the north-central states, and the canvasback duck (*Aythya valisineria*) in the prairie potholes region. The Pacific Northwest has several state-listed threatened and endangered species that rely on wetlands during their life cycle. Without management, purple loosestrife could contribute to local extirpations or declines of these vulnerable species.

Plants: Table 3 lists the threatened and endangered wetland plant species in Washington state (WDOE 1991a, b) and summarizes the vulnerability of their habitats to purple loosestrife infestation. Based on their preferred habitat and plant associates, Kalm's lobelia (*Lobelia kalmii*) and howellia (*Howellia aquatilis*) are at relatively high risk from purple loosestrife infestation. The degree of risk depends on the vigor of the endangered population, the level of disturbance within the habitat, and the proximity and accessibility of purple loosestrife infestations. Populations of yellow lady's slipper (*Cypripedium calceolus* var. *parviflorum*) in vernal pool habitats in the channeled scablands are moderately vulnerable and may be jeopardized by nearby purple loosestrife infestations. Species listed in Table 3 as having low vulnerability to purple loosestrife occupy habitats typically not exploited by purple loosestrife.

Animals: The Western pond turtle (*Clemmys marmorata*) (state-listed threatened) is the only state-listed or candidate endangered or threatened species confined to wetland habitats in Washington state (WDOE 1991a, b). However, three state endangered species, the peregrine falcon (*Falco peregrinus*), sandhill crane (*Grus canadensis*), and Columbian white-

Table 3. State threatened and endangered plant species that may be found in wetlands in Washington.

SPECIES	GENERAL HABITAT	VULNERABILITY TO PURPLE LOOSESTRIFE
Threatened		
Calamagrostis crassiglumis (thick-glume reedgrass)	Coastal freshwater marsh; lakeshores. One historical record from brackish marsh.	MODERATE. Purple loosestrife does inhabit freshwater marshes and lakeshores.
Corydalis aquae-gelidae (Clackamas corydalis)	Mostly >2500 feet elev. in Pacific silver fir zone. Forested boggy areas, small streams, seeps, and springs	LOW. Purple loosestrife does not occur in these habitats.
Lobelia kalmii (Kalm's lobelia)	Marshlands or springs, along shores and in other wet areas	HIGH. Close to existing infestations.
Platanthera chorisiana (Choriso bog orchid)	Moist areas, especially at the edge of subalpine streams or bogs	LOW. Purple loosestrife rarely occurs in subalpine zone.
Sisyrinchium sarmentosum (pale blue-eyed grass)	Perimeter of moist meadows in mixed coniferous forests, 2500-3000 feet elev.	LOW. Purple loosestrife rarely occurs at these elevations.
Endangered		
Cypripedium calceolus var. *parviflorum* (yellow lady's slipper)	Shady areas. Bogs to damp, mossy woods at high elevations. Mostly forested habitats or in vernal pools in channeled scabland.	MODERATE. Purple loosestrife not very shade tolerant. Some populations close to existing purple loosestrife infestations in vernal pools in channeled scabland.
Delphinium viridescens (Wenatchee larkspur)	Highly restricted in boggy meadowland in Wenatchee Mountains	LOW. Isolated and at higher elevation than most purple loosestrife infestations.
Howellia aquatilis (howellia)	In vernal or permanent ponds and lakes, channeled scablands, and Turnbull Wildlife Refuge	HIGH. Usually associated with potential purple loosestrife habitat.
Liparis loeselii (twayblade)	Mostly around springs and in Sphagnum bogs	LOW. Unlikely purple loosestrife habitat.
Polemonium pectinatum (Washington polemonium)	Intermittent streams, frequently found on upland sites	LOW. Habitat too dry or shady for purple loosestrife.
Rorippa columbiae (persistentsepal yellowcress)	Moist sandy soil in the periodically inundated zone of the Columbia River, usually in areas subjected to wave action	LOW. Unlikely purple loosestrife habitat.

Some habitat information and vulnerability assessments provided by John Gamon (pers. comm., 1992) tailed deer (*Odocoileus virginianus leucurus*), use wetlands in addition to other habitats (WDOE 1991a, b).

Found in western and south-central Washington, the Western pond turtle inhabits marshes, sloughs, moderately deep ponds, and slow-moving portions of creeks and rivers (Nussbaum *et al.* 1983). This omnivorous species feeds on water lily pods, fish, worms, and other invertebrates. An important habitat requirement is available basking sites, such as partially submerged logs, vegetation mats, rocks, and mud banks. Western pond turtle populations in wetlands invaded by purple loosestrife could be negatively affected by the loss of native food plants and suitable basking sites. Nesting sites appear to be less vulnerable because nests are usually sunny, dry fields or banks hundreds of meters from the water, although sometimes the turtles will nest in sandy banks near the water (Nussbaum *et al.* 1983).

Peregrine falcons utilize riparian and wetland areas as forage areas. These habitats yield higher densities of prey such as ducks and shorebirds. If these forage areas are infested by purple loosestrife, prey species populations may decline.

Sandhill cranes nest in persistent emergent wetlands. They build platform nests utilizing native vegetation such as cattail and bulrush (*Scirpus* spp.). If these native plant species were displaced by purple loosestrife, populations could be negatively affected because purple loosestrife is not suitable nesting material (Tressler 1992). Sandhill cranes forage in agricultural fields and wetlands and feed on plants, frogs, and invertebrates. The impact of purple loosestrife infestations on small vertebrate and invertebrate populations in wetlands is not known. However, invertebrate populations in wetlands may be adversely affected in a monospecific stand of purple loosestrife. The decrease in vegetative diversity and loss of open water would cause a decrease of available microhabitats.

Columbian white-tailed deer use riparian areas and forested scrub-shrub wetlands for reproduction in spring and early summer (Tressler 1992). These are vital habitats because the deer can find food and cover within a relatively small area—an important criterion when movements are restricted during fawning. It is possible that purple loosestrife could displace native plant species in these habitats and lower forage values.

Fish: Only one candidate fish species, the Olympic mudminnow (*Novumbra hubbsi*), is likely to be dependent upon wetland habitat (WDOE 1991a, b). This fish occurs in coastal lowlands of the western Olympic peninsula. This species is generally found in quiet waters with mud substrate and dense aquatic or riparian vegetation. Pre-existing dense vegetation may make these areas less susceptible to purple loosestrife invasion. Impact of purple loosestrife infestation on this fish species is unknown.

4.2.1.4 Areas Specifically Designated for Preservation

Impacts of purple loosestrife infestations on areas that were specifically established to preserve and protect the natural environments and flora and fauna of the state, such as wildlife refuges, natural areas and reserves, are of particular concern. Continued spread of purple loosestrife is likely to occur under this no action alternative. Negative impacts to some components of natural ecosystems will likely occur with continued spread. Such impacts may not be compatible with the purposes for which areas such as refuges or natural reserves were established.

4.2.1.5 Recreation and Aesthetics

If the spread and colonization of purple loosestrife is not controlled, several recreational activities will be negatively affected. Hunting and fishing are activities most directly affected. Reduction of bird populations and duck productivity are concomitant with purple loosestrife infestations. Additionally, purple loosestrife does not provide blind (cover) opportunity for hunters. Decline in bird species diversity will affect bird-watching activities commonly associated with wetland habitats. Dense stands of purple loosestrife make shore-fishing difficult because the vegetation creates an impenetrable wall to access (Brookreson 1991). Boating will also be adversely affected by dense stands of purple loosestrife because water level may decrease and maneuverability will become more difficult.

Purple loosestrife is an aesthetically pleasing plant. Showy mid- and late-summer flowering displays have attracted attention from photographers, hikers, and so forth. The no action alternative would problably result in increased areal extents of this species.

4.2.2 Agricultural/Aquacultural Environment

4.2.2.1 Sediment Composition and Dynamics

Increased sediment deposition in irrigation ditches may result from purple loosestrife infestations.

4.2.2.2 Water Quality and Movement

There are extensive purple loosestrife infestations in irrigation systems in eastern Washington. Dense stands of this species interfere with ditchbank integrity and channel capacity (Sorby 1991). Under the no action alternative, irrigation systems will continue to be infested. They are particularly susceptible to purple loosestrife infestations because they are regularly maintained (disturbed) and are part of an interconnected waterway that serves as an efficient dispersal route for floating seeds and propagules. It is also probable that flow rates within these channels and irrigation water available to agricultural producers will continue to decrease.

4.2.2.3 Affected Species and Practices

Purple loosestrife will probably not pose much of a threat to cultivated upland crops. The rootstocks lie with the upper 30 cm of the soil profile and are thereby susceptible to any form of soil and weed management that includes tillage and herbicide application (Thompson *et al.* 1987). However, infestations of purple loosestrife may reduce the quality of wetland pasture. Native forage species such as grasses and sedges are more palatable to livestock than mature purple loosestrife plants. The young foliage of purple loosestrife is palatable to livestock but mature plants are less frequently grazed, giving purple loosestrife a growth advantage over the more heavily grazed native forages (Thompson *et al.* 1987). Additionally, pastures with heavy purple loosestrife infestations are difficult to mow and manage (Malecki *et al.* 1991). Western Washington has extensive wetland pasture suitable for occupation by purple loosestrife. Conservative economic loss figures for agriculture due to *L. salicaria* in North America have been estimated to exceed 2.6 million dollars annually (Thompson, pers. comm. as cited in Malecki *et al.* 1991).

Due to water-dependent cultivation practices, cranberry bogs and wild rice beds in the Pacific Northwest may be highly susceptible to purple loosestrife invasion (Parker and Burrill 1992). At present, there are no reports of purple loosestrife in cranberry bogs or wild rice beds in Washington (Hovanic 1992, pers. comm.).

Cutting of wild hay is an agricultural practice in the Pacific Northwest that may be affected by the spread of purple loosestrife. According to Thompson *et al.* (1987), wild hay meadows in the Pacific Northwest have been invaded by purple loosestrife, although specific locations are not discussed. Wild hay fields that are mowed annually are apparently more resistant to invasion by purple loosestrife if the native species are not selectively grazed. More information is needed on the extent of cultivated wild hay pastures in Pacific Northwest before the impacts of purple loosestrife infestations on this practice can be fully evaluated.

4.2.2.4 Economic Impacts

Purple loosestrife infestations may have significant economic impacts in areas with interconnected irrigation and drainage systems. In eastern Washington, this species interferes with ditchbank integrity and channel capacity of the irrigation canals. Infestations create additional operating and maintenance expenses for irrigation districts (Sorby 1991). Channel capacity is reduced because vegetation slows down the velocity of water through the canals, allowing sediment to accumulate around the base of the purple loosestrife plants. As sediment accumulation progresses, berms form in the canals. The subsequent decrease in carrying capacity and flow velocity would restrict the delivery of irrigation water and elevate the canal water level and adjacent water table. A back up of the delivery system would saturate adjacent irrigated areas, potentially damaging crops and making irrigation of circles difficult (McEachen 1992, pers. comm.). Further spread of purple loosestrife within irrigation delivery systems would also increase the amount of seed present in the irrigation water. Perennial, moist crops—such as mint and alfalfa—could become infested with purple loosestrife if irrigated with seed-laden water (McEachen 1992, pers. comm.). An increase in purple loosestrife seed in irrigation water could also affect seed crop growers. Purple loosestrife seed is a noxious contaminant that would decrease the quality and value of the seed crop.

More information is needed to estimate the long- and short-term impacts of purple loosestrife infestation on native wetland pastures and wild hay meadows. Studies are needed to evaluate the susceptibility and current extent of purple loosestrife infestation in cranberry bogs and wild rice beds.

4.2.2.5 Sensitive Agricultural Areas

Cranberry bogs and wild rice beds in the Pacific Northwest may be susceptible to purple loosestrife invasion (Parker and Burrill 1992).

4.2.3 Built Environment

4.2.3.1 Sediment Composition and Dynamics

The no action alternative may result in an increase in sediment deposition in waterways and wetlands associated with private residences, commercial properties, parks, etc., because of purple loosestrife infestations.

4.2.3.2 Water Quality and Movement

The quality of public water supplies may be adversely affected if detention basins become infested with purple loosestrife. Colonization of purple loosestrife could affect the ability of the detention basin to store water. If the basin functions as a biofiltration area for stormwater and roadside runoff, the decreased storage capacity and decreased efficacy for biofiltration could adversely affect water quality. Purple loosestrife infestations may also decrease the water storage capacity of wetland areas by encouraging the accretion of sediments. Impacts associated with the decreased storage capacity include decreased groundwater quality and quantity (see Section 4.2.1.2).

Purple loosestrife infestations also affect ditch systems used to convey stormwater and roadside runoff away from urban areas. Adverse impacts of infestations on ditch systems include: (1) increased rate of sediment accretion and the associated frequency and cost of cleaning; (2) reduced storage capacity; (3) increased vegetation control requirements; (4) damaged roadways resulting from impedence of drainage; and (5) dispersal of the plant within a drainage system.

4.2.3.3 Recreation, Aesthetic, and Cultural Resources

Expanding populations of purple loosestrife in wetlands, streams, and rivers may be perceived negatively and positively. In addition, recreational opportunities on private or park lands may be altered. Opportunities for hunting, fishing, boating, bird-watching, etc. may be adversely affected. Potentially, sediment deposition in culturally sensitive areas may serve to bury or protect these sites.

4.2.3.4 Economic Impacts

Property damage from flooding may occur in some areas as a result of decreased storage and carrying capacity of detention basins, wetlands, and ditch systems. Additional public costs could accrue due to the increased maintenance frequency and costs associated with infestations in stormwater facilities. Costs may be incurred by private property owners if they wish to control purple loosestrife infestations in their wet pastures (to maintain forage values) or wild hay fields.

4.3 ECONOMIC USES OF PURPLE LOOSESTRIFE

The beneficial uses of purple loosestrife are largely unexplored. The useful functions of purple loosestrife appear to be horticultural, apicultural, and herbal/medicinal. The economic uses of *L. salicaria* have contributed to the spread of this species.

5.0 MISCELLANEOUS SPECIES

This section summarizes the efficacy and impacts of the no action alternative for managing infestations of garden loosestrife (*Lysimachia vulgaris*), dotted loosestrife (*L. punctata*), giant hogweed (*Heracleum mantegazzianum*), and indigobush (*Amorpha fruticosa*) in Washington. For each of these species, information is scant regarding the environmental impacts of an infestation and the potential for the development of natural stabilization and/or dieback. For some of these species, such as *L. punctata*, the nature of the threat of infestation in Washington is unknown. The discussion presented below is based on the few pertinent references in the literature and subjective appraisals of some possible impacts.

5.1 EFFICACY OF THE NO ACTION ALTERNATIVE AS A MANAGEMENT TOOL

These species have been observed to be invading wetland and/or riparian plant communities in Washington and/or other parts of the world with temperate climates. Each of these species has been observed to colonize these habitats aggressively via seed and/or asexual means. Abundant suitable habitat exists for each of these species in Washington, which suggests these species could become much more widely distributed in the future.

The no action alternative could effectively control all or some of these species in Washington if natural processes stabilize or decrease their distributions at the local and regional levels. The extent to which existing infestations of these species have experienced stabilization or dieback in Washington or other areas of infestation has not been investigated. Pysek (1991) mentions the abundance of giant hogweed has decreased in the region of its original introduction in Czechoslovakia, but provides no further description or explanation for this situation. Other instances of dieback for giant hogweed or the other species are not mentioned in existing literature.

The lack of information on stabilizing factors and potentials for natural dieback precludes a full evaluation of the efficacy of the no action alternative as a management tool for these species.

5.2 IMPACTS

5.2.1 Natural Environment

5.2.1.1 Sediment Composition and Dynamics

Potential impacts of infestations on the sediment dynamics of wetlands or shorelines are mostly unknown. Impacts potentially posed by garden and dotted loosestrife might be considered similar to those posed by purple loosestrife because they tend to infest similar wetland habitat classes as the latter and are similar vegetatively (that is, both have deciduous, numerous upright stems and an aggressive, rhizomatous habit). However, little is known of the biology of infestation for these species in Washington.

Giant hogweed is able to successfully shade out competing vegetation, but dies back each year to expose bare soil to the elements. Infestations have thus been observed to result in increased erosional rates, particularly in riverine/riverbank environments (Wright 1985).

5.2.1.2 Water Quality and Movement

Potential impacts of infestations on water quality and movement have not been investigated. Increased erosion caused by giant hogweed could adversely impact water quality of wetlands, streams, and rivers during storm or high-flow events. Infestations of indigobush along shorelines of streams and rivers may alter the speed and currents of water in those bodies, thus suggesting possible flooding effects. No information is available on erosional rates within stands of indigobush. Additionally, there is some indication that indigobush transpires large amounts of soil moisture during the growing season (Bragg 1991, pers. comm.), which could adversely impact wetland and stream hydrologic regimes and effect subsequent changes in associated biota.

5.2.1.3 Biota

Specific potential impacts of infestations of these species to the biotic communities in Washington or elsewhere have not been investigated. Giant hogweed is well-known to shade out competing native vegetation (Dawe and White 1979; Wright 1984). Both are able to displace large areas of native plant habitat in uplands, wetlands, and riparian zones. Indigobush also colonizes shorelines and riparian habitats in Washington, to the near-complete exclusion of other native vegetation. As mentioned in Section 5.2.1.2, indigobush is also perceived to transpire large amounts of water, which could adversely impact associated biota.

Plant competition posed by infesting species could lead to the localized extirpation of unusual or rare plant species because many rare plants in Washington inhabit wetland and riparian communities (WDOE 1991a, b). The elimination or reduction in plant species diversity in habitats could also result in the elimination of natural foods and cover essential to many wetland wildlife inhabitants, possibly including rare species.

5.2.1.4 Areas Specifically Designated for Preservation

Impacts of noxious weed infestations are of particular concern for areas that are specifically set aside to preserve and protect natural habitats and flora and fauna, such as wildlife refuges, natural areas, and reserves. Impacts to some components of natural ecosystems and habitats will occur if infestations continue to spread and develop. Such impacts may not be compatible with the intended missions or roles for which these areas were established, and may adversely impact biota of specific importance in those areas.

5.2.2 Agricultural Environment

Based on available information, the species most likely to impact agricultural areas is giant hogweed, which has been observed to invade pastures, gardens, and orchards (Wright 1984). Giant hogweed is known to be consumed by cows and hogs without ill effect. However, goats have been observed to be susceptible to the photodermititis caused by giant hogweed (Andrew *et al.* 1985). It is unknown if horses consume giant hogweed.

Since giant hogweed produces massive, deciduous leaves that effectively shade out all competing vegetation, infestations on pasture lands would likely lead to increased erosion on bare ground exposed after the growing season.

5.2.3 Built Environment

5.2.3.1 Sediment Composition/Dynamics and Water Quality

Impacts of infestations of these species on the built environment are mostly unknown. Infestations of giant hogweed may result in increased rates of erosion on some residential and commercial properties due to their massive but deciduous leaves. Such increased erosion would adversely impact water quality, particularly during storm or high-flow events.

5.2.3.2 Recreation, Aesthetic, and Cultural Resources

Conversion of emergent wetlands and shorelines to habitats dominated by one of these species may be perceived aesthetically as either positive or negative. Adverse effects of infestations for any of the species might include loss of beach or shoreline habitat, reduced access to water, beaches or streams, and loss of navigation routes.

Recreation opportunities on private, commercial, or public lands may be altered, particularly by infestations of giant hogweed and indigobush. Giant hogweed has been shown to cause phytophotodermatitis in susceptible humans (Drever and Hunter 1970; Hyppio and Cope 1982). The clear, watery sap found in roots and shoots contains the furanocoumarins that sensitize skin to ultraviolet radiation. This causes disintegration of intra-epidermal cells, resulting in severe blistering and painful dermatitis. After the blisters subside, skin may remain abnormally pigmented for several months. Giant hogweed is considered a significant threat to human health (Hyypio and Cope 1982). Indigobush forms dense impenetrable dams along rivers and streams and presents a nuisance to anglers and boaters. In contrast, restricted access to some rivers, streams, and wetlands might be viewed positively by some individuals .

Areas of significant archaeological importance may also be adversely affected by infestations of these species.

5.3 ECONOMIC USES

Flavenol glucosides extracted from *Lysimachia vulgaris* var. *davurica* are used in Asian folk medicines to treat high blood pressure. In addition, *L. vulgaris* produces triterpene saponins that have anti-microbial activity. These chemicals suggest that there could be a beneficial economic use of this species.

Giant hogweed produces furanocoumarins, a family of glycosides that has high phytotoxicity towards insects and fungi, suggesting plants may provide potentially useful natural insecticides and fungicides.

Indigobush produces insecticidal, acaricidal (kills spiders and ticks), antimicrobial, and anti-viral flavenoids such as the rotenoid amorphigenin, amorphin, dehydrotoxicarol, and 11-hydroxytephrosin. Some of these chemicals may prove to be effective natural pesticides.

6.0 INFORMATION AND RESEARCH NEEDS

Additional information on the efficacy and impacts of the no action alternative may be required for development of an effective management plan for noxious species. Suggestions for further information and research are described below.

6.1 *Spartina*

It is unlikely that the no action alternative will result in control of *Spartina* infestations in Washington within the next several decades. Impacts to the physical, chemical, and biotic characteristics of environments could result from *Spartina* infestations. While ability of

Spartina to alter the physical characteristics of its surroundings through trapping of sediment is well-documented, few studies have been undertaken evaluating the impacts of *Spartina* to fish and wildlife. The following studies may provide more information on the efficacy and impacts of the no action alternative:

- **Monitoring the occurrence of dieback and determining the environmental conditions under which it occurs.** This information may help to better evaluate the potential for extensive natural dieback.

- **Long-term monitoring of dieback areas** to determine whether the areas are re-colonized by *Spartina* or native species, or whether a return to pre-invasion conditions occurs.

- **Investigations to determine whether *Spartina* species are displacing native salt marsh and eelgrass communities.**

- **Local studies to determine the habitat value of *Spartina* and the impacts, positive or negative, to resident and migratory fish and wildlife occurring from infestations.**

  Little research has been done evaluating the impacts of *Spartina* infestations to local fish and wildlife. The Washington State Department of Fisheries Willapa Shellfish Research Lab has proposed a study to directly evaluate the impacts of *Spartina* colonization to clams and other shellfish (Dumbauld 1992, pers. comm.).

  A study funded by Washington Sea Grant (anticipated to begin on January 1, 1993) will compare the characteristics of tidal flats and differing size stands of *S. alterniflora* in Willapa Bay, including epibenthic invertebrate and algal colonization, fish use, prey utilization by fish, rates of sediment accretion, and detrital export/retention. Charles Simenstad of the University of Washington will act as principal investigator (Fresh 1992, pers. comm.).

  Steve Herman of Evergreen State College conducted shorebird habitat availability and utilization studies in Willapa Bay and Grays Harbor in the early 1980s. He indicated it may be possible to gather comparative data to determine changes in habitat suitability for shorebirds resulting from *Spartina* infestations (Herman 1992, pers. comm.).

- **Long-term productivity studies** to determine the potential impacts of *Spartina* to food webs in estuaries of the Pacific Northwest.

- **Evaluation of potential agricultural, aquacultural, or other uses of *Spartina*.** Development of new economic enterprises could offset some of the economic impacts that may result from *Spartina* infestations.

6.2 PURPLE LOOSESTRIFE

It is unlikely that the no action management scenario would result in effective management of purple loosestrife infestations in Washington within the next several decades. Impacts to the biotic and physical characteristics of environments could result from purple loosestrife infestations. The following studies may provide more information on the efficacy and impacts of the no action alternative. Many of the conclusions drawn in this report are based on research done in Europe and northeastern and midwestern North America. Thus, local studies would be of particular relevance.

- **Monitoring of the various stages of purple loosestrife infestations and assessment of subsequent impacts on the structure, functions, and productivity of infested wetlands.**

- **Monitoring of known populations of threatened and endangered plant and animal species in habitats susceptible to purple loosestrife invasion.** This will provide information on the vulnerability of declining populations, direct and indirect impacts on these populations, and possible management strategies for protecting these species.

- **Investigations on the impacts of purple loosestrife infestations on fish, wildlife, and invertebrate populations.** Local studies are necessary to determine the habitat value of purple loosestrife to resident and migratory fish, wildlife, and invertebrate populations. In addition, studies are needed to determine the importance of purple loosestrife as a food source for birds and mammals.

- **Long-term productivity studies** to determine the potential impacts of purple loosestrife to food webs in the Pacific Northwest.

- **Comparative genetic studies of European and North American purple loosestrife populations** to address the differential stability of North American populations compared to European populations. For example, do local populations remain as monospecific stands for decades because there are more vigorous and adaptive North American biotypes?

- **Evaluation of the susceptibility of Washington agricultural lands to purple loosestrife infestation.** Regional studies are of particular importance to estimate the potential impacts of *L. salicaria* on various agricultural practices in Washington (wetland pastures, wild hay meadows, cranberry bogs, and wild rice beds). Studies should focus on assessment of the current extent of infestations, susceptibility of habitats, existing damages, and current and projected economic impacts.

- **Evaluation of impacts of purple loosestrife on irrigation practices,** particularly emphasizing economic impacts and the role of interconnected irrigations systems as dispersal corridors for loosestrife.

- **Documentation of changes in sediment deposition as a function of purple loosestrife infestations.** This information is needed to assess impacts to streams, rivers, and wetlands, as well as constructed features such as ditches and irrigation canals.

- **Local quantification of the distribution of purple loosestrife relative to salinity gradients.** One study (Hutchinson undated) reports purple loosestrife in habitats on the Fraser River delta foreshore where it experienced salinity values up to 8 parts per thousand for short periods, early in the growing season. Salinity tolerance of purple loosestrife has important implications on the susceptibility of certain habitats.

7.0 LITERATURE CITED

SPARTINA SPECIES

ACOE (United States Army Corps of Engineers). 1992. Draft Report to Washington State Department of Ecology. Characterization of *Spartina* spp. communities in western Washington. Seattle District Office, Environmental Resources Section.

Allard, D.J. 1991. The occurrence of fish in cordgrass and mudflat habitats on Willapa national wildlife refuge. U.S. Fish and Wildlife Service, Lower Columbia River Fishery Resource Office, Vancouver, Washington.

Atkinson, J.B. 1992. A preliminary investigation of benthic invertebrates associated with intertidal mudflats and intertidal mudflats infested with *Spartina* at one location in Willapa Bay, Washington. U.S. Fish and Wildlife Service Willapa National Wildlife Refuge.

Bertness, M.D. 1984. Ribbed mussels and *Spartina alterniflora* production in a New England salt marsh. Ecology 65(6):1794-1807.

Bertness, M.D. 1985. Fiddler crab regulation of *Spartina alterniflora* production on a New England salt marsh. Ecology 66(3):1042-1055.

Bertness, M.D. 1991. Zonation of *Spartina patens* and *Spartina alterniflora* in a New England salt marsh. Ecology 72(1):138-148.

Bertness, M.D. and A.M. Ellison. 1987. Determinants of pattern in a New England salt marsh plant community. Ecological Monographs 57(2):129-147.

Bold, H.C., C.J. Alexopoulos, and T. Delevoryas. 1980. Morphology of Plants and Fungi. Harper and Row, Publishers, New York. 819 pp.

Calloway, J.C. and M.N. Josselyn. 1992. The introduction and spread of smooth cordgrass (*Spartina alterniflora*) in south San Francisco Bay. Estuaries 15(2):218-226.

Charman, K. 1990. The current status of *Spartina anglica* in Britain. Pp. 11-14 in A.J. Gray and P.E.M. Benham (eds.) *Spartina anglica* - a research review. Institute of Terrestrial Ecology, Natural Environment Research Council. 80 pp.

Cheney, D.P., G.L. Bonacker, and R.E. Noble. 1986. Aquaculture in Willapa Bay: A plan for development. Report to Willapa Development Corporation and Port of Willapa Harbor. 88 pp.

Corkhill, P. 1984. *Spartina* at Lindisfame NNR and details of recent attempts to control its spread. Pp. 60-63 in P. Doody (ed.) *Spartina anglica* in Great Britain. Nature Conservancy Council. 71 pp.

Chung, C.H.C. 1985. The effects of introduced *Spartina* grass on coastal morphology in China. Annals of Geomorphology Suppl.-Bd. 57: 169-174.

Daiber, F.C. 1974. Salt marsh plants and future coastal salt marshes in relation to animals. Pp. 475-508 in R.J. Reimold and W.H. Queen (eds.) Ecology of halophytes. Academic Press, New York and London.

DeLaune, R.D., C.J. Smith, and W.H. Patrick, Jr. 1983. Relationship of marsh elevation, redox potential, and sulfide to *Spartina alterniflora* productivity. Soil Science Society of America Journal 47:930-935.

Doody, P. 1984. *Spartina* and its effects on nature conservation. P. 36 in P. Doody (ed.) *Spartina anglica* in Great Britain. Nature Conservancy Council. 71 pp.

Doody, P. 1990. *Spartina* - friend or foe? A conservation viewpoint. Pp. 77-79 in A.J. Gray and P.E.M. Benham (eds.) *Spartina anglica* - a research review. Institute of Terrestrial Ecology, Natural Environment Research Council. 80 pp.

Franko, G.D., A.L. MacKenzie and P.A. Gillespie. 1985. Report on the Environmental Implications of the Proposed Herbicide Spraying of *Spartina* in Waimea Inlet, Nelson Province.

Frenkel, R.E. and T.R. Boss. 1988. Introduction, establishment and spread of *Spartina patens* on Cox Island, Siuslaw Estuary, Oregon. Wetlands 8:33-45.

Frenkel, R.E. and L.M. Kunze. 1984. Introduction and spread of three *Spartina* species in the Pacific Northwest. Paper presented at the Annual Meeting of the Association of American Geographers, Washington, D.C. April 22-25. 1984.

Goodman, P.J., E.M. Braybrooks, and J.M. Lambert 1959. Investigations into 'die-back' in *Spartina townsendii* agg. I. The present status of *Spartina townsendii* in Britain. Journal of Ecology 48:711-724.

Goodman, P.J. and W.T. Williams. 1961. Investigations into "die-back" in *Spartina townsendii*. 3. Physiological correlates of die-back. Journal of Ecology 49:391-398.

Goss-Custard, J.C. and M.E. Moser. 1988. Rates of change in the numbers of dunlin, *Calidris alpina*, wintering in British estuaries in relation to the spread of *Spartina anglica*. Journal of Applied Ecology 25:95-109.

Gray, A.J., P.E.M. Benham, and A.F. Raybould. 1990. *Spartina anglica*-the evolutionary and ecological background. Pp. 5-10 in A.J. Gray and P.E.M. Benham (eds.) *Spartina anglica* - a research review. Institute of Terrestrial Ecology, Natural Environment Research Council. 80 pp.

Harrison, P.G. 1979. Reproductive strategies in intertidal populations of two co-occurring seagrasses (*Zostera* spp.). Canadian Journal of Botany 57:2635-2638.

Hettler, W.F., Jr. 1989. Nekton use of regularly-flooded saltmarsh cordgrass habitat in North Carolina, USA. Marine Ecology Progress Series 56:111-118.

Hill, M.I. 1984. Population studies on the Dee Estuary. Pp. 53-58 in P. Doody (ed.) *Spartina anglica* in Great Britain. Nature Conservancy Council. 71 pp.

Hubbard, J.C. 1965. *Spartina* marshes in Southern England. VI. Pattern of invasion in Poole Harbour. Journal of Ecology 53:799-813.

King, G.M., M.J. Klug, R.G. Wiegert and A.G. Chalmers. 1982. Relation of soil water movement and sulfide concentration to *Spartina alterniflora* production in a Georgia salt marsh. Science 218:61-63.

Kraus, M.L. 1989. A review of the role of saltmarsh cordgrass, *Spartina alterniflora*, in estuarine heavy metal export. Pp. 222-225 in J.A. Kusler and S. Daly (eds.) Proceedings of an International Symposium on Wetlands and River Corridor Management. Association of Wetland Managers.

Landin, M.C. 1990. Growth habits and other considerations of smooth cordgrass, *Spartina alterniflora* Loisel. Pp. 15-20 in T.F. Mumford, P. Peyton, J.R. Sayce and S. Harbell (eds.) *Spartina* Workshop Record. Washington Sea Grant Program, University of Washington, Seattle. 73 pp.

LaSalle, M.W., M.C. Landin, and J.G. Sims. 1991. Evaluation of the flora and fauna of a *Spartina alterniflora* marsh established on dredged material in Winyah Bay, South Carolina. Wetlands 11(2)191-208.

Lee, W.G. and T.R. Partridge. 1983. Rates of spread of *Spartina anglica* and sediment accretion in the New River Estuary, Invercargill, New Zealand. New Zealand Journal of Botany 21:231-236.

Linthurst, R.A. and E.D. Seneca. 1980. Dieback of salt-water cordgrass)*Spartina alterniflora* Loisel.) in the lower Cape Fear estuary of North Carolina: an experimental approach to re-establishment. Environmental Conservation 7(1):59-66.

Marinucci, A.C. 1982. Trophic importance of *Spartina alterniflora* production and decomposition to the marsh-estuarine ecosystem. Biological Conservation 22:35-58.

Mayer, J.R. 1989. Potential impact of agricultural pesticide runoff on *Zostera marina* and *Zostera japonica* (eelgrass communities) in Padilla Bay, Washington. Final report to Water Quality Financial Assistance Program, Non point section, Washington State Department of Ecology. 19pp.

Mendelssohn, I.A. and K.L. McKee. 1988. *Spartina alterniflora* die-back in Louisiana: time-course investigation of soil waterlogging effects. Journal of Ecology 76:509-521.

Millard, A.V. and P.R. evans. 1984. Colonization of mudflats by *Spartina anglica*: some effects on invertebrates and shorebirds at Lindisfarne. Pp. 41-48 in P. Doody (ed.) *Spartina anglica* in Great Britain. Nature Conservancy Council. 71 pp.

Phillips, R.C. 1974. Temperate grass flats. Pp. 244-299 in H.T. Odum, B.J. Copeland, and E.A. McMahan (eds.) Coastal ecological systems of the United States. The Conservation Foundation.

Phillips, R.C. 1984. The ecology of eelgrass meadows in the Pacific Northwest: A community profile. U.S. Fish and Wildlife Service. FWS/OBS-84/24. 85 pp.

Pomeroy, W.M. 1977. Benthic algal ecology and primary pathways of energy flow on the Squamish River delta, British Columbia. Ph.D. dissertation, University of British Columbia, Vancouver.

Ranwell, D.S. 1964. *Spartina* marshes in Southern England, 2. rates and seasonal pattern of sediment accretion. Journal of Ecology 52:79-94.

Ranwell, D.S. 1967. World resources of *Spartina townsendii (sensu lato)* and economic use of *Spartina* marshland. Journal of Applied Ecology 4(1):239-256.

Ranwell, D.S. and B.M. Downing. 1960. The use of dalaphon and substituted area herbicides for control of seed bearing *Spartina* (cordgrass) in intertidal zones of estuarine marsh. Weeds 8:78-88.

Raybould, A.F., A.J. Gray, M.J. Lawrence, and D.F. Marshall. 1991. The evolution of *Spartina anglica* C.E. Hubbard (Graminae): genetic variation and status of the parental species in Britain. Biological Journal of the Linnean Society 43(2):370-380.

Redfield A.C. 1972. Development of a New England salt marsh. Ecological Monographs 42(2):201-237.

Riggs, S.R. 1992. Distribution of *Spartina alterniflora* in Padilla Bay, Washington, in 1991. Washington State Department of Ecology, Padilla Bay National Estuarine Research Reserve Technical Report No. 3, Mount Vernon, Washington. 63 pp.

Rountree, R.A. and K.W. Able. 1992. Fauna of polyhaline subtidal marsh creeks in southern New Jersey: composition, abundance, and biomass. Estuaries 15(2):171-185.

Sayce, K. 1988. Introduced cordgrass, *Spartina alterniflora* Loisel. in salt marshes and tidelands of Willapa Bay, Washington. U.S. Fish and Wildlife Service, Willapa National Wildlife Refuge, Ilwaco, Washington. 70 pp.

Scholten, M. and J. Rozema. 1990. The competitive ability of *Spartina anglica* on Dutch marshes. Pp. 39-47 in A.J. Gray and P.E.M. Benham (eds.) *Spartina anglica* - a research review. Institute of Terrestrial Ecology, Natural Environment Research Council. 80 pp.

Simenstad, C.A. and R.C. Wissmar. 1985. δ ^{13}C evidence of the origins and fates of organic carbon in estuarine and nearshore food webs. Marine Ecology Progress Series. 22:141-152.

Strong, D. 1990. Insect herbivores that feed on *Spartina alterniflora*. Pp. 20-21 in T.F. Mumford, P. Peyton, J.R. Sayce and S. Harbell (eds.) *Spartina* Workshop Record. Washington Sea Grant Program, University of Washington, Seattle. 73 pp.

Thom, R.M. 1987. The biological importance of Pacific Northwest estuaries. Northwest Environment Journal 3:21-42.

Thompson, J.D. 1991. The biology of an invasive plant. What makes *Spartina anglica* so successful? BioScience 41(6):393-401.

Tubbs, C. 1984. *Spartina* on the South Coast: an introduction. Pp. 3-4 in P. Doody (ed.) *Spartina anglica* in Great Britain. Nature Conservancy Council. 71 pp.

University of Hull - Institute of Estuarine and Coastal Studies. 1987. The distribution and status of *Spartina anglica* in the Humber Estuary. A Report Prepared Under Contract to the Nature Conservancy Council - Peterborough.

Vanderzee, M. 1992. The biology of *Spartina*. Pp. 9-13 in G. Fraser-Quick and A. Philips (eds) The sea has weeds too! Proceedings of a Conference on the problem of *Spartina*. Environmental Monitoring Branch, National Parks and Public Land, Department of Conservation and Evnironment. 33pp.

Wilsey, B.J., K.L.McKee, and I.A. Mendelssohn. 1992. Effects of increased elevation and macro- and micronutrient additions on *Spartina alterniflora* transplant success in salt-marsh dieback areas in Louisiana. Environmental Management 16(4):505-511.

WSNWCB (Washington State Noxious Weed Control Board). 1991. Report of the Washington state noxious weed control board with recommendations for the continued best used of state funds. Catherine Hovanic (ed.).

Zedler, J.B. 1982. The ecology of southern California coastal salt marshes: a community profile. U.S. Fish and Wildlife Service, Biological Services Program, Washington D.C. FWS/OBS-81/54 110pp.

PERSONAL COMMUNICATION

Atkinson, Jim. 1992. Biologist. United States National Park Service, formerly with Willapa National Wildlife Refuge, U.S. Fish and Wildlife Service.

Bulthuis, Doug. 1992. Padilla Bay Estuarine Reserve, Mount Vernon, Washington.

Civille, Janie. 1992. *Spartina* Coordinator, Washington State Department of Natural Resources. Olympia, Washington. November 22 Memo to K. Rokstad, Washington State Department of Ecology.

Dumbauld, Brett. 1992. Biologist. Washington State Department of Fisheries Willapa Shellfish Research Lab.

Friebaum, Janice. 1992. Washington State Department of Natural Resources, Olympia, Washington.

Fresh, Kurt. 1992. Biologist. Washington State Department of Fisheries, Olympia, Washington.

Gamon, John. 1992. Botanist. Washington Natural Heritage Program, Olympia, Washington.

Herman, Steve. 1992. Professor, Evergreen State College, Olympia, Washington.

Hidy, Jim. 1992. Willapa National Wildlife Refuge, U.S. Fish and Wildlife Service.

Iten, Connie. 1992. Biologist, Washington State Department of Wildlife. December 8 Memo to K. Rokstad, Washington State Department of Ecology.

Kilbride, Kevin. 1992. Biologist, U.S. Fish and Wildlife Service, Vancouver Field Station.

Kunze, Linda. 1992. Washington State Department of Natural Resources, Olympia, Washington.

Morris, Kevin. 1992. Biologist. Seattle, Washington.

OYSTER GROWERS

Hayes, Tom. 1992. Coast Seafood Company, South Bend, Washington.

Nesbitt, Dave. 1992. Nesbitt Oyster. Bay Center, Washington.

Sheldon, Dick. 1992. Bay Center, Washington.

Taylor, Harry. 1992. East Point Seafoods, Bay Center, Washington.

Weigardt, Lee. 1992. Bay Center, Washington.

Wilson, Dick. 1992. Bay Center Mariculture, Bay Center, Washington.

Paveglio, Fred. 1992. Wildlife biologist, U.S. Fish and Wildlife Service, Vancouver Field Station. November 23 letter to J. Civille, Washington Department of Natural Resources.

Riggs, Sharon. 1992. Padilla Bay Estuarine Reserve, Mount Vernon, Washington.

Sayce, Kathleen. 1992. Botanist, Willapa Bay, Washington

Strong, Donald. 1992. Professor, Bodega Bay Marine Lab, Bodega Bay, California.

Thom, Ronald. 1992. Marine/Wetlands Resources Group Manager, Battelle Pacific Northwest Laboratories, Sequim, Washington.

Wiegardt, Andrew. 1992. Citizen, Washington.

PURPLE LOOSESTRIFE

Bender, J. 1988. The Nature Conservancy Element Stewardship Abstract for *Lythrum salicaria* - purple loosestrife, May 3.

Benedict, J. and L. Grim. 1989. Purple loosestrife control plan. Voyageurs National Park, Minnesota.

Bodrogkozy, G., and I. Horvath. 1977. Connection between stand pattern and the organic matter production in the marshlands of the inundation area at Kortvelyes. Tiscia 12:65-70.

Bodrogkozy, G., and I. Horvath. 1979. Effect of lasting floods on the species composition and organic-matter production of the marshy meadow-lands in the floodplains of the Tisza. Tisca 14:81-88.

Brookreson, B. 1991. Purple loosestrife control effects: A ten-year perspective. Washington State Department of Agriculture and Washington State Department of Wildlife, Olympia, Washington.

Heidorn, R. and B. Anderson. 1991. Vegetation management guideline: Purple loosestrife (*Lythrum salicaria* L.). Natural Areas Journal 11:172-173.

Henderson, R. 1987. Status and control of purple loosestrife in Wisconsin. Research Management Findings. Number 4.

Hight, S.D. 1989. Available feeding niches in populations of *Lythrum salicaria* (purple loosestrife) in the northeastern United States, pp. 269-278. in: E. Delfosse (ed.) Proceedings of the VII International Symposium on Biological Control of Weeds 6-11 March 1988, Rome, Italy. 1st Sper. Patol. Veg. (MAF).

Hutchinson, I. Undated. Salinity tolerance of plants of estuarine wetlands and associated uplands in: Washington State Shorelands and Coastal Zone Management Program, Wetlands Section, Department of Ecology, Olympia, Washington.

Isabelle, P.S., L. J. Fooks, P.A. Keddy, and S.D. Wilson. 1987. Effects of roadside snowmelt on wetland vegetation: an experimental study. Journal of Environmental Management 25:57-60.

Malecki, R.A., S.D. Hight, J. J. Drea, L.T. Kok, D. Schroeder, B. Blossey, and J.R. Coulson. 1991. Host plant specificity testing of *Hylobius transversovittatus* (Goeze) (Curculionidae), *Galerucella calmariensis* (L.), and *G. pusilla* (Duftschmidt) (Chrysomelidae) for use in the biological control of *Lythrum salicaria* L. (purple loosestrife) in North America. Petition.

Nussbaum, R.A., E.D. Brodie, Jr., and R.M. Storm. 1983. Amphibians and reptiles of the Pacific Northwest. University of Idaho Press, Moscow, Idaho. 332 pp.

Parker, R., and L. C. Burrill. 1992. Purple loosestrife (*Lythrum salicaria* L.). Pacific Northwest Extension publication PNW 380. 3 pp.

Rawinski, T.J. 1982. The ecology and management of purple loosestrife (*Lythrum salicaria* L.) in central New York. M.S. thesis, Cornell University, Ithaca, New York. 88 pp.

Rawinski, T.J. and R.A. Malecki. 1984. Ecological relationships among purple loosestrife, cattail and wildlife at the Montezuma National Wildlife Refuge. New York Fish and Game Journal 31:81-87.

Shamsi, S.R.A. and F.H. Whitehead. 1974. Comparative eco-physiology of *Epilobium hirsutum* L. and *Lythrum salicaria* L. 1. General biology, distribution, and germination. Journal of Ecology 62:279-290.

Sorby, S. 1991. The Purple Plague. Aquatic Plant News No.36, February.

Stuckey, R.L. 1980. Distributional history of *Lythrum salicaria* (purple loosestrife) in North America. Bartonia 47:3-20.

Thompson, D.Q., R.L Stuckey, and E.B. Thompson. 1987. Spread, impact, and control of purple loosestrife (*Lythrum salicaria*) in North America. Fish and Wildlife Leaflet-2. U.S. Department of the Interior, Fish and Wildlife Service, Washington, 55 pp.

Thompson, D.Q. 1989. Control of purple loosestrife. Fish and Wildlife Leaflet 13.4.11:1-6.

Timmerman, K. 199X (date unknown). Purple Loosestrife. Idaho Wildlife 26-? pp.

WDOE (Washington State Department of Ecology). 1991a. Washington state wetlands rating system for western Washington. Publication 91-57. October.

WDOE (Washington State Department of Ecology). 1991b. Washington state wetlands rating system for eastern Washington. Publication 91-58. October.

Wilcox, D.A. 1989. Migration and control of purple loosestrife (*Lythrum salicaria* L.) along highway corridors. Environmental Management 13(3):365-370.

PERSONAL COMMUNICATION

Gamon, John. 1992. Botanist, Washington Natural Heritage Program, Olympia, Washington.

Hovanic, Catherine. 1992. Executive Secretary, Washington State Noxious Weed Control Board. November 10 Memo to K. Rokstad, Washington State Department of Ecology.

McEachen, Hugh. 1992. Columbia Basin Project Agronomist, Southeast and Quincy Irrigation District.

Piper, Gary. 1992. Entomologist, Washington State University.

Tressler, Ron. 1992. Wildlife Biologist, Ebasco Environmental, Bellevue, Washington.

MISCELLANEOUS SPECIES

Andrews, A.H., C.J. Giles, and L.R. Thomsett. 1985. Suspected poisoning of a goat by giant hogweed. Veterinary Record 116:205-207.

Dawe, N.K. and E.R. White. 1979. Giant cow parsnip (*Heracleum mantegazzianum*) on Vancouver Island, British Columbia. Canadian Field Naturalist 93:82-83.

Drever, J.C. and J.A.A. Hunter. 1970. Giant hogweed dermatitis. Scottish Medical Journal 15:315.

Hyypio, Peter and Edward Cope. 1982. Giant hogweed: *Heracleum mantegazzianum*. Cornell University Cooperative Extension Misc. Bull. 123.

Pysek, P. 1991. *Heracleum mantegazzianum* in the Czech Republic: dynamics of spreading from the historical perspective. Folia Geobotanica and Phytotaxonomica 26:439-454.

WDOE (Washington State Department of Ecology). 1991a. Washington state wetlands rating system for Western Washington. Publication 91-57. October.

WDOE (Washington State Department of Ecology. 1991b. Washington state wetlands rating system for Eastern Washington. Publication 91-58. October.

Wright, M. 1984. Giant Hogweed: time for action is now. New Scientist 101:44.

PERSONAL COMMUNICATION

Bragg, David. 1991. Garfield County Extension Chair. December 23 letter to Bill Brookreson, Washington State Department of Agriculture.