Fire effects on selected bryophytes, lichens and herbs in Garry oak and associated ecosystems

prepared by

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Introduction

According to Fuchs (2001),* 694 plant taxa, 7 amphibians, 7 reptiles, 104 birds, and 33 mammals have been identified in Garry oak and associated ecosystems in British Columbia. A significant number of these species are considered to be at risk at national and global scales.

The First Nations peoples long employed prescribed burning to maintain open vegetation conditions and favour the camas plants, *Camassia quamash* and *Camassia leichtlinii*, which were their primary plant foodstuffs. The Garry Oak Ecosystems Recovery Team (GOERT) identified the role of fire as one of three essential ecosystem characteristics associated with the composition, structure, and function of Garry oak ecosystems. The other two are spatial and biotic integrity.

Fuchs (2001) also noted that minimal research had been conducted in B.C.’s Garry oak ecosystems, making ecological theory and information from other localities especially important. A valuable source of information on fire ecology is the Fire Effects Information System (FEIS), an online database maintained by the USDA Forest Service (see http://www.fs.fed.us/database/feis/).

For the Garry oak ecosystem species listed in Fuchs (2001) which are contained in the FEIS database, the fire ecology, fire effects, and fire management content was extracted and the associated references reformatted to author – date format. Content varies species by species, depending on how much is known about each.

Many of the species found in the Garry oak and associated ecosystems have no entries in the FEIS but it is hoped that this extract will be useful for those interested in the species which are included. The FEIS database contains additional information about distribution and occurrence, management considerations, and botanical and ecological characteristics for most species. The FEIS database should be consulted if these items are of interest and, of course, to verify information contained in this extract.

New species and new information are added to the FEIS periodically and so it should be consulted from time to time for such updates.

John Parminter
July 2006

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<td>Poa pratensis</td>
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<td>312</td>
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Bryophyte & Lichen Layer

*Cladina portentosa* and *Cladina rangiferina*
Reindeer lichens

FIRE ECOLOGY OR ADAPTATIONS

Reindeer lichens are not well adapted to fire. They are highly flammable and may take 30 to 100 years or more to recover to pre-fire densities (Scotter 1971).

IMMEDIATE FIRE EFFECT ON PLANT

Reindeer lichens can survive cool fires but are almost always killed by severe fire (Kelsall 1957, Viereck and Schandelmeier 1980, Miller and See 1981). In the black spruce zone, lichens generally burn poorly in early morning or near sunset, even on hot days, whereas at mid-day they flare up with almost incredible heat and flame. Humidity changes in the microclimate at ground level and dehydration of the lichens appear to be the most likely factors involved. Fifty percent or more of the lichens may survive if there is not much organic litter to retain the fire (Kelsall 1957). Following fire in a black spruce community, all the *Cladonia* spp. survived a light burn, whereas none survived a heavy burn (Viereck and Dyrness 1979).

PLANT RESPONSE TO FIRE

Reindeer lichens recover very slowly after fire (Lutz 1953). The length of time required for full recovery varies with species, the extent and intensity of the fires, and site and microclimatic condition, but an average of 40 to 50 years appears to be a conservative estimate (Lutz 1953 and Viereck and Schandelmeier 1980). Based on annual growth rates of 4.1 and 4.9 mm for *C. alpestris* and *C. rangiferina*, it has been estimated that these species would require nearly a century to reach pre-fire abundance (Loughrey and Kelsall 1970). After fire the first reindeer lichen to become established is *Cladonia mitis*. The second reindeer lichen phase is generally dominated by *C. alpestris*, *C. rangiferina*, or *C. arbuscula* (Lutz 1953 and Viereck and Schandelmeier 1980).

DISCUSSION AND QUALIFICATION OF PLANT RESPONSE

One hundred and forty years after a severe wildfire in Sweden, *Cladonia alpestris* still showed no recovery, but did show good recovery only 20 years after a light, controlled burn. Slow recovery rates were reported from the Seward Peninsula of Alaska, while rapid recovery rates have been reported from the open lichen woodlands of Newfoundland, where the climate is warmer than in Alaska (Viereck and Schandelmeier 1980).
In a black spruce and jack pine woodland in northwestern Manitoba and northeastern Saskatchewan, *Cladonia mitis* became established in less than 40 years after fire, while *C. alpestris* and *C. rangiferina* first appeared in stands greater than 40 years of age. A dense growth of reindeer lichens was found in stands that had not burned for at least 150 years. In south-central Alaska it took 30 to 40 years for *C. rangiferina* and *C. arbuscula* to recover after fire (Miller 1976). In a post-fire black spruce – lichen vegetation type of interior Alaska, cover values for *Cladonia* species were 32 percent in 26 – 50-year-old stands, and 41 percent in stands greater than 100 years (Miller 1976). In a lightly-burned 75-year-old stand with an open canopy of spruce and occasional jack pine, *C. alpestris* made up 32.8 percent of the total ground cover and *C. rangiferina* made up 1.7 percent (Kelsall 1957).

**FIRE MANAGEMENT CONSIDERATIONS**

Reindeer lichens are highly flammable. *Cladonia rangiferina* collected from Ely, Minnesota, and oven-dried had a heat value of 4360 cal/g (Hough 1969). They dry rapidly during periods of low atmospheric humidity because of the absence of roots, water storage tissues, and low resistance to water loss. Reindeer lichens resemble dead litter more than live tissue in their susceptibility to fire. Continuous mats of reindeer lichen present an uninterrupted surface along which a fire spreads. Lichen mats also typically accumulate tree and shrub litter which adds to the flammability (Auclair 1983). In black spruce – *Cladonia alpestris* woodland, litter suspended in the lichen mat added 20.5 percent dry weight to the total combustible material present above the soil (Auclair 1983).

While the destruction of reindeer lichens may have an immediate effect on the winter range of caribou, some studies indicate that at least infrequent fire is necessary to maintain optimum lichen cover (Viereck and Schandelmeier 1980). In the northern boreal lichen belt, lichen supplies could be increased by burning *Sphagnum fuscum* peatlands, treeless bogs, or wooded muskegs. The result of severe fire is an almost solid reindeer lichen stand in some 40 to 50 years. Because black spruce and mosses regenerate more slowly than lichen on these sites, good lichen growth persists for at least 100 years (Ahti and Hepburn 1967). Light burning has been suggested as a method to improve reindeer range in Scandinavia (Viereck and Schandelmeier 1980).

**LITERATURE CITED**


Hylocomium splendens
Mountain fern moss, step moss

FIRE ECOLOGY OR ADAPTATIONS

Mountain fern moss is not well adapted to fire. It typically occurs in wet stands of white or black spruce that have a fire regime of 200 to 400 years (Viereck et al. 1992). When they do burn, the moss/lichen layer provides the major carrier fuels. These fuels take only minutes to reach equilibrium moisture content when the relative humidity changes; therefore, they are very flammable (Norum 1982).

IMMEDIATE FIRE EFFECT ON PLANT

Mountain fern moss is generally killed by fire, although small patches may survive low-severity fire (Viereck et al. 1992). Some moss species on burned areas can survive as fragments in the soil (Ahlgren 1974).

PLANT RESPONSE TO FIRE

Mountain fern moss takes many years to recover following fire. Although small patches may survive fire, it is not until a closed or nearly closed canopy is established that mountain fern moss can spread and become the dominant ground cover (Viereck et al. 1992). Ten to thirty years after fire, mountain fern moss will replace the early successional mosses and liverworts. In mesic, high-nutrient habitats, mountain fern moss generally appears 30 to 50 years after fire and quickly becomes the most abundant ground cover (Johnson 1981). However, in Finland, mountain fern moss began appearing 10 years after fire. Recovery was slow and 50 years after fire, this moss still had not reached preburn levels (Ahlgren 1974).

LITERATURE CITED


Pleuroziunm schreberi
Schreber's moss, Red-stemmed feathermoss

FIRE ECOLOGY OR ADAPTATIONS

Schreber’s moss is not well adapted to fire. It typically occurs in wet stands of white or black spruce that have a fire regime of 200 to 400 years (Viereck and Schandelmeier 1980). When they do burn, the moss/lichen layer provides the major source of fuels. These fuels take only minutes to reach equilibrium moisture content when the relative humidity changes; therefore, they are very flammable (Norum 1982).

IMMEDIATE FIRE EFFECT ON PLANT

Schreber’s moss is generally killed by fire because it often lacks connection with the substrate (Uemura et al. 1990, Uggla 1959). Some moss species can survive on burned sites as fragments in the soil (Ahlgren 1974).

PLANT RESPONSE TO FIRE

Schreber’s moss recovery after fire is very slow (Uggla 1959, Viereck and Schandelmeier 1980). It is not until favorable edaphic conditions and a closed or nearly closed canopy is established that Schreber’s moss can spread and form a continuous moss cover. It therefore often takes several decades before Schreber’s moss will recover to pre-burn densities (Uggla 1959).

DISCUSSION AND QUALIFICATION OF PLANT RESPONSE

Twenty-four years after a fire in a northern Swedish forest, Schreber’s moss was still very rare in the severely burned areas (Uggla 1959). The percent cover values of Schreber’s moss in a jack pine (Pinus banksiana) - black spruce forest in northeastern Minnesota at different intervals after fire were as follows (Ahlgren 1974):

<table>
<thead>
<tr>
<th>Years after fire</th>
<th>1-4</th>
<th>5</th>
<th>10</th>
<th>15</th>
<th>20</th>
<th>30</th>
<th>50</th>
<th>80</th>
</tr>
</thead>
<tbody>
<tr>
<td>% cover</td>
<td>0</td>
<td>1</td>
<td>2</td>
<td>3</td>
<td>3</td>
<td>3</td>
<td>9</td>
<td>5</td>
</tr>
</tbody>
</table>

LITERATURE CITED


Specific information on the fire ecology of *T. ruralis* is lacking. However, poikilohydric plants such as *T. ruralis* dry quickly during periods of low relative humidity because of their absence of roots and water storage tissue, and low resistance to water loss. It is therefore assumed that *T. ruralis* is highly flammable under dry conditions.

**IMMEDIATE FIRE EFFECT ON PLANT**

Rangeland fires can severely damage all components of soil crusts, including *T. ruralis* (Johansen and St. Clair 1986).

**PLANT RESPONSE TO FIRE**

Information on the response of *T. ruralis* to fire is sparse. One study in Scotland reported that *T. ruralis* was present in burned heaths within the first few years following fire (Hobbs and Gimingham 1984).

Following a 1975 fire in a shadscale (*Atriplex confertifolia*)-black greasewood (*Sarcobatus vermiculatus*) community in Camp Floyd State Park, Utah, an unnamed *Tortula* species was absent from burned sites for at least 7 years. The species had an average cover in unburned controls of 7.2 and 6.1 percent in 1980 and 1982, respectively (Johansen *et al.* 1984).

**LITERATURE CITED**

Herb Layer

*Achillea millefolium*
Western yarrow

**FIRE ECOLOGY OR ADAPTATIONS**

The life cycle of western yarrow in grasslands is completed by the onset of the summer drought and fire season in July (Antos et al. 1983). Following fire, regeneration is from rapid rhizome spread (Volland and Dell 1981) and wind dispersal of seeds onto burned sites from adjacent unburned areas (Howe 1994).

Western yarrow occurs in plant communities with a variety of fire regimes. The range of fire intervals reported for some species that dominate communities where western yarrow occurs are listed below.

<table>
<thead>
<tr>
<th>Community dominant</th>
<th>Range of fire interval (yr)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Interior ponderosa pine</td>
<td>20 - 42</td>
</tr>
<tr>
<td><em>(Pinus ponderosa var. scopulorum)</em></td>
<td></td>
</tr>
<tr>
<td>Rocky Mt. Douglas-fir</td>
<td>10 - 30</td>
</tr>
<tr>
<td><em>(Pseudotsuga menziesii var. glauca)</em></td>
<td></td>
</tr>
<tr>
<td>Trembling aspen</td>
<td>7 - 10</td>
</tr>
<tr>
<td><em>(Populus tremuloides)</em></td>
<td></td>
</tr>
<tr>
<td>Rough fescue</td>
<td>5 - 10</td>
</tr>
<tr>
<td><em>(Festuca altaica)</em></td>
<td></td>
</tr>
</tbody>
</table>

**POST-FIRE REGENERATION STRATEGY**

Rhizomatous herb, rhizome in soil; initial-offsite colonizer (off-site, initial community).
Immediate Fire Effect on Plant

Western yarrow’s rhizomes and mycorrhizae are usually only slightly damaged by fire (Berch et al. 1988, Higgins and Mack 1987, Smith and Busby 1981), although western yarrow is susceptible to fire-kill and reduction by severe fire (Mitchell 1984).

Western yarrow is not highly flammable. Out of 14 species commonly found in boreal forests, western yarrow has the lowest potential ignitability based on chemical characteristics measured on live stem, live leaf, and dead leaf tissues. These rankings rely primarily on total ash, silica-free ash, and energy content (Hoggenbirk and Sarrazin-Delay 1995). Ignitability is measured as time to ignition.

Plant Response to Fire


Discussion and Qualification of Plant Response

The initial surge of western yarrow is probably caused by extensive rhizome sprouting; mineral soil exposure and the resulting favorable seedbed; less competition from tree, grass and shrub cover; and nutrient release (Eichhorn and Watts 1984, Mueggler 1976).

A burn was conducted each April for at least 24 years on a rough fescue (Festuca scabrella) grassland in a quaking aspen parkland in east-central Alberta. Average frequency and canopy cover for western yarrow were as follows (Anderson and Bailey 1980):

<table>
<thead>
<tr>
<th>% Frequency</th>
<th>% Cover</th>
</tr>
</thead>
<tbody>
<tr>
<td>burned</td>
<td>unburned</td>
</tr>
<tr>
<td>36</td>
<td>23</td>
</tr>
</tbody>
</table>
Density and crown area of western yarrow (per 180,000 in$^2$) following an August wildfire of moderate severity in a northeastern California range dominated by bitterbrush (*Purshia tridentata*) and various perennial bunchgrasses were as follows (Countryman and Cornelius 1957):

<table>
<thead>
<tr>
<th>Number of plants</th>
<th>Crown area (in$^2$)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Unburned plots</td>
<td>99</td>
</tr>
<tr>
<td>post-fire yr 1</td>
<td>3</td>
</tr>
<tr>
<td>post-fire yr 2</td>
<td>9</td>
</tr>
<tr>
<td>post-fire yr 3</td>
<td>88</td>
</tr>
<tr>
<td>post-fire yr 4</td>
<td>269</td>
</tr>
<tr>
<td>post-fire yr 5</td>
<td>48</td>
</tr>
</tbody>
</table>

Productivity values (kg/ha) of western yarrow before and after a late August fire in western Wyoming quaking aspen communities are listed below for plots of different burn intensities (Bartos and Mueggler 1981):

- Before burning: 14 kg/ha
- After a “light” burn: 40 kg/ha
- After a “moderate” burn: 16 kg/ha
- After a “heavy” burn: 14 kg/ha

**FIRE MANAGEMENT CONSIDERATIONS**

Western yarrow’s good sprouting ability, high germination percentages, and competitive seedlings result in a remarkable persistence under fire disturbance. Western yarrow often appears in the first stages of succession (Bourdot *et al.* 1985, Stickney 1989); however, no consistent trends relative to age of burns seem evident for the western yarrow (Anderson *et al.* 1970, Raunkiaer 1934).

Western yarrow has low ignitability, and can be used as a fire barrier, created by replacing highly flammable vegetation with species that are less likely to burn (Howe 1994). Planting less-flammable vegetation in fire-prone areas, or around property and fire-sensitive areas, may help prevent ignition or slow fire spread (Hoggenbirk and Sarrazin-Delay 1995).

**LITERATURE CITED**


*Agrostis exarata*
Spike bentgrass

FIRE ECOLOGY OR ADAPTATIONS

No information was available in the literature concerning spike bentgrass fire ecology or adaptations. However, a similar species, ticklegrass (*Agrostis scabra*), colonizes bare mineral soil on recently burned sites and may store seeds in the soil for short durations, allowing for early establishment of areas burned in the spring.

POST-FIRE REGENERATION STRATEGY

Tussock graminoid; initial-offsite colonizer (off-site, initial community).

IMMEDIATE FIRE EFFECT ON PLANT

Grasses are generally top-killed by fire so spike bentgrass is probably top-killed by fire. Specific fire effects, however, are not described in the literature.

PLANT RESPONSE TO FIRE

No specific information on spike bentgrass response to fire is available in the literature. Ticklegrass, a similar species, increases in abundance in response to fire.
FIRE MANAGEMENT CONSIDERATIONS

Since spike bentgrass is considered a decreaser species when overgrazed (Ratliff 1985), fire plans may have to be coordinated with grazing management to ensure seedling establishment.

LITERATURE CITED


Agrostis gigantea (= A. alba, in part)
Redtop

FIRE ECOLOGY OR ADAPTATIONS

Redtop is fairly resilient to fire because of its rhizomes and buried seed. Most natural fires in redtop stands probably occur in the fall when the grass has dried out.

POST-FIRE REGENERATION STRATEGY

Rhizomatous herb, rhizome in soil; ground residual colonizer (on-site, initial community).

IMMEDIATE FIRE EFFECT ON PLANT

Fire top-kills redtop. Rhizomes probably survive most fires, but they can be severely damaged by the shallow burning of peat (Frolik 1941). Seeds buried in soil probably survive most fires.

PLANT RESPONSE TO FIRE

Fire generally favors redtop. Rhizomes sprout after fire and buried seed may germinate.

Redtop was present in the post-fire vegetation of the Sundance Burn in northern Idaho. On several sites it was present and flowered in post-fire years 1 and 2, but on other sites it did not appear until more than 10 years after the fire (Stickney 1985).

Redtop was not present on the Curtis Prairie, Wisconsin, in 1951, but after 10 years of biennial dormant season burning, it had 8 percent frequency (Anderson 1972).
The Hayden Prairie in northeastern Iowa was subject to early spring fires. Redtop seedstalks, inventoried in August, did not differ substantially between burned and unburned sites. Redtop seedstalk density on sites burned 2 and 3 consecutive years was not significantly different from that on sites burned a single time (Ehrenreich and Aikman 1963). Early spring fires in southeastern Iowa pastureland dominated by exotic cool-season grasses had no significant (p < 0.05) effect on redtop cover (Rosburg and Glenn-Lewin 1992).

In south-central New York, little bluestem (Schizachyrium scoparium) fields and goldenrod-poverty oatgrass (Danthonia spicata) fields burned by spring wildfires were compared to adjacent unburned sites. Redtop increased with burning; on little bluestem fields, redtop frequency averaged 17 percent on the unburned plots and 38 percent on the burned plots. On goldenrod fields, redtop frequency was 25 percent on unburned plots and 39 percent on burned plots (Swan 1970).

However, redtop decreased with 17 years of early April annual and biennial burning of little bluestem fields in Connecticut (Rosburg and Glenn-Lewin 1992). The repeated burning may have stressed redtop, or the species present may have actually been a nonrhizomatous form of creeping bentgrass, which may be more susceptible to fire than redtop.

**FIRE MANAGEMENT CONSIDERATIONS**

Redtop has been seeded onto burns with other grasses to prevent erosion (Evanko 1953, Neary and Currier 1982, Slinkard et al. 1970). In northeastern Washington, redtop excelled on northeast-facing slopes where moisture was high. Redtop was not as vigorous on southwest-facing exposures but was still present 4 years after the seeding (Evanko 1953).

Prescribed burning rejuvenates redtop fields and is recommended to enhance prairie chicken cover in the Midwest. Fields should be burned 3 to 4 years after seeding (either in August or in March prior to nesting season) to remove duff, improve redtop vigor, and control weeds. A second fire may be desirable 3 years after the first fire if the area is not too densely invaded by forbs (Westemeier 1973).

Early spring fire followed by the application of the herbicide atrazine significantly (p < 0.05) reduced redtop in most treatments (Rosburg and Glenn-Lewin 1992).

**LITERATURE CITED**


Agrostis scabra
Tickle grass, Hair bentgrass

FIRE ECOLOGY OR ADAPTATIONS

Wind-dispersed tickle grass seeds readily colonize bare mineral soil on recently burned sites (Carroll and Bliss 1982, Johnson 1975, Smith 1970). Seeds may also be stored for short durations in the soil, allowing for early establishment of areas burned in the spring (Fyles 1989).

POST-FIRE REGENERATION STRATEGY

Tussock graminoid; initial-offsite colonizer (off-site, initial community).

IMMEDIATE FIRE EFFECT ON PLANT

Fire generally top-kills tickle grass. Specific effects on tickle grass mortality, however, are not well documented.

PLANT RESPONSE TO FIRE

In general, tickle grass increases in abundance in response to fire. Seedlings immediately colonize recently burned areas, provided a favorable seedbed has been established (Lutz 1956, Stickney 1985). Annual spring burns in aspen stands in Alberta caused an increase in tickle grass inflorescence production. In unburned areas, there was an average of ten flower head per square metre, but on burned sites 110 flower heads per metre were produced (Anderson 1959). In interior Alaska, seedlings were not found in burned plots where the organic layer had not been completely removed, although a seed source was nearby. Seedlings were, however, abundant on adjacent firelines (Viereck 1982).

LITERATURE CITED


Agrostis stolonifera (= A. alba, in part)
Creeping bentgrass

FIRE ECOLOGY OR ADAPTATIONS

Creeping bentgrass has fair tolerance to fire (Wasser 1982). No information was available in the literature concerning creeping bentgrass fire ecology or adaptations. However, a similar species, ticklegrass (Agrostis scabra), colonizes bare mineral soil on recently burned sites and may store seeds in the soil for short durations, allowing for early establishment of areas burned in the spring. The stolons are probably killed by moderately severe and severe fires.

POST-FIRE REGENERATION STRATEGY

Tussock graminoid; initial-offsite colonizer (off-site, initial community); secondary colonizer – on-site seed.

IMMEDIATE FIRE EFFECT ON PLANT

Creeping bentgrass is probably top-killed by fire, as are most grasses. Specific fire effects on creeping bentgrass are not described in the literature.

PLANT RESPONSE TO FIRE

In 1972, prescription burning at the Buffalo River State Park in northwest New Mexico was initiated as a tallgrass prairie management and restoration technique. The response of creeping bentgrass to burning varied with the site. On a nearly level
mesic site in a badly disturbed prairie, stimulation of flowering occurred at post-fire
year 1. Inhibition of flowering occurred, however, on a wet swale site in an

In 1950, a fire burned 400 000 ha of woodland in Alberta and British Columbia.
Creeping bentgrass established on plots where seeded species did not produce full
stands (Anderson and Elliott 1957).

FIRE MANAGEMENT CONSIDERATIONS

Since creeping bentgrass is considered an increaser species when overgrazed, fire
plans may have to be coordinated with grazing management to ensure seedling
establishment or inhibition.

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_Alliaria petiolata_ (= _Alliaria officinalis_
Garlic mustard

FIRE ECOLOGY OR ADAPTATIONS

Although garlic mustard plants are readily top-killed when exposed to fire, they may
Ecological conditions that permit sprouting are not well understood and it is unclear
to what extent resprouted plants are capable of producing viable seed.
At the population level, garlic mustard may be adapted to perpetuate itself in mixed-severity or low-severity surface fire regimes, although this has not been quantified. Even though individual plants may be killed by fire, post-fire conditions may be favorable for rapid population expansion because of increases in the area of disturbed habitat and, depending on the extant community, temporary reductions in interspecific competition. Additionally, garlic mustard seed banks may facilitate rapid recolonization of disturbed areas (Byers and Quinn 1998). For example, 3 consecutive years of prescribed burning in a central Illinois black oak forest, which were described as “hot and fast” with flame lengths to 1.2 m, failed to eradicate garlic mustard populations. This was attributable, in part, to the protection afforded a small number of plants by refugia such as the lee of a downed log or an area of damp litter (Nuzzo et al. 1996). The ability of individual plants to escape mortality will depend upon fire severity and the heterogeneity of the fire landscape.

Garlic mustard may be found within understory surface, stand-replacement, mixed-severity fire, and nonfire regimes (Brown 2000). Because garlic mustard has become established only relatively recently in most areas in North America, and because natural fire regimes have been substantially altered in many of these areas, predicting the response of garlic mustard to any particular fire regime is speculative. In some areas colonized by garlic mustard, estimated mean fire return intervals may be longer than the time in which garlic mustard has been present.

As natural areas and preserve managers reintroduce fire into locations where natural and anthropogenic fire has been suppressed in recent times, the response of this and many other species may become better understood. Those who intend to reintroduce fire where it has been absent for a substantial period are encouraged to plan and implement research and monitoring programs and share their findings.

Fire regimes of some of the plant communities in which garlic mustard occurs are summarized below.

<table>
<thead>
<tr>
<th>Community or Ecosystem</th>
<th>Dominant Species</th>
<th>Fire Return Interval Range (years)</th>
</tr>
</thead>
<tbody>
<tr>
<td>maple-beech-birch</td>
<td><em>Acer-Fagus-Betula</em></td>
<td>&gt; 1000</td>
</tr>
<tr>
<td>silver maple-American elm</td>
<td><em>A. saccharinum-Ulmus americana</em></td>
<td>&lt; 35 to 200</td>
</tr>
<tr>
<td>sugar maple</td>
<td><em>A. saccharinum</em></td>
<td>&gt; 1000</td>
</tr>
<tr>
<td>sugar maple-basswood</td>
<td><em>A. saccharinum-Tilia americana</em></td>
<td>&gt; 1000 [1]</td>
</tr>
<tr>
<td>bluestem prairie</td>
<td><em>Andropogon gerardii var. gerardii-Schizachyrium scoparium</em></td>
<td>&lt; 10 [2,3]</td>
</tr>
<tr>
<td>sugarberry-America elm-green ash</td>
<td><em>Celtis laevigata-U. americana-Fraxinus pennsylvanica</em></td>
<td>&lt; 35 to 200</td>
</tr>
<tr>
<td>beech-sugar maple</td>
<td><em>Fagus spp.-A. saccharum</em></td>
<td>&gt; 1000</td>
</tr>
<tr>
<td>black ash</td>
<td><em>Fraxinus nigra</em></td>
<td>&lt; 35 to 200 [1]</td>
</tr>
<tr>
<td>tamarack</td>
<td><em>Larix laricina</em></td>
<td>35-200 [3]</td>
</tr>
<tr>
<td>Plant Group</td>
<td>Species</td>
<td>Interval</td>
</tr>
<tr>
<td>-------------------------------------</td>
<td>----------------------------------------------</td>
<td>----------</td>
</tr>
<tr>
<td>yellow-poplar</td>
<td><em>Liriodendron tulipifera</em></td>
<td>&lt; 35</td>
</tr>
<tr>
<td>eastern white pine-northern red oak</td>
<td><em>Pinus strobus-Quercus rubra-A. rubrum</em></td>
<td>35-200</td>
</tr>
<tr>
<td>red oak-red maple</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Virginia pine-oak</td>
<td><em>P. virginiana-Quercus spp.</em></td>
<td>10 to &lt; 35</td>
</tr>
<tr>
<td>sycamore-sweetgum-American elm</td>
<td><em>Platanus occidentalis-Liquidambar styraciflua-U. americana</em></td>
<td>&lt; 35 to 200 [1]</td>
</tr>
<tr>
<td>eastern cottonwood</td>
<td><em>Populus deltoides</em></td>
<td>&lt; 35 to 200 [3]</td>
</tr>
<tr>
<td>aspen-birch</td>
<td><em>P. tremuloides-Betula papyrifera</em></td>
<td>35 to 200 [4,1]</td>
</tr>
<tr>
<td>black cherry-sugar maple</td>
<td><em>Prunus serotina-A. saccharum</em></td>
<td>&gt; 1000</td>
</tr>
<tr>
<td>oak-hickory</td>
<td><em>Quercus-Carya spp.</em></td>
<td>&lt; 35</td>
</tr>
<tr>
<td>northeastern oak-pine</td>
<td><em>Quercus-Pinus spp.</em></td>
<td>10 to &lt; 35</td>
</tr>
<tr>
<td>southeastern oak-pine</td>
<td><em>Quercus-Pinus spp.</em></td>
<td>&lt; 10</td>
</tr>
<tr>
<td>white oak-black oak-northern red oak</td>
<td><em>Q. alba-Q. velutina-Q. rubra</em></td>
<td>&lt; 35</td>
</tr>
<tr>
<td>northern pin oak</td>
<td><em>Q. ellipsoidalis</em></td>
<td>&lt; 35</td>
</tr>
<tr>
<td>bur oak</td>
<td><em>Q. macrocarpa</em></td>
<td>&lt; 10 [1]</td>
</tr>
<tr>
<td>oak savanna</td>
<td><em>Q. macrocarpa/Andropogon gerardii-Schizachyrium scoparium</em></td>
<td>2-14 [3,1]</td>
</tr>
<tr>
<td>chestnut oak</td>
<td><em>Q. prinus</em></td>
<td>3-8</td>
</tr>
<tr>
<td>northern red oak</td>
<td><em>Q. rubra</em></td>
<td>10 to &lt; 35</td>
</tr>
<tr>
<td>post oak-blackjack oak</td>
<td><em>Q. stellata-Q. marilandica</em></td>
<td>&lt; 10</td>
</tr>
<tr>
<td>black oak</td>
<td><em>Q. velutina</em></td>
<td>&lt; 35 [1]</td>
</tr>
<tr>
<td>elm-ash-cottonwood</td>
<td><em>Ulmus-Fraxinus-Populus spp.</em></td>
<td>&lt; 35 to 200 [4,1]</td>
</tr>
</tbody>
</table>


POST-FIRE REGENERATION STRATEGY

Caudex/herbaceous root crown, growing points in soil; ground residual colonizer (on-site, initial community); initial off-site colonizer (off-site, initial community); secondary colonizer (on-site or off-site seed sources).

IMMEDIATE FIRE EFFECT ON PLANT

Garlic mustard is often top-killed when exposed to fire. A prescribed burn in the understory of a northern Illinois hardwood forest removed all aboveground garlic mustard biomass (Hintz 1996). Prescribed burns in a central Illinois black oak forest conducted both in the fall and in mid-spring removed nearly all garlic mustard rosettes (Nuzzo et al. 1996). Although there was no immediate post-fire survey of plants mentioned in the article, Luken and Shea (2000) suggest that garlic mustard “plants are readily killed by mid-intensity dormant season fires.” Emergent seedlings may also be killed by fire (Nuzzo et al. 1996).

DISCUSSION AND QUALIFICATION OF FIRE EFFECT

It has been suggested that dense stands of garlic mustard may be able to resist low-severity fire, such that “abundant green garlic mustard plants”...may “literally extinguish fires” (Nuzzo 1991), but detailed descriptions of the direct effects of fire on garlic mustard plants (or visa versa) are scarce. Such observations may be confounded by the inherently patchy nature of mixed-severity fire regimes in many eastern deciduous forests where garlic mustard may commonly be found.

PLANT RESPONSE TO FIRE

Garlic mustard has at least some ability to sprout from the root crown following damage by fire. By excavating charred rosettes, Nuzzo et al. (1996) found that adult plants resprouted from adventitious buds on the root crown located just below the soil surface following a mid-spring burn. In a northern Illinois oak woodland, garlic mustard reportedly resprouted several weeks following complete top removal by a prescribed fire conducted in late March (Hintz 1996). Repeated fall burning (2-3 annual burns) did not reduce abundance or relative importance of garlic mustard in an eastern mesophytic forest understory in Kentucky (Luken and Shea 2000).

DISCUSSION AND QUALIFICATION OF PLANT RESPONSE

There is some indication that garlic mustard is capable of sprouting following fire, but several questions remain. To what extent is post-fire sprouting in garlic mustard influenced by fire severity? What, if any, physiological conditions promote or constrain post-fire root crown sprouting? To what extent are resprouting plants successful at producing seed?
Nuzzo (1996) reported that a fall burn in a central Illinois black oak forest removed 79% of the litter layer, and very few adult garlic mustard plants were encountered in these plots the following spring. Conversely, many garlic mustard plants resprouted following a mid-spring burn at the same site that resulted in removal of only 32% of the litter layer. Spring burn plots retained a damp 1-2 cm layer of litter which seems to have protected the root crowns of top-killed plants, fostering survival via sprouting of multiple secondary shoots from adventitious buds located just below the soil surface (Nuzzo et al. 1996).

Hintz (1996) conducted a late-March prescribed burn in a mesic upland oak-hickory forest in northern Illinois. Garlic mustard established following the fire, although it is unclear whether these were sprouting burned plants or new spring seedlings. The burn was conducted near the time when seedling emergence might be expected, leaving some question as to which life-cycle stage was observed to be “sprouting.” There is reference to “very little” garlic mustard producing seed that summer, intimating that at least some adult plants were present both prior to and after the fire.

Luken and Shea (2000) conducted a prescribed fire experiment in a northern Kentucky mesic deciduous forest in which they showed that garlic mustard plants could be removed by a fall burn. Yet it was also apparent from this experiment that populations can persist following even repeated burns. Garlic mustard remained the dominant species in the herb layer of both burned and unburned plots through 3 seasons of fall burning, and beyond. The authors proposed 3 possible explanations. First, persistence of individual garlic mustard plants immediately following fire may result from the patchy nature of many understory or mixed-severity burns. Under such conditions some extant plants may escape damage, and because of its ability to self-pollinate (Anderson et al. 1996, Cavers et al. 1979, Cruden and McClain 1996), the survival of a single plant may be sufficient to perpetuate a population. Second, the data of Luken and Shea (2000) showed that burning resulted in higher densities of flowering stems compared with control plots. They speculated this as being due to either resprouting or release from competition. No observations of sprouting were reported. Third, even if all plants are killed, the existing seed bank may remain viable for several years (Baskin and Baskin 1992, Byers and Quinn 1998), requiring subsequent annual burns to completely eradicate the population.

FIRE MANAGEMENT CONSIDERATIONS

Control of invasive garlic mustard populations using prescribed fire, especially as a single management tool, appears to be difficult. Some temporary control is likely, but difficulties sustaining long-term control are confounded by a) the patchiness of understory and mixed-severity fires, b) the biennial nature of the species, c) the moderately persistent seed bank, and d) garlic mustard’s propensity for rapid population increase (Luken and Shea 2000, Nuzzo et al. 1996, Schwartz and Heim 1996).
It may be possible to substantially diminish the number of individuals in a garlic mustard population with repeated burn treatments. But prescribed burning, especially during the growing season, could actually increase the relative importance of garlic mustard (Anderson et al. 1996, Luken and Shea 2000, Nuzzo et al. 1996). A prescribed burn conducted in May in a northern Illinois dry-mesic upland deciduous forest effectively reduced cover of garlic mustard, from a pre-burn 29.4% cover to 2.3% cover, post-fire year 1.

But May burning also damaged the native forb community, where total stem density of major herbs and small shrubs was reduced by 32% and average number of species per plot was reduced by 35%, post-fire year 1. Although native plants subsequently showed gradual recovery, these effects were detectable for 3 years, most notably for Jack-in-the-pulpit and stickywilly. Garlic mustard recovery was more rapid. Within three years following burning garlic mustard had rebounded to 17.3% cover compared with a pre-burn level of 29.4% (Schwartz and Heim 1996).

Dormant-season burns, while less likely to have negative effects on indigenous flora, also appear to be less effective at killing garlic mustard rosettes. After 3 years following a March prescribed burn at the above location, both garlic mustard and native herb cover had returned to approximate pre-burn levels (Schwartz and Heim 1996).

It has been suggested that a narrow window of time exists during early spring in some areas and in some years, during which garlic mustard may be more effectively controlled by fire without damaging native plants. This hypothesis remains untested as of this writing (Schwartz and Heim 1996). Also, spring burns may increase seedling survival. Fires of insufficient severity may spare a sizable fraction of seedlings protected by the unburned portion of the litter layer. Additionally, a spring burn timed too early may permit survival of garlic mustard seedlings that germinate after treatment. In addition to greater initial seedling survival, removal of a portion of the litter layer may also provide a more favorable environment for growth and development of garlic mustard rosettes (Nuzzo et al. 1996).

Apparently not all fires are equally effective at top-killing garlic mustard. The effectiveness of prescribed spring and fall burn treatments in reducing garlic mustard populations in an oak (Quercus spp.)-dominated dry-mesic upland forest in northern Illinois was directly related to fire “intensity.” “Low-intensity” burns, with flame lengths up to 3 cm, were patchy and frequently extinguished within plots. These “low intensity” burns had little to no effect on garlic mustard plants, whether seedlings or adults, regardless of season of burning. It was suggested that abundant green garlic mustard plants frequently extinguished the “low intensity” fires. “Mid-intensity” burns, with flame lengths up to 15 cm, burned through most of the plots and significantly reduced the presence of garlic mustard. Adult plant densities were reduced by both spring and fall burns, as well as repeated fires, although single spring burns were most effective (Nuzzo 1991).
In areas with long fire-return intervals where favorable conditions for conducting effective prescribed burns may be rare to nonexistent, especially repeated annual burns, or where fire-sensitive native species exist, prescribed fire may be unsuitable as a management tool. Nevertheless, in areas with a fire-tolerant native flora, frequent prescribed burning may deter garlic mustard invasion by both directly killing invading plants, and perhaps in some areas by enhancing growth of native herbaceous competitors and thereby reducing habitat for garlic mustard colonization (Nuzzo 1991, Weber and Wittmann 1996). It is highly likely that managers who use fire to control garlic mustard may need to augment burn treatments with 1 or more additional control methods, such as pulling or herbicide use to achieve acceptable levels of control.

LITERATURE CITED


Antennaria microphylla
Rosy pussytoes

FIRE ECOLOGY OR ADAPTATIONS

Rosy pussytoes colonizes bare mineral soil from light, wind-dispersed seed (Simpson 1990, Steele and Geier-Hayes 1993).

POST-FIRE REGENERATION STRATEGY

Initial-offsite colonizer (off-site, initial community); secondary colonizer - off-site seed.

IMMEDIATE FIRE EFFECT ON PLANT

Rosy pussytoes is probably killed by moderate or severe fires. However, no marked rosy pussytoes plants were killed by low-severity spring or fall prescribed fires in mountain big sagebrush (Artemisia tridentate spp. vaseyana)/rough fescue (Festuca scabrella), mountain big sagebrush/Kentucky bluegrass (Poa pratense), or Douglas-fir/mountain big sagebrush vegetation types in the Helena National Forest, Montana (Schwecke and Hann 1989).
PLANT RESPONSE TO FIRE

The response of rosy pussytoes to fire probably depends on site characteristics and fire severity. It is a major early seral species following fires in subalpine fir/beargrass habitat types in central Idaho (Simpson 1990). Rosy pussytoes was first observed in post-fire year 3 following the severe Sundance Forest Fire in northern Idaho (Stickney 1986). In Douglas-fir stands in the Deerlodge National Forest, Montana, rosy pussytoes decreased 22.5 percent in the first 2 post-fire years after spring fires (Bushey 1985). In sagebrush habitats in Idaho rosy pussytoes decreased the first years following September prescribed fires, but then increased and regained much of its original cover. Production was greater on lightly burned or moderately burned sites than on either unburned or severely burned sites 15 years after the fires (Blaisdell 1953).

LITERATURE CITED


Artemisia absinthium
Absinth wormwood

FIRE ECOLOGY OR ADAPTATIONS

Although top-killed by fire, absinth wormwood probably re-establishes after fire by sprouting from undamaged perennating buds (Steuter and Plumb 1988) or regenerating from buried seed.

POST-FIRE REGENERATION STRATEGY

Caudex, growing points in soil; ground residual colonizer (on-site, initial community).

IMMEDIATE FIRE EFFECT ON PLANT

Low-severity fire readily top-kills absinth wormwood and may completely kill some plants. Because absinth wormwood perennating buds are at or near the soil surface, they are susceptible to fire (Steuter and Plumb 1988).

PLANT RESPONSE TO FIRE

Absinth wormwood perennating buds will sprout if they survive fire. Annual early spring prescribed fires were conducted in a South Dakota prairie infested with absinth wormwood. Dormant fine fuels ranged from 2000 to 2400 kg/ha.

Nearly two-thirds of the absinth wormwood survived the first fire and regrew, but four consecutive annual spring fires reduced absinth wormwood by 96 percent (Steuter and Plumb 1988).

FIRE MANAGEMENT CONSIDERATIONS

Spring fire can reduce absinth wormwood on northern mixed prairie sites during years with adequate fuel. However, abundant green herbaceous material can reduce fire severity during the spring and thus reduce fire damage to absinth wormwood (Steuter and Plumb 1988).

LITERATURE CITED

Asarum caudatum
Wild ginger

FIRE ECOLOGY OR ADAPTATIONS

To date (2004) no primary literature has been located that specifically addresses wild ginger’s fire adaptations. Wild ginger was found the first postburn year following a high-severity fire in northern Idaho (Stickney 1986). In a decade-long study following a high-severity, stand-replacing fire in western redcedar-western hemlock forests of northern Idaho, wild ginger was classified as a residual colonizer, establishing from on- or off-site seed or fruit (Stickney 1989, Stuart 1987). However, wild ginger has rhizomes that sprout new individuals (Jankovsky-Jones et al. 1999, Raunkiaer 1948), and while no study specifically states that wild ginger sprouts from rhizomes following disturbance or fire, this adaptation should not be overlooked as a potential postdisturbance or post-fire regeneration strategy.

The fire regime for wild ginger is dictated by the overstory community. Prior to 1900, grand fir communities are characterized as having mixed and stand-replacing fire regimes (Arno and Fischer 1995). Smith and Fischer (1997) highlights the extreme variation within grand fir habitat types. This includes frequent fires that create persistent shrub communities and areas where no evidence of past fire has been located (Smith and Fischer 1997). Western hemlock/wild ginger habitat types experienced infrequent, high-severity fires at 100 to 200 year intervals (Lu 1982). Presettlement fire regimes in northern Idaho for the western redcedar and western hemlock/wild ginger habitat types have been described as stand replacing with long fire return intervals that have been attributed to the moist understory conditions and the build-up of continuous fuels (Smith and Fischer 1997). However, small understory burns have been described as well.

The following list provides fire return intervals for plant communities and ecosystems where wild ginger may be found. It may not be inclusive.

<table>
<thead>
<tr>
<th>Community or Ecosystem</th>
<th>Dominant Species</th>
<th>Fire Return Interval Range (years)</th>
</tr>
</thead>
<tbody>
<tr>
<td>silver fir-Douglas-fir</td>
<td>Abies amabilis-Pseudotsuga menziesii var. menziesii</td>
<td>&gt; 200</td>
</tr>
<tr>
<td>grand fir</td>
<td>Abies grandis</td>
<td>35-200 [1]</td>
</tr>
<tr>
<td>tamarack</td>
<td>Larix laricina</td>
<td>35-200 [2]</td>
</tr>
<tr>
<td>western larch</td>
<td>Larix occidentalis</td>
<td>25-350 [3,4,5]</td>
</tr>
<tr>
<td>western white pine*</td>
<td>Pinus monticola</td>
<td>50-200</td>
</tr>
<tr>
<td>Pacific ponderosa pine*</td>
<td>Pinus ponderosa var. ponderosa</td>
<td>1-47 [1]</td>
</tr>
<tr>
<td>interior ponderosa pine*</td>
<td>Pinus ponderosa var. scopulorum</td>
<td>2-30 [1,6,7]</td>
</tr>
<tr>
<td>Species Type</td>
<td>Scientific Name</td>
<td>Maximum Height</td>
</tr>
<tr>
<td>------------------------------------</td>
<td>----------------------------------------------------------------------------------</td>
<td>----------------</td>
</tr>
<tr>
<td>Rocky Mountain Douglas-fir*</td>
<td><em>Pseudotsuga menziesii</em> var. <em>glauc</em>a</td>
<td>25-100 [1,8,9]</td>
</tr>
<tr>
<td>coastal Douglas-fir*</td>
<td><em>Pseudotsuga menziesii</em> var. <em>menziesii</em></td>
<td>40-240 [1,10,11]</td>
</tr>
<tr>
<td>California mixed evergreen</td>
<td><em>Pseudotsuga menziesii</em> var. <em>menziesii</em> - <em>Lithocarpus densiflorus-Arbutus menziesii</em></td>
<td>&lt; 35 [1]</td>
</tr>
<tr>
<td>redwood</td>
<td><em>Sequoia sempervirens</em></td>
<td>5-200 [1,12,13]</td>
</tr>
<tr>
<td>western redcedar-western hemlock</td>
<td><em>Thuja plicata-Tsuga heterophylla</em></td>
<td>&gt; 200</td>
</tr>
<tr>
<td>western hemlock-Sitka spruce</td>
<td><em>Tsuga heterophylla-Picea sitchensis</em></td>
<td>&gt; 200</td>
</tr>
<tr>
<td>mountain hemlock*</td>
<td><em>Tsuga mertensiana</em></td>
<td>35 to &gt; 200 [1]</td>
</tr>
</tbody>
</table>

*fire return interval varies widely; trends in variation are noted in the species review

POST-FIRE REGENERATION STRATEGY (Stickney 1989)

Secondary colonizer (on-site or off-site seed sources)

IMMEDIATE FIRE EFFECT ON PLANT

Wild ginger is commonly killed by even low-severity fires and is described as very sensitive to fire in a literature review by Smith and Fischer (1997).

DISCUSSION AND QUALIFICATION OF FIRE EFFECT

Wild ginger may be fire tolerant but its presence in a community is likely dictated by its shade requirement. When studying the successional development of the grand fir/wild ginger habitat type, Green and Jensen (1991) found that following low intensity burns in which the forest canopy remained intact, wild ginger was still present in the community. However, when fires removed canopy vegetation wild ginger was not among the first species to colonize (Green and Jensen 1991).

PLANT RESPONSE TO FIRE

Wild ginger’s post-fire response depends on the severity on the fire and its effect on the overstory community. Following moderate severity slash burns in a clearcut Douglas-fir community near Oakridge, Oregon, burned and unburned plots were compared (Stickney 1985). Wild ginger, an uncommon understory species in this community, was not found on burned plots sampled 11 to 16 years following the fires. It did occur on clearcut, unburned sites, where vine maple contributed to greater shading of the area (Stickney 1985). Similarly, in a study of single and multiple broadcast burns that removed all tree cover except an occasional western larch, wild ginger was significantly (p = 0.05) more frequent on sites that had not burned (Muir 1995). Data were not provided about time since or between burns of the sites.

The first year following a high-severity, stand-replacing fire in northern Idaho forests, wild ginger was found on just 1 of 21 postburn plots (Stickney 1986). The wild ginger found was a seedling and did not flower that first postburn year (Stickney 1986). In a study designed to determine successional pathways following disturbance within the grand fir/wild ginger habitat type, wild ginger was found in all but the earliest seral communities following fire and clearcutting that produce fires of varying severities (Halvorsen 1986).
LITERATURE CITED


FIRE ECOLOGY OR ADAPTATIONS

Information regarding the fire ecology of California brome is scant. Based upon the few data available, fire appears to have little long-term effect on California brome. Coverage of California brome is slightly reduced from pre-fire levels for several years after fire, then returns to approximate pre-fire levels (Brown and DeByle 1989, Leege and Godbolt 1985, Simmerman et al. 1991).

POST-FIRE REGENERATION STRATEGY

Tussock graminoid.

IMMEDIATE FIRE EFFECT ON PLANT

California brome is top-killed by fire. In the Pacific Northwest, it routinely survives fall burning of pastureland and sprouts from surviving root crowns the next growing season (Young et al. 1984).

PLANT RESPONSE TO FIRE

California brome appears to recover from fire within a few years. It was one of three grasses to establish in the first 2 years after stand-replacing wildfire in two-needle pinyon-Utah juniper woodland in Mesa Verde National Park, Colorado (Erdman 1969).
Recovery of California brome seems to be similar after both early- and late-season burning. Data on effects of summer burning are not available. Neither spring nor fall prescribed fire after logging had much effect on California brome cover in northern Idaho. After shelterwood cutting in a ponderosa pine (Pinus ponderosa) forest on the Priest River Experimental Forest, two prescribed fires were set to reduce fuels: a June 1, 1989, fire on moist duff (moist burn), and a September 13 and 14, 1989, fire on dry duff (dry burn). California brome percentage cover a year before logging and at post-fire year 1 was as follows (Simmerman et al. 1991):

<table>
<thead>
<tr>
<th>Control (no burn)</th>
<th>Moist Burn</th>
<th>Dry Burn</th>
</tr>
</thead>
<tbody>
<tr>
<td>pre-fire</td>
<td>post-fire</td>
<td>pre-fire</td>
</tr>
<tr>
<td>0.7</td>
<td>1.4</td>
<td>0.4</td>
</tr>
</tbody>
</table>

Similarly, early fall prescribed burning in pure quaking aspen and mixed quaking aspen-conifer forests had little effect on California brome.

In mixed forests on the Caribou National Forest of Idaho and the Bridger-Teton National Forest of Wyoming, California brome was categorized as providing 5 percent or less cover on both burned and unburned plots at post-fire years 1 and 2. In pure quaking aspen on the Caribou and Bridger-Teton National Forests, California brome was categorized as providing from 6 to 25 percent cover on burned and control plots at post-fire years 1 and 2 (Brown and DeByle 1989).

California brome was significantly reduced (p = 0.05) for at least 1 year by spring prescribed fire in southwestern Montana. Average basal cover of California brome in the first post-fire growing season was 0.55 sq dm/sq dm on burned plots and 0.72 sq dm/sq dm on unburned plots (Nimir and Payne 1978).

Season of burning affected California brome dominance on the Helena National Forest of Montana. A year after fall prescribed fire in a mountain big sagebrush/Kentucky bluegrass community, California brome was codominant with Kentucky bluegrass. After spring prescribed fire in the same community, dandelion (Taraxacum officinale) was codominant with Kentucky bluegrass (Schwecke and Hann 1989).

FIRE MANAGEMENT CONSIDERATIONS

Post-fire seeding: California brome is sometimes seeded in after fire and often establishes good cover. It can help stabilize soil after fire but may preclude establishment of other species including conifers (Root and Habeck 1972, Steele and Geier-Hayes 1990). Steele and Geier-Hayes (1990) stated that seeding was not necessary on conifer/pinegrass types where good pinegrass cover was present before fire.
California brome developed good coverage following post-fire seeding on two Douglas-fir/ninebark (*Physocarpus malvaceus*) habitat types in central Idaho. In northern Idaho, California brome coverage on several seeded burn sites varied from “vigorous” to “poor.” Fall seedings resulted in better grass coverage than spring seedings (Slinkard et al. 1970). Fall seeding on prescribed burned ponderosa pine sites in north-central Washington resulted in good California brome coverage (Weaver 1951).

Bradley et al. (1992) describe a quaking aspen/California brome community type that appears to be maintained by long-term grazing. Fire is difficult to sustain in this type due to discontinuity of fuels. Rate of spread may be one-tenth that of ungrazed stands. Quaking aspen/grass types will not sustain fire spread unless flame lengths are 0.3 - 0.5 m, which requires at least 50 percent cured herbaceous vegetation (Brown and Simmerman 1986).

Brown et al. (1989) found that time of year was a simple, reliable indicator of moisture content of herbaceous fuels, including California brome, in quaking aspen understories. In a quaking aspen/western coneflower (*Rudbeckia occidentalis*) community type on the Bridger-Teton National Forest, Wyoming, perennial grasses in the understory cured at a slow, steady rate beginning early in the growing season. Dominant understory grasses measured for seasonal change in moisture content included California brome, blue wildrye (*Elymus glaucus*), and slender wheatgrass. Gradual curing of perennial grasses occurred in both 1981, a dry summer, and 1982, a wet summer. Moisture content of grasses corresponded “reasonably well” to the National Fire Danger Rating System (NFDRS) (Deeming et al. 1974) during mid- to late summer. In contrast, forb moisture content decreased slowly in early summer, then accelerated rapidly.

Forb moisture content related poorly to the NFDRS, fluctuating through a much greater range of moisture contents than allowed by the NFDRS model (Brown et al. 1989).

Fire spread may be difficult to obtain in mountain big sagebrush/California brome or other mountain big sagebrush/grass types due to wide spacing of shrubs and mesic conditions (Hironaka et al. 1983).

LITERATURE CITED


FIRE ECOLOGY OR ADAPTATIONS

Summer and fall fires have no direct effect on soft chess. Soft chess has usually senesced and shattered seed when the fire season starts. The seed is not killed until fire temperatures rise above approximately 93º C. Since grassland fires are usually fast-burning and relatively “cool,” soft chess seed is usually not damaged by fire (McNaughton 1968, Sampson et al. 1951).

Fire can affect relative abundance of soft chess in the post-fire plant community, however (Keeley and Keeley 1984, Seymour 1982). Fire removes mulch, which favors annual forbs over soft chess. Some soft chess germinates the fall after fire, but best germination occurs in mid-succession, when mulch layer is moderate (Bartolome 1979, Bartolome 1987).

Fire regimes: California native grassland - data are lacking to quantify intensity and frequency of fire in pristine California prairie. It is generally accepted that lightning-caused fire was part of the evolutionary history of California prairie. The California Division of Forestry reported an average of 312 lightning-ignited fires per year in its fire protection area, which is 43 percent woodland-annual grassland. Frequency of lightning-caused fires was probably at least as great in the presettlement era (Heady 1987).
Native Americans may have used frequent fire to enhance production of edible perennial bunchgrass seeds (Bean and Lawton 1973). Fire enhances flowering and seedling recruitment for some perennial bunchgrasses native to California prairie including purple needlegrass (Kuchler 1964) and bottlebrush squirreltail. Both species show mass flowering after fire and require mineral soil for establishment (Glenn-Lewin et al. 1990, Keeley 1990).

Annual grassland - since California annual grassland has existed for less than two hundred years, it has no evolutionary history of fire. Like the perennial grassland that preceded it, however, California annual grassland is a fire-tolerant ecosystem (Keeley and Keeley 1984). Studies attempting to promote native perennial bunchgrasses over exotic annuals by using prescribed fire have had mixed results.

**IMMEDIATE FIRE EFFECT ON PLANT**

Fire has little direct effect on soft chess. Wildland and prescribed fires usually occur after soft chess has dried and shattered seed (Heady 1973, Heady 1977, Heady et al. 1977).

**PLANT RESPONSE TO FIRE**

Fire may reduce soft chess in the short term (Hervey 1949). Species composition in the post-fire plant community is difficult to predict, however. Year-to-year plant composition in annual grassland is highly dependent upon local weather patterns, and even slight differences in annual precipitation can alter species assemblages (Keeley and Keeley 1984). Fall weather patterns, especially interactions of precipitation and temperature after rainfall, appear to be overriding factors in soft chess establishment (Heady 1977, Heady et al. 1992, Kay 1987).

Fire affects plant species composition in annual grasslands largely by removing mulch, which affects germination and seedling establishment rates of soft chess relative to associated herbaceous species. Bartolome (1979, 1987) found that soft chess reached highest densities when mulch biomass was at intermediate levels. Little quaking grass (*Briza minor*) was favored when mulch biomass was low, as it would be in the immediate post-fire environment. Fescues (*Vulpia and Festuca* spp.) were favored when mulch biomass was high. Heady (1977) reported that without heavy grazing the mulch layer usually recovers by post-fire year 3, and soft chess and other annual bromes regain dominance.

Decreases with fire: Hansen (1986) found that fall prescribed fire in Tulare County, California, significantly increased dominance of annual forbs relative to soft chess. Greatest reduction soft chess and other annual grasses (and greatest increase of annual forbs) was achieved by 3 years of successive fall burning. Response of native grasses was similar to that of soft chess: Native grasses were reduced by fall burning, with greatest reduction achieved after 3 years of consecutive fall burning. Percent cover of soft chess the spring after fall burning follows.
A July 1947 prescription fire reduced soft chess on ungrazed annual grassland near Berkeley, California. Precipitation in the fall and winter of 1947-1948 was slightly below average for the area 51.8 cm with the average being 57.4 cm. Average height and yield of soft chess on two burned and two unburned sites in May of 1948 was as follows (Hervey 1949):

<table>
<thead>
<tr>
<th>Year</th>
<th>Burned</th>
<th>Unburned</th>
</tr>
</thead>
<tbody>
<tr>
<td>1982</td>
<td>10</td>
<td>&lt;1</td>
</tr>
<tr>
<td>1983</td>
<td>8</td>
<td>5</td>
</tr>
<tr>
<td>1984</td>
<td>23</td>
<td>44</td>
</tr>
<tr>
<td>1985</td>
<td>12</td>
<td>23</td>
</tr>
</tbody>
</table>

Mixed effects: Chaparral and oak woodland - Density of soft chess increased greatly from pre-fire levels 5 years after prescribed fall burning in a nonsprouting manzanita-Lemmon ceanothus (*Arctostaphylos* spp.-*Ceanothus lemmonii*) community in Mendocino County. However, density of soft chess had changed little 5 years after prescribed fall fires in nearby nonsprouting manzanita (*Arctostaphylos* spp.)-Lemmon ceanothus and interior live oak-blue oak (*Quercus wislezenii*-*Q. douglasii*) woodland communities. Average density (plants/milacre) of soft chess was (Sampson *et al.* 1951):

<table>
<thead>
<tr>
<th>Community</th>
<th>Pre-fire</th>
<th>1</th>
<th>2</th>
<th>3</th>
<th>4</th>
<th>5</th>
</tr>
</thead>
<tbody>
<tr>
<td>nonsprouting manzanita-ceanothus</td>
<td>0.0</td>
<td>2.8</td>
<td>7.3</td>
<td>11.2</td>
<td>24.6</td>
<td>30.3</td>
</tr>
<tr>
<td>sprouting manzanita-ceanothus</td>
<td>0.3</td>
<td>4.1</td>
<td>6.5</td>
<td>3.8</td>
<td>5.1</td>
<td>2.8</td>
</tr>
<tr>
<td>live oak-blue oak</td>
<td>1.5</td>
<td>6.6</td>
<td>6.7</td>
<td>5.8</td>
<td>3.0</td>
<td>1.3</td>
</tr>
</tbody>
</table>

No effect: neither spring nor fall prescribed fire had significant effect on soft chess in annual grassland of Sequoia National Park, California. Precipitation averaged about 200 percent of normal during post-fire years 1 to 4. Soft chess formed an important component of the vegetation (between 10 and 27%) on plots measured before fire and on spring-burned, fall-burned, and unburned plots measured 4 years after fire (Radford *et al.* 1968).
Sagebrush steppe - in central Idaho, fire had little effect on soft chess coverage in either the long term or the short term. A long-term study was conducted above the Snake River Canyon, after a July wildfire occurred 1961 in a rubber rabbitbrush (Chrysothamnus nauseosus)-cheatgrass community. At post-fire year 12, soft chess had declined on both burned and adjacent unburned plots. (Weather data were not given.) Soft chess coverage was as follows (Daubenmire 1975):

<table>
<thead>
<tr>
<th>Post-fire year</th>
<th>Unburned</th>
<th>Burned</th>
</tr>
</thead>
<tbody>
<tr>
<td>2</td>
<td>4.80</td>
<td>trace</td>
</tr>
<tr>
<td>4</td>
<td>1.45</td>
<td>trace</td>
</tr>
<tr>
<td>12</td>
<td>trace</td>
<td>trace</td>
</tr>
</tbody>
</table>

A short-term study was conducted nearby when an August 1972 wildfire occurred in a rubber rabbitbrush-cheatgrass stand within the Snake River Canyon. The following spring, soft chess frequency was 21 percent on unburned plots and 18 percent on burned plots (Daubenmire 1975).

**FIRE MANAGEMENT CONSIDERATIONS**

California: annual grassland - use of prescribed fire to increase the balance of natives relative to non-natives such as soft chess has had mixed results. In all cases, “remnant” California prairie contains exotic annuals, and attempts to eliminate the exotics have been unsuccessful (Keeley and Keeley 1984). However, fire sometimes tips the balance toward natives. Perennial bunchgrasses are well adapted to frequent fire (Clements 1934, Welsh et al. 1987). Some authors have reported that fire favors native bunchgrassses over exotic annuals (Ahmed 1983, McKell et al. 1962).

However, Garcia and Lathrop (1984) reported no increase in purple needlegrass after burning, and Lathrop and Martin (1982) found that native deer grass (Muhlenbergia rigens) decreased under some burning regimes. In view of the differences in phenology and life histories between perennial bunchgrasses and annual grasses such as soft chess, it would be instructive to know how burning in different seasons affects the ratio of native to non-natives. Since annual grasses produce seed about a month earlier than perennial grasses, precise timing of burning may alter the balance of reproductive success between annual and perennial grasses (Keeley and Keeley 1984).

When used with prescribed grazing, fire may favor purple needlegrass and reduce soft chess and other annual grasses. Langstrotti (1991) found that on the Jepson Prairie (a relict perennial grassland reserve in Solano County, California), short-term, intensive grazing by domestic sheep in early spring (late March or early April) combined with late summer (early September) prescribed fire favored tillering and seedling establishment of purple needlegrass over exotic annual grasses including soft chess. Purple needlegrass had been declining on the reserve for a number of years. Frequency of soft chess was significantly reduced (p = 0.05) by early spring grazing and late summer fire. The treatments reduced soft chess cover to less than 2 percent. Early spring grazing reduced average seed mass, and the number of soft
chess seeds was reduced by 76 percent (p = 0.25). Late summer fire reduced soft chess cover by 50 percent (p < 0.001). Summer grazing and late summer fire also reduced soft chess, but not as much. Data from the spring grazing/late summer fire treatments follow.

<table>
<thead>
<tr>
<th></th>
<th>grazed-burnt</th>
<th>ungrazed-unburnt</th>
</tr>
</thead>
<tbody>
<tr>
<td>soft chess frequency (%)</td>
<td>39.7</td>
<td>3.0</td>
</tr>
<tr>
<td>soft chess seeds/sq dm</td>
<td>198</td>
<td>1,343</td>
</tr>
<tr>
<td>soft chess seed mass (mg)</td>
<td>0.57</td>
<td>0.97</td>
</tr>
</tbody>
</table>

Effects of post-fire seeding of ryegrass on soft chess: seeding Italian ryegrass (*Lolium multiflorum*) to reduce post-fire erosion had little effect on post-fire growth of soft chess and other exotic bromes in southern California chaparral. Coverage of annual bromes was similar on unseeded plots and on plots seeded with Italian ryegrass (*Beyers et al.* 1994).

Oregon: big sagebrush - prescribed fire had little effect on soft chess in a basin big sagebrush/bluebunch wheatgrass community in John Day Fossil Beds National Monument, Oregon. Weather patterns occurring after fire greatly influenced plant community composition, however. One study area was prescribed burned on September 25, 1987; an adjacent study area was prescribed burned on May 24, 1988. Prescription burning was followed by 3 years of drought, which appeared to greatly reduce soft chess cover. By the third post-fire year, soft chess was absent from all treatments including the unburned control.

Density of other annual grass species was also greatly reduced on all treatments including the unburned control. Density of annual forbs increased on all plots, and density of native perennial grasses did not change. Density of woody shrub species was greatly reduced on burned plots but did not change on control plots. Average density of soft chess (plants/m²) on unburned control, fall-burned, and spring-burned plots is given below. Numbers in parenthesis are the standard errors of the mean; different letters denote a significant difference between years (p < 0.1) (*Seymour 1982*).

<table>
<thead>
<tr>
<th></th>
<th>1987</th>
<th>1988</th>
<th>1989</th>
</tr>
</thead>
<tbody>
<tr>
<td>control</td>
<td>160a (87)</td>
<td>0b (0)</td>
<td>0b (0)</td>
</tr>
<tr>
<td>fall burn</td>
<td>82a (28)</td>
<td>10b (8)</td>
<td>0b (0)</td>
</tr>
<tr>
<td>spring burn</td>
<td>--</td>
<td>37a (16)</td>
<td>0b (0)</td>
</tr>
</tbody>
</table>

LITERATURE CITED


_Bromus inermis (= Bromus pumpellianus)_

Pumpelly brome

FIRE ECOLOGY OR ADAPTATIONS

Since Pumpelly brome can reproduce vegetatively (Hitchcock 1951, Hulten 1968), it probably sprouts from rhizomes after aerial portions are burned.

POST-FIRE REGENERATION STRATEGY

Rhizomatous herb, rhizome in soil

IMMEDIATE FIRE EFFECT ON PLANT

Pumpelly brome culms are probably killed by fire during the growing season.

LITERATURE CITED


Bromus tectorum
Cheatgrass

FIRE ECOLOGY OR ADAPTATIONS

Cheatgrass establishes from soil-stored and transported seed after fire (e.g. Evans and Young 1987, Hassan and West 1986, Hulbert 1955, Humphrey and Schupp 2001, Young et al. 1969, Young et al. 1972). It has long been known that cheatgrass is highly adapted to a regime of frequent fires (Leopold 1941, Pickford 1932).

Cheatgrass has a very fine structure, tends to accumulate litter, and dries completely in early summer, thus becoming a highly flammable, often continuous fuel (Billings 1952, Peters and Bunting 1994, Stewart and Hull 1949, Young 1989, Young 1991). By the time of burning most cheatgrass seeds are already on the ground, and those not near the heat of burning shrubs can survive and allow cheatgrass to pioneer in the newly burned area (Billings 1952). Even if fire comes when cheatgrass plants are still green and kills them before they can set seed, there may be enough viable cheatgrass seed in litter and upper layers of soil for plants to reestablish (e.g. Eckert and Evans 1967, Harris and Goebel 1976, Hull et al. 1974, Young et al. 1969).

Cheatgrass is a strong competitor in the post-fire environment, where it takes advantage of increased resource availability and produces an abundant seed crop (Billings 1994, Keeley 2001, Young and Evans 1978). A cheatgrass population may average around 10,750 per m$^2$ prior to burning. During a wildfire, most of the cheatgrass seeds beneath the canopy of sagebrush plants are killed by the heat associated with the burning of the shrub. Some cheatgrass seeds located in the interspaces among shrubs are also consumed, while those that are buried or lying in cracks in the soil will likely survive. The next season, surviving seeds germinate and establish at a density of about 11/m$^2$. These plants are released from competition, and have more water and nutrients available to them. The cheatgrass plants in this sparse population can produce abundant tillers, each supporting many flowers, thus producing a large seed crop (Young et al. 1987).

Young et al. (1987) provide an illustration of cheatgrass fire adaptations with an example from a big sagebrush ecosystem which suggests that hybridization in post-fire populations contributes to the success of cheatgrass in these ecosystems. Studies by Novak (e.g. Novak 1994, Novak and Mack 1993, Novak et al. 1991) and by Pyke and Novak (1994) suggest, however, that “the success of cheatgrass throughout many areas in western North America is not due to genetic variation but perhaps due to phenotypic plasticity.”

Fire facilitates cheatgrass dominance on some sites by interrupting successional trajectories of post-fire plant communities, and cheatgrass facilitates fire and can thus shorten the interval between fires (Billings 1994, Mack 1981, Stewart and Hull 1949, West 1994, Whisenant 1990, Young et al. 1987). This grass/fire cycle is a serious ecological threat on sites where most native plant species are poorly adapted to fire.
(Brooks 1999) and is recognized in many ecosystems worldwide (D’Antonio and Vitousek 1992). This cycle has been documented in the Great Basin since the 1930s (Pickford 1932, Piemeisel 1951, Whisenant 1990, Young and Evans 1978), and has been reported in the Mojave and Sonoran deserts beginning in the early 1980s (Brooks and Pyke 2001). The result is a type conversion from native shrub and perennial grasslands to annual grasslands adapted to frequent fires.

Fire regimes: Cheatgrass expansion has dramatically changed fire regimes and plant communities over vast areas of western rangelands by creating an environment where fires are easily ignited, spread rapidly, cover large areas, and occur frequently (Young and Evans 1978). An estimated 80,000 km$^2$ of primarily shrubland and grassland communities in the Great Basin have fire regimes that have been seriously altered because of the presence of cheatgrass. Approximately 67% of this area is in ecosystems that historically experienced mixed-severity fires at intervals of 35 to 100+ years; and about 25% is in areas that historically experienced low-severity fires at intervals of 0 to 35 years (Menakis et al. 2003).

Cheatgrass promotes more frequent fires by increasing the biomass and horizontal continuity of fine fuels that persist during the summer lightning season and by allowing fire to spread across landscapes where fire was previously restricted to isolated patches (Beatley 1966, Billings 1952, Billings 1994, Brooks and Pyke 2001, Bunting et al. 1987, D’Antonio 2000, Knick and Rotenberry 1997, Stewart and Hull 1949, Whisenant 1990, Young 1989, Young and Evans 1978). Fire in these habitats can have severe effects on native species of plants and animals, although the impact of fire regime changes may differ by region and ecosystem type due to differences in the composition and structure of the invaded plant communities (Daubenmire 1970, Ott et al. 2001, Whisenant 1990, Wright et al. 1979) and to climatic differences such as occurrence of summer thunderstorms (Billings 1994, Knapp 1998).

A review by D’Antonio (2000) suggests that species that alter the disturbance regime of a site are those that are qualitatively different from the rest of the species in a community (i.e. they have no functional analogues in the invaded system). Where invaders are similar in overall life form to natives, they tend to alter primarily fuel biomass per unit area of ground. This in turn has the potential to influence fire intensity, or slightly modify the existing fire regime, as may be the case with cheatgrass invasion in the more mesic temperate grasslands of North America (Grace et al. 2001). Where invaders have no functional analogues (in terms of fuel characteristics) in the invaded system, they have the potential to alter fire frequency and even to introduce fire to ecosystems where it had no evolutionary role, resulting in a complete alteration of that community (D’Antonio 2000, Grace et al. 2001). This has been the case with the introduction of cheatgrass in sagebrush grasslands, desert shrublands, and pinyon-juniper woodlands over extensive areas in the Columbia and Great basins and other areas the Intermountain West. In these systems, cheatgrass fills spaces between widely spaced vegetation and dries earlier than most native species. Thus, from the time plants dry until the onset of fall rains, cheatgrass stands present a fire hazard not usually found in vegetation native to the areas where it is most invasive.
Many dry temperate conifer forests have become susceptible to severe wildfires because of the dense forest structure that results from a century of fire exclusion and past management practices (e.g. Arno et al. 1995). Fires in these ecosystems, especially fires of high severity, can lead to invasion and dominance of cheatgrass.

At Sequoia-Kings Canyon National Park, prescribed burning in ponderosa pine in the Cedar Grove section appears to have promoted vigorous invasion of cheatgrass (Keeley 2001). Cheatgrass had higher cover on severely burned sites, compared to less severely burned sites, in ponderosa pine in Arizona (Crawford et al. 2001). The presence of cheatgrass-dominated ecosystems adjacent to these dense forests is also likely to cause larger, more frequent, and more severe wildfires (Harrod and Reichard 2001). Cheatgrass fueled a large wildfire in the ponderosa pine forest type in Oregon as early as 1938 (Weaver 1959). Fire effects on many species, and the effects of invasives on disturbance regimes in temperate and boreal forests, are still poorly understood (Harrod and Reichard 2001).

In temperate grasslands of North America, fire has historically been an important selective force, and native communities are well adapted to frequent fires in most cases. Cheatgrass is more commonly found in the northern portion of these temperate grasslands. In more arid habitats with low natural fire frequencies cheatgrass is able to replace native species. In mesic grasslands, however, cheatgrass does not compete as successfully against native perennial grasses, and it does not appear to pose as great a threat to native communities. The effects of new species that create greater fuel loads and/or increase the probability of fire or the rate of fire spread are expected to have less dramatic effects in these communities (Grace et al. 2001).

A review by Grace and others (2001) suggests that cheatgrass is favored by occasional burning at study sites within shortgrass steppe and mixed-grass prairies. Smith and Knapp (1999) provide evidence that cheatgrass and other nonnative species are less frequent on tallgrass prairie sites at Konza Prairie, Kansas, that are annually burned than they are on unburned sites. Across the broad range of conditions and circumstances that occur in temperate grasslands, a complex interplay of contemporary and historical factors will ultimately determine how fire interrelates with invasive species (Grace et al. 2001).

Cheatgrass fire regime: Cheatgrass often dominates post-fire plant communities, and once established, cheatgrass-dominated grasslands greatly increase the potential and recurrence of wildfires. Cheatgrass fires tend to burn fast and cover large areas, with a fire season from 1 to 3 months longer than that of native rangeland (Paysen et al. 2000, Roberts 1991). The average fire-return interval for cheatgrass-dominated stands is less than 10 years (Paysen et al. 2000), and is about 3 to 6 years on the Snake River Plain as reported by Whisenant (1990) and Peters and Bunting (1994). This adaptation to and promotion of frequent fires is what gives cheatgrass its greatest competitive advantage in ecosystems that evolved with less frequent fires.
The cheatgrass-fire cycle is self-promoting, as it reduces the ability of many perennial grasses and shrubs to re-establish and furthers the dominance of cheatgrass (Pellant 1990, Peters and Bunting 1994). Moisture availability can affect cheatgrass productivity and thus affect fuel loads on a site. Drought years may reduce the dominance of cheatgrass in both recently burned and unburned areas, thus decreasing fuel loads and the chance of fire (Knapp 1998).

The following table provides some fire regime intervals for ecosystems in which cheatgrass may occur.

<table>
<thead>
<tr>
<th>Community or Ecosystem</th>
<th>Dominant Species</th>
<th>Fire Return Interval Range (years)</th>
</tr>
</thead>
<tbody>
<tr>
<td>grand fir</td>
<td><em>Abies grandis</em></td>
<td>35-200 [1]</td>
</tr>
<tr>
<td>California chaparral</td>
<td><em>Adenostoma</em> and/or <em>Arctostaphylos</em> spp.</td>
<td>&lt; 35 to &lt; 100 [2]</td>
</tr>
<tr>
<td>bluestem prairie</td>
<td><em>Andropogon gerardii</em> var. <em>gerardii</em>- <em>Schizachyrium</em></td>
<td>&lt; 10 [3,2]</td>
</tr>
<tr>
<td>Nebraska sandhills prairie</td>
<td><em>A. g. var. paucipilus</em>- <em>S. scoparium</em></td>
<td>&lt; 10</td>
</tr>
<tr>
<td>bluestem-Sacahuista prairie</td>
<td><em>A. littoralis</em>- <em>Spartina</em> <em>spartinae</em></td>
<td>&lt; 10 [2]</td>
</tr>
<tr>
<td>silver sagebrush steppe</td>
<td><em>Artemisia cana</em></td>
<td>5-45 [4,5,6]</td>
</tr>
<tr>
<td>sagebrush steppe</td>
<td><em>A. tridentata</em>- <em>Pseudoroegneria</em> <em>spicata</em></td>
<td>20-70 [2]</td>
</tr>
<tr>
<td>basin big sagebrush</td>
<td><em>A. t. var. tridentata</em></td>
<td>12-43 [7]</td>
</tr>
<tr>
<td>mountain big sagebrush</td>
<td><em>A. t. var. vasyana</em></td>
<td>15-40 [8,9,10]</td>
</tr>
<tr>
<td>Wyoming big sagebrush</td>
<td><em>A. t. var. wyomingensis</em></td>
<td>10-70 (40**)[11,12]</td>
</tr>
<tr>
<td>coastal sagebrush</td>
<td><em>A. californica</em></td>
<td>&lt; 35 to &lt; 100</td>
</tr>
<tr>
<td>saltbush-greasewood</td>
<td><em>Atriplex confertifolia</em>- <em>Sarcobatus</em> <em>vermiculatus</em></td>
<td>&lt; 35 to &lt; 100</td>
</tr>
<tr>
<td>desert grasslands</td>
<td><em>Bouteloua eriopoda</em> and/or <em>Pleuraphis</em> <em>mutica</em></td>
<td>5-100</td>
</tr>
<tr>
<td>plains grasslands</td>
<td><em>Bouteloua</em> <em>spp.</em></td>
<td>&lt; 35</td>
</tr>
<tr>
<td>blue grama-needle-and-thread grass-western wheatgrass</td>
<td><em>B. gracilis</em>- <em>Hesperotis</em> <em>comata</em>- <em>Pascopyrum smithii</em></td>
<td>&lt; 35</td>
</tr>
<tr>
<td>blue grama-buffalo grass</td>
<td><em>B. gracilis</em>- <em>Buchloe</em> <em>dactyloides</em></td>
<td>&lt; 35</td>
</tr>
<tr>
<td>grama-galleta steppe</td>
<td><em>B. gracilis</em>- <em>Pleuraphis</em> <em>jamesii</em></td>
<td>&lt; 35 to &lt; 100</td>
</tr>
<tr>
<td>blue grama-tobosa prairie</td>
<td><em>B. gracilis</em>- <em>P. mutica</em></td>
<td>&lt; 35 to &lt; 100 [2]</td>
</tr>
<tr>
<td>cheatgrass</td>
<td><em>Bromus</em> <em>tectorum</em></td>
<td>&lt; 10 [13,14]</td>
</tr>
<tr>
<td>California montane chaparral</td>
<td><em>Ceanothus</em> and/or <em>Arctostaphylos</em> <em>spp.</em></td>
<td>50-100 [2]</td>
</tr>
<tr>
<td>curlleaf mountain-mahogany*</td>
<td><em>Cercocarpus</em> <em>ledifolius</em></td>
<td>13-1000 [15,16]</td>
</tr>
<tr>
<td>mountain-mahogany-Gambel oak scrub</td>
<td><em>C. ledifolius</em>- <em>Quercus</em> <em>gambelii</em></td>
<td>&lt; 35 to &lt; 100</td>
</tr>
<tr>
<td>Plant Type</td>
<td>Scientific Name</td>
<td>Height Range</td>
</tr>
<tr>
<td>----------------------------------</td>
<td>----------------------------------------------</td>
<td>----------------</td>
</tr>
<tr>
<td>Blackbrush</td>
<td>Coleogyne ramosissima</td>
<td>&lt; 35 to &lt; 100</td>
</tr>
<tr>
<td>Arizona cypress</td>
<td>Cupressus arizonica</td>
<td>&lt; 35 to 200</td>
</tr>
<tr>
<td>Northern cordgrass prairie</td>
<td>Distichlis spicata-Spartina spp.</td>
<td>1-3</td>
</tr>
<tr>
<td>California steppe</td>
<td>Festuca-Danthonia spp.</td>
<td>&lt; 35</td>
</tr>
<tr>
<td>Juniper-oak savanna</td>
<td>Juniperus ashei-Quercus virginiana</td>
<td>&lt; 35</td>
</tr>
<tr>
<td>Ashe juniper</td>
<td>J. ashei</td>
<td>&lt; 35</td>
</tr>
<tr>
<td>Western juniper</td>
<td>J. occidentalis</td>
<td>20-70</td>
</tr>
<tr>
<td>Rocky Mountain juniper</td>
<td>J. scopulorum</td>
<td>&lt; 35 [2]</td>
</tr>
<tr>
<td>Western larch</td>
<td>Larix occidentalis</td>
<td>25-100 [1]</td>
</tr>
<tr>
<td>Creosotebush</td>
<td>Larrea tridentata</td>
<td>&lt; 35 to &lt; 100</td>
</tr>
<tr>
<td>Ceniza shrub</td>
<td>L. tridentata-Leucophyllum frutescens-Prosopis glandulosa</td>
<td>&lt; 35</td>
</tr>
<tr>
<td>Wheatgrass plains grasslands</td>
<td>Pascopyrum smithii</td>
<td>&lt; 35</td>
</tr>
<tr>
<td>Pinyon-juniper</td>
<td>Pinus-Juniperus spp.</td>
<td>&lt; 35 [2]</td>
</tr>
<tr>
<td>Mexican pinyon</td>
<td>P. cembroides</td>
<td>20-70 [17,18]</td>
</tr>
<tr>
<td>Rocky Mountain lodgepole pine*</td>
<td>P. contorta var. latifolia</td>
<td>25-300+ [16,1,19]</td>
</tr>
<tr>
<td>Sierra lodgepole pine*</td>
<td>P. c. var. murrayana</td>
<td>35-200 [1]</td>
</tr>
<tr>
<td>Colorado pinyon</td>
<td>P. edulis</td>
<td>10-400+ [20,21,22,2]</td>
</tr>
<tr>
<td>Jeffrey pine</td>
<td>P. jeffreyi</td>
<td>5-30</td>
</tr>
<tr>
<td>Western white pine*</td>
<td>P. monticola</td>
<td>50-200</td>
</tr>
<tr>
<td>Pacific ponderosa pine*</td>
<td>P. ponderosa var. ponderosa</td>
<td>1-47 [1]</td>
</tr>
<tr>
<td>Interior ponderosa pine*</td>
<td>P. p. var. scopulorum</td>
<td>2-30 [1,23,24]</td>
</tr>
<tr>
<td>Arizona pine</td>
<td>P. p. var. arizonica</td>
<td>2-15 [23,25,26]</td>
</tr>
<tr>
<td>Galleta-threeawn shrubsteppe</td>
<td>Pleuraphis jamesii-Aristida purpurea</td>
<td>&lt; 35 to &lt; 100  [2]</td>
</tr>
<tr>
<td>Quaking aspen (west of the Great Plains)</td>
<td>Populus tremuloides</td>
<td>7-120 [1,27,28]</td>
</tr>
<tr>
<td>Mesquite</td>
<td>Prosopis glandulosa</td>
<td>&lt; 35 to &lt; 100  [29,2]</td>
</tr>
<tr>
<td>Mesquite-buffalo grass</td>
<td>P. glandulosa-Buchloe dactyloides</td>
<td>&lt; 35</td>
</tr>
<tr>
<td>Texas savanna</td>
<td>P. glandulosa var. glandulosa</td>
<td>&lt; 10 [2]</td>
</tr>
<tr>
<td>Mountain grasslands</td>
<td>Pseudoroegneria spicata</td>
<td>3-40 (10**) [16,1]</td>
</tr>
<tr>
<td>Rocky Mountain Douglas-fir*</td>
<td>Pseudotsuga menziesii var. glauca</td>
<td>25-100 [1,8,21]</td>
</tr>
<tr>
<td>California mixed evergreen</td>
<td>P. m. var. m.-Lithocarpus densiflorus-Arbutus menziesii</td>
<td>&lt; 35</td>
</tr>
<tr>
<td>California oakwoods</td>
<td>Quercus spp.</td>
<td>&lt; 35 [1]</td>
</tr>
<tr>
<td>Oak-juniper woodland (Southwest)</td>
<td>Quercus-Juniperus spp.</td>
<td>&lt; 35 to &lt; 200  [2]</td>
</tr>
<tr>
<td>Coast live oak</td>
<td>Q. agrífolia</td>
<td>&lt; 35 to 200</td>
</tr>
<tr>
<td>Species</td>
<td>Common Name</td>
<td>Return Interval</td>
</tr>
<tr>
<td>----------------------------------------------</td>
<td>----------------------------------</td>
<td>-----------------------</td>
</tr>
<tr>
<td><em>Q. chrysolepis</em></td>
<td>canyon live oak</td>
<td>&lt; 35 to 200</td>
</tr>
<tr>
<td><em>Q. douglasii-Pinus sabiniana</em></td>
<td>blue oak-foothills pine</td>
<td>&lt; 35</td>
</tr>
<tr>
<td><em>Q. garryana</em></td>
<td>Oregon white oak</td>
<td>&lt; 35 [1]</td>
</tr>
<tr>
<td><em>Q. kelloggi</em></td>
<td>California black oak</td>
<td>5-30 [2]</td>
</tr>
<tr>
<td><em>Q. macrocarpa/Andropogon gerardii-Schizachyrium scoparium</em></td>
<td>oak savanna</td>
<td>2-14 [2,30]</td>
</tr>
<tr>
<td><em>Q. wislizenii</em></td>
<td>interior live oak</td>
<td>&lt; 35 [1]</td>
</tr>
<tr>
<td><em>S. scoparium-Nassella leucotricha</em></td>
<td>blackland prairie</td>
<td>&lt; 10</td>
</tr>
<tr>
<td><em>S. scoparium-Bouteloua spp.</em></td>
<td>little bluestem-grama prairie</td>
<td>&lt; 35 [2]</td>
</tr>
<tr>
<td><em>Thuja plicata-Tsuga heterophylla</em></td>
<td>western redcedar-western hemlock</td>
<td>&gt; 200 [1]</td>
</tr>
<tr>
<td><em>Ulmus-Fraxinus-Populus spp.</em></td>
<td>elm-ash-cottonwood</td>
<td>&lt; 35 to 200 [31,30]</td>
</tr>
</tbody>
</table>

*fire return interval varies widely; trends in variation are noted in the species summary
**mean


scar data from a ponderosa pine ecosystem in the central Rocky Mountains,
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cooperation with: Rocky Mountain Region, National Park Service, US Department of
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James K. and Smith, Jane Kapler, eds. Wildland fire in ecosystems: effects of fire on
Department of Agriculture Forest Service, Ogden, UT.  pp. 35-51.

POST-FIRE REGENERATION STRATEGY

Ground residual colonizer (on-site, initial community), initial off-site colonizer (off-site,
initial community) and secondary colonizer (on-site or off-site seed sources).

IMMEDIATE FIRE EFFECT ON PLANT

Live cheatgrass plants are susceptible to heat kill, as with a flame thrower or handled
propane torch, though they are difficult to burn when green. When cheatgrass plants
are dry enough to burn, they are already dead, and have already set seed. Fire will
then reduce cheatgrass plants to ash.

Cheatgrass seeds are also susceptible to heat kill, but can survive fires of low-
severity if the entire litter layer is not consumed or if seeds are buried deeply enough
to be insulated from the heat (Young 1991). The amount of litter or ash left on a site
is a good indicator of the amount of cheatgrass seed surviving on that site (Young et
al. 1976). Low density of cheatgrass immediately following fire (Young and Evans 1978) indicates either low numbers of cheatgrass seed in the seed bank, or poor survival of seeds during fire (Keeley et al. 1981).

DISCUSSION AND QUALIFICATION OF FIRE EFFECT

The effects of fire on cheatgrass plants and seeds vary with timing and severity of fire and the composition and density of the pre-fire plant community. If fire occurs when seed remains in panicles above ground, most seeds will be killed and cheatgrass density will decline immediately following fire (Brooks 2002, Brooks and Pyke 2001, Pyke 1994). The chances of seed surviving fire are enhanced once they have dispersed onto or beneath the soil surface (Brooks 2002, Daubenmire 1968). In sagebrush communities, most of the litter and cheatgrass seeds are found under the canopies of sagebrush plants (Young and Evans 1975). The woody biomass of the shrub, plus litter accumulations, provide sufficient fuel to elevate temperatures high enough for a long enough period to consume cheatgrass seeds on these microsites. Some cheatgrass seeds in the interspace zones are also consumed by fire, but many survive even though the cheatgrass herbage is completely consumed (Young 1991). Fire from herbaceous fuel alone is not usually hot enough to consume cheatgrass seeds (Evans and Young 1987). Although fires in pure cheatgrass stands, without woody fuel, are less severe, cheatgrass seed banks can be substantially reduced after fire (Young et al. 1976). For example, after a fire in a community dominated by cheatgrass, tumblemustard, and Russian-thistle in Utah, post-fire density of cheatgrass seeds in the seed bank was < 3% of that on unburned plots. Densities of cheatgrass seeds were higher on a low-severity burn compared with a high-severity burn. Nonetheless, the seed bank recovered to preburn levels after 1 growing season, even on the more severely burned site (Humphrey and Schupp 2001).

PLANT RESPONSE TO FIRE

Cheatgrass establishes from soil-stored and transported seed after fire (e.g. Evans and Young 1987, Hassan and West 1986, Hulbert 1955, Humphrey and Schupp 2001, Young et al. 1969, Young et al. 1972). On preferred sites where cheatgrass thrives in the Intermountain West, cheatgrass can establish or maintain dominance in the post-fire environment (e.g. Clements and Young 1996, Conrad and Poulton 1966, McArthur et al. 1990), sometimes maintaining dominance for decades (e.g. Billings 1994), even after artificial seeding with competitive plants (e.g. Bethlenfalvay and Dakessian 1984, Goodrich and Rooks 1999, Ott et al. 2001).

Cheatgrass may also invade recently burned sites where it does not usually dominate or did not previously occur (e.g. Crawford et al. 2001, Johnson and Strang 1983, Keeley 2001, Lyon 1971) if there is an available seed source. For example, pinyon-juniper woodlands with large, continuous tree canopies limit herbaceous understory and facilitate severe summer fires that promote invasion of nonnative species such as cheatgrass and red brome (West et al. 1998). Koniak and Everett (1982) observed that most of the seed bank in a mature singleleaf pinyon stand in California consisted of annuals, many of which were not present in the community as
mature plants. Fire can promote germination of these dormant seeds by consuming litter containing allelopathic compounds (Ott et al. 2001), and/or altering the nutrient, water, microbial, temperature, and light regimes of the seedbed (Blank, Abraham, and Young 1994, Blank, Allen, and Young 1994a, Blank, Allen, and Young 1994b, Koniak 1985, Ott et al. 2001).

There are some examples in the literature reporting decreased density of cheatgrass in the first post-fire year (e.g. Akinsoji 1988, Poreda and Wullstein 1994, White and Currie 1983). Others report increased cover of cheatgrass the first post-fire year in ponderosa pine (Arno 1996, Merrill et al. 1980), sagebrush (Akinsoji 1988), shadscale (Groves and Steenhof 1988), bluebunch wheatgrass (Johnson 1998, Stucker and Peek 1984), and cheatgrass communities (Christensen 1964). These studies provide no additional information on plant community changes in subsequent years.

More common are reports that cheatgrass density decreases the first post-fire year and approximately equals preburn density by the second or third post-fire year in sagebrush (Daubenmire 1968, Daubenmire 1970, Hull and Pechanec 1947, Pechanec and Hull 1945, Sapsis 1990, Young and Evans 1978), desert shrub (Callison et al. 1985), cheatgrass (Daubenmire 1975), and antelope bitterbrush/cheatgrass communities (Bunting et al. 1994). This is because many cheatgrass seeds are killed by fire (Hassan and West 1986, Humphrey and Schupp 2001, Sapsis 1990): A majority of cheatgrass seeds are found under shrub canopies which tend to be the microsites that experience the greatest fire severity (Hassan and West 1986, Young et al. 1976). In a northern Nevada study, fire reduced cheatgrass seed density approximately 96% to 99%, from 5000 to 8000 seeds per square metre to 20 to 300 seeds per square metre (Young et al. 1976). Thus the number and density of cheatgrass plants are reduced the first post-fire year. These plants, however, respond to the released environmental potential resulting from the reduction in plant density and increased water and nutrient availability, and can become large plants with many tillers and high seed production (Brooks 2002, Daubenmire 1970, Humes 1960, Merrill et al. 1980, Young and Evans 1978).

Young and Evans (1978) recorded 10 cheatgrass plants per square metre 1 year after fire, and 10 000 plants per square metre 3 years after fire. On an unburned control plot, the maximum number of seeds on a cheatgrass plant was 250. On burned plots, the lowest seed production per plant 1 year after fire was 960. While plant and seed bank density may decrease the first post-fire year, biomass and seed production may equal or exceed that of the pre-fire population, resulting in increased plant and seed bank density by the second or third post-fire year (Hassan and West 1986, Humphrey and Schupp 2001, Ott et al. 2001, Young 1991, Young and Evans 1978). The increase to peak population density of cheatgrass after fire may require several years on some sites (Keeley et al. 1981). These increases in cheatgrass plant and seed bank density can prevent the establishment of natives and predispose the vegetation to recurring wildfires (Young and Evans 1978).
A few studies found that cheatgrass increased in the early post-fire years and then decreased over time. These include a wildfire in a northeastern California antelope bitterbrush/bottlebrush squirreltail community (Countryman and Cornelius 1957); fires on sites dominated by some combination of mountain big sagebrush, antelope bitterbrush, Saskatoon serviceberry (*Amelanchier alnifolia*), Wyoming big sagebrush, and true mountain-mahogany (*Cercocarpus montanus*) in south-central Wyoming (Cook *et al.* 1994); and a wildfire under ponderosa pine in Idaho (Merrill *et al.* 1980). A study of several burns in Utah juniper-singleleaf pinyon communities in west-central Utah found that cheatgrass cover varied from 12.6% in a 3-year-old burn to 0.9% in the oldest stands (85-90 years without fire). On these sites, cheatgrass was one of the initial dominant annuals, reaching maximum density in the first 3 to 4 years (Barney 1972). Cheatgrass declined in cover the first 22 years after fire, then leveled off and stayed about the same for the remainder of the invasion sequence (Barney and Frischknecht 1974). Similarly, in California chamise (*Adenostoma fasciculatum*) chaparral sites, cheatgrass was most abundant 3 to 5 years after fire, and its abundance tapered off as brush cover closed. Prior to burning, cheatgrass and other annual grasses were found only along trails, firebreaks, and openings (Horton and Kraebel 1955).

Examples where fire seems to have had little or no effect on cheatgrass populations include several sagebrush/bunchgrass sites in the northern Great Basin, where cheatgrass populations were unchanged by prescribed fire treatments over 3 years of the study (Bushey 1987). In a mountain meadow bordering Jeffrey pine in the Sierra Nevada, cheatgrass was present in both burned and unburned dry meadow plots but showed no apparent response to prescribed fires (Boyd *et al.* 1993). In a mountain big sagebrush-antelope bitterbrush community with a healthy understory of perennial bunchgrasses in Oregon that was subjected to fall and spring prescribed fire, there was no increase of cheatgrass after fire. Perennial bunchgrasses increased to > 50% of total vegetation cover 1 to 2 years after the fire (Pyle and Crawford 1996).

**DISCUSSION AND QUALIFICATION OF PLANT RESPONSE**

Cheatgrass response to fire depends on plant community and seed bank composition, density, and spatial distribution; season of burning; fire severity, frequency and patchiness; scale of consideration; post-fire management; and climatic conditions. Generalizations are difficult because each combination of climate, vegetation, and soil must be considered separately (Bunting *et al.* 1987, Kauffman *et al.* 1997, Mueggler 1976), as well as considerations of environmental differences both at the time of burning and during subsequent plant reestablishment (Tausch *et al.* 1995).

Timing of fire: If burned during a crucial time during seed ripening, fire can greatly reduce the density of the succeeding cheatgrass stand (Mueggler 1976); however, post-fire seed production may equal or exceed that of the pre-fire population, resulting in increased density the following year (Hassan and West 1986, Humphrey...
and Schupp 2001, Ott *et al.* 2001, Young 1991, Young and Evans 1978). Timing of fire is important also because of variable damage to potential competitors in the native community. For example, cool-season perennial grasses such as bluebunch wheatgrass and western wheatgrass may be less damaged by late-summer wildfires than by fires earlier in the growing season (Ott *et al.* 2001).

Fire size and frequency: Nonnative, invasive grasses generally benefit from fire and promote recurrent fire. Fire kills biologic soil crusts (Belnap *et al.* 2001, Johansen *et al.* 1982, Ott 2001), thereby allowing more germination sites for cheatgrass for several years or even decades, as crusts are slow to recover (Belnap *et al.* 2001, Callison *et al.* 1985). Recurrent fires also tend to enhance cheatgrass dominance because native species cannot usually persist under a regime of frequent fires. Native plant assemblages are thus converted to nonnative annual grasslands. Frequency and size of fires is then further increased (Brooks and Pyke 2001, Pechanec *et al.* 1954, Young 2000).

Explanations of why individual cheatgrass plants thrive in the post-fire environment have been explored, but remain unclear. Blank and Young (1998) found that after exposing cheatgrass seeds to smoke of burning big sagebrush, the rates of new leaf production and leaf elongation in cheatgrass seedlings were significantly (*p* < 0.05) greater than cheatgrass seedlings from untreated seed. Blank and others (1994c) also found that cheatgrass had significantly (*p* < 0.05) greater aboveground mass when grown in post-wildfire soil than plants grown in unburned soil. Individual cheatgrass plants may thrive in the post-fire environment due to temporary increases in the availability of soil nutrients, especially inorganic nitrogen, after fire. Soil disturbance and fire are known to accelerate mineralization of nitrogen (Blank *et al.* 1996, Covington *et al.* 1991, Covington and Sackett 1992, Stark and Hart 1999, Stubbs 2001). Cheatgrass displays flexibility in effectively using both patches of nitrogen and early pulses of nitrogen which may contribute to its effectiveness in competing with perennials (Duke and Caldwell 2001). It has been suggested that changes in nitrogen availability, and in the abundance of NO3- relative to NH4+, play important roles in allowing cheatgrass stands to persist in much of the western United States (Stark and Hart 1999, Young *et al.* 1995). Studies by Blank and others (1994, 1996) further explore changes in soil chemistry after fire under sagebrush and cheatgrass. These differential changes may help to explain post-fire succession of cheatgrass in different microsites.

Microsite effects: Cheatgrass’ post-fire response varies between microsites and appears to be related to microsite fuel gradients and subsequent differences in fire severity between microsites. Following wildfires in sagebrush steppe, cheatgrass first dominated the interspaces between shrubs, and later occupied shrub subcanopy microsites (Young and Evans 1978, Young *et al.* 1976). A similar post-fire successional pattern was observed in big sagebrush and Colorado pinyon-Utah juniper communities in west-central Utah (Ott *et al.* 2001). Cheatgrass became dominant in the interspaces between burned trees by the second post-fire year. In the subcanopy zones of burned trees, cheatgrass did not become dominant until the third year following fire, and was preceded by exotic annual forbs.
In creosotebush scrub vegetation in the Mojave Desert, annual brome (red brome, cheatgrass, and Chilean chess (*Bromus berterianus*) pre-fire biomass was 3 times higher under the shrub canopy than in drip line microsites. Following experimental fires, annual bromes had poor recovery the first four post-fire years (Brooks 2002). The observed patterns are probably due to higher temperatures and longer duration of heat exposure under canopies (i.e. higher-severity fire), where pre-fire fuel loads tend to be higher. The result is greater consumption of litter and seeds in subcanopy zones, leaving an unfavorable and unlikely site for cheatgrass germination (Brooks 2002, 329). Results presented by Blank and others (Blank, Abraham, and Young 1994, Blank, Allen, and Young 1994b) suggest that because of cooler, briefer fires in cheatgrass microsites compared with sagebrush microsites, post-fire soil qualities differ in concentrations of organic and inorganic anions, which, in turn, may influence seed germination, plant establishment, and mineral nutrition (i.e. post-fire succession). In some cases, recolonization by cheatgrass after fire may be delayed by drought or in areas where interspace cheatgrass densities are low and seeds must disperse from adjacent unburned areas (Brooks 2002). Additional research is needed to understand how these changes affect post-fire succession in specific plant communities.

Effects of fire severity on post-fire dominance of cheatgrass at the site scale were observed in ponderosa pine forests in Arizona that burned in early summer. Two years after fire, cheatgrass had < 0.5% cover in unburned sites, ~19% cover in high-severity burn sites, and ~ 3% in moderate-severity sites (Crawford et al. 2001).

Plant community effects: The exact nature of secondary succession following fire in sagebrush/bunchgrass ecosystems is not clearly understood due to the highly variable response of subspecific populations of big sagebrush (Bunting et al. 1987, Kauffman et al. 1997). Mountain big sagebrush is the most mesic of the big sagebrush subspecies, usually occurs at the highest elevations, is probably the best adapted to fire, and tends to have the most productive herbaceous component of the big sagebrush communities. It is well adapted to post-fire establishment via seeds that are stimulated to germinate by light and heat. It also grows rapidly and reaches reproductive maturity within 3 to 5 years, and populations may return to preburn levels within 15 to 20 years, or 30 years after a severe fire (Bunting et al. 1987). Young and Evans (Young and Evans 1978) described big sagebrush communities at higher elevations, under more mesic conditions, and/or long distances from livestock water, as those where cheatgrass was still absent in 1978. Mountain big sagebrush is probably the most likely to resist invasion by cheatgrass, but cheatgrass may initially dominate the post-fire community, especially if post-fire cover of perennial grasses is low (Barrington et al. 1988). Similarly, cheatgrass is often found with threetip sagebrush, but seldom becomes a problem following fire or other types of disturbance (Bunting et al. 1987).
Another subspecies of big sagebrush with limited distribution, xeric big sagebrush (Artemisia tridentata ssp. xericensis), sometimes referred to as an ecotype of mountain big sagebrush, is found primarily in western Idaho and eastern Oregon and is restricted to a zone where the annual precipitation exceeds 300 mm, elevation is less than 1360 m, and the summers are relatively warm. Many of these communities are on relatively steep slopes, have a high potential for human and lightning-caused fires, and burn frequently. These frequently burned areas are often dominated by cheatgrass and medusahead (Bunting et al. 1987). Similarly, at Craters of the Moon National Monument, Idaho, cheatgrass is most likely to occur in mountain big sagebrush communities on xeric sites, such as those growing at lower elevations or on cinder-derived soils, where mountain big sagebrush is less competitive (Barrington et al. 1988).

The majority of the historic area of basin big sagebrush is now under intensive agricultural cultivation, so these communities are now restricted primarily to field edges, swales, and along water drainage ways in areas dominated by other sagebrush species. Sites formerly dominated by basin big sagebrush are susceptible to invasion by cheatgrass, and as such, vast areas of this type have been converted to annual grasslands. Basin big sagebrush does not sprout, and repeated fires have eliminated it from many of the remaining sites (Bunting et al. 1987, Countryman and Cornelius 1957). In eastern Oregon, basin big sagebrush was completely eliminated by fall burning, while spring burning resulted in an 84% decrease in density. Burning in both seasons reduced cheatgrass density at post-fire year 1 compared to pre-fire density (Sapsis 1990).

Wyoming big sagebrush is the most arid of the big sagebrush types, occurring on sites with annual precipitation of less than 178 mm in some places. Wyoming big sagebrush and its associated perennial grasses are not well adapted to fire, as they evolved in low-productivity communities with few herbaceous species and infrequent fires. Wyoming big sagebrush establishes readily from seed after fire, but repeated fires rapidly diminish the on-site seed source and reduce opportunities for establishment. Cheatgrass predominates in early successional stands of the Wyoming big sagebrush series in western Idaho, northern Nevada, and Oregon, thus increasing the likelihood of fire and subsequent dominance of cheatgrass. Once a site is dominated by cheatgrass, the risk of wildfire increases and the possibility of succession to perennial grasses or shrubs by natural regeneration greatly decreases. Many Wyoming big sagebrush sites have been burned repeatedly by wildfire, resulting in a conversion to nearly pure stands of cheatgrass (Bunting et al. 1987).

Cheatgrass dominance may be avoided on sites that have sufficient cover of native perennials, proper management of livestock, and favorable climatic conditions for post-fire recovery (Barney and Frischknecht 1974). Three years after a severe wildfire on an ungrazed foothill mountain grassland in western Montana, cheatgrass cover was lower in burned patches than in unburned patches, and cheatgrass showed no indication of invading the burn. At that time Idaho fescue and bluebunch wheatgrass cover were similar to unburned levels, and rough fescue cover was slightly below unburned levels (Antos et al. 1983).
After a mid-summer, lightning-caused wildfire on a good condition sagebrush-grass site (Wyoming big sagebrush, black sagebrush, and bluebunch wheatgrass) in central Utah, cheatgrass dominated the site the first post-fire year. Livestock were kept off the site, and with favorable precipitation, perennial bunchgrasses returned to nearly their preburn cover the second year. Although cheatgrass had highest cover of grasses, the authors conclude that sagebrush-grass sites in good condition may be improved for cattle production with a few years of livestock exclusion following wildfire (West and Hassan 1985). On a similar site, perennial grasses came to dominate plant cover over time, especially in ungrazed plots. Cheatgrass became locally almost absent during a 3-year intense drought, so the threshold to an annual-dominated site was not crossed (West and Yorks 2002). Similarly, a mesic site dominated by Colorado pinyon, Gambel oak, true mountain-mahogany, and mountain snowberry (*Symphoricarpus oreophilus*) was seeded with nonnative grasses, including crested wheatgrass and smooth brome (*Bromus inermis*), after fire. Seeded sites had lower cover of cheatgrass than drier sites that were not seeded (Ott *et al.* 2001). A tallgrass prairie community in Oklahoma was burned under prescription in mid-April for 3 consecutive years. At post-fire year 3, cheatgrass cover was significantly (p < 0.05) lower on burned sites than on unburned sites, both with and without grazing (Collins 1987).

**FIRE MANAGEMENT CONSIDERATIONS**

As a management tool, fire can be used to either kill unwanted species or to simulate historic fire regimes and promote desired species. Historic fire regimes did not occur in the presence of many invasive plants that are currently widespread, and the use of fire may not be a feasible or appropriate management action if fire-tolerant invasive plants are present. For example, while fire may be an important natural component of the Great Basin ecosystem, its reintroduction by land mangers is complicated by the presence of invasive plants such as cheatgrass (Brooks and Pyke 2001). Fire management should be conducted in ways that prevent establishment of invasive species (Harrod and Reichard 2001), and the management of fire and invasive plants must be closely integrated for each to be managed effectively (Brooks and Pyke 2001).

Rasmussen (1994) presents considerations (e.g. species composition, fuel load, fuel continuity, and weather) to be addressed when using prescribed fire in sagebrush steppes, and general prescriptions that could be used. When precipitation is below 300 mm, caution should be used to ensure desired plant response. If the objective is to maintain the perennial herbaceous vegetation, prescribed burning is most effective when used before sagebrush dominates the site and effectively excludes perennial herbaceous plants. Such timing reduces the need for seeding following a burn. If the objective is to maintain the sagebrush, prescribed burning has very limited applicability (Rasmussen 1994).
Fire as a control agent for cheatgrass: In sagebrush ecosystems, prescribed burning alone will generally decrease cheatgrass cover only in the short term, so in areas where cheatgrass dominates the understory, fire may best be used as a seedbed preparation technique prior to seeding desirable species (Evans 1988, Evans and Young 1987, Rasmussen 1994, Stewart and Hull 1949). Burning of mixed shrub-cheatgrass stands generates enough heat to kill most cheatgrass seeds and may offer a one-season window for the establishment of perennial seedlings (Evans and Young 1977, Young and Evans 1978, Young 2000). The abundance of viable seeds of cheatgrass after a burn can be judged by examining seeds in the ash. Even if the lemma and palea are charred, if entire caryopses can be identified some seed will be viable and germinate (Young et al. 1976).

The period of reduction of cheatgrass density (1-2 years) is not usually long enough to allow for the establishment of perennial seedlings (Bunting 1998). Cheatgrass plants that do establish the first post-fire year tend to produce so much seed per plant that total post-fire cheatgrass seed production for a site may actually increase by a factor of 100 over preburn production (Mosely et al. 1999, Young 1983). Unless desirable species establish and outcompete cheatgrass, density of cheatgrass plants may exceed preburn levels within 1 to 5 years (Mosely et al. 1999, Wright et al. 1979). On range sites in Washington, seeded grasses were successfully established on sites where cheatgrass density was reduced to less than 90 seedlings/m$^2$ with summer burning and to less than 40/m$^2$ with burning combined with fall spraying of herbicide (Haferkamp et al. 1987).

If fire is used as a pretreatment to seeding in sagebrush communities depleted of perennial herbs, and levels of annual grasses are low at the time of the fire, perennial seedlings may establish before the annuals dominate the site if perennials are seeded the first year after fire (Bunting 1998). Seeding burned areas immediately after fire is likely to reduce the “influence” of cheatgrass but is not likely to exclude it. A closed-canopy Colorado pinyon-Utah juniper site in the Green River corridor in Utah burned in 1976. Response of native plants was low (because of sparse seed bank), and within a decade cheatgrass and musk thistle (Carduus nutans) dominated the site. The site was then burned under prescription in late June 1990, when cheatgrass seed was mature but not yet shattered. The site was aerially seeded in fall 1990 with aggressive, introduced grasses including crested wheatgrass, intermediate wheatgrass (Thinopyrum intermedium), orchardgrass (Dactylis glomerata), and smooth brome (Bromus inermis). Some of the burned area was not seeded, and cheatgrass established in unseeded areas. There were fewer and smaller cheatgrass plants in the seeded area (Goodrich and Rooks 1999). However, cover of cheatgrass was slightly higher in seeded versus unseeded plots following an August wildfire in sagebrush steppe in Idaho (Ratzlaff and Anderson 1995).

Late spring or early summer burns, before cheatgrass seed matures, may effectively control cheatgrass (Astroth and Frischknecht 1984, Mueggler 1976, Rasmussen 1994); however, burning before the seed is ripe is difficult because the plants are still green (Astroth and Frischknecht 1984). This timing is also a period of high sensitivity to fire damage for cool-season perennial grasses. In areas where native warm-
season grasses are desired, a prescribed fire that kills cheatgrass seedlings and reduces the surface seed bank may be effective (Young 2000). A site in Oregon that was dominated by cheatgrass and annual forbs was burned under prescription in July. The density of cheatgrass decreased and bottlebrush squirreltail increased in burned areas (Young and Miller 1985). Prescribed burning prior to herbicide application may increase the effectiveness and/or reduce the application rate required for effective control of cheatgrass (Shaw and Monsen 2000). Preliminary results from a site in Oregon indicate that glyphosate treatment or mowing 1 year following summer prescribed burning were equally effective at reducing medusahead and cheatgrass cover (Ponzetti 1997).

In all cases where invasive species are targeted for control, the potential for other invasive species to fill their void must be considered (Brooks and Pyke 2001, Harris 1990, Roberts 1999).

Fire as a control agent for shrubs and trees: Prescribed burning to reduce cover of shrubs and trees has been practiced for decades (e.g. Astroth and Frischknecht 1984, Blaisdell 1953, Blaisdell et al. 1982, Countryman and Cornelius 1957, Miller and Tausch 2001, Pechanec et al. 1954), and was sometimes successful at enhancing desirable species without increases in cheatgrass (Blaisdell 1953, Countryman and Cornelius 1957). The resulting plant community is dependent on several variables, however, including the composition of the plant community and seed bank before burning, the conditions of the fire, post-fire management, and climatic conditions.

The cheatgrass problem on rangeland dominated by sagebrush was probably exacerbated by efforts, beginning in the 1930s, to control sagebrush and increase grazing capacity. A report by Pechanec and others (1954) recommends against burning range with little understory of perennial grasses unless it is to be reseeded the first fall following the burn. They note that where cheatgrass is present, burning the range is likely to increase cheatgrass and damage desirable perennial species. Proper grazing management following burning is essential in maintaining desirable species.

Blaisdell and others (1982) suggest that each situation be carefully examined and evaluated before burning can be prescribed as a plant control measure, and emphasize that areas with a poor stand of desirable perennials prior to burning will probably require post-fire seeding to provide satisfactory forage production and delay return of sagebrush or other unwanted species such as cheatgrass, halogeton, and medusahead (Blaisdell et al. 1982). Burning in Wyoming big sagebrush will remove brush, but it will not provide more perennial grass where cheatgrass has become dominant (Bunting et al. 1987). They suggest prescribed burning in areas with 10 to 15% cover of sagebrush and desirable plants present “in a density that will allow a favorable postfire response.” Young and Evans (1975, 1978) determined that 21.5 perennial grass plants per square metre is the minimum necessary to preempt invasion by nonnative annual species and/or shrub seedlings.
A review by Bunting (1998) suggests many sites are difficult to burn because herbaceous productivity is inherently low. Sites with less than 600 kg/ha of fine fuels will be difficult to burn. Even if sites with low amounts of herbaceous plant cover are burned, it may take many years for the desirable perennials to establish, which leaves the site open to erosion and invasion by nonnative plants. If the fine fuel load is sufficient but it is composed of annuals, establishment of perennials may still be prevented, and fire return intervals may decrease. Thus, the site may be dominated by annuals indefinitely. Then the management concern is for fire prevention as a means to increase the fire-free interval until desirable perennial vegetation can become established (Bunting 1998).

Post-fire colonization potential: Cheatgrass can invade recently burned sites from offsite seed sources (Arno 1996, Floyd-Hanna and Hanna 1999, Fraas et al. 1992, Rice and Randall 1999), or may establish from seeds in the soil seed bank (e.g. Evans 1988), even if plants are absent from the site at the time of the fire (e.g. Koniak and Everett 1982). Predicting a site’s susceptibility to invasion may be difficult. Management techniques to help reduce post-fire cheatgrass invasion may include elimination of nonnative seed sources from roadsides and other disturbance areas adjacent to burn sites, and increasing size of burns to increase the distance from seed sources. Increasing size of burns does, however, also increase distance from native seed sources (Keeley 2001).

Excessive or poorly timed grazing after burning can also increase cheatgrass dominance (Pechanec et al. 1954). The optimum amount of grazing rest and deferment that is needed following fire in sagebrush steppe and pinyon-juniper vegetation continues to be controversial, but varies with vegetal composition, site potential, objectives of the burn, and environmental conditions following fire (Bunting 1998). It has been suggested that grazing be deferred on seeded lands for a minimum of 2 years with nonnative seeding, and a minimum of 5 to 8 years with native seedings (Evans and Young 1978, Knick 1999, Young et al. 1987).

Cheatgrass fires: The majority of grassland fires in the Intermountain West are small (< 4 ha) and represent < 1% of total acreage burned. Large fires (> 2008 ha) are infrequent, but represent > 70% of the acreage burned (Knapp 1998). These fires are of particular concern for rehabilitation efforts (Roberts 1999), and predicting their occurrence and behavior would be beneficial to land managers in assigning resource allocation prior to the fire season (Knapp 1998). Knapp (1995, 1998) suggests that these large fires have distinct spatial patterns, and their occurrence can be predicted based on antecedent moisture conditions. Summer moisture conditions in the year preceding that of large fire years tend to be near-normal or wetter. Conversely, < 20% of all the large fires occur when the previous summer’s moisture conditions were below normal (Knapp 1995, Knapp 1998). Other researchers have suggested this relationship between precipitation in the preceding winter months and large fires during the following summers (Billings 1994, Young et al. 1987). Moisture conditions in the summer in which the large fires occur appear to have less influence on the likelihood of those fires, suggesting that fuel moisture conditions are secondary to fine-fuel amounts for promoting large fires on rangelands (Knapp 1995, Knapp 1998).
Knapp (1998) examined the spatial and temporal occurrence of large grassland fires in the Intermountain West for the period 1980 through 1995. He found that large fires tended to occur in areas dominated by annual grass cover (> 50% herbaceous cover), at lower elevations than smaller fires (1341 m average), and during a shorter fire season (July and August). More than half of all large fires were on relatively flat terrain (i.e. basins and foothills) that was historically more susceptible to invasion by nonnatives (i.e. ranchlands) and concurrently dominated by annual grasses. Because the annual grass/wildfire cycle is driven by positive feedback, these areas are also likely to experience large fires in the future. Large fires also occurred in more physiographically discrete regions than did smaller fires, with 8 specific regions representing approximately 60% of the overall Intermountain area, and representing the optimal combination of fuel amounts and fuel continuity for large fires.

Cheatgrass fuels: In the absence of grazing, grass biomass during the fire season may represent 2 years of fuel accumulation, which appears to be optimal for grassland fires (Knapp 1998). Abundant, continuous cover of cheatgrass can lead to rapid spread of wildfires so that under conditions of high temperatures, low humidity, and wind, the fires are very difficult to suppress (Young 1991).

Brooks (1999) compared the roles of nonnative annual grasses and other annual plants in facilitating the spread of fires in the Mojave Desert. Landscapes dominated by nonnative annual grasses, especially annual bromes (Bromus spp.), are more flammable than those dominated by native forbs. Possible explanations for this include higher surface-to-volume ratio of grasses compared to forbs; more continuous vegetative cover; and the ability of alien annual grasses to remain rooted and upright longer than native forbs, allowing them to persist as flammable fuels into the summer when the threat of fire is highest Brooks (1999). Thick layers of annual plant litter accumulate, and litter decomposes especially slowly in desert regions (Brooks 1999, Young et al. 1987). Accumulations of litter led to particularly hot temperatures, long flame residence times, and continuous burn patterns in experimental fires in the Mojave Desert (Brooks 1999).

Cheatgrass provides a flammable link between open grasslands and forests. It cures early in the fire season and ignites readily during dry periods because of its finely divided stems and pedicels, and it responds readily to changes in atmospheric moisture because of its fine structure. Moisture content is the single most important factor influencing cheatgrass flammability, and varies with plant phenology and color change as follows (Mutch 1967):

<table>
<thead>
<tr>
<th>Plant color</th>
<th>Moisture content (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>green</td>
<td>&gt; 100</td>
</tr>
<tr>
<td>purple</td>
<td>30-100</td>
</tr>
<tr>
<td>straw</td>
<td>&lt; 30</td>
</tr>
</tbody>
</table>
Since there is considerable variation in plant coloration in a stand, close inspection is necessary to determine the predominant coloration. Cheatgrass is not readily ignitable until it reaches the straw-colored stage. The time required for the moisture content to drop from 100% to 30% ranged from 8 days on a northern exposure in western Montana, to 23 days on a southern exposure in different years, with an average of 14 days. The onset of purple coloring forewarns of hazardous fire conditions within about 2 weeks (Mutch 1967).

Cheatgrass ignites and burns easily when dry, regardless of quantity, and can support rapid rate of fire spread (Brown 1969, Klemmedson and Smith 1964). When cured and at 9% moisture content, each gram of cheatgrass material is capable of producing 3900 calories to contribute to the spread of fire (Richards 1940). Flammability of cheatgrass fuels depends primarily on moisture content, weight, and porosity. Anderson (1990b) provides figures for equilibrium moisture content of cheatgrass litter under different conditions of relative humidity and temperature. Moisture diffusivity and response time in cheatgrass as fuel are given for different stand densities by Anderson (1990a). When the moisture content reaches low levels (5 to 10% dry weight), variations in flammability are probably primarily caused by fuel weight and bulk density. Estimation of bulk density (weight per unit of volume of the fuel bed) is a practical aid in assessing the flammability of cheatgrass. Average bulk density for cheatgrass in western Montana was 0.00032 g/cm$^3$. More details are given by Brown (1970a). Ratio of surface area to volume for several fine fuels is explored by Brown (1970b), and physical fuel properties of a cheatgrass fuel complex are given. Surface area:volume ratio for cheatgrass was 145 cm$^2$/cm$^3$.

Some mineral content and volatilization characteristics of cheatgrass leaves are given below (Philpot 1970):

<table>
<thead>
<tr>
<th>Property</th>
<th>Value</th>
</tr>
</thead>
<tbody>
<tr>
<td>Silica-free ash (%)</td>
<td>1.04</td>
</tr>
<tr>
<td>Total ash (%)</td>
<td>5.27</td>
</tr>
<tr>
<td>Volatilization at 175-350 °C (%)</td>
<td>73</td>
</tr>
<tr>
<td>Maximum volatilization rate (μg °C)</td>
<td>68</td>
</tr>
<tr>
<td>Organic residue at 400 °C (%)</td>
<td>26</td>
</tr>
</tbody>
</table>

A basic procedure for evaluating the grass fuel models of the National Fire Danger Rating System is provided by Sneeuwjagt (1974), with several examples from cheatgrass-dominated sites.

Fuel management/fire prevention: On areas where cheatgrass is abundant, special measures may be necessary to prevent recurrent fires, and thus prevent the elimination of fire-sensitive perennial grasses and forbs (Blaisdell et al. 1982) and other potential adverse impacts. Fire suppression can discourage invasion and spread of cheatgrass (Rice and Randall 1999). Grazing management to reduce fuel loads and greenstripping are 2 methods employed to prevent large recurrent fires in areas dominated by cheatgrass. Additionally, herbicides are being tested for effectiveness in creating fuelbreaks in cheatgrass-dominated range (Synergy Resource Solutions, Inc. 2002).
Cattle grazing can reduce the accumulation of cheatgrass litter and thus lessen the fire hazard on a site (Emmerich et al. 1993, Pellant 1990, Young and Tipton 1990). Grazing cheatgrass in winter can reduce cheatgrass herbage and seeds while protecting the dormant perennial grasses (Emmerich et al. 1993). Davison (1996) provides more detailed information on using livestock grazing to reduce fuel loads and subsequent fire occurrence and severity in cheatgrass-dominated rangelands.

Greenstripping is a method of establishing fuel breaks to impede the flow of wildfires and thereby increase the fire-free interval on a site dominated by cheatgrass. These fuel breaks are 10-120 m wide, and are seeded with fire-resistant vegetation. As of 1994, 6588 hectares of experimental and operational greenstrips had been established in Idaho. The effectiveness of greenstrips, or any fuels modification project, in reducing wildfire spread is enhanced by 3 factors: 1) disrupting fuel continuity (e.g. by replacing cheatgrass with caespitose grasses such as crested wheatgrass, which have large spaces between individual shrubs); 2) reducing fuel accumulations and volatility (e.g. shrub stands are thinned to maintain a minimum distance of 3 m between plants); and 3) increasing the density of plants with high moisture and low volatile oil content, thus reducing both the potential for ignition and rate of fire spread (Pellant 1990, Pellant 1994). Plants used in greenstrips remain green and moist into late summer, making the greenstrip area less flammable for a longer time. Wildfire speed may slow when entering a greenstrip, thus allowing fire suppression crews to extinguish the fire. Some wildfires burn into greenstrips and extinguish (Mosely et al. 1999). Native plants in the Great Basin generally do not meet firebreak criteria (Brooks and Pyke 2001). Crested wheatgrass and forage kochia are effective in retarding wildfire spread, compete well in a weedy environment, and have been the most successful species in greenstrips (MacDonald 1999, Monsen 1994, Mosely et al. 1999, Pellant 1994). Both plants can, however, be invasive and spread into areas where cheatgrass is being managed with prescribed fire (Brooks and Pyke 2001, Monsen 1994).

Revegetation after cheatgrass fires: After wildfires or when planning prescribed burning in areas where cheatgrass is present, managers must decide whether the burned area should be seeded or whether sufficient perennial grasses are present to revegetate a site and successfully compete with cheatgrass (Roberts 1999, Young et al. 1987). Seeding may not be necessary or desirable if native plant species are able to recover after fire (Ott 2001, Ratzlaff and Anderson 1995). Cheatgrass-dominated communities tend to have extremely sparse perennial seed banks, however, and the cheatgrass seed bank generally recovers by the second post-fire year (Humphrey and Schupp 2001, Young et al. 1987). In Utah, natural revegetation (no seeding) is most effective at higher elevations where sufficient moisture and a diverse population of perennial vegetation exist, especially on north- and east-facing slopes. Below 1820 m and in much of Utah’s arid environment, cheatgrass and other weedy species readily invade and dominate burned areas (MacDonald 1999). Seeding following fire may be needed to prevent cheatgrass dominance in Wyoming big sagebrush and pinyon-juniper communities, but not in mountain big sagebrush communities (Goodrich et al. 1999).
Revegetation of burned areas is desirable to assure forage for livestock and wildlife, and to minimize the potential for erosion and/or invasion by nonnative species. Ideally, wildfire rehabilitation should enhance the recovery of native vegetation through the seeding of native plants adapted to local environmental conditions. Native plants such as basin wildrye (*Leymus cinereus*), bluebunch wheatgrass, western wheatgrass, Indian ricegrass, big sagebrush, and fourwing saltbush (*Atriplex canescens*) have been used in rehabilitation seedings (Ott 2001). Early seral species such as bottlebrush squirreltail may provide managers with native plant materials that can successfully germinate and establish in the presence of invasive annuals (Jones 1998, Young and Miller 1983) and do well after subsequent fire (McArthur et al. 1990). Bottlebrush squirreltail deserves consideration as a post-wildfire revegetation species because in greenhouse experiments, it has substantially greater growth in post-wildfire soil compared with unburned soil, and exhibits relatively higher growth rates in post-wildfire soil compared to cheatgrass (Blank et al. 1994a). Restoration projects using native species mixes to provide a variety of above- and belowground growth forms, and sowing at high densities, may increase establishment of desirable plants while providing adequate competition against invasive plants (Brooks and Pyke 2001).

Monsen (2000) discusses seed, seeding technology, and microenvironmental requirements for the reestablishment of big sagebrush weed-infested sites. Wyoming big sagebrush establishes readily from seed (Bunting et al. 1987). Establishment of bareroot Wyoming big sagebrush seedlings is most successful on fine-textured soils at Hanford in eastern Washington (Durham et al. 2001). Colonization of sagebrush roots by mycorrhizae is much lower in burned sites compared with unburned sites; therefore, burning itself may impede the reestablishment of sagebrush over cheatgrass after fire (Gurr and Wicklow-Howard 1994). Mycorrhizae are reduced by high-severity fires for about 2 years; therefore, establishment of sagebrush may be more successful 2 years after high-severity fires (Lyon 1971). Reducing levels of available nitrogen immediately after fire may increase the rate of establishment by native plants and reduce the dominance of invasive annuals. Sucrose has been used experimentally to reduce nitrogen availability by increasing soil microbial biomass. Such treatments have reduced the growth of invasive plants while enhancing the establishment and composition of late-seral native plants in a semiarid ecosystem (McLendon and Redente 1994). More research is needed to identify cost-effective techniques for reducing available nitrogen and enhancing the success of native plants (Brooks and Pyke 2001). Hardegree and others (1996) discuss the development of technology to characterize seedbed microclimate as it pertains to the restoration and maintenance of native plant communities in sagebrush steppe, and how to use this information to design optimal planting scenarios for establishment of desirable native species and develop mitigating strategies to minimize competition from cheatgrass. Re-inoculation of components of biological soil crusts is also being explored in restoration efforts (Jones et al. 1999).
Federal policy currently encourages the use of native plant materials on public lands; but because the primary objective of wildfire rehabilitation on public lands is not ecological restoration but rather prevention of erosion and invasion by undesirable nonnative species, and because of the limited availability of native seeds, the use of native species is not mandatory for revegetation (Brooks and Pyke 2001, Ott 2001, Roberts 1999). Roberts (1999) summarizes some budgetary, ecological, and managerial concerns with regard to cheatgrass and pinyon-juniper fires. Because of difficulties related to cost, handling, and reliability of native seed supplies in wildfire rehabilitation situations, many managers prefer nonnative plant materials and traditional seeding methods.

Many large areas have been seeded with nonnative, herbaceous forage species (Brooks and Pyke 2001, Ott 2001, Young et al. 1987) including crested wheatgrass, intermediate wheatgrass, tall wheatgrass (Thinopyrum ponticum), Russian wildrye (Psathyrostachys juncea), smooth brome, alfalfa, and yellow sweetclover (Melilotus officinalis). Seeds for these species are readily available and responsive to standard seeding methods; plants establish and grow rapidly, and have wide environmental tolerances. Many cultivars are also drought tolerant, grazing tolerant, and competitive against other, less desirable nonnative species (Boltz et al. 1987, Ott 2001). The most reliable and persistent grass for low-elevation, drought-prone areas of the Intermountain West is crested wheatgrass. It establishes rapidly even under relatively dry conditions and tends to persist for many years (Kay 1960, Monsen 1994, Ott 2001), although some sites seeded to crested wheatgrass return to cheatgrass dominance over time (Bethlenfalvay and Dakessian1984). Grasses that are most competitive against cheatgrass include “Hycrest” crested wheatgrass, “Luna” intermediate wheatgrass, “Bozoisky” Russian wildrye, and smooth brome (Aguirre and Johnson 1991a, Aguirre and Johnson 1991b, Francis and Pyke 1996). The competitive advantage for establishment of crested wheatgrass seedlings is lost if burned areas are not seeded the year of the fire (Evans and Young 1987). Forbs such as alfalfa tend to have low persistence in rehabilitation seedings (Monsen 1994, Ott 2001).

Cheatgrass suppression was best met by aerial seeding followed by chaining or drill seeding after large wildfires on rangeland in Utah. These methods can still result in large amounts of cheatgrass in the interspaces between seeded grasses (MacDonald 1999, Ott 2001). Rehabilitation of cheatgrass burns has been attempted with diskng, plowing, and other mechanical methods to reduce cheatgrass competition prior to seeding, but these methods obliterate any remaining native plants, especially perennial grasses and microbiotic soil crusts. Thus, these methods are not appropriate for wildland settings. As for soil stabilization, cheatgrass establishes more rapidly than seeded grasses and therefore may provide better soil protection in the short term (Ott 2001). Seeded bunchgrasses may not control soil erosion because of the large amount of bare soil in the interspaces between plants (Lesica and DeLuca 1996). While perennial grasses and forbs reduce potential for recurring fire (Monsen 1994, Pellant 1994), perennial seedings are not immune to burning, especially when there is cheatgrass in the interspaces (Ott 2001).
Finally, seeding nonnative species into burned areas should be carefully considered because nonnative species may have negative impacts on native vegetation and wildlife (Harrod and Reichard 2001, Monsen and Shaw 2000). See Cultural control for more information on the relative merits of native versus nonnative plant materials for revegetation on rangelands.

Current goals of making wildfire rehabilitation objectives compatible with other management objectives on public lands may require careful planning of treatments and some modifications of standard practices, such as greater use of native plants (Ott 2001). The identification and use of competitive native perennial plants for aridland rehabilitation has become a priority for managers and researchers (Monsen 1994). In big fire years - such as 1996, when millions of acres burned - the scale of the demand for seed greatly exceeds the supply of native plant seed, especially of local genotypes. The competitive ability of nonnative species and the relatively low cost and high availability of their seed will continue to appeal to those faced with of large-scale burns in cheatgrass-prone areas (Goodrich and Rooks 1999). If managers are able to predict large fires in advance (as per Knapp 1998), perhaps more efforts could be made to have more native seed available for specific sites.

LITERATURE CITED


FIRE ECOLOGY OR ADAPTATIONS

Columbia brome has basal culm buds which may sprout after aerial portions are burned (Halverson 1986, Topik and Hemstrom 1982). If thick tufts form, they may protect the basal buds from fire damage.

POST-FIRE REGENERATION STRATEGY

Tussock graminoid and ground residual colonizer (on-site, initial community).

IMMEDIATE FIRE EFFECT ON PLANT

Columbia brome culms are probably killed by fire.

PLANT RESPONSE TO FIRE

Columbia brome usually decreases (Armour et al. 1984) or is neutral (Simmerman et al. 1991, Vogl and Ryder 1969) in response to fire. However, occasional increases occur (Edgerton 1987, Freedman and Habeck 1985).

DISCUSSION AND QUALIFICATION OF PLANT RESPONSE

Several studies of the effects of logging and burning include information on Columbia brome response.

In a ponderosa pine (Pinus ponderosa) forest in northern Idaho, Columbia brome decreased after fire (Armour et al. 1984).

In seral shrub communities in the cedar-hemlock (Thuja-Tsuga spp.) zone of northern Idaho, Columbia brome was significantly more frequent in unburned stands than in broadcast burned stands. Presence of Columbia brome in stands with different disturbance histories was as follows (Mueggler 1961):

<table>
<thead>
<tr>
<th>Percent Frequency</th>
</tr>
</thead>
<tbody>
<tr>
<td>Closed stand, no disturbance</td>
</tr>
<tr>
<td>Logged, no Burn</td>
</tr>
<tr>
<td>Logged, piled and burned</td>
</tr>
<tr>
<td>Single broadcast burn</td>
</tr>
<tr>
<td>Multiple broadcast burns</td>
</tr>
</tbody>
</table>
Columbia brome was considered neutral with respect to fire in mixed forests of Rocky Mountain Douglas-fir, grand fir, and western redcedar (*Thuja plicata*) on the Priest River Experimental Forest in northern Idaho. Columbia brome was present in plots that had been logged and then given treatments of no fire, a moist fuels underburn in June 1989, or a dry fuels underburn in September 1989. Pretreatment cover estimates were made during the summer before logging began. Posttreatment cover estimates were made for both fire and no fire units in the summer of the year after the fires. Columbia brome was present with the following percent cover (Simmerman *et al.* 1991):

<table>
<thead>
<tr>
<th></th>
<th>No Fire</th>
<th>Moist Fuels</th>
<th>Dry Fuels</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>pre-logging</td>
<td>post-logging</td>
<td>pre-logging</td>
</tr>
<tr>
<td>fire</td>
<td>1.9</td>
<td>0.2</td>
<td>1.7</td>
</tr>
</tbody>
</table>

Vegetation in plots on burned slash piles in the Mission Mountains in northwestern Montana was compared with vegetation in plots adjacent to the slash piles which had not burned. Each logging site had 40 burned and 40 unburned quadrats. The slash piles had been burned 2 to 15 years (average 8.8 years) before evaluation. Average frequency of Columbia brome was 5.6 percent higher on burned plots than on unburned plots, but this change was not statistically significant; Columbia brome was considered neutral with respect to fire (Vogl and Ryder 1969).

Columbia brome in the Swan Valley of northwestern Montana was apparently favored in mixed coniferous stands which had been logged, logged and burned, or burned only. Columbia brome occurred in undisturbed forests with presence of 36 percent and cover of 2 percent. In treated plots (all treatments considered together), Columbia brome had presence of 46 percent and cover of 3 percent. Percent presence and average percent cover were based on plots of occurrence (Freedman and Habeck 1985).

Columbia brome was also favored by disturbance in grand fir/pachistima and grand fir/twinflower (*Linnaea borealis*) forests in the Blue Mountains of northeastern Oregon. Stands were logged and broadcast burned to reduce slash. Columbia brome germination and establishment was enhanced in those areas with deeply churned soils and heavily burned spots (Edgerton 1987).

**LITERATURE CITED**


Camassia quamash
Common camas

FIRE ECOLOGY OR ADAPTATIONS

Soil insulates the meristematic tissue in common camas bulbs from damage by fire (Turner and Kuhnlein 1983).

POST-FIRE REGENERATION STRATEGY

Geophyte, growing points deep in soil.

IMMEDIATE FIRE EFFECT ON PLANT

Fire presumably top-kills common camas.

PLANT RESPONSE TO FIRE

Common camas on the palouse prairie of eastern Washington increases with frequent fire (Antieau and Gaynor 1990). Data regarding common camas post-fire recovery are lacking.

FIRE MANAGEMENT CONSIDERATIONS

Because growth and flowering occur in spring and early summer, short-interval fires in spring or early summer would probably reduce common camas populations.

Northwest Coast Indians reportedly set fires annually. This optimized common camas production by maintaining an open prairie (Turner and Bell 1971, Turner and Bell 1973).

LITERATURE CITED


Ross’ sedge survives fire through buried seed with long-term viability (Steele et al. 1983). These seeds germinate after heat treatment (Bradley et al. 1992).

Ross’ sedge’s rhizomes survive low- to moderate-severity fires (Bradley et al. 1992, Zamora 1975).

POST-FIRE REGENERATION STRATEGY

Rhizomatous herb, rhizome in soil; ground residual colonizer (onsite, initial community); secondary colonizer - onsite seed; and secondary colonizer - offsite seed.

IMMEDIATE FIRE EFFECT ON PLANT

Fire top-kills Ross’ sedge.

PLANT RESPONSE TO FIRE

Ross’ sedge is resistant to fire. It regenerates through rhizomes and seed germination (Ash and Lasko 1990, Zamora 1975). Recovery is rapid to moderate, taking 2 to 10 years to return to preburn frequency (Zamora 1975).

Ross’ sedge increases after fires that heat the soil but do not completely consume duff (Bradley et al. 1992). After the Sundance Fire of northern Idaho, Ross’ sedge was one of the most widely distributed plants (Strickler and Edgerton 1976). Ross’ sedge occurred on burned and grass-seeded plots but was more prevalent on unseeded burn plots (Leege and Godbolt 1985). Ross’ sedge increased for 4 years following the fire (Zamora 1975). Season of fire does not appear to have a major effect on plant recovery (Leege and Godbolt 1985, Zamora 1975).

At some sites, Ross’ sedge may be part of the pre-fire vegetation but may exist as “residual seed in the ground awaiting fire to create the proper germination conditions” (Ash and Lasko 1990).

FIRE MANAGEMENT CONSIDERATIONS

Development of Ross’ sedge cover is best at lower elevations (below 1150 m) (Zamora 1975).

Vigor of first-year plants after early spring or summer burns may be reduced by grazing (Volland 1974).
LITERATURE CITED


Prion’s pine is a fire-sensitive species that is very susceptible to damage and often shows a strong decline following fire (Halpern 1989, McLean 1968, Spies 1991, Stickney 1991). Survival probably depends to a great extent on damage to rhizomes, so it depends on depth of rhizomes, fire severity, and consumption of duff (Shearer and Stickney 1991, Stickney 1991). Loss of the long-lived evergreen leaves may also reduce survival. Post-fire vegetative recovery depends primarily on the survival of scattered individuals in undisturbed microsites (Halpern 1989).

**POST-FIRE REGENERATION STRATEGY**

Rhizomatous low woody plant, rhizome in organic mantle; rhizomatous herb, rhizome in soil.

**IMMEDIATE FIRE EFFECT ON PLANT**

Prince’s pine has a moderate to high probability of being killed by fire (Hopkins 1979, Stickney 1986). Low-severity fires that do not consume the organic mantle may only top-kill it.

**PLANT RESPONSE TO FIRE**

Post-fire response of prince’s pine is variable and is probably most dependent on fire severity and the uniformity of the burn. Some studies have reported prince’s-pine surviving fire. In mixed western hemlock-Douglas-fir-western redcedar (*Thuja plicata*) stands in North Cascades National Park, Washington, prince’s pine was considered a residual species following a July wildfire. Its frequency in post-fire years 1, 2, and 3 was 65.3, 52.1, and 52.1 percent, respectively (Miller and Miller 1976).

Prince’s-pine appeared to survive on moderately burned sites following the Waterfalls Canyon Fire in Grand Teton National Park in July, 1974, but was eliminated from severely burned sites. The pre-fire vegetation was spruce-fir with lodgepole pine (*Pinus contorta*) and whitebark pine (*P. albicaulis*). Prince’s pine had the following percent frequency and cover as measured in 1975 (Barmore *et al.* 1976):

<table>
<thead>
<tr>
<th>Unburned sites</th>
<th>Sites burned in 1932</th>
<th>Moderately burned sites</th>
<th>Severely burned sites</th>
</tr>
</thead>
<tbody>
<tr>
<td>Frequency</td>
<td>Cover</td>
<td>Frequency</td>
<td>Cover</td>
</tr>
<tr>
<td>52</td>
<td>5</td>
<td>2</td>
<td>trace</td>
</tr>
<tr>
<td>17</td>
<td>1</td>
<td>0</td>
<td>0</td>
</tr>
</tbody>
</table>
In the northern Rocky Mountains, slow recovery after fire has been reported. Prince’s pine was eliminated from initial post-fire communities by a severe wildfire in western larch (*Larix occidentalis*)-Douglas-fir stands on the Flathead National Forest, Montana (Stickney 1982). In western larch-fir (*Abies grandis* and *A. lasiocarpa*) stands on the Flathead and Lolo National Forests, Montana, prince’s-pine had not recovered by post-fire year 9 following logging and broadcast burning (Stickney 1980). Prince’s pine was also absent 10 months after a late-summer wildfire in lodgepole pine stands in the Chamberlain Basin, Idaho. It was found on adjacent unburned sites and was present on burned sites 5 years after the fire, but had less biomass production than on unburned sites (Phillips 1973).

Variable responses to fire have been reported for prince’s pine in Minnesota. It survived the Little Sioux Wildfire in May, 1971, in mixed conifer-hardwood stands in northeastern Minnesota. Number of individuals (on seventy 0.605 sq m plots) and aboveground average dry weight per individual prince’s pine were measured at the end of each growing season for the first 5 post-fire years (Ohmann and Grigal 1966):

<table>
<thead>
<tr>
<th>Year</th>
<th>No. of individuals</th>
<th>Ave. dry wt. (g)</th>
</tr>
</thead>
<tbody>
<tr>
<td>1971</td>
<td>15</td>
<td>.07</td>
</tr>
<tr>
<td>1972</td>
<td>no data</td>
<td>no data</td>
</tr>
<tr>
<td>1973</td>
<td>57</td>
<td>.33</td>
</tr>
<tr>
<td>1974</td>
<td>30</td>
<td>.29</td>
</tr>
<tr>
<td>1975</td>
<td>7</td>
<td>.46</td>
</tr>
</tbody>
</table>

Prince’s pine responded more slowly after wildfires in second-growth mixed conifer-hardwood forests in northeastern Minnesota. It was not present in post-fire years 3, 5, or 14 after the April Heartlake Fire. It was not present on the Kelley Creek Burn, resulting from a July fire, at post-fire year 2 but had a frequency of 3 percent in post-fire years 5 and 11 (Krefting and Ahlgren 1974).

**FIRE MANAGEMENT CONSIDERATIONS**

Prince’s pine is a component in many subzones in which guidelines for prescribed burning and tree species selection have been developed in the Coast Forest Region, British Columbia (Klinka 1977).

**LITERATURE CITED**


Canada thistle is adapted to both survive fire on site, and to colonize recently burned sites with exposed bare soil. The extensive root system gives it the ability to survive major disturbances as observed, for example, at Mt. St. Helens, where Canada thistle was part of the initial community after the 1980 eruption. It survived landslides and resprouted from root and stem fragments after the blast (Adams et al. 1987, Dale 1989, Titus et al. 1998). Similarly, the roots can survive fires of varying severity and produce new shoots (Romme et al. 1995). Additionally, there are numerous examples from the literature where Canada thistle seedlings established from wind-deposited seed, anywhere from 2 to 9 years after fire (Ahlgren 1979, Doyle et al. 1998, Lafferty 1970, McKell 1950, Neiland 1958, Rowe 1983, Turner et al. 1997, Willard 1995).

Canada thistle may change the fire ecology of the site in which it occurs by its abundant, flammable aboveground biomass. For example, in boreal wet-meadows, Canada thistle has the potential to increase fire frequency and perhaps severity as a result of its abundant and readily ignited litter (Hogenbirk and Wein 1995).
The following table provides some historic fire regime intervals for habitats in which Canada thistle may occur:

<table>
<thead>
<tr>
<th>Community or Ecosystem</th>
<th>Dominant Species</th>
<th>Fire Return Interval Range (years)</th>
</tr>
</thead>
<tbody>
<tr>
<td>silver fir-Douglas-fir</td>
<td>Abies amabilis-Pseudotsuga menziesii var. menziesii</td>
<td>&gt; 200</td>
</tr>
<tr>
<td>grand fir</td>
<td>A. grandis</td>
<td>35-200 [1]</td>
</tr>
<tr>
<td>maple-beech-birch</td>
<td>Acer-Fagus-Betula</td>
<td>&gt; 1000</td>
</tr>
<tr>
<td>silver maple-American elm</td>
<td>A. saccharinum-Ulmus americana</td>
<td>&lt; 35 to 200</td>
</tr>
<tr>
<td>sugar maple</td>
<td>A. saccharinum</td>
<td>&gt; 1000</td>
</tr>
<tr>
<td>sugar maple-basswood</td>
<td>A. s.-Tilia americana</td>
<td>&gt; 1000 [2]</td>
</tr>
<tr>
<td>bluestem prairie</td>
<td>Andropogon gerardii var. gerardii-Schizachyrium scoparium</td>
<td>&lt; 10 [3,4]</td>
</tr>
<tr>
<td>Nebraska sandhills prairie</td>
<td>A. g. var. paucipilus-S. s.</td>
<td>&lt; 10</td>
</tr>
<tr>
<td>bluestem-Sacahuista prairie</td>
<td>A. littoralis-Spartina spartinae</td>
<td>&lt; 10</td>
</tr>
<tr>
<td>sagebrush steppe</td>
<td>Artemisia tridentata/Pseudoroegneria spicata</td>
<td>20-70 [4]</td>
</tr>
<tr>
<td>basin big sagebrush</td>
<td>A. t. var. tridentata</td>
<td>12-43 [5]</td>
</tr>
<tr>
<td>mountain big sagebrush</td>
<td>A. t. var. vaseyana</td>
<td>20-60 [6,7]</td>
</tr>
<tr>
<td>Wyoming big sagebrush</td>
<td>A. t. var. wyomingensis</td>
<td>10-70 (40**) [8,9]</td>
</tr>
<tr>
<td>coastal sagebrush</td>
<td>A. californica</td>
<td>&lt; 35 to &lt; 100</td>
</tr>
<tr>
<td>plains grasslands</td>
<td>Bouteloua spp.</td>
<td>&lt; 35</td>
</tr>
<tr>
<td>blue grama-needle-and-thread grass-western wheatgrass</td>
<td>B. gracilis-Hesperostipa comata-Pascopyrum smithii</td>
<td>&lt; 35</td>
</tr>
<tr>
<td>blue grama-buffalo grass</td>
<td>B. g.-Buchloe dactyloides</td>
<td>&lt; 35</td>
</tr>
<tr>
<td>cheatgrass</td>
<td>Bromus tectorum</td>
<td>&lt; 10</td>
</tr>
<tr>
<td>California montane chaparral</td>
<td>Ceanothus and/or Arctostaphylos spp.</td>
<td>50-100 [4]</td>
</tr>
<tr>
<td>sugarberry-America elm-green ash</td>
<td>Celtis laevigata-Ulmus americana-Fraxinus pennsylvanica</td>
<td>&lt; 35 to 200 [2]</td>
</tr>
<tr>
<td>curlleaf mountain-mahogany*</td>
<td>Cercocarpus ledifolius</td>
<td>13-1000 [10,11]</td>
</tr>
<tr>
<td>mountain-mahogany-Gambel oak scrub</td>
<td>C. l.-Quercus gambelii</td>
<td>&lt; 35 to &lt; 100</td>
</tr>
<tr>
<td>northern cordgrass prairie</td>
<td>Distichlis spicata-Spartina spp.</td>
<td>1-3 [4]</td>
</tr>
<tr>
<td>beech-sugar maple</td>
<td>Fagus spp.-Acer saccharum</td>
<td>&gt; 1000 [2]</td>
</tr>
<tr>
<td>California steppe</td>
<td>Festuca-Danthonia spp.</td>
<td>&lt; 35 [4]</td>
</tr>
<tr>
<td>Plant Type</td>
<td>Scientific Name</td>
<td>Geographic Range</td>
</tr>
<tr>
<td>----------------------------------</td>
<td>----------------------------------</td>
<td>------------------</td>
</tr>
<tr>
<td>black ash</td>
<td><em>Fraxinus nigra</em></td>
<td>&lt; 35 to 200 [2]</td>
</tr>
<tr>
<td>juniper-oak savanna</td>
<td><em>Juniperus ashei-Quercus virginiana</em></td>
<td>35</td>
</tr>
<tr>
<td>Ashe juniper</td>
<td><em>J. ashei</em></td>
<td>35</td>
</tr>
<tr>
<td>western juniper</td>
<td><em>J. occidentalis</em></td>
<td>20-70</td>
</tr>
<tr>
<td>Rocky Mountain juniper</td>
<td><em>J. scopulorum</em></td>
<td>35</td>
</tr>
<tr>
<td>tamarack</td>
<td><em>Larix laricina</em></td>
<td>35-200 [4]</td>
</tr>
<tr>
<td>western larch</td>
<td><em>L. occidentalis</em></td>
<td>25-100 [1]</td>
</tr>
<tr>
<td>yellow-poplar</td>
<td><em>Liriodendron tulipifera</em></td>
<td>&lt; 35 [2]</td>
</tr>
<tr>
<td>wheatgrass plains grasslands</td>
<td><em>Pascopyrum smithii</em></td>
<td>&lt; 35 [4]</td>
</tr>
<tr>
<td>Great Lakes spruce-fir</td>
<td><em>Picea-Abies</em> spp.</td>
<td>35 to &gt; 200</td>
</tr>
<tr>
<td>northeastern spruce-fir</td>
<td><em>P.-A. spp.</em></td>
<td>35-200 [12]</td>
</tr>
<tr>
<td>Engelmann spruce-subalpine fir</td>
<td><em>P. engelmannii-A. lasiocarpa</em></td>
<td>35 to &gt; 200 [1]</td>
</tr>
<tr>
<td>black spruce</td>
<td><em>P. mariana</em></td>
<td>35-200</td>
</tr>
<tr>
<td>conifer bog*</td>
<td><em>P. m.-Larix laricina</em></td>
<td>35-200 [12]</td>
</tr>
<tr>
<td>blue spruce*</td>
<td><em>P. pungens</em></td>
<td>35-200 [1]</td>
</tr>
<tr>
<td>red spruce*</td>
<td><em>P. rubens</em></td>
<td>35-200 [12]</td>
</tr>
<tr>
<td>pine-cypress forest</td>
<td><em>Pinus-Cupressus</em> spp.</td>
<td>&lt; 35 to 200 [1]</td>
</tr>
<tr>
<td>pinyon-juniper</td>
<td><em>P.-Juniperus</em> spp.</td>
<td>&lt; 35 [4]</td>
</tr>
<tr>
<td>whitebark pine*</td>
<td><em>P. albicaulis</em></td>
<td>50-200 [1]</td>
</tr>
<tr>
<td>jack pine</td>
<td><em>P. banksiana</em></td>
<td>&lt; 35 to 200 [12]</td>
</tr>
<tr>
<td>Rocky Mountain lodgepole pine*</td>
<td><em>P. contorta var. latifolia</em></td>
<td>25-300+ [13,1,14]</td>
</tr>
<tr>
<td>Sierra lodgepole pine*</td>
<td><em>P. c. var. murrayana</em></td>
<td>35-200 [1]</td>
</tr>
<tr>
<td>shortleaf pine</td>
<td><em>P. echinata</em></td>
<td>2-15</td>
</tr>
<tr>
<td>shortleaf pine-oak</td>
<td><em>P. e.-Quercus</em> spp.</td>
<td>&lt; 10 [2]</td>
</tr>
<tr>
<td>Colorado pinyon</td>
<td><em>P. edulis</em></td>
<td>10-49 [4]</td>
</tr>
<tr>
<td>South Florida slash pine</td>
<td><em>P. elliottii var. densa</em></td>
<td>1-5 [15,2]</td>
</tr>
<tr>
<td>Jeffrey pine</td>
<td><em>P. jeffreyi</em></td>
<td>5-30</td>
</tr>
<tr>
<td>western white pine*</td>
<td><em>P. monticola</em></td>
<td>50-200</td>
</tr>
<tr>
<td>Pacific ponderosa pine*</td>
<td><em>P. ponderosa var. ponderosa</em></td>
<td>1-47</td>
</tr>
<tr>
<td>interior ponderosa pine*</td>
<td><em>P. p. var. scopulorum</em></td>
<td>2-10</td>
</tr>
<tr>
<td>Arizona pine</td>
<td><em>P. p. var. arizonica</em></td>
<td>2-10 [1]</td>
</tr>
<tr>
<td>Table Mountain pine</td>
<td><em>P. pungens</em></td>
<td>&lt; 35 to 200 [2]</td>
</tr>
<tr>
<td>red pine (Great Lakes region)</td>
<td><em>P. resinosa</em></td>
<td>10-200 (10**) [12,16]</td>
</tr>
<tr>
<td>red-white-jack pine*</td>
<td><em>P. r.-P. strobus-P. banksiana</em></td>
<td>10-300 [12,17]</td>
</tr>
<tr>
<td>pitch pine</td>
<td><em>P. rigida</em></td>
<td>6-25 [18,19]</td>
</tr>
<tr>
<td>eastern white pine</td>
<td><em>P. strobus</em></td>
<td>35-200</td>
</tr>
<tr>
<td>eastern white pine-eastern hemlock</td>
<td><em>P. s.-Tsuga canadensis</em></td>
<td>35-200</td>
</tr>
<tr>
<td>Forest Type</td>
<td>Species/Genus</td>
<td>Age Range</td>
</tr>
<tr>
<td>------------------------------------------------</td>
<td>-----------------------------------</td>
<td>-----------</td>
</tr>
<tr>
<td>eastern white pine-northern red oak-red maple</td>
<td>P. s.-Quercus rubra-Acer rubrum</td>
<td>35-200</td>
</tr>
<tr>
<td>loblolly pine</td>
<td>P. taeda</td>
<td>3-8</td>
</tr>
<tr>
<td>loblolly-shortleaf pine</td>
<td>P. t.-P. echinata</td>
<td>10 to &lt; 35</td>
</tr>
<tr>
<td>Virginia pine</td>
<td>P. virginiana</td>
<td>10 to &lt; 35</td>
</tr>
<tr>
<td>Virginia pine-oak</td>
<td>P. v.-Quercus spp.</td>
<td>10 to &lt; 35 [2]</td>
</tr>
<tr>
<td>eastern cottonwood</td>
<td>Populus deltoides</td>
<td>&lt; 35 to 200 [4]</td>
</tr>
<tr>
<td>aspen-birch</td>
<td>P. tremuloides-Betula papyrifera</td>
<td>35-200 [12,2]</td>
</tr>
<tr>
<td>quaking aspen (west of the Great Plains)</td>
<td>P. tremuloides</td>
<td>7-120 [1,20,21]</td>
</tr>
<tr>
<td>black cherry-sugar maple</td>
<td>Prunus serotina-Acer saccharum</td>
<td>&gt; 1000 [2]</td>
</tr>
<tr>
<td>mountain grasslands</td>
<td>Pseudoroegneria spicata</td>
<td>3-40 (10**) [13,1]</td>
</tr>
<tr>
<td>Rocky Mountain Douglas-fir*</td>
<td>Pseudotsuga menziesii var. glauca</td>
<td>25-100 [1]</td>
</tr>
<tr>
<td>coastal Douglas-fir*</td>
<td>P. m. var. menziesii</td>
<td>40-240 [1,22,23]</td>
</tr>
<tr>
<td>California mixed evergreen</td>
<td>P. m. var. m.-Lithocarpus densiflorus-Abutus m.</td>
<td>&lt; 35</td>
</tr>
<tr>
<td>California oakwoods</td>
<td>Quercus spp.</td>
<td>&lt; 35 [1]</td>
</tr>
<tr>
<td>oak-hickory</td>
<td>Q.-Carya spp.</td>
<td>&lt; 35[2]</td>
</tr>
<tr>
<td>oak-juniper woodland (Southwest)</td>
<td>Q.-Juniperus spp.</td>
<td>&lt; 35 to &lt; 200 [4]</td>
</tr>
<tr>
<td>northeastern oak-pine</td>
<td>Q.-Pinus spp.</td>
<td>10 to &lt; 35 [2]</td>
</tr>
<tr>
<td>coast live oak</td>
<td>Q. agrifolia</td>
<td>&lt; 35 to 200 [1]</td>
</tr>
<tr>
<td>canyon live oak</td>
<td>Q. chrysolepis</td>
<td>&lt; 35 to 200</td>
</tr>
<tr>
<td>blue oak-foothills pine</td>
<td>Q. douglasii-P. sabiana</td>
<td>&lt; 35 [1]</td>
</tr>
<tr>
<td>northern pin oak</td>
<td>Q. ellipsoidalis</td>
<td>&lt; 35 [2]</td>
</tr>
<tr>
<td>Oregon white oak</td>
<td>Q. garryana</td>
<td>&lt; 35 [1]</td>
</tr>
<tr>
<td>bear oak</td>
<td>Q. ilicifolia</td>
<td>&lt; 35 &gt; [2]</td>
</tr>
<tr>
<td>California black oak</td>
<td>Q. kelloggii</td>
<td>5-30 [4]</td>
</tr>
<tr>
<td>bur oak</td>
<td>Q. macrocarpa</td>
<td>&lt; 10</td>
</tr>
<tr>
<td>chestnut oak</td>
<td>Q. prinus</td>
<td>3-8</td>
</tr>
<tr>
<td>northern red oak</td>
<td>Q. rubra</td>
<td>10 to &lt; 35</td>
</tr>
<tr>
<td>post oak-blackjack oak</td>
<td>Q. stellata-Q. marilandica</td>
<td>&lt; 10</td>
</tr>
<tr>
<td>black oak</td>
<td>Q. velutina</td>
<td>&lt; 35</td>
</tr>
<tr>
<td>live oak</td>
<td>Q. virginiana</td>
<td>10 to&lt; 100 [2]</td>
</tr>
<tr>
<td>interior live oak</td>
<td>Q. wislizenii</td>
<td>&lt; 35 [1]</td>
</tr>
<tr>
<td>blackland prairie</td>
<td>Schizachyrium scoparium-</td>
<td>&lt; 10</td>
</tr>
<tr>
<td>Plant Community</td>
<td>Dominant Species</td>
<td>Abundance</td>
</tr>
<tr>
<td>------------------------------</td>
<td>-------------------------------------------------------</td>
<td>-----------</td>
</tr>
<tr>
<td>Fayette prairie</td>
<td><em>Nassella leucotricha</em></td>
<td>S. s.-<em>Buchloe dactyloides</em></td>
</tr>
<tr>
<td>little bluestem-grama prairie</td>
<td><em>S. s.-Bouteloua</em> spp.</td>
<td>&lt; 35</td>
</tr>
<tr>
<td>tule marshes</td>
<td><em>Scirpus</em> and/or <em>Typha</em> spp.</td>
<td>&lt; 35 [4]</td>
</tr>
<tr>
<td>redwood</td>
<td><em>Sequoia sempervirens</em></td>
<td>5-200 [1,23,25]</td>
</tr>
<tr>
<td>western redcedar-western hemlock</td>
<td><em>Thuja plicata</em>-<em>Tsuga heterophylla</em></td>
<td>&gt; 200 [1]</td>
</tr>
<tr>
<td>eastern hemlock-yellow birch</td>
<td><em>T. canadensis</em>-<em>Betula alleghaniensis</em></td>
<td>&gt; 200 [2]</td>
</tr>
<tr>
<td>western hemlock-Sitka spruce</td>
<td><em>T. h.-Picea sitchensis</em></td>
<td>&gt; 200</td>
</tr>
<tr>
<td>mountain hemlock*</td>
<td><em>T. mertensiana</em></td>
<td>35 to &gt; 200 [1]</td>
</tr>
<tr>
<td>elm-ash-cottonwood</td>
<td><em>Ulmus</em>-<em>Fraxinus</em>-<em>Populus</em> spp.</td>
<td>&lt; 35 to 200 [12,2]</td>
</tr>
</tbody>
</table>

*fire return interval varies widely; trends in variation are noted in the species summary

**mean**


POST-FIRE REGENERATION STRATEGY (Stickney 1989)

Geophyte, growing points deep in soil; ground residual colonizer (on-site, initial community); initial off-site colonizer (off-site, initial community).

IMMEDIATE FIRE EFFECT ON PLANT

Fire kills the aboveground portion of Canada thistle plants, while the roots can survive severe fires (Young 1986).

PLANT RESPONSE TO FIRE

Canada thistle is slightly damaged to enhanced by fire (Young 1986). It can survive fire and sprout vegetatively from its extensive perennial root system (Romme et al. 1995), or colonize bare ground via seedling establishment after fire (Ahlgren 1979, Doyle et al. 1998, Rowe 1983). For example, in Yellowstone National Park, Canada thistle is rare in unburned forests but locally abundant in burned areas (Despain 1990). When sites supporting Canada thistle are burned, its response is variable, and may be affected by season of burn, burn severity, site conditions, and plant community composition and phenology before and after the fire. Existing research provides no clear correlations with these variables.

DISCUSSION AND QUALIFICATION OF PLANT RESPONSE

Several studies have indicated the presence of Canada thistle in burned areas where it was absent from the pre-fire community and/or adjacent unburned areas (Lafferty 1970, McKell 1950, Neiland 1958, Ossinger 1983, Schoenberger and Perry 1982). In Grand Teton National Park, Wyoming, Canada thistle did not occur in unburned forest and was not part of the initial post-fire vegetation after a mixed-severity wildfire. It established 2 years after fire on a moderate-severity site, and 9 years after fire on a severe site. On both sites, it decreased to < 1% cover by post-fire year 17 as cover of tree saplings increased (Doyle et al. 1998). Seedlings were found in a red pine forest in Minnesota, 3 years after fire, but not on adjacent unburned forest (Ahlgren 1979). Canada thistle established 3 years after mixed-severity fires in sedge meadows in Glacier National Park (Willard et al. 1995).

In Yellowstone National Park, Canada thistle established after 1988 fires and increased in density over time, 2 to 5 years after fire, in all burn severities. Density was lowest in the low-severity burns and highest in the stand-replacing burns (Turner et al. 1997). Canada thistle established on both bulldozer lines and burned areas after a 1988 wildfire in Glacier National Park, but was not present in comparable undisturbed sites (Benson and Kurth 1995).
Response of established Canada thistle plants to fire is unclear, as there are mixed reports in the literature. A Canada thistle clone in a mid-boreal wetland site was not noticeably changed when burned in the spring with a propane torch to simulate both light and deep burns (Hogenbirk and Wein 1991). The authors concluded that there is a moderate to high probability that Canada thistle and other Eurasian xerophytic species will dominate these wet-meadows in the short term after fire, and that they will continue to dominate small areas for longer periods (Hogenbirk and Wein 1995). There were no significant differences ($p < 0.05$) in Canada thistle cover after spring burning in the prairie pothole region of Iowa (Messinger 1974). In Mesa Verde National Park, Colorado, populations of Canada thistle that were well established before an August wildfire resprouted immediately after the burn, and spread downstream in the canyons. Canada thistle and other non-native species (e.g., cheatgrass ($Bromus tectorum$) and musk thistle ($Carduus nutans$) continued to dominate the severely burned areas and expanded their area by 260% 6 years after the wildfire (Floyd-Hanna et al. 1997, Floyd-Hanna et al. 1993).

In a native mixed-grass prairie in North Dakota, late-spring and late-summer burning increased seed production and seedling numbers in Canada thistle, but fewer thistles were observed during the years following the burn than before or during the year of the burn (Smith 1985). Dormant season (winter and early spring) burning in eastern Oregon resulted in fewer total and fewer functional flowerheads on reproductive shoots of Canada thistle when compared to unburned control. Also, Canada thistle plants on burned sites grew more slowly and associated vegetation was more productive than on control sites. It was concluded that burning reduced the relative abundance of Canada thistle and may be useful as a means of halting its invasion or spread by maintaining a productive stand of native vegetation (Young 1986). The discrepancy in these reports is probably due to the large number of variables that can affect the response of Canada thistle to fire, including fire severity, for which we lack a standard nomenclature in the literature. Other important variables include vegetation and site characteristics, frequency, and season of burning.

Site differences such as soil moisture content, plant community, and slope aspect can influence fire severity and may influence the response of Canada thistle to fire. In a northwestern Minnesota prairie site, prescribed burning on a nearly level mesic site in badly disturbed prairie had no effect on Canada thistle flowering while flowering was inhibited on a level, wet-mesic site in badly disturbed prairie after burning (Pemble et al. 1981). On a forested site in western Montana that was harvested and burned, Canada thistle seems to have increased with both light and severe burning in the fall, with larger increases on south aspects compared with others (Lafferty 1970). Olson (1975) provided evidence that prescribed burning in the spring either reduced or did not change canopy cover of Canada thistle in Minnesota. Results differed between sites, which differed primarily in plant community type and in time and frequency of burning.
Frequency, severity and season of burning may have a considerable effect on Canada thistle response. In a study conducted on a mesic tallgrass prairie site in Colorado, plots that were burned frequently (5 times over 7 years) had lower density of Canada thistle than did and area that was burned only twice during the same period. Results were inconclusive, however, since the final season of the study saw increased spread of Canada thistle from the surrounding area, probably due to clonal growth from existing plants (Morghan et al. 2000). In a prairie site at Pipestone National Monument, Minnesota, 5 years of annual spring burning in mid- to late April, with fires of low to moderate severity, reduced the frequency of Canada thistle over time until it was absent after the fifth year (Becker 1989). Similarly, observations in tallgrass prairie sites in South Dakota indicate that late spring prescribed burning (when native species are still dormant) on a 4 to 5 year rotation (as per the historic fire regime) encourages the growth of native plants and discourages the growth of Canada, bull and musk thistles. Livestock use must be carefully timed following burning, since grazing early in the growing season can potentially negate beneficial effects of prescribed fire (Dailey 2001). On a common reed marsh in Manitoba, Canada thistle response to burning varied with season of burn. Aboveground biomass, stem density, and seedling density were unchanged on spring burns, but increased on both summer and fall burns (Thompson and Shay 1989). Results are presented below:

<table>
<thead>
<tr>
<th></th>
<th>Biomass (g/m²)</th>
<th>density (stems of nonseedling shoots/m²)</th>
<th>density of seedlings (stems/m²)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Control</td>
<td>5.0±7.0</td>
<td>0.9±0.9</td>
<td>0.2±0.4</td>
</tr>
<tr>
<td>Spring</td>
<td>5.3±4.8</td>
<td>4.9±3.1</td>
<td>0.4±0.2</td>
</tr>
<tr>
<td>Summer</td>
<td>63.3±39.4</td>
<td>20±3.9</td>
<td>1.5±3.3</td>
</tr>
<tr>
<td>Fall</td>
<td>27.6±48.6</td>
<td>9.5±12.5</td>
<td>1.4±2.6</td>
</tr>
</tbody>
</table>

FIRE MANAGEMENT CONSIDERATIONS

Abundant evidence of post-fire establishment of Canada thistle (Benson and Kurth 1995, McKell 1950, Ossinger 1983, Shoenberger and Perry 1982) suggests that managers need to be aware of this possibility, especially if a known seed source is in the area, and take measures to prevent the establishment of Canada thistle after prescribed burning and wildfires. Seeding with aggressive, introduced grasses such as crested wheatgrass, intermediate wheatgrass, orchardgrass, and smooth brome following a prescribed burn in Utah pinyon-juniper communities prevented establishment of Canada thistle, whereas unseeded areas supported Canada thistle seedlings (Goodrich and Rooks 1999). Similarly, in disturbed forest sites where Canada thistle becomes established, it may be shaded out over time as trees reestablish (Doyle et al. 1998).
Research in this report suggests that response of Canada thistle to fire is variable and it depends on vegetation and site characteristics, as well as frequency, severity and season of burning. Prescribed burns may be effective at stimulating growth of native species and thereby discouraging the growth of invasives such as Canada thistle (Rice and Randall 2001), and may be best if timed to emulate the natural fire regime of a site (Dailey 2001). Hutchison (1992) states that prescribed burning is a “preferred treatment” for the control of Canada thistle, and that late spring burns effectively discourage this species, whereas early spring burns can increase sprouting and reproduction. During the first 3 years of control efforts, he recommends that burns be conducted annually (Hutchinson 1992).

Season of burn is an important consideration for prescribed burning, as the timing of the burn will determine species composition and cover in the post-fire community (Howe 1994a, Howe 1994b). Dormant season burning may be a preferred treatment method in some areas, because in many habitats it stimulates growth of native vegetation that subsequently competes with Canada thistle (Nuzzo 2000, Young 1986). However, dormant season burning may not be as effective as late spring burning (Hutchinson 1992). Controlled studies comparing the effects of these variables in different natural areas are currently lacking in the literature.

Equations for estimating fuel loading of forb communities including Canada thistle are available (Brown and Marsden 1976).

The USDA Forest Service’s “Guide to Noxious Weed Prevention Practices” (USDA Forest Service 2001) provides several fire management considerations for weed prevention in general that apply to Canada thistle. To prevent invasion after wildfires and prescribed burns, re-establish vegetation on bare ground as soon as possible using either natural recovery or artificial techniques as appropriate to site objectives. When reseeding burn areas, use only certified weed-free seed. Monitor burn sites and associated disturbed areas after the fire and the following spring for emergence of Canada thistle, and treat to eradicate any emergent Canada thistle plants. Regulate human, pack animal, and livestock entry into burned areas at risk for weed invasion until desirable site vegetation has recovered sufficiently to resist weed invasion.

When planning a prescribed burn, preinventory the project area and evaluate cover and phenology of any Canada thistle present on or adjacent to the site, and avoid ignition and burning in areas at high risk for Canada thistle establishment or spread due to fire effects. Avoid creating soil conditions that promote weed germination and establishment. Discuss weed status and risks in burn rehabilitation plans. Wildfire managers might consider including weed prevention education and providing weed identification aids during fire training; avoiding known weed infestations when locating fire lines, monitoring camps, staging areas, helibases, etc., to be sure they are kept weed free; taking care that equipment is weed free; incorporating weed prevention into fire rehabilitation plans; and acquiring restoration funding. Additional guidelines and specific recommendations and requirements are available (USDA Forest Service 2001).
LITERATURE CITED


**Cirsium vulgare**  
Bull thistle

**FIRE ECOLOGY OR ADAPTATIONS**

Bull thistle reproduces by abundant seed, some of which may disperse over moderate distances by wind and some of which may remain dormant in the soil for several years (research thus far suggests up to 5). Fire creates conditions that are favorable for establishment (i.e. open canopy, reduced competition, areas of bare soil), so if bull thistle seeds are present and competition minimal, bull thistle may be favored in the post-fire community. This is supported by several examples of bull thistles establishment within a few years after fire (Arno 1996, Arno 1999, Ashton 1981, Benson and Kurth 1995, Shearer and Stickney 1991, Simmerman *et al.* 1991, Stickney 1980). More research is needed regarding adaptations of bull thistle to fire.

Bull thistle is the most common and widespread thistle of pastures and rangelands in western North America, so it occurs in a large number of ecosystems with different fire regimes. Introduced species can alter the rate of fire spread, the probability of fire occurrence, and the intensity of fire in an ecosystem (D’Antonio 2000). It is unclear how the presence of bull thistle alters the fire regime of a given site, and it is unclear how a historical fire regime might affect the presence or abundance of bull thistle at a given site. Dominant species of forest communities in which bull thistle has been noted as a primary or secondary colonizer after disturbance are described in the Habitat Types and Plant Communities section of this report. Bull thistle also occurs in tallgrass prairie ecosystems, where fire can stimulate flowering of native grasses (Dailey 2001). In Kansas, frequent burning of tallgrass prairie is said to be effective in keeping out exotic plants on sites where prairie grasses are vigorous (Hulbert 1986). Bull thistle did not occur in any of these communities at the time in which historic fire regimes were functioning, but has established since fire exclusion began. It is unclear how the presence of bull thistle might affect fire regimes in these communities.

Because it is so widespread and has broad ecological tolerances, it is difficult to exclude many ecosystems as potential hosts of bull thistle plants or populations. The following table provides fire regime intervals for several plant communities in which bull thistle may be found.

<table>
<thead>
<tr>
<th>Community or Ecosystem</th>
<th>Dominant Species</th>
<th>Fire Return Interval Range (years)</th>
</tr>
</thead>
<tbody>
<tr>
<td>silver fir-Douglas-fir</td>
<td><em>Abies amabilis-Pseudotsuga menziesii</em> var. <em>menziesii</em></td>
<td>&gt; 200</td>
</tr>
<tr>
<td>grand fir</td>
<td><em>A. grandis</em></td>
<td>35-200 [1]</td>
</tr>
<tr>
<td>maple-beech-birch</td>
<td><em>Acer-Fagus-Betula</em></td>
<td>&gt; 1000</td>
</tr>
<tr>
<td>silver maple-American elm</td>
<td><em>A. saccharinum-Ulmus americana</em></td>
<td>&lt; 35 to 200</td>
</tr>
<tr>
<td>Ecosystem</td>
<td>Species/Community</td>
<td>Abundance</td>
</tr>
<tr>
<td>-----------------------------------</td>
<td>--------------------------------------------</td>
<td>-----------</td>
</tr>
<tr>
<td>sugar maple</td>
<td><em>A. saccharum</em></td>
<td>&gt; 1000</td>
</tr>
<tr>
<td>sugar maple-basswood</td>
<td><em>A. s.-Tilia americana</em></td>
<td>&gt; 1000 [2]</td>
</tr>
<tr>
<td>California chaparral</td>
<td><em>Adenostoma and/or Arctostaphylos spp.</em></td>
<td>&lt; 35 to &lt; 100 [3]</td>
</tr>
<tr>
<td>bluestem prairie</td>
<td><em>Andropogon gerardii var. gerardii-Schizachyrium scoparium</em></td>
<td>&lt; 10 [4,3]</td>
</tr>
<tr>
<td>Nebraska sandhills prairie</td>
<td><em>A. g. var. paucilus-Schizachyrium scoparium</em></td>
<td>&lt; 10</td>
</tr>
<tr>
<td>bluestem-Sacahuista prairie</td>
<td><em>A. littoralis-Spartina spartinae</em></td>
<td>&lt; 10</td>
</tr>
<tr>
<td>sagebrush steppe</td>
<td><em>Artemisia tridentata/Pseudoroegneria spicata</em></td>
<td>20-70 [3]</td>
</tr>
<tr>
<td>basin big sagebrush</td>
<td><em>A. t. var. tridentata</em></td>
<td>12-43 [5]</td>
</tr>
<tr>
<td>mountain big sagebrush</td>
<td><em>A. t. var. vaseyana</em></td>
<td>15-40 [6,7,8]</td>
</tr>
<tr>
<td>Wyoming big sagebrush</td>
<td><em>A. t. var. wyomingensis</em></td>
<td>10-70 (40**) [9,10]</td>
</tr>
<tr>
<td>coastal sagebrush</td>
<td><em>A. californica</em></td>
<td>&lt; 35 to &lt; 100</td>
</tr>
<tr>
<td>plains grasslands</td>
<td><em>Bouteloua spp.</em></td>
<td>&lt; 35</td>
</tr>
<tr>
<td>cheatgrass</td>
<td><em>Bromus tectorum</em></td>
<td>&lt; 10</td>
</tr>
<tr>
<td>California montane chaparral</td>
<td><em>Ceanothus and/or Arctostaphylos spp.</em></td>
<td>50-100 [3]</td>
</tr>
<tr>
<td>sugarberry-America elm-green ash</td>
<td><em>Celtis laevigata-Ulmus americana-Fraxinus pennsylvanica</em></td>
<td>&lt; 35 to 200 [2]</td>
</tr>
<tr>
<td>curlleaf mountain-mahogany*</td>
<td><em>Cercocarpus ledifolius</em></td>
<td>13-1000 [11,12]</td>
</tr>
<tr>
<td>mountain-mahogany-Gambel oak scrub</td>
<td><em>C. l.-Quercus gambelii</em></td>
<td>&lt; 35 to &lt; 100 [3]</td>
</tr>
<tr>
<td>Atlantic white-cedar</td>
<td><em>Chamaecyparis thyoides</em></td>
<td>35 to &gt; 200 [2]</td>
</tr>
<tr>
<td>Arizona cypress</td>
<td><em>Cupressus arizonica</em></td>
<td>&lt; 35 to 200</td>
</tr>
<tr>
<td>northern cordgrass prairie</td>
<td><em>Distichlis spicata-Spartina spp.</em></td>
<td>1-3 [3]</td>
</tr>
<tr>
<td>beech-sugar maple</td>
<td><em>Fagus spp.-Acer saccharum</em></td>
<td>&gt; 1000 [2]</td>
</tr>
<tr>
<td>California steppe</td>
<td><em>Festuca-Danthonia spp.</em></td>
<td>&lt; 35 [3]</td>
</tr>
<tr>
<td>black ash</td>
<td><em>Fraxinus nigra</em></td>
<td>&lt; 35 to 200 [2]</td>
</tr>
<tr>
<td>juniper-oak savanna</td>
<td><em>Juniperus ashei-Quercus virginiana</em></td>
<td>&lt; 35</td>
</tr>
<tr>
<td>Ashe juniper</td>
<td><em>J. ashei</em></td>
<td>&lt; 35</td>
</tr>
<tr>
<td>western juniper</td>
<td><em>J. occidentalis</em></td>
<td>20-70</td>
</tr>
<tr>
<td>Rocky Mountain juniper</td>
<td><em>J. scopulorum</em></td>
<td>&lt; 35</td>
</tr>
<tr>
<td>cedar glades</td>
<td><em>J. virginiana</em></td>
<td>3-7</td>
</tr>
<tr>
<td>tamarack</td>
<td><em>Larix laricina</em></td>
<td>35-200 [3]</td>
</tr>
<tr>
<td>western larch</td>
<td><em>L. occidentalis</em></td>
<td>25-100 [1]</td>
</tr>
<tr>
<td>yellow-poplar</td>
<td><em>Liriodendron tulipifera</em></td>
<td>&lt; 35 [2]</td>
</tr>
<tr>
<td>wheatgrass plains grasslands</td>
<td><em>Pascopyrum smithii</em></td>
<td>&lt; 35 [3]</td>
</tr>
<tr>
<td>Forest Type</td>
<td>Species</td>
<td>Age Range</td>
</tr>
<tr>
<td>------------------------------------------------</td>
<td>--------------------------------------------------</td>
<td>-----------</td>
</tr>
<tr>
<td>Great Lakes spruce-fir</td>
<td><em>Picea-Abies</em> spp.</td>
<td>35 to &gt; 200</td>
</tr>
<tr>
<td>northeastern spruce-fir</td>
<td><em>P.-A.</em> spp.</td>
<td>35-200 [13]</td>
</tr>
<tr>
<td>southeastern spruce-fir</td>
<td><em>P.-A.</em> spp.</td>
<td>35 to &gt; 200 [2]</td>
</tr>
<tr>
<td>Engelmann spruce-subalpine fir</td>
<td><em>P. engelmannii-</em> A. lasiocarpa</td>
<td>35 to &gt; 200 [1]</td>
</tr>
<tr>
<td>black spruce</td>
<td><em>P. mariana</em></td>
<td>35-200 [13]</td>
</tr>
<tr>
<td>blue spruce*</td>
<td><em>P. pungens</em></td>
<td>35-200 [1]</td>
</tr>
<tr>
<td>red spruce*</td>
<td><em>P. rubens</em></td>
<td>35-200 [13]</td>
</tr>
<tr>
<td>pine-cypress forest</td>
<td><em>Pinus-Cupressus</em> spp.</td>
<td>&lt; 35 to 200 [1]</td>
</tr>
<tr>
<td>pinyon-juniper</td>
<td><em>P.-Juniperus</em> spp.</td>
<td>&lt; 35 [3]</td>
</tr>
<tr>
<td>whitebark pine*</td>
<td><em>P. albicaulis</em></td>
<td>50-200 [1]</td>
</tr>
<tr>
<td>jack pine</td>
<td><em>P. banksiana</em></td>
<td>&lt; 35 to 200 [13]</td>
</tr>
<tr>
<td>Mexican pinyon</td>
<td><em>P. cembroides</em></td>
<td>20-70 [14,15]</td>
</tr>
<tr>
<td>Rocky Mountain lodgepole pine*</td>
<td><em>P. contorta var. latifolia</em></td>
<td>25-300+ [1,16,17]</td>
</tr>
<tr>
<td>Sierra lodgepole pine*</td>
<td><em>P. c. var. murrayana</em></td>
<td>35-200 [1]</td>
</tr>
<tr>
<td>shortleaf pine</td>
<td><em>P. echinata</em></td>
<td>2-15</td>
</tr>
<tr>
<td>slash pine</td>
<td><em>P. elliottii</em></td>
<td>3-8 [2]</td>
</tr>
<tr>
<td>Jeffrey pine</td>
<td><em>P. jeffreyi</em></td>
<td>5-30</td>
</tr>
<tr>
<td>western white pine*</td>
<td><em>P. monticola</em></td>
<td>50-200 [1]</td>
</tr>
<tr>
<td>longleaf-slash pine</td>
<td>*P. palustris-<em>P. elliottii</em></td>
<td>1-4 [18,2]</td>
</tr>
<tr>
<td>Pacific ponderosa pine*</td>
<td><em>P. ponderosa var. ponderosa</em></td>
<td>1-47 [1]</td>
</tr>
<tr>
<td>interior ponderosa pine*</td>
<td><em>P. p. var. scopulorum</em></td>
<td>2-30 [1,19,20]</td>
</tr>
<tr>
<td>Arizona pine</td>
<td><em>P. p. var. arizonica</em></td>
<td>2-10 [1]</td>
</tr>
<tr>
<td>Table Mountain pine</td>
<td><em>P. pungens</em></td>
<td>&lt; 35 to 200 [2]</td>
</tr>
<tr>
<td>red pine (Great Lakes region)</td>
<td><em>P. resinosa</em></td>
<td>10-200 (10**)[13,21]</td>
</tr>
<tr>
<td>red-white-jack pine*</td>
<td>*P. r.-P. strobus-*P. banksiana</td>
<td>10-300 [13,22]</td>
</tr>
<tr>
<td>pitch pine</td>
<td><em>P. rigida</em></td>
<td>6-25 [23,24]</td>
</tr>
<tr>
<td>pocosin</td>
<td><em>P. serotina</em></td>
<td>3-8</td>
</tr>
<tr>
<td>eastern white pine</td>
<td><em>P. strobos</em></td>
<td>35-200</td>
</tr>
<tr>
<td>eastern white pine-eastern hemlock</td>
<td><em>P. s.-Tsuga canadensis</em></td>
<td>35-200</td>
</tr>
<tr>
<td>eastern white pine-northern red oak-   red maple</td>
<td>*P. s.-Quercus rubra-*Acer rubrum</td>
<td>35-200</td>
</tr>
<tr>
<td>loblolly pine</td>
<td><em>P. taeda</em></td>
<td>3-8</td>
</tr>
<tr>
<td>loblolly-shortleaf pine</td>
<td><em>P. t.-P. echinata</em></td>
<td>10 to &lt; 35</td>
</tr>
<tr>
<td>Virginia pine</td>
<td><em>P. virginiana</em></td>
<td>10 to &lt; 35</td>
</tr>
<tr>
<td>Virginia pine-oak</td>
<td><em>P. v.-Quercus</em> spp.</td>
<td>10 to &lt; 35</td>
</tr>
<tr>
<td>sycamore-sweetgum-American elm</td>
<td>*Platanus occidentalis-*Liquidambar styraciflua-*Ulmus americana</td>
<td>&lt; 35 to 200 [2]</td>
</tr>
<tr>
<td>galleta-threeawn shrubsteppe</td>
<td>*Pleuraphis jamesii-*Aristida purpurea</td>
<td>&lt; 35 to &lt; 100</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Plant Type</th>
<th>Scientific Name</th>
<th>Density/Range</th>
</tr>
</thead>
<tbody>
<tr>
<td>eastern cottonwood</td>
<td><em>Populus deltoides</em></td>
<td>&lt; 35 to 200 [3]</td>
</tr>
<tr>
<td>aspen-birch</td>
<td><em>P. tremuloides-Betula papyrifera</em></td>
<td>35-200 [13,2]</td>
</tr>
<tr>
<td>quaking aspen (west of the Great Plains)</td>
<td><em>P. tremuloides</em></td>
<td>7-120 [1,25,26]</td>
</tr>
<tr>
<td>mesquite</td>
<td><em>Prosopis glandulosa</em></td>
<td>&lt; 35 to &lt; 100 [27,3]</td>
</tr>
<tr>
<td>black cherry-sugar maple</td>
<td><em>Prunus serotina-Acer saccharum</em></td>
<td>&gt; 1000 [2]</td>
</tr>
<tr>
<td>mountain grasslands</td>
<td><em>Pseudoroegneria spicata</em></td>
<td>3-40 (10**) [16,1]</td>
</tr>
<tr>
<td>Rocky Mountain Douglas-fir*</td>
<td><em>Pseudotsuga menziesii var. glauca</em></td>
<td>25-100 [1]</td>
</tr>
<tr>
<td>coastal Douglas-fir*</td>
<td><em>P. m. var. menziesii</em></td>
<td>40-240 [1,28,29]</td>
</tr>
<tr>
<td>California mixed evergreen</td>
<td><em>P. m. var. m.-Lithocarpus densiflorus-Arbutus menziesii</em></td>
<td>&lt; 35</td>
</tr>
<tr>
<td>California oakwoods</td>
<td><em>Quercus spp.</em></td>
<td>&lt; 35 [1]</td>
</tr>
<tr>
<td>oak-hickory</td>
<td><em>Q.-Carya spp.</em></td>
<td>&lt; 35[2]</td>
</tr>
<tr>
<td>oak-juniper woodland (Southwest)</td>
<td><em>Q.-Juniperus spp.</em></td>
<td>&lt; 35 to &lt; 200 [3]</td>
</tr>
<tr>
<td>northeastern oak-pine</td>
<td><em>Q.-Pinus spp.</em></td>
<td>10 to &lt; 35 [2]</td>
</tr>
<tr>
<td>oak-gum-cypress</td>
<td><em>Q.-Nyssa-spp.-Taxodium distichum</em></td>
<td>35 to &gt; 200 [18]</td>
</tr>
<tr>
<td>coast live oak</td>
<td><em>Q. agrifolia</em></td>
<td>&lt; 35 to 200 [1]</td>
</tr>
<tr>
<td>canyon live oak</td>
<td><em>Q. chrysolepis</em></td>
<td>&lt; 35 to 200</td>
</tr>
<tr>
<td>blue oak-foothills pine</td>
<td><em>Q. douglasii-Pinus sabiana</em></td>
<td>&lt; 35 [1]</td>
</tr>
<tr>
<td>northern pin oak</td>
<td><em>Q. ellipsoidalis</em></td>
<td>&lt; 35 [2]</td>
</tr>
<tr>
<td>Oregon white oak</td>
<td><em>Q. garryana</em></td>
<td>&lt; 35 [1]</td>
</tr>
<tr>
<td>bear oak</td>
<td><em>Q. ilicifolia</em></td>
<td>&lt; 35 &gt; [2]</td>
</tr>
<tr>
<td>California black oak</td>
<td><em>Q. kelloggii</em></td>
<td>5-30 [3]</td>
</tr>
<tr>
<td>bur oak</td>
<td><em>Q. macrocarpa</em></td>
<td>&lt; 10 [2]</td>
</tr>
<tr>
<td>oak savanna</td>
<td><em>Q. m./Andropogon gerardii-Schizachyrium scoparium</em></td>
<td>2-14 [3,2]</td>
</tr>
<tr>
<td>chestnut oak</td>
<td><em>Q. prinus</em></td>
<td>3-8</td>
</tr>
<tr>
<td>northern red oak</td>
<td><em>Q. rubra</em></td>
<td>10 to &lt; 35</td>
</tr>
<tr>
<td>post oak-blackjack oak</td>
<td><em>Q. stellata-Q. marilandica</em></td>
<td>&lt; 10</td>
</tr>
<tr>
<td>black oak</td>
<td><em>Q. velutina</em></td>
<td>&lt; 35</td>
</tr>
<tr>
<td>live oak</td>
<td><em>Q. virginiana</em></td>
<td>10 to &lt; 100 [2]</td>
</tr>
<tr>
<td>interior live oak</td>
<td><em>Q. wislizenii</em></td>
<td>&lt; 35 [1]</td>
</tr>
<tr>
<td>blackland prairie</td>
<td><em>Schizachyrium scoparium-Nassella leucotricha</em></td>
<td>&lt; 10</td>
</tr>
<tr>
<td>Fayette prairie</td>
<td><em>S. s.-Buchloe dactyloides</em></td>
<td>&lt; 10</td>
</tr>
<tr>
<td>little bluestem-grama prairie</td>
<td><em>S. s.-Bouteloua spp.</em></td>
<td>&lt; 35 [3]</td>
</tr>
<tr>
<td>redwood</td>
<td><em>Sequoia sempervirens</em></td>
<td>5-200 [1,30,31]</td>
</tr>
<tr>
<td>Forest Association</td>
<td>Common/Latin Names</td>
<td>Fire Return Interval</td>
</tr>
<tr>
<td>--------------------</td>
<td>-------------------</td>
<td>---------------------</td>
</tr>
<tr>
<td>Western redcedar-western hemlock</td>
<td>Thuja plicata-Tsuga heterophylla</td>
<td>&gt; 200 [1]</td>
</tr>
<tr>
<td>Eastern hemlock-yellow birch</td>
<td>Tsuga canadensis-Betula alleghaniensis</td>
<td>&gt; 200 [2]</td>
</tr>
<tr>
<td>Western hemlock-Sitka spruce</td>
<td>Thuja heterophylla-Picea sitchensis</td>
<td>&gt; 200</td>
</tr>
<tr>
<td>Mountain hemlock*</td>
<td>T. mertensiana</td>
<td>35 to &gt; 200 [1]</td>
</tr>
<tr>
<td>Elm-ash-cottonwood</td>
<td>Ulmus-Fraxinus-Populus spp.</td>
<td>&lt; 35 to 200 [13,2]</td>
</tr>
</tbody>
</table>

*Fire return interval varies widely; trends in variation are noted in the species summary

**Mean


**POST-FIRE REGENERATION STRATEGY**

Initial off-site colonizer (off-site, initial community); secondary colonizer (on-site or off-site seed sources).

**IMMEDIATE FIRE EFFECT ON PLANT**

More research is needed to determine the immediate effects of fire on bull thistle plants and seeds. Bull thistle may or may not be killed by fire. In south-central Idaho on a Douglas-fir site where bull thistle was present before prescribed burning, bull thistle frequency declined immediately following burning, then increased 3 years after the burn (Lyon 1971). Musk thistle, a biennial thistle with a similar life history, may be killed by high-severity fires that kill the root crown, but may survive low-severity fires. It has been suggested that combustion would only readily take place on mature thistle plants, from which seed would have already dispersed (Popay and Medd 1990).

It is also unclear what effects fire has on bull thistle seeds in the soil. Incidents of rapid colonization after fire (Arno 1996, Arno 1999, Ashton 1981, Benson and Kurth 1995, Messinger 1974) suggest that either bull thistle seeds were present in the soil at the time of the fire and survived to germinate after the overstory was removed, or that bull thistle seeds were dispersed after fire from off-site seed sources. However, when experimental heat treatments including 6 combinations of temperature, duration, and soil moisture were applied to bull thistle seeds from an old-growth Douglas-fir forest seed bank, researchers concluded that even low-severity fire could kill bull thistle seeds. Seed survival was lower in wet soil than in dry soil. In wet soil, 35% of the bull thistle seeds tested survived 50 °C for 60 minutes, and 0 seeds survived 75 °C or 100 °C for 15 minutes. In dry soil, 44% survived 50 °C for 60 minutes, 32% survived 75 °C for 15 minutes, and 6% survived 100 °C for 15 minutes (Clark and Wilson 1994).
PLANT RESPONSE TO FIRE


Fire creates conditions that are favorable to the establishment of bull thistle (i.e. open canopy, reduced competition, areas of bare soil), so if bull thistle seeds are present and competition is minimal, bull thistle may be favored in the post-fire community. Bull thistle densities increased dramatically after a prescribed burn in Yosemite Valley, leading managers to believe that burning promotes thistle populations. It is unclear, however, whether prescribed burning alone caused the increase in bull thistle cover (Randall 1991).

DISCUSSION AND QUALIFICATION OF PLANT RESPONSE

Response of bull thistle to fire depends on the conditions of the fire such as fire severity, time of burning, prior and subsequent weather conditions (D’Antonio 2000), site conditions (e.g. soil moisture content) and composition of the preburn community and seedbank.

Observations in tallgrass prairie sites in South Dakota indicate that late spring prescribed burning on a 4- to 5-year rotation encourages the growth of native plants and discourages the growth of Canada thistle (*Cirsium arvense*), musk thistle, and bull thistle (Dailey 2001, Rice and Randall 2001). Additionally, Hulbert (1986) suggests that late spring burning in these ecosystems results in fewer forbs but greater grass production than fall or early spring burning.

A spring prescribed fire following clearcutting in 1968 on Miller Creek in western Montana (when the lower half of the duff was still wet from snowmelt and rain) left a continuous, intact duff mantle as a seedbed and killed the aerial portions of understory herbs and shrubs. Forest succession then began with regrowth of heartleaf arnica (*Arnica cordifolia*) and beargrass (*Xerophyllum tenax*) and establishment of the offsite colonizers fireweed (*Epilobium angustifolium*) and bull thistle. Other sites in the area that were harvested during the same time period but either unburned or burned in summer or fall did not have bull thistle in the postdisturbance plant community (Shearer and Stickney 1991).

The Research Project Summary Vegetation response to restoration treatments in ponderosa pine-Douglas-fir forests provides information on prescribed fire and post-fire response of plant community species, including bull thistle, that was not available when this species review was originally written.
More research is needed on short- and long-term secondary effects of fire on bull thistle. See “Post-fire colonization potential” below for more details.

FIRE MANAGEMENT CONSIDERATIONS

Fire as a control agent: Research is needed regarding the potential of prescribed burning to control bull thistle. Observations in tallgrass prairie sites in South Dakota indicate that a program of prescribed burning designed to simulate the historic fire regime encourages the growth of native plants and discourages the growth of invasive thistles (Dailey 2001, Rice and Randall 2001). However, poorly timed grazing (i.e. early in the growing season) can potentially negate beneficial effects of prescribed fire on these sites (Dailey 2001).

Post-fire colonization potential: General precautions should be followed to prevent bull thistle establishment after fire. The USDA Forest Service’s “Guide to noxious weed prevention practices” (USDA Forest Service 2001) provides several fire management considerations for weed prevention in general that can be applied to bull thistle. Fire managers might consider including weed prevention education and providing weed identification aids during fire training; avoiding known weed infestations when locating firelines, monitoring camps, staging areas, and helibases to be sure they are kept weed free; taking care that equipment is weed free; incorporating cost of weed prevention and management into fire rehabilitation plans; and acquiring restoration funding (USDA Forest Service 2001). Careful post-fire vigilance to identify and record the establishment of new populations is critical. About 1 month after fire, survey for signs of new or resprouting weeds. Repeated surveys will be needed, with the frequency and intensity guided by local conditions (Asher et al. 2001).

Potential weed problems must be addressed during pre-fire planning of prescribed burns, and following both wild and prescribed fires. When planning a prescribed burn, preinventory the project area and evaluate cover and phenology of any bull thistle present on or adjacent to the site, and evaluate the potential for increased bull thistle populations in the area (Asher et al. 2001). Avoid ignition and burning in areas at high risk for weed establishment or spread, and/or plan for follow-up treatments in succeeding years. Avoid creating soil conditions that promote weed germination and establishment. Discuss weed status and risks in burn rehabilitation plans (USDA Forest Service 2001). To prevent infestations, re-establish vegetation on bare ground as soon after fire as possible, using either natural recovery or artificial techniques as appropriate to site conditions and objectives. When reseeding after wildfires and prescribed burns, use only certified weed-free seed. Monitor the burn site and associated disturbed areas after the fire and the following spring for emergence of bull thistle, and treat to eradicate any emergent bull thistle plants. Regulate human, pack animal, and livestock entry into burned areas at risk for weed invasion until desirable site vegetation has recovered sufficiently to resist weed invasion. Additional guidelines and specific recommendations and requirements are available (Asher et al. 2001, Goodwin and Sheley 2001, USDA Forest Service 2001).
LITERATURE CITED


FIRE ECOLOGY OR ADAPTATIONS

Miner’s-lettuce has long-lived seeds that are stored in the soil (Stickney 1993) and germinate following fire (Toth 1991). It is a prolific seeder (Lawrence 1966); mass flowering in the years immediately following a fire recharges the seedbank (Stickney 1993). Miner’s-lettuce can develop high cover on exposed soil in full sun (Steele and Geier-Hayes 1992).

POST-FIRE REGENERATION STRATEGY

Ground residual colonizer (on-site, initial community), secondary colonizer - on-site seed.

IMMEDIATE FIRE EFFECT ON PLANT

Miner’s-lettuce is probably killed by fire.
PLANT RESPONSE TO FIRE

Miner’s-lettuce was present in the first growing season after the stand-destroying Marble-Cone wildfire in the Santa Lucia Range of California in August 1977. Peak cover was reached in post-fire year 2 and declined by post-fire year 3. Percent frequency of miner’s-lettuce on two study sites that had been dominated by Coulter pine follows (Griffin 1982):

<table>
<thead>
<tr>
<th>Site</th>
<th>1978</th>
<th>1979</th>
<th>1980</th>
</tr>
</thead>
<tbody>
<tr>
<td>Chews Ridge site 1</td>
<td>9</td>
<td>36</td>
<td>8</td>
</tr>
<tr>
<td>Chews Ridge site 2</td>
<td>7</td>
<td>48</td>
<td>2</td>
</tr>
</tbody>
</table>

Miner’s-lettuce is common in recently burned chaparral (Keeley and Keeley 1986). A year after a fire in chaparral in the Sierra Nevada foothills, miner’s-lettuce had high seed production on moist north-east slopes. Post-fire cover quickly exceeded pre-fire levels (Lawrence 1966). Miner’s-lettuce was also present the year following a severe fire in a chaparral riparian zone in the Los Padres National Forest, California, but its frequency was reduced by post-fire year 2 (Davis et al. 1989).

Miner’s-lettuce is also common after fire in more northern portions of its range. It was present in the first growing season after a fall wildfire in ponderosa pine (*Pinus ponderosa*) stands in the Selway-Bitterroot Wilderness, Idaho, and had increased in frequency by post-fire year 3 (Merrill et al. 1980). In burned ponderosa pine shelterwood cut units in Idaho, miner’s-lettuce was present in post-fire year 1 on sites burned with dry fuels, but was not present on sites burned with moist fuels. It also was not present in the preburn vegetation or in unburned control plots (Simmerman et al. 1991).

Miner’s-lettuce was present in the first growing season following the stand-destroying Pattee Canyon wildfire in a Douglas-fir (*Pseudotsuga menziesii*)/ninebark (*Physocarpus malvaceus*) habitat type in west-central Montana (Crane et al. 1983). It was still present in the herbaceous layer 10 years later (Toth 1991).

Basin big sagebrush (*Artemisia tridentata* spp. *tridentata*)-Idaho fescue (*Festuca idahoensis*)-bluebunch wheatgrass communities at the John Day Fossil Beds National Monument in eastern Oregon were burned in the spring and fall. Although not in the preburn vegetation, miner’s-lettuce was present in trace amounts (less than 2% frequency) the summer after the fall prescribed fire. It was not present after the spring fire or in control plots (Sapsis 1990).

Miner’s-lettuce establishes after fire in disturbed and climax grasslands in southeastern Washington (Daubenmire 1975).
FIRE MANAGEMENT CONSIDERATIONS

Rapid growth of miner’s-lettuce after fire in chaparral in the Sierra Nevada foothills contributes to an increased food supply for flocking bird species such as mourning dove and western meadowlark (Lawrence 1966).

LITERATURE CITED


Griffin, James R. 1982. Pine seedlings, native ground cover, and *Lolium multiflorum* on the Marble Cone burn, Santa Lucia Range, California. Madrono 29(3):177-188.


*Clintonia uniflora*

Queencup beadlily

FIRE ECOLOGY OR ADAPTATIONS

Fire adaptations: Queencup beadlily is an onsite survivor that regrows from underground rhizomes (Stickney 1989). As of this writing, there are no reports in the literature of queencup beadlily establishing from seed within a year after a fire.

<table>
<thead>
<tr>
<th>Community or Ecosystem</th>
<th>Dominant Species</th>
<th>Fire Return Interval Range (years)</th>
</tr>
</thead>
<tbody>
<tr>
<td>silver fir-Douglas-fir</td>
<td><em>Abies amabilis-Pseudotsuga menziesii var. menziesii</em></td>
<td>&gt; 200</td>
</tr>
<tr>
<td>grand fir</td>
<td><em>A. grandis</em></td>
<td>35-200 [1]</td>
</tr>
<tr>
<td>western larch</td>
<td><em>Larix occidentalis</em></td>
<td>25-350 [2,3,4]</td>
</tr>
<tr>
<td>Engelmann spruce-subalpine fir</td>
<td><em>Picea engelmannii-A. lasiocarpa</em></td>
<td>35 to &gt; 200 [1]</td>
</tr>
<tr>
<td>Rocky Mountain lodgepole pine*</td>
<td><em>Pinus contorta var. latifolia</em></td>
<td>25-340 [5,3,6]</td>
</tr>
<tr>
<td>western white pine*</td>
<td><em>P. monticola</em></td>
<td>50-200 [1]</td>
</tr>
<tr>
<td>interior ponderosa pine*</td>
<td><em>P. ponderosa var. scopulorum</em></td>
<td>2-30 [1,7,8]</td>
</tr>
<tr>
<td>Rocky Mountain Douglas-fir*</td>
<td><em>Pseudotsuga menziesii var. glauca</em></td>
<td>25-100 [1,9,10]</td>
</tr>
<tr>
<td>coastal Douglas-fir*</td>
<td><em>P. menziesii var. menziesii</em></td>
<td>40-240 [1,11,12]</td>
</tr>
<tr>
<td>western redcedar-western hemlock</td>
<td><em>Thuja plicata-Tsuga heterophylla</em></td>
<td>&gt; 200</td>
</tr>
<tr>
<td>western hemlock-Sitka spruce</td>
<td><em>T. heterophylla-Picea sitchensis</em></td>
<td>&gt; 200</td>
</tr>
<tr>
<td>mountain hemlock*</td>
<td><em>T. mertensiana</em></td>
<td>35 to &gt; 200 [1]</td>
</tr>
</tbody>
</table>

*fire return interval varies widely; trends in variation are noted in the species review


POST-FIRE REGENERATION STRATEGY (Stickney 1991)

Rhizomatous herb, rhizome in soil.

IMMEDIATE FIRE EFFECT ON PLANT

Fire top-kills (Stickney 1991) or kills queencup beadlily.

DISCUSSION AND QUALIFICATION OF FIRE EFFECT

The extent of damage inflicted on queencup beadlily depends on several factors including root and rhizome depth, fire severity, and whether the plant is rooted in soil or organic material (Hartford and Frandsen 1992, Smith and Fischer 1997, Stickney 1991). Fires with high degree of soil heating, long durations, or those that occur in areas with low moisture content can cause greater mortality (DeByle 1981, Hartford and Frandsen 1992, Simmerman et al. 1991).

PLANT RESPONSE TO FIRE

Queencup beadlily typically declines in frequency and coverage due to fire (Hamilton and Peterson 2003, Miller and Miller 1976, Neiland 1958, Simmerman et al. 1991, Vogl and Ryder 1969). Declines can be small and short lived. For example, 1 year after a fall experimental burn in a cold, wet area of the Engelmann spruce-subalpine fir zone of south-central British Columbia, mean queencup beadlily coverage dropped from 2.13% before fire to 0.61%.
However, subsequent monitoring after 2, 3, 5, and 11 post-fire years showed queencup beadlily coverage was equal to or greater than pre-fire coverage (Hamilton and Peterson 2003). On a western larch-Douglas-fir site in western Montana, Halvorson (1982) listed queencup beadlily as a common species 4 years after a fire on damp fuels that did not affect 65% of the vegetation and only charred the duff in affected areas. However, occurrence of queencup beadlily before the fire was unknown.

Much larger and lasting effects of fire have also been reported for queencup beadlily. For instance, 10 years after the Tillamook Fire in northwestern Oregon, frequency of queencup beadlily was 6% in a burned area compared to 68% within an island of unburned Douglas-fir, western hemlock, and western redcedar forest (Neiland 1958). Miller and Miller (1976) listed queencup beadlily as an herb typical of unburned western hemlock-Douglas-fir-western redcedar stands, but reported it as absent from burned areas during 3 years of post-fire monitoring after lightening started several wildfires in the North Cascades National Park of north-central Washington. After the Sundance wildfire in northern Idaho, an area containing Douglas-fir, western larch, western redcedar, and western hemlock took 10 or more years for queencup beadlily to reach coverages of at least 1% on many sites (Stickney and Campbell 2000). Again, coverages before the fire were unknown. It is possible that light and moisture conditions after severe fires are unfavorable for queencup beadlily persistence or establishment.

DISCUSSION AND QUALIFICATION OF PLANT RESPONSE

Much of the above variation may be explained by the severity of the fire and moisture content of the forest floor at the time of the fire. Neiland (1958) was investigating the Tillamook Burn area, a region exposed to 3 severe fires within 12 years. The Sundance Burn was also a severe wildfire (Stickney 1986), while queencup beadlily responses reported by Halvorson (1982) and Hamilton and Peterson (2003) were after low-severity fires. Hamilton and Peterson’s study design included logging followed by no burn, spring burn, and fall burn treatments. Although both were low-severity fires, drier conditions occurred on the spring burn. This fire resulted in 24.9% duff reduction compared to 10.4% duff reduction in the fall burn. The dry, spring burn had the largest effect on queencup beadlily. Coverage dropped from 2.30% before treatment to 0.23% the year after treatment. After 11 years queencup beadlily coverage at the site (1.68%) had yet to reach pretreatment level (2.30%). It is important to note that the treatments in this study were pseudoreplicated. Thus, results should not be extrapolated to other areas.
Simmerman and others (1991) investigated the response of vegetation after a shelterwood cut and 3 burn treatments (dry burn, moist burn, and a no burn control) on a Priest River Experimental Forest site in northern Idaho. Overstory was about 51% Douglas-fir, 33% ponderosa pine, and the remainder a mixture of western larch, western redcedar, and western white pine. They also found a larger effect from a dry burn. Queencup beadlily coverage was reduced from 2.4% to 0.3% compared to a reduction from 3.0% to 1.4% on the moist burn and a decrease from 3.6% to 2.7% on the site where only the shelterwood cut was performed.

In addition to fire severity and moisture content at the time of the burn, the substrate in which queencup beadlily is rooted can have a large effect. Queencup beadlily can root in soil or in organic layers (Sharpe 1956, Stickney 1991). Plants rooted in organic layers are much more susceptible to fire (Smith and Fischer 1997, Stickney 1991). The rhizomes and roots of plants growing within the organic layer are consumed as a fire burns through, while those in the soil are protected. Rooting depth in the soil is also an important factor, as rhizomes and roots that are deeper are more protected from fire (Hartford and Frandsen 1992).

FIRE MANAGEMENT CONSIDERATIONS

A decline in queencup beadlily coverage and frequency can be expected after fire. The extent and duration of the effect depends on many factors including fire severity, rooting depth and substrate, and site conditions, such as moisture content of the organic mantle, before and after the fire. Effects on queencup beadlily may be mitigated by performing low-severity prescribed burns when the forest floor is moist (Hamilton and Peterson 2003, Simmerman et al. 1991).

LITERATURE CITED


Hartford, Roberta A. and Frandsen, William H. 1992. When it’s hot, it’s hot...or maybe it’s not! (Surface flaming may not portend extensive soil heating). International Journal of Wildland Fire 2(3):139-144.


**POST-FIRE REGENERATION STRATEGY**

Ground residual colonizer (on-site, initial community); secondary colonizer - on-site seed; and secondary colonizer - off-site seed.
IMMEDIATE FIRE EFFECT ON PLANT

Fire generally kills pink corydalis.

PLANT RESPONSE TO FIRE


The species is generally not found in areas that have not been recently burned, although seed may be present in the soil (Ahlgren 1979, Krefting and Ahlgren 1974). Pink corydalis has been reported to increase in lightly, moderately, and heavily burned areas (Dymness et al. 1986, Viereck and Dymness 1979). In one case, it grew better after a summer fire (on a warm, dry forest floor) than after a spring fire (on a cool, wet floor) (Ohmann and Grigal 1981). In another case, seedlings of pink corydalis and of Carolina geranium (Geranium carolinianum) emerged soon after a severe fire in the Appalachian Mountains of New Jersey and were well established 1 year later (Torrey 1932).

FIRE MANAGEMENT CONSIDERATIONS

Pink corydalis requires fire to thrive. Since the species declines 3 to 5 years after a fire, it is likely to become uncommon in areas where fires are suppressed. Controlled burning in areas where this species was formerly present would likely cause it to increase if viable seed were still present in the soil. Recently burned areas may be monitored for this species to determine whether the potential for viable populations remains.

LITERATURE CITED


*Dactylis glomerata*

Orchardgrass

POST-FIRE REGENERATION STRATEGY

Tussock graminoid; caudex, growing points in soil; ground residual colonizer (onsite, initial community); secondary colonizer - offsite seed.

IMMEDIATE FIRE EFFECT ON PLANT

In general, bunchgrasses with large accumulations of dead material can generate high temperatures for long periods of time after the fire has passed. This can reduce fire survival for older plants (Wright and Bailey 1982).

PLANT RESPONSE TO FIRE

Orchardgrass is reported to increase or remain stable after burning (Cocking *et al.* 1979, Pase and Granfelt 1977).

FIRE MANAGEMENT CONSIDERATIONS

Orchardgrass is frequently seeded onto areas disturbed by fire to control soil erosion. Concern has been raised that the increase of grass species in the area, especially summer-dormant grasses such as orchardgrass, could increase the risk of fast-spreading, low-intensity fires that could set back the rate of tree and shrub regeneration. The application of seed to reduce erosion is, therefore, not always beneficial (Crane *et al.* 1983, Helvey and Fowler 1979).

Orchardgrass mixtures are recommended in the conversion of chaparral to grassland to reduce fire intensity and frequency (Bentley 1967).

LITERATURE CITED


Danthonia intermedia
Timber oatgrass

FIRE ECOLOGY OR ADAPTATIONS

In the Pacific Northwest, timber oatgrass is “moderately resistant” to fire (Volland and Dell 1981). Post-fire regeneration occurs through seed and recovery is generally complete within 5 to 10 years (Lent 1984, Volland and Dell 1981).

POST-FIRE REGENERATION STRATEGY

Tussock graminoid; initial-offsite colonizer (off-site, initial community).

IMMEDIATE FIRE EFFECT ON PLANT

Timber oatgrass is often top-killed by fire. However, some individuals may survive (Volland and Dell 1981).
PLANT RESPONSE TO FIRE

Timber oatgrass is “moderate” in post-fire regeneration response in the Pacific Northwest, (having a 35 to 64% chance that at least 50% of the population will survive or reestablish after fire). In the Pacific Northwest, it takes 5 to 10 years to approximate preburn frequency or coverage (Volland and Dell 1981). In Washington, levels of timber oatgrass and other grasses were not affected immediately after fire in a timbered area. However, several grasses, including timber oatgrass, began to increase 2 years after the burn. By the fifth year after fire, timber oatgrass was the dominant grass on burned plots (Minore et al. 1979).

Timber oatgrass increased after a mid-summer fire in the Pacific Northwest (Volland and Dell 1981). Two seasons after a May burn in northwestern Montana, timber oatgrass cover increased by 7.5% over preburn levels (Bushey 1985).

LITERATURE CITED


Danthonia spicata
Mountain oatgrass

FIRE ECOLOGY OR ADAPTATIONS

Mountain oatgrass occurs in oak (*Quercus* spp.) woods and grassy mountain meadows which occasionally experience either lightning or human-caused fire. Lightning commonly strikes the peaks and ridges of the southern Appalachian Mountains from April through August (Barden and Woods 1974). Grassy balds in the Pisgah National Forest of North Carolina respond favorably to fire, becoming thick and lush. The low-severity fires burn very little of the surface detritus (Lindsay and Bratton 1979). Mountain oatgrass basal buds and dormant seeds probably survive low-severity fire.

POST-FIRE REGENERATION STRATEGY

Tussock graminoid; ground residual colonizer (on-site, initial community); secondary colonizer - off-site seed.

IMMEDIATE FIRE EFFECT ON PLANT

Fire probably kills the culms and leaves of mountain oatgrass.

PLANT RESPONSE TO FIRE

Dormant seeds in the soil or litter germinate after fire. Mountain oatgrass seedlings began growing 1 week after a spring fire on Gregory Bald in the Great Smoky Mountains National Park (Lindsay and Bratton 1979). Poverty grass, a close relative of mountain oatgrass, regenerated from a seedbank after fire on an upland site in Michigan (Scheiner 1988).

FIRE MANAGEMENT CONSIDERATIONS

Prescribed fire is used to maintain grassy balds in the southern Appalachian Mountains. Woody species are invading many grassy balds because of fire suppression and decreased grazing (Lindsay and Bratton 1979). Prescribed burning was as effective as mowing in preventing woody species establishment in Big Meadows, Shenandoah National Park, Virginia.

Biomass in the area prescribed burned in April was equal to the unburned control by the end of the summer (Cocking *et al.* 1979).

Grassy balds respond well to fall fires. The fuel is less compact and favorable weather conditions last longer in the fall than in the spring (Lindsay and Bratton 1979).
LITERATURE CITED


*Deschampsia cespitosa*
Tufted hairgrass

FIRE ECOLOGY OR ADAPTATIONS

Tufted hairgrass generally survives all but the most severe fires (DeBenedetti and Parsons 1979). It usually sprouts from the root crown after aerial portions are burned. Tufts formed by the leaves (Great Plains Flora Association 1986) often protect basal buds from fire damage. Tufted hairgrass seeds occur in the seedbank (Chambers 1993) and after fire tufted hairgrass may regenerate from soil-stored seed.

POST-FIRE REGENERATION STRATEGY

Tussock graminoid; ground residual colonizer (on-site, initial community).

IMMEDIATE FIRE EFFECT ON PLANT

Tufted hairgrass culms and leaves are often killed by fire, though dense tufts may protect some green biomass during low-severity fire. Tufted hairgrass root crowns usually survive all but the most severe fires (DeBenedetti and Parsons 1984).
PLANT RESPONSE TO FIRE

Within just a few years tufted hairgrass usually recovers to pre-fire levels (DeBenedetti and Parsons 1984).

In the Medicine Bow Mountains of Wyoming, tufted hairgrass occurs in wet or dry subalpine meadows that were produced when forests were burned in 1871. Above 3000 m elevation burned areas remain open for 50 to 100 years after stand-replacing fire. After a century or more, the drier meadows usually are covered by young spruce (*Picea*)-fir (*Abies*) forests and tufted hairgrass declines. However, tufted hairgrass in wet meadows above 3000 m may remain dominant for centuries (Billings 1969).

In the same area, tufted hairgrass is a component of successional tundra meadow that developed following a severe 1809 crown fire in ribbon forest (Billings 1969).

LITERATURE CITED


Blue wildrye can survive fire. It typically forms small bunches that rarely exceed 10 cm in diameter, and mature aboveground growth generally consists of coarse leaves and stems (Hitchcock et al. 1969, Vallentine 1961). Such attributes suggest that this bunchgrass burns rather quickly, with little heat transferred down into the root crown (Wright and Bailey 1982). As a result, basal buds located at or just below the ground surface are not subjected to prolonged heating, and may survive and sprout. In Idaho, Wyoming, and Utah, blue wildrye is survives fire by sprouting from the root crown and establishing from on-site seeds (Bradley et al. 1992, Simmerman et al. 1991).

Because blue wildrye is a short-lived perennial that generally does not compete well with surrounding vegetation, severity and frequency of fire or other types of disturbance greatly influence the recovery and maintenance of this species. In northern Idaho, Mueggler (1965) observed highest frequencies of blue wildrye on sites that had been subjected to multiple broadcast burns 2 or more times in the previous 30 years.

Blue wildrye occurs in plant communities with varying fire regimes. The range of fire intervals reported for some species that dominate communities where blue wildrye occurs are listed below.

<table>
<thead>
<tr>
<th>Community dominant</th>
<th>Range (yrs)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Pacific ponderosa pine</td>
<td>1-40</td>
</tr>
<tr>
<td>(Pinus ponderosa var. ponderosa)</td>
<td></td>
</tr>
<tr>
<td>Rocky Mountain lodgepole pine</td>
<td>25-300+</td>
</tr>
<tr>
<td>(P. contorta var. latifolia)</td>
<td></td>
</tr>
<tr>
<td>Rocky Mountain Douglas-fir</td>
<td>40-140</td>
</tr>
<tr>
<td>(Pseudotsuga menziesii var. glauca)</td>
<td></td>
</tr>
<tr>
<td>quaking aspen</td>
<td>7-80</td>
</tr>
<tr>
<td>(Populus tremuloides)</td>
<td></td>
</tr>
<tr>
<td>chamise</td>
<td>2-90</td>
</tr>
<tr>
<td>(Adenostoma fasciculatum)</td>
<td></td>
</tr>
</tbody>
</table>
POST-FIRE REGENERATION STRATEGY

Ground residual colonizer (on-site, initial community); initial off-site colonizer (off-site, initial community); secondary colonizer - on-site seed; tussock graminoid; surface rhizome/chamaephytic root crown.

IMMEDIATE FIRE EFFECT ON PLANT

Blue wildrye mortality following fire has not been widely documented. Indirect evidence indicates that it may be somewhat susceptible to fire. Leege and Godbolt (1985) reported reduced frequencies of blue wildrye 1 year after a spring burn in seral brushfields in a grand fir/pachistima (Abies grandis/Pachistima myrsinites) habitat type in north-central Idaho. However, blue wildrye densities showed little change after fire on chaparral sites in California where nonsprouting forms of manzanita (Arctostaphylos spp.) and ceanothus (Ceanothus spp.) comprised most of the pre-fire overstory vegetation (Sampson 1944).

PLANT RESPONSE TO FIRE

Blue wildrye is often “an abundant and characteristic species of old burns and cutover areas” (Sampson 1944, USDA Forest Service 1937, Vallentine 1961), but few studies have dealt specifically with the post-fire response of this species. Limited information indicates that blue wildrye depends in part on residual plant survival and subsequent seed regeneration for post-fire establishment. Powell (1994) reports that that fire creates an excellent seedbed following moderate-severity burns in mixed-conifer forests, and that most post-fire regeneration in those forests may be from surviving seedbank propagules.

Tillering can occur from surviving basal buds located on the root crown. Plants in the Great Plains may also regenerate via short rhizomes (Great Plains Flora Association 1986). Blue wildrye in the Pacific Northwest is rarely rhizomatous (Barkworth 2000, Gage 2000, Hitchcock and Cronquist 1973).

DISCUSSION AND QUALIFICATION OF PLANT RESPONSE

Results of laboratory experiments suggest that blue wildrye seeds may be able to survive ground surface temperatures generated by moderate-intensity fires. Approximately 39% of seeds exposed for 5 minutes to temperatures ranging from 82-93º C germinated, and 17% exposed to temperatures ranging from 93-99º C germinated. This research indicates that on sites where blue wildrye occurs in the pre-fire community, viable seed may be available for establishment in the first post-fire growing season (Sampson 1944).

Although blue wildrye was not recorded as a component of the preburn vegetation on seral brushfield sites in northern Idaho, it was recorded on sample plots the fourth growing season after burning (Leege and Godbolt 1985) and in both pre- and post-fire quaking aspen stands in Colorado.
Fire creates seedbeds that appear to be conducive to the successful germination and rapid establishment of blue wildrye (Carter and Law 1948, Hanson and Whitman 1938, Raunkiaer 1934, Stickney 1989). Seedlings develop rapidly on sites where competing vegetation is greatly reduced. In general, cover of blue wildrye increases for the first few years following fire (Brown and DeByle 1989), but abundance and vigor may decline after 3 or 4 years. On broadcast-seeded burns in the mountain-brush zone of Utah, blue wildrye established readily and gave high yields for 4 post-fire years, but then was suppressed by smooth brome (Bromus inermis) (Frischknecht and Plummer 1955). Mean density of blue wildrye in a California chaparral community of nonsprouting manzanita (Arctostaphylos spp.) and ceanothus (Ceanothus spp.) is presented below (Sampson 1944):

<table>
<thead>
<tr>
<th>Post-fire year</th>
<th>pre-fire</th>
<th>1</th>
<th>2</th>
<th>3</th>
<th>4</th>
<th>5</th>
</tr>
</thead>
<tbody>
<tr>
<td>plants/thousandth acre</td>
<td>0.6</td>
<td>0.8</td>
<td>1.8</td>
<td>2.6</td>
<td>3.9</td>
<td>4.7</td>
</tr>
</tbody>
</table>

**FIRE MANAGEMENT CONSIDERATIONS**

Blue wildrye is recommended for seeding mixtures for revegetating burn sites because it exhibits good germination and establishes rapidly (Frischknecht and Plummer 1955, Sampson 1944). To reintroduce blue wildrye and associated grassland species native to California, McClaran (1981) recommends site preparation either by tillage or fire, which should be timed in accordance with the emergence of exotic annual seedlings. In McClaran’s study, previously unnoticed remnant natives including blue wildrye showed a flush of growth and an increase in seed set in response to burning. However, blue wildrye may be quickly suppressed by other commonly seeded species. On a site in a grand-fir/pachistima habitat type in north-central Idaho, blue wildrye occurred in pretreatment stands but was essentially eliminated from the burn and seed treatment plots within 1 year. Increased competition from seeded species may have been responsible for its decline. Four years after treatment, 46% of the total herbage production on this site consisted of seeded orchard grass (Dactylis glomerata) and slender wheatgrass (Leege and Godbolt 1985).

Blue wildrye was among 5 grasses measured for canopy coverage before and after moist-fuel and dry-fuel underburns in an Idaho ponderosa pine forest. The underburns were conducted in experimental shelterwood logging units. No-burn, moist-burn, and dry-burn treatments represented a progression of heat treatments on the soil and surface vegetation. Total woody fuel consumption was 24% in the moist burn and 57% in the dry burn. Duff reduction ranged from a low of 10% in the moist sites to a high a 90% in the dry sites. Preburn vegetation was measured prior to logging. In the moist-burn treatment all grass cover, including blue wildrye, was similar to the original vegetation. Also in the moist-burn treatment, blue wildrye responded similarly to the other grasses by showing an increase in canopy cover the summer following burning.
In the dry-burn treatment, unlike the other grasses, blue wildrye had disappeared from the test plots by the summer after burning. Canopy cover of grasses other than blue wildrye was reduced or maintained, with the exception of rhizomatous pinegrass (*Calamagrostis rubescens*), which increased (Simmerman *et al.* 1991).

In a Wyoming study on a quaking aspen site, blue wildrye was among 3 dominant grasses tested for seasonal changes in understory live fuel moisture. Other dominant grasses were California brome and slender wheatgrass. Percent moisture content decreased “as expected” during the summers of 1981 and 1982, but 1982 had 6 times more precipitation than the previous year, leading to considerable variation in overall fuel moisture between the 2 years. Also, in the drier year, moisture content of the grasses averaged 41% higher in a closed stand than in the adjacent open stand, with curing time lagging behind the open stand by 3 weeks (Brown *et al.* 1989).

Although blue wildrye forage quality generally improves during the first post-fire growing season (DeByle *et al.* 1989), this may not always occur. In a Wyoming study of the effect of prescribed burning on nutritional status (crude protein and in vitro dry matter) of understory species in quaking aspen, nutritional content of blue wildrye did not differ between burned and unburned plots 3 years after prescribed burning (Canon *et al.* 1987).

**LITERATURE CITED**


_Elymus repens (= Agropyron repens)_

Quackgrass

FIRE ECOLOGY OR ADAPTATIONS

Quackgrass is adapted to certain seasonal fires because of its rhizomes.

POST-FIRE REGENERATION STRATEGY

Rhizomatous herb, rhizome in soil.

IMMEDIATE FIRE EFFECT ON PLANT

Late spring fires generally reduce quackgrass cover, flowering and biomass, while early spring fires can increase these.
DISCUSSION AND QUALIFICATION OF FIRE EFFECT

A May burn in oak savannas of Wisconsin significantly reduced quackgrass and halted flowering (Diboll 1986). Similar results (reduction in biomass and cover) have been shown for other areas (Halvorsen and Anderson 1983, Hughes 1985). Burning quackgrass on a biennial schedule for several years has been effective in eradicating this species (Anderson 1973, Bailey 1978).

PLANT RESPONSE TO FIRE

Quackgrass cover can increase following fire.

DISCUSSION AND QUALIFICATION OF PLANT RESPONSE

Five annual late April to early May burns in Minnesota resulted in a decrease in quackgrass height but an increase in cover (Becker 1989). Plant vigor was reduced and flowering stopped, but quackgrass continued to spread into adjacent areas. At the time of the April burns, plant height was between 10-15 cm, and during the May burn, heights were between 15-25 cm. May and June burns on North Dakota grasslands “harmed” quackgrass in the first postburn season, but quackgrass recovered to almost preburn levels by the second postburn season. Following the late June fire, quackgrass showed a slight increase in cover, height, shoot density, production, and flowering (Olson 1975). Wisconsin grassland fires in March caused an increase in seed production by July and August (Halvorsen and Anderson 1983).

FIRE MANAGEMENT CONSIDERATIONS

Cool-season grasses such quackgrass are best eliminated with early spring burns (Glenn-Lewin et al. 1990, Kucera 1981, Linne 1978). Cool-season grasses can grow in the fall following summer dormancy; therefore, fall burns might also help reduce undesirable cool-season grasses (Risser et al. 1981).

LITERATURE CITED


Erodium cicutarium
Cutleaf filaree, Stork’s bill

FIRE ECOLOGY OR ADAPTATIONS
Seed driven into the soil by the styles is usually protected from fire (Young et al. 1976).

The prostrate stems of cutleaf filaree aid in spreading groundfire. Dead plants contribute to fuel loads.

POST-FIRE REGENERATION STRATEGY

Initial-offsite colonizer (off-site, initial community); secondary colonizer - off-site seed.

IMMEDIATE FIRE EFFECT ON PLANT

Moderate fire kills mature plants (Heady 1977). Grass fires are typically light to moderate, and very young seedlings can survive fires of that severity. Dennis (1981) found that newly germinated cutleaf filaree seedlings just beneath the litter layer were not harmed by a moderate grass fire in Mendocino National Forest, California. Cutleaf filaree seed in the litter layer remains viable following light fire, and seed just under the litter layer remains viable following moderate fire. Severe fire will kill seed unless it is buried 1.25 cm or more deep (Ryan and Noste 1985, Zedler et al. 1983).

PLANT RESPONSE TO FIRE

During the first post-fire growing season, density of cutleaf filaree is reduced, but biomass increases (Cave 1982). Seed production is highest at post-fire year 1, with cutleaf filaree populations peaking at post-fire year 2. Callison et al. (1985) reported cutleaf filaree as providing an absolute cover value of 0.2 percent in an unburned area in the Beaver Dam Mountains of southwestern Utah. Following a prescribed burn, the cover value was 11.1 percent in the first post-fire growing season, and 11.5 percent in the second. Cover value declined from post-fire year 3 and after. By post-fire year 12, cutleaf filaree was no longer visible in the plant community.

FIRE MANAGEMENT CONSIDERATIONS

Frequent prescribed burning favors cutleaf filaree and other forbs over annual grasses (Biswell and Gilman 1961, Heady 1977). This is desirable when the climax grass provides poor forage, such as ripgut brome. Grassland fire typically destroys very few seeds or other organic matter in the soil (Heady 1977). It does destroy the overlying mulch layer that inhibits germination of cutleaf filaree seeds (Biswell and Gilman 1961, Griffin 1974).
LITERATURE CITED


Idaho fescue is a small bunchgrass that can survive light-severity fires. It is usually harmed by more severe fire (Boyer and Dell 1980, Cattelino 1980, Smith and Busby 1981, Wright et al. 1979). Fires burning at 10- to 25- year intervals have neutral to negative effects on Idaho fescue (Agee 1996). Rapid tillering occurs when root crowns are not killed and soil moisture is favorable (Johnson et al. 1994, Robberecht and Defosse 1995). Plants may re-establish from seed after fire if temperatures are low enough to allow for survival of seed (Clark et al. 1994, Warg 1938).

Native ranges and forests in which Idaho fescue occurs have historically been subjected to fires at varying intervals. Native Americans were probably an important ignition source in prehistoric Idaho fescue grasslands (Agee 1996). Maintenance of grasslands in the Intermountain West is dependent, in part, on periodic fires to remove dry matter and invading shrubs and trees (Antos et al. 1983, Arno and Gruell 1986, Burkhardt and Tisdale 1976, Butler 1986, Koterba and Habeck 1971, Patten 1963). A decrease in or loss of dominant seral species such as Idaho fescue due to fire exclusion has been noted in many areas (Greene and Evenden 1996).

The following table provides some fire regime intervals for communities in which Idaho fescue occurs:

<table>
<thead>
<tr>
<th>Community or Ecosystem</th>
<th>Dominant Species</th>
<th>Fire Return Interval Range in Years</th>
</tr>
</thead>
<tbody>
<tr>
<td>silver fir-Douglas-fir</td>
<td><em>Abies amabilis-Pseudotsuga menziesii var. menziesii</em></td>
<td>&gt; 200</td>
</tr>
<tr>
<td>grand fir</td>
<td><em>A. grandis</em></td>
<td>35-200</td>
</tr>
<tr>
<td>California chaparral</td>
<td><em>Adenostoma and/or Arctostaphylos spp.</em></td>
<td>&lt; 35 to &lt; 100</td>
</tr>
<tr>
<td>sagebrush steppe</td>
<td><em>Artemisia tridentata/Pseudoroegneria spicata</em></td>
<td>20-70 [1]</td>
</tr>
<tr>
<td>basin big sagebrush</td>
<td><em>A. t. var. tridentata</em></td>
<td>12-43 [2]</td>
</tr>
<tr>
<td>mountain big sagebrush</td>
<td><em>A. t. var. vaseyana</em></td>
<td>5-15 [3]</td>
</tr>
<tr>
<td>Wyoming big sagebrush</td>
<td><em>A. t. var. wyomingensis</em></td>
<td>10-70 (40)**[4,3]</td>
</tr>
<tr>
<td>coastal sagebrush</td>
<td><em>A. californica</em></td>
<td>&lt; 35 to &lt; 100</td>
</tr>
<tr>
<td>cheatgrass</td>
<td><em>Bromus tectorum</em></td>
<td>&lt; 10</td>
</tr>
<tr>
<td>California montane chaparral</td>
<td><em>Ceanothus and/or Arctostaphylos spp.</em></td>
<td>50-100 [1]</td>
</tr>
<tr>
<td>curlleaf mountain-mahogany*</td>
<td><em>Cercocarpus ledifolius</em></td>
<td>13-1000 [5,6]</td>
</tr>
<tr>
<td>Vegetation Type</td>
<td>Species</td>
<td>Fire Return Interval</td>
</tr>
<tr>
<td>---------------------------------------</td>
<td>--------------------------------------</td>
<td>----------------------</td>
</tr>
<tr>
<td>Mountain-mahogany-Gambel oak scrub</td>
<td>C. l.-Quercus gambelli</td>
<td>&lt; 35 to &lt; 100</td>
</tr>
<tr>
<td>California steppe</td>
<td>Festuca-Danthonia spp.</td>
<td>&lt; 35</td>
</tr>
<tr>
<td>Western juniper</td>
<td>Juniperus occidentalis</td>
<td>20-70</td>
</tr>
<tr>
<td>Rocky Mountain juniper</td>
<td>J. scopulorum</td>
<td>&lt; 35</td>
</tr>
<tr>
<td>Western larch</td>
<td>Larix occidentalis</td>
<td>25-100</td>
</tr>
<tr>
<td>Engelmann spruce-subalpine fir</td>
<td>Picea engelmannii-Abies lasiocarpa</td>
<td>35 to &gt; 200</td>
</tr>
<tr>
<td>Black spruce</td>
<td>P. mariana</td>
<td>35-200</td>
</tr>
<tr>
<td>Pinyon-juniper</td>
<td>Pinus-Juniperus spp.</td>
<td>&lt; 35</td>
</tr>
<tr>
<td>Whitebark pine*</td>
<td>P. albicaulis</td>
<td>50-200 [1]</td>
</tr>
<tr>
<td>Rocky Mountain lodgepole pine*</td>
<td>P. contorta var. latifolia</td>
<td>25-300+ [7,8]</td>
</tr>
<tr>
<td>Colorado pinyon</td>
<td>P. edulis</td>
<td>10-49</td>
</tr>
<tr>
<td>Jeffrey pine</td>
<td>P. jeffreyi</td>
<td>5-30</td>
</tr>
<tr>
<td>Western white pine*</td>
<td>P. monticola</td>
<td>50-200</td>
</tr>
<tr>
<td>Pacific ponderosa pine*</td>
<td>P. ponderosa var. ponderosa</td>
<td>1-47</td>
</tr>
<tr>
<td>Rocky Mountain ponderosa pine*</td>
<td>P. p. var. scopulorum</td>
<td>2-10 [1]</td>
</tr>
<tr>
<td>Quaking aspen (west of the Great Plains)</td>
<td>Populus tremuloides</td>
<td>7-120 [1,9,10]</td>
</tr>
<tr>
<td>Mountain grasslands</td>
<td>Pseudoroegneria spicata</td>
<td>3-40 (10)** [7]</td>
</tr>
<tr>
<td>Rocky Mountain Douglas-fir*</td>
<td>Pseudotsuga menziesii var. glauca</td>
<td>25-100 [1]</td>
</tr>
<tr>
<td>Coastal Douglas-fir*</td>
<td>P. m. var. menziesii</td>
<td>40-240 [1,11,12]</td>
</tr>
<tr>
<td>California oakwoods</td>
<td>Quercus spp.</td>
<td>&lt; 35</td>
</tr>
<tr>
<td>Oak-juniper woodland (Southwest)</td>
<td>Q.-Juniperus spp.</td>
<td>&lt; 35 to &lt; 200</td>
</tr>
<tr>
<td>Western redcedar-western hemlock</td>
<td>Thuja plicata-Tsuga heterophylla</td>
<td>&gt; 200</td>
</tr>
<tr>
<td>Western hemlock-Sitka spruce</td>
<td>T. heterophylla-Picea sitchensis</td>
<td>&gt; 200 [1]</td>
</tr>
</tbody>
</table>

*Fire return interval varies widely; trends in variation are noted in the species summary
**Mean**

POST-FIRE REGENERATION STRATEGY (Stickney 1989)

Tussock graminoid ; secondary colonizer (on-site or off-site seed sources).

IMMEDIATE FIRE EFFECT ON PLANT

Idaho fescue grows in a dense, fine-leaved tuft. Fires tend to burn within the accumulated fine leaves at the base of the plant and may produce temperatures sufficient to kill some of the root crown (Agee 1996]. Mature Idaho fescue plants are commonly reported to be severely damaged by fire in all seasons (Boyer and Dell 1980, Cattelino 1980, Smith and Busby 1981, Wright et al. 1979). Initial mortality may be high (in excess of 75%) on severe burns, but usually varies from 20 to 50% (Barrington et al. 1988). Idaho fescue is commonly reported to be more sensitive to fire than bluebunch wheatgrass (Blaisdell 1953, Concannon 1978, Conrad and Poulton 1996, Johnson et al. 1994, Wright et al. 1979); however Robberecht and Defosse (1995), using special instrumentation to control the intensity and duration of fire treatment for individual plants, suggested the latter was more sensitive. They observed culm and biomass reduction with moderate fire severity in bluebunch wheatgrass, whereas a high fire severity was required for this reduction in Idaho fescue. Also, given the same fire severity treatment, post-fire culm production was initiated earlier and more rapidly in Idaho fescue (Robberecht and Defosse 1995).

DISCUSSION AND QUALIFICATION OF FIRE EFFECT

Fire effects vary with condition and size of the plant, season and severity of fire, and ecological conditions. Rapid burns leave little damage to root crowns, and new tillers are produced with onset of fall moisture (Johnson et al. 1994). This was the case with small bunches of Idaho fescue when a “hot” June wildfire caused incomplete burning of the root crowns on a western Montana grassland (Antos et al. 1983). After decades of fire exclusion and/or the absence of grazing, the thicker duff layer and dense, dry crowns burn longer, so heat penetrates deeper and may kill the plants (Armour et al.).
The dense culms may provide enough fuel to burn for hours after the fire has passed, reaching temperatures as high as 538º C, thus killing or seriously injuring the plant regardless of the intensity of the fire (Wright et al. 1979). Bunting (1984) also points out that Idaho fescue is sometimes rooted in the litter under western juniper (*Juniperus occidentalis*) which will, of course, burn.

**PLANT RESPONSE TO FIRE**

Most studies show that fire initially reduces frequency (Bushey 1985, Kuntz 1982, Schwecke and Hann 1989) and basal area (Beardall and Sylvester 1976, Schwecke and Hann 1989) of Idaho fescue. Recovery to pre-fire levels may require more than 25 years if density is severely reduced (Barrington et al. 1988), as was observed by Harniss and Murray (1973) in Idaho, where return to pre-fire cover took over 30 years after summer prescribed burning in a big sagebrush/Idaho fescue range. Effects of most fires are not, however, so extreme, and show a wide range of responses. Idaho fescue plants burned in Nevada in the spring of 1973 still showed signs of damage in 1976, but they were recovering (Beardall and Sylvester 1976). Idaho fescue frequency was not significantly (p < 0.05) different between burn and control plots 2 and 4 years after prescribed burning on a mountain big sagebrush/Idaho fescue site in Idaho (Mangan and Autenrieth 1985).

On a lodgepole pine site in Idaho, Idaho fescue plants had regained vigor by the third year and showed improved vigor by the fifth year (Phillips 1973). Idaho fescue resprouted after spring prescribed burns in central Oregon, and within 3 months more than 80% of Idaho fescue plants had vigorous growth, with greater production in burned areas than in adjacent unburned areas (Adams 1980). Forage biomass of burned Idaho fescue grassland had surpassed that of unburned grassland following fires in Yellowstone National Park in both grazed (by elk and some bison) and ungrazed portions, 2 years after burning (Singer and Harter 1996).

Idaho fescue on burned areas may have more protein than those on unburned areas (Beardall and Sylvester 1976). Singer and Harter (1996) found that digestibility of Idaho fescue was enhanced (for 1 year) on grazed but not on ungrazed sites following the 1988 fires in Yellowstone National Park. Dry matter digestibility was higher in Idaho fescue plants the first year following burning in Yellowstone, and both digestibility and percent protein were higher the second year (Norland et al. 1996). Similarly, crude protein in Idaho fescue increased from 0.6 to 2.6% after spring burning in Douglas-fir and limber pine in central Montana (Keown 1982).

**DISCUSSION AND QUALIFICATION OF PLANT RESPONSE**

Plant response varies with fire severity, season of burn, ecological condition at the time of the burn, and management activities before and after burning (Agee 1996). Vigor of surviving Idaho fescue plants is reduced by high severity fires because the root crown’s budding zone is confined to a relatively small area at or above the surface of the ground where it may be exposed to higher temperatures for prolonged periods (Conrad and Poulton 1996).
Spring prescribed burns in a Nevada big sagebrush/grassland were classified as “hot” and “cool” fires, and cool fires resulted in only a small decrease in Idaho fescue cover, while hot burns resulted in a significant (p < 0.05) decrease that recovered to pre-fire levels in 3 years (Kuntz 1982).

Idaho fescue is sensitive to severe burns in late summer and early fall in eastern Oregon (Johnson and Simon 1987). Such fires favor succession to forbs in Idaho fescue plant associations (Agee 1996). Both number of plants and basal crown area were severely reduced in Idaho fescue following an August wildfire on northern California perennial range and remained reduced 5 years later (Countryman and Cornelius 1957). A hot June wildfire in a Montana grassland reduced biomass and cover of Idaho fescue. The damaged clumps failed to produce much autumn growth, so Idaho fescue cover remained low in the following spring in favor of forb species. Idaho fescue recovered completely (98% of unburned cover) 3 years after the fire (Antos et al. 1983). Spring and late fall burns on sites with good soil moisture and favorable Idaho fescue root reserves are thought to injure plants less (Beardall and Sylvester 1976, Wright et al. 1979, Young 1983). Britton and others (1983) observed greater plant damage with late August than mid-October burning; however, they also found that plants watered immediately before or after burning had the greatest basal area reduction and produced the least re-growth. They explained that with increasing water content, thermal conductivity increases, and therefore the potential for the heat pulse to reach the grass’s meristematic tissue faster and remain at lethal temperatures longer exists when soils are wet.

Idaho fescue is tolerant of late-season burning (Agee 1996, Agee and Maruoka 1994, Wright and Klemmedson 1965), but again, results are varied. Armour and others (n.d.) saw recovery to preburn levels of Idaho fescue in 3 years after fall prescribed burning in Douglas-fir/ninebark (Physocarpus malvaceus) habitat type in Idaho. Britton and others (1978) compared early May, mid-June, and mid-October burns in eastern Oregon and found highest mortality in early May (30%), and no mortality in mid-June or mid-October. Corresponding basal area reductions were 48% in May, 52% in June, and 34% in October (Britton et al. 1978). A comparison of spring and fall burning in Idaho fescue grassland in Oregon showed no difference for season of burn (Tveten and Fonda 1999). A significant (p < 0.05) decrease in Idaho fescue cover occurred in both seasons, although frequency was not reduced, and Idaho fescue remained the dominant prairie species (Tveten and Fonda 1999).

Conversely, Schwecke and Hann (1989) observed 25% kill of Idaho fescue after a spring burn, compared with 40% kill after a fall burn, in a Douglas-fir and big sagebrush/grass mosaic in western Montana. There was a similar decrease in basal crown sizes for both burns, but the canopy cover of surviving fescue (Festuca spp.) plants almost doubled compared to pre-fire canopy cover. Sagebrush sites became dominated by fescues 1 year after fire in both cases (Schwecke and Hann 1989).
Another comparison of spring and fall burning found that fall burning killed 20% of the Idaho fescue population and reduced basal area by 23% the first year. Spring burning resulted in no significant change in basal area and only 3.5% mortality. Plants recovered to 90% of their preburn size by the second year after the fall burn (Sapsis 1990). Idaho fescue is sensitive to burning in any season in areas where it is at the margins of its ecological range (Blaisdell 1953, Conrad and Poulton 1996, Wright et al. 1979).

Fire in water-limited environments generally reduces the productivity of grasses during the first post-fire growing season (Blaisdell 1953, Daubenmire 1968, Robberecht and Defosse 1995, Wright 1974), and in many cases reduces productivity of Idaho fescue for several years to come (Harniss and Murray 1973). Defosse and Robberecht (1996) used a special device to apply similar fire severity levels inside the meristematic root crown region to several Idaho fescue and bluebunch wheatgrass plants and followed the treatment with different levels of competition simulated by removing varying amounts of aboveground biomass of neighboring potential competitors. Idaho fescue did show meristematic damage after the fire, but no mortality was observed. Regrowth occurred within 15 days - more rapidly than bluebunch wheatgrass. Subsequent competition reduced root production and restricted aboveground productivity by 115% in Idaho fescue, and by 70% for bluebunch wheatgrass. These results suggest that survival and productivity following fires is related to subsequent soil water availability. A species with roots concentrated in upper soil layers (e.g., Idaho fescue) will experience a decline of water availability when compared with a deeper rooted species (e.g., bluebunch wheatgrass), thus affecting subsequent growth (Defosse and Robberecht 1996). This may help explain why many studies show that Idaho fescue is more severely damaged by fire than bluebunch wheatgrass (Agee 1996).

Conrad and Poulton (1966) observed that Idaho fescue basal diameter reduction was less after fire in grazed conditions (27%) than in ungrazed conditions (40%). Idaho fescue basal area was reduced equally by burning and clipping (an average of 48%) in May and June in eastern Oregon (Britton et al. 1990). Other treatment-date combinations (late summer and fall) did not significantly (p < 0.05) reduce basal area, suggesting that it is less susceptible to late-season defoliation than reported previously.

Recovery of Idaho fescue frequency is also a function of seed production and germination after a fire. Sapsis (1990) found higher numbers of vegetative culms in burned plants compared with unburned plants. Seed production of Idaho fescue plants subjected to fall prescribed burning in the sagebrush/grassland region in Idaho and Oregon was not different from seed production on unburned controls in post-fire years 1 and 3, but was greater on a 5-year-old burn (Patton et al. 1988). Both severe and lower-severity fire treatments reduce emergence of Idaho fescue from seed (Chaplin and Winward 1982).
Warg (1938) cites a study in which seeds of Idaho fescue are exposed to temperatures of 80, 100, 125, and 150º C for periods of 5, 15, 30, and 60 minutes. Germination was good for seeds exposed to 80, 100 and 125º C for 5 minutes, but did not occur beyond that temperature or time period. Clark and others (1994) studied the effects of fire on seed banks and found the LD 50 for most seeds was between 70 to 85º C.

FIRE MANAGEMENT CONSIDERATIONS

Fire suppression coupled with grazing pressure has changed the structure of Idaho fescue communities, often by increasing cover of woody species (Agee 1996, Arno and Gruell 1986). Prescribed burning can be an effective management tool for all types of Idaho fescue communities. Early spring burning is preferred in some cases e.g., (Lancaster et al. 1987, Schwecke and Hann 1989), late season burning in others (e.g., Agee and Maruoka 1994, Britton et al. 1978, Sapsis 1990, Wright and Klemmedson 1965). Beardall and Sylvester (1976) recommend burning of big sagebrush/grasslands before or just after the plants have broken dormancy, when root reserves remain high, to improve survival of perennial species. Johnson and Simon (1987) suggest that cool, light burns in late winter or early spring, when plant moisture levels are high, help protect root crowns from damage. Similarly, Wright (1974) suggests conducting burns when preferred plants are dormant, and includes that it is better to burn during wet years and never during extended dry periods, so as to not magnify drought stress on plants.

Bunting and others (1998) concluded that post-fire plant mortality and productivity might be related to the length of time grazing is excluded during post-fire regeneration period. Early spring fire alone resulted in low mortality, and early season defoliation (simulated grazing) after fire resulted in 50 % mortality for Idaho fescue. Detrimental effects were lessened when defoliation was delayed by 1 growing season after fire (Bunting et al. 1998). In a big sagebrush/grassland in Idaho burned once in September of 1933, again in August of 1936, and subsequently “conservatively” grazed after 1 full year of protection, Blaisdell (1953) observed no significant differences in total grass production on any severity of burn 15 years after burning. Idaho fescue was, however, significantly reduced, achieving pre-fire levels within 12 years after a light-severity burn, and at only 77 and 53% of pre-fire levels 12 years after a moderate and heavy burn, respectively. All other grasses had recovered beyond pre-fire levels by post-fire year (Blaisdell 1953).

LITERATURE CITED


Festuca rubra
Red fescue

FIRE ECOLOGY OR ADAPTATIONS

Red fescue probably sprouts from rhizomes after aerial portions are burned.

POST-FIRE REGENERATION STRATEGY

Rhizomatous herb, rhizome in soil; tussock graminoid.

IMMEDIATE FIRE EFFECT ON PLANT

Red fescue culms and leaves are probably killed by fire.

PLANT RESPONSE TO FIRE

No information was available on short-term response of red fescue to fire.

A wildfire on the border of northern British Columbia and Yukon Territory burned 8000 ha over a 13-day period in July 1988. The dominant tree species were lodgepole pine (Pinus contorta) and black spruce (Picea mariana) which were about 120 years old. By the fifth post-fire year, red fescue was the only herb on the wet lower slope providing more than 1 percent cover, although it had spotty distribution. Red fescue was present only in trace amounts at other sites (Oswald and Brown 1990).

FIRE MANAGEMENT CONSIDERATIONS

Red fescue can be used to revegetate burns. Red fescue was broadcast seeded on November 1, 1944, on a burned area near Priest River, Idaho. Elevation was 762 m. The area had been covered with Douglas-fir, western larch (Larix occidentalis), and grand fir (Abies grandis). After seeding, plots were fenced; light grazing was allowed after 1945. Red fescue established fair to excellent stands. Red fescue prevented brush encroachment, and ponderosa pine (Pinus ponderosa) seedlings were abundant by 1955. By 1960 trees dominated the seeded area and few grasses or legumes were left (Slinkard et al. 1970).
Red fescue and other seeds were used to revegetate burned land in the Peace River region of northern Canada. Wildfires burned 404,700 ha of wooded land in September 1950. Following the fire, depth of ash ranged from 2.5-7.6 cm. Seeding was done in October 1950 and in early April the following spring. Snow covered the ash at both seeding times and the burned soil and debris were disturbed as little as possible. Time of seeding did not influence the establishment of red rescue. Percent ground cover was determined in the first and fifth growing seasons after seeding. Red fescue cover varied from 72 to 83 percent in 1952 and from 17 to 80 percent in 1955 (Anderson and Elliott 1957).

Discussion of effects of seeding after fire on recovery of other native species is not available in the literature. Since red fescue can outcompete other native species on disturbed sites in both temperate and arctic communities (Densmore 1992, Wilson 1989), it may also do so on burned sites.

The red fescue cultivar “Clatsop” was selected from native vegetation on the coastal dunes of Oregon. “Clatsop” grows during the summer as well as during the cooler seasons; this continued growth reduces the hazard of wildfires on dunes (Hafenrichter et al. 1968).

Red fescue seed-producing fields can be burned after harvest to kill weed seeds, discourage diseases and harmful insects, and prevent red fescue stands from becoming too thick (Hardison 1980). For successful burning, soil and sod should be dry and the plants in semidormancy. Weather should be hot and dry, with enough wind to produce a quick, thorough fire. Flammable material should be well distributed to prevent hot spot fires. Burning should be done each year; old, thick sods burn slowly and with too much heat for plant survival (Hardison 1980, Wheeler and Hill 1957).

LITERATURE CITED


FIRE ECOLOGY OR ADAPTATIONS

Fire adaptations: Stickywilly recolonizes burned sites through seed germination. Seed may be from on-site and/or off-site sources (see Fire Effects).

Fire regimes: Many diverse communities provide stickywilly habitat. The fire regimes are dictated by the overstory community. Stickywilly experiences extreme ranges in fire frequencies. Vegetation in Quebec’s Huntingdon Marsh that includes stickywilly burns almost every fall or early spring. Researchers found evidence of previous growing season fires in 28% to 50% of the quadrats sampled, and 14% to 25% of quadrats burned in the last 2 or 3 years (Auclair et al. 1973). Western Montana’s rough fescue (Festuca altaica)-dominated grasslands that are also stickywilly habitat tolerate fire frequencies of between 5 and 10 years. Researchers based estimated fire frequencies on this community’s post-fire vegetation recovery (Antos et al. 1983). In the East, stickywilly is common in sugar maple communities where fires are exceptionally rare, occurring at greater than 1,000-year intervals (Wade et al. 2000). This range of fire regimes tolerated by stickywilly suggests that this species is fire tolerant but not fire dependent.

The following table provides fire return intervals for plant communities and ecosystems where stickywilly is important. For further information, see the FEIS review of the dominant species listed below. This list may not be inclusive for all plant communities in which stickywilly occurs.
<table>
<thead>
<tr>
<th>Community or Ecosystem</th>
<th>Dominant Species</th>
<th>Fire Return Interval Range (years)</th>
</tr>
</thead>
<tbody>
<tr>
<td>silver fir-Douglas-fir</td>
<td>Abies amabilis-Pseudotsuga menziesii var. menziesii</td>
<td>&gt; 200</td>
</tr>
<tr>
<td>grand fir</td>
<td>Abies grandis</td>
<td>35-200 [1]</td>
</tr>
<tr>
<td>maple-beech-birch</td>
<td>Acer-Fagus-Betula spp.</td>
<td>&gt; 1,000</td>
</tr>
<tr>
<td>silver maple-American elm</td>
<td>Acer saccharinum-Ulmus americana</td>
<td>&lt; 35 to 200</td>
</tr>
<tr>
<td>sugar maple</td>
<td>Acer saccharum</td>
<td>&gt; 1,000</td>
</tr>
<tr>
<td>sugar maple-basswood</td>
<td>Acer saccharum-Tilia americana</td>
<td>&gt; 1,000 [2]</td>
</tr>
<tr>
<td>California chaparral</td>
<td>Adenostoma and/or Arctostaphylos spp.</td>
<td>&lt; 35 to &lt; 100 [3]</td>
</tr>
<tr>
<td>bluestem prairie</td>
<td>Andropogon gerardii var. gerardii-Schizachyrium scoparium</td>
<td>&lt; 10 [4,3]</td>
</tr>
<tr>
<td>Nebraska sandhills prairie</td>
<td>Andropogon gerardii var. paucipilus-Schizachyrium scoparium</td>
<td>&lt; 10</td>
</tr>
<tr>
<td>bluestem-Sacahuista prairie</td>
<td>Andropogon littoralis-Spartina spartinae</td>
<td>&lt; 10 [3]</td>
</tr>
<tr>
<td>silver sagebrush steppe</td>
<td>Artemisia cana</td>
<td>5-45 [5,6,7]</td>
</tr>
<tr>
<td>sagebrush steppe</td>
<td>Artemisia tridentata/Pseudoroegneria spicata</td>
<td>20-70 [3]</td>
</tr>
<tr>
<td>basin big sagebrush</td>
<td>Artemisia tridentata var. tridentata</td>
<td>12-43 [8]</td>
</tr>
<tr>
<td>mountain big sagebrush</td>
<td>Artemisia tridentata var. vaseyana</td>
<td>15-40 [9,10,11]</td>
</tr>
<tr>
<td>Wyoming big sagebrush</td>
<td>Artemisia tridentata var. wyomingensis</td>
<td>10-70 (40**) [12,13]</td>
</tr>
<tr>
<td>coastal sagebrush</td>
<td>Artemisia californica</td>
<td>&lt; 35 to &lt; 100 [3]</td>
</tr>
<tr>
<td>plains grasslands</td>
<td>Bouteloua spp.</td>
<td>&lt; 35 [3,7]</td>
</tr>
<tr>
<td>cheatgrass</td>
<td>Bromus tectorum</td>
<td>&lt; 10 [14,15]</td>
</tr>
<tr>
<td>California montane chaparral</td>
<td>Ceanothus and/or Arctostaphylos spp.</td>
<td>50-100 [3]</td>
</tr>
<tr>
<td>sugarberry-America elm-green ash</td>
<td>Celtis laevigata-Ulmus americana-Fraxinus pennsylvanica</td>
<td>&lt; 35 to 200 [2]</td>
</tr>
<tr>
<td>mountain-mahogany-Gambel oak scrub</td>
<td>Cercocarpus ledifolius-Quercus gambelii</td>
<td>&lt; 35 to &lt; 100 [3]</td>
</tr>
<tr>
<td>beech-sugar maple</td>
<td>Fagus spp.-Acer saccharum</td>
<td>&gt; 1,000</td>
</tr>
<tr>
<td>black ash</td>
<td>Fraxinus nigra</td>
<td>&lt; 35 to 200 [2]</td>
</tr>
<tr>
<td>juniper-oak savanna</td>
<td>Juniperus ashei-Quercus virginiana</td>
<td>&lt; 35</td>
</tr>
<tr>
<td>Ashe juniper</td>
<td>Juniperus ashei</td>
<td>&lt; 35</td>
</tr>
<tr>
<td>western juniper</td>
<td>Juniperus occidentalis</td>
<td>20-70</td>
</tr>
<tr>
<td>Rocky Mountain juniper</td>
<td>Juniperus scopulorum</td>
<td>&lt; 35 [3]</td>
</tr>
<tr>
<td>cedar glades</td>
<td>Juniperus virginiana</td>
<td>3-22 [16,3]</td>
</tr>
<tr>
<td>tamarack</td>
<td>Larix laricina</td>
<td>35-200 [3]</td>
</tr>
<tr>
<td>Plant Type</td>
<td>Scientific Name</td>
<td>Range</td>
</tr>
<tr>
<td>------------------------------------</td>
<td>----------------------------------------</td>
<td>----------------</td>
</tr>
<tr>
<td>western larch</td>
<td><em>Larix occidentalis</em></td>
<td>25-350 [17,18,19]</td>
</tr>
<tr>
<td>yellow-poplar</td>
<td><em>Liriodendron tulipifera</em></td>
<td>&lt; 35 [2]</td>
</tr>
<tr>
<td>Great Lakes spruce-fir</td>
<td><em>Picea-Abies</em> spp.</td>
<td>35 to &gt; 200</td>
</tr>
<tr>
<td>northeastern spruce-fir</td>
<td><em>Picea-Abies</em> spp.</td>
<td>35-200 [20]</td>
</tr>
<tr>
<td>southeastern spruce-fir</td>
<td><em>Picea-Abies</em> spp.</td>
<td>35 to &gt; 200 [2]</td>
</tr>
<tr>
<td>Engelmann spruce-subalpine fir</td>
<td><em>Picea engelmannii</em>-<em>Abies lasiocarpa</em></td>
<td>35 to &gt; 200</td>
</tr>
<tr>
<td>black spruce</td>
<td><em>Picea mariana</em></td>
<td>35-200</td>
</tr>
<tr>
<td>conifer bog*</td>
<td><em>Picea mariana</em>- <em>Larix laricina</em></td>
<td>35-200 [20]</td>
</tr>
<tr>
<td>pinyon-juniper</td>
<td><em>Pinus-Juniperus</em> spp.</td>
<td>&lt; 35 [3]</td>
</tr>
<tr>
<td>Rocky Mountain lodgepole pine*</td>
<td><em>Pinus contorta</em> var. <em>latifolia</em></td>
<td>25-340 [21,18,22]</td>
</tr>
<tr>
<td>Sierra lodgepole pine*</td>
<td><em>Pinus contorta</em> var. <em>murrayana</em></td>
<td>35-200 [4]</td>
</tr>
<tr>
<td>shortleaf pine</td>
<td><em>Pinus echinata</em></td>
<td>2-15</td>
</tr>
<tr>
<td>slash pine-hardwood</td>
<td>*Pinus elliottii-*variable</td>
<td>&lt; 35</td>
</tr>
<tr>
<td>longleaf pine-scrub oak</td>
<td>*Pinus palustris-<em>Quercus</em> spp.</td>
<td>6-10 [2]</td>
</tr>
<tr>
<td>Pacific ponderosa pine*</td>
<td><em>Pinus ponderosa</em> var. <em>ponderosa</em></td>
<td>1-47 [4]</td>
</tr>
<tr>
<td>interior ponderosa pine*</td>
<td><em>Pinus ponderosa</em> var. <em>scopulorum</em></td>
<td>2-30 [4,23,24]</td>
</tr>
<tr>
<td>Arizona pine</td>
<td><em>Pinus ponderosa</em> var. <em>arizonica</em></td>
<td>2-15 [23,25,26]</td>
</tr>
<tr>
<td>eastern white pine</td>
<td><em>Pinus strobus</em></td>
<td>35-200</td>
</tr>
<tr>
<td>eastern white pine-eastern hemlock</td>
<td><em>Pinus strobus</em>-<em>Tsuga canadensis</em></td>
<td>35-200</td>
</tr>
<tr>
<td>eastern white pine-northern red oak-red maple</td>
<td><em>Pinus strobus</em>-<em>Quercus rubra</em>- <em>Acer rubrum</em></td>
<td>35-200</td>
</tr>
<tr>
<td>Virginia pine</td>
<td><em>Pinus virginiana</em></td>
<td>10 to &lt; 35</td>
</tr>
<tr>
<td>Virginia pine-oak</td>
<td>*Pinus virginiana-<em>Quercus</em> spp.</td>
<td>10 to &lt; 35</td>
</tr>
<tr>
<td>sycamore-sweetgum-American elm</td>
<td><em>Platanus occidentalis-</em> <em>Liquidambar styraciflua-</em> <em>Ulmus americana</em></td>
<td>&lt; 35 to 200 [2]</td>
</tr>
<tr>
<td>eastern cottonwood</td>
<td><em>Populus deltoides</em></td>
<td>&lt; 35 to 200 [3]</td>
</tr>
<tr>
<td>aspen-birch</td>
<td><em>Populus tremuloides</em>- <em>Betula papyrifera</em></td>
<td>35-200 [20,2]</td>
</tr>
<tr>
<td>quaking aspen (west of the Great Plains)</td>
<td><em>Populus tremuloides</em></td>
<td>7-120 [4,27,28]</td>
</tr>
<tr>
<td>black cherry-sugar maple</td>
<td><em>Prunus serotina-</em> <em>Acer saccharum</em></td>
<td>&gt; 1,000 [2]</td>
</tr>
<tr>
<td>mountain grasslands</td>
<td><em>Pseudoroegneria spicata</em></td>
<td>3-40 (10**) [29,4]</td>
</tr>
<tr>
<td>Rocky Mountain Douglas-fir*</td>
<td><em>Pseudotsuga menziesii</em> var. <em>glaucum</em></td>
<td>25-100 [4,9,30]</td>
</tr>
<tr>
<td>coastal Douglas-fir*</td>
<td><em>Pseudotsuga menziesii</em> var. <em>menziesii</em></td>
<td>40-240 [4,31,32]</td>
</tr>
<tr>
<td>California mixed evergreen</td>
<td><em>Pseudotsuga menziesii</em> var. <em>menziesii</em>- <em>Lithocarpus densiflorus-</em> <em>Arbutus menziesii</em></td>
<td>&lt; 35</td>
</tr>
<tr>
<td>Vegetation Type</td>
<td>Tree Species</td>
<td>Age Range</td>
</tr>
<tr>
<td>---------------------------------------</td>
<td>---------------------------------------</td>
<td>-----------</td>
</tr>
<tr>
<td>California oakwoods</td>
<td><em>Quercus spp.</em></td>
<td>&lt; 35 [4]</td>
</tr>
<tr>
<td>oak-hickory</td>
<td><em>Quercus-Carya spp.</em></td>
<td>&lt; 35 [2]</td>
</tr>
<tr>
<td>oak-juniper woodland (Southwest)</td>
<td><em>Quercus-Juniperus spp.</em></td>
<td>&lt; 35 to &lt; 200 [3]</td>
</tr>
<tr>
<td>northeastern oak-pine</td>
<td><em>Quercus-Pinus spp.</em></td>
<td>10 to &lt; 35 [2]</td>
</tr>
<tr>
<td>oak-gum-cypress</td>
<td><em>Quercus-Nyssa-spp.-Taxodium distichum</em></td>
<td>35 to &gt; 200 [33]</td>
</tr>
<tr>
<td>southeastern oak-pine</td>
<td><em>Quercus-Pinus spp.</em></td>
<td>&lt; 10 [2]</td>
</tr>
<tr>
<td>coast live oak</td>
<td><em>Quercus agrifolia</em></td>
<td>2-75 [34]</td>
</tr>
<tr>
<td>white oak-black oak-northern red oak</td>
<td><em>Quercus alba-Q. velutina-Q. rubra</em></td>
<td>&lt; 35 [2]</td>
</tr>
<tr>
<td>canyon live oak</td>
<td><em>Quercus chrysolepis</em></td>
<td>&lt; 35 to 200</td>
</tr>
<tr>
<td>blue oak-foothills pine</td>
<td><em>Quercus douglasii-P. sabiniana</em></td>
<td>&lt; 35 [4]</td>
</tr>
<tr>
<td>northern pin oak</td>
<td><em>Quercus ellipsoidalis</em></td>
<td>&lt; 35 [2]</td>
</tr>
<tr>
<td>Oregon white oak</td>
<td><em>Quercus garryana</em></td>
<td>&lt; 35 [4]</td>
</tr>
<tr>
<td>bear oak</td>
<td><em>Quercus ilicifolia</em></td>
<td>&lt; 35 [2]</td>
</tr>
<tr>
<td>California black oak</td>
<td><em>Quercus kelloggii</em></td>
<td>5-30 [3]</td>
</tr>
<tr>
<td>bur oak</td>
<td><em>Quercus macrocarpa</em></td>
<td>&lt; 10 [2]</td>
</tr>
<tr>
<td>oak savanna</td>
<td><em>Quercus macrocarpa/Andropogon gerardii-Schizachyrium scoparium</em></td>
<td>2-14 [3,2]</td>
</tr>
<tr>
<td>chestnut oak</td>
<td><em>Quercus prinus</em></td>
<td>3-8</td>
</tr>
<tr>
<td>northern red oak</td>
<td><em>Quercus rubra</em></td>
<td>10 to &lt; 35</td>
</tr>
<tr>
<td>post oak-blackjack oak</td>
<td><em>Quercus stellata-Q. marilandica</em></td>
<td>&lt; 10</td>
</tr>
<tr>
<td>black oak</td>
<td><em>Quercus velutina</em></td>
<td>&lt; 35</td>
</tr>
<tr>
<td>live oak</td>
<td><em>Quercus virginiana</em></td>
<td>10 to&lt; 100 [2]</td>
</tr>
<tr>
<td>interior live oak</td>
<td><em>Quercus wislizenii</em></td>
<td>&lt; 35 [4]</td>
</tr>
<tr>
<td>blackland prairie</td>
<td><em>Schizachyrium scoparium-Nassella leucotricha</em></td>
<td>&lt; 10</td>
</tr>
<tr>
<td>Fayette prairie</td>
<td><em>Schizachyrium scoparium-Buchloe dactyloides</em></td>
<td>&lt; 10 [2]</td>
</tr>
<tr>
<td>little bluestem-grama prairie</td>
<td><em>Schizachyrium scoparium-Bouteloua spp.</em></td>
<td>&lt; 35 [3]</td>
</tr>
<tr>
<td>redwood</td>
<td><em>Sequoia sempervirens</em></td>
<td>5-200 [4,35,36]</td>
</tr>
<tr>
<td>baldcypress</td>
<td><em>Taxodium distichum var. distichum</em></td>
<td>100 to &gt; 300</td>
</tr>
<tr>
<td>pondcypress</td>
<td><em>Taxodium distichum var. nutans</em></td>
<td>&lt; 35 [33]</td>
</tr>
<tr>
<td>western redcedar-western hemlock</td>
<td><em>Thuja plicata-Tsuga heterophylla</em></td>
<td>&gt; 200 [4]</td>
</tr>
<tr>
<td>eastern hemlock-yellow birch</td>
<td><em>Tsuga canadensis-Betula alleghaniensis</em></td>
<td>&gt; 200 [2]</td>
</tr>
<tr>
<td>western hemlock-Sitka spruce</td>
<td><em>Tsuga heterophylla-Picea sitchensis</em></td>
<td>&gt; 200</td>
</tr>
<tr>
<td>Species</td>
<td>Details</td>
<td>Fire Return Interval</td>
</tr>
<tr>
<td>---------------------</td>
<td>------------------------------</td>
<td>----------------------</td>
</tr>
<tr>
<td>mountain hemlock*</td>
<td>Tsuga mertensiana</td>
<td>35 to &gt; 200 [4]</td>
</tr>
<tr>
<td>elm-ash-cottonwood</td>
<td>Ulmus-Fraxinus-Populus spp.</td>
<td>&lt; 35 to 200 [20,2]</td>
</tr>
</tbody>
</table>

*fire return interval varies widely; trends in variation are noted in the species review

**mean


POST-FIRE REGENERATION STRATEGY (Stickney 1989)

Initial off-site colonizer (off-site, initial community); secondary colonizer (on-site or off-site seed sources).

IMMEDIATE FIRE EFFECT ON PLANT

Fire kills stickywilly when it is actively growing (Stickney and Campbell 2000). Fall germinating seedlings were killed by both early winter and spring fires in tallgrass prairie wetlands of northeastern Kansas (Johnson and Knapp 1995). Fires late in the growing season may only affect stored seed as stickywilly senescences following fruiting (Moore 1975).
DISCUSSION AND QUALIFICATION OF FIRE EFFECT

Survival of stored seed following fire likely depends on depth of burial and fire intensity. Some suggest that seed in the litter layer is killed by fire (Pratt et al. 1984), while others suggest recolonization of an area is by germination of on-site seed (Stickney and Campbell 2000). Pratt and others (1984) found heat significantly reduced (p < 0.05) stickywilly germination. See Germination for study specifics.

PLANT RESPONSE TO FIRE

Stickney and Campbell [2000) tentatively classified stickywilly as a nonsurvivor that colonizes burned sites from on-site seed. Due to a limited number of fire effect observations for this species, researchers were tentative in their description of stickywilly’s post-fire response.

DISCUSSION AND QUALIFICATION OF PLANT RESPONSE

Single fires: Stickywilly is typically present in post-fire communities. Coverage, frequency, and/or density are often reduced immediately following fire; however, stickywilly’s reduction or absence post-fire is likely short lived.

Coniferous forests: The following studies indicate that stickywilly is often absent from the first post-fire year conifer communities. Several fires burned in 2 northeastern Oregon forests (Douglas-fir and subalpine fir) where stickywilly occurs. Moderately severe fires partially consumed the litter and woody debris, blackened shrub stems, and charred and partially burned tree trunks. Severe fires deeply charred tree trunks, consumed most branches, consumed litter and duff, and left a white ash layer. Stickywilly coverage in the fifth post-fire year surpassed pre-fire coverages in moderate and severe burns. Pre- and post-fire percent coverages for stickywilly are provided below (Johnson 1998):

<table>
<thead>
<tr>
<th>Vegetation association</th>
<th>pre-fire year 1</th>
<th>post-fire year 1</th>
<th>post-fire year 5</th>
<th>post-fire year 5</th>
</tr>
</thead>
<tbody>
<tr>
<td>moderate burn</td>
<td></td>
<td></td>
<td>1</td>
<td></td>
</tr>
<tr>
<td>(n = 4)</td>
<td></td>
<td></td>
<td>1</td>
<td></td>
</tr>
<tr>
<td>severe burn</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>(n = 2)</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Subalpine fir/menziesia (Menziesia ferruginea)</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>partial burn</td>
<td>0</td>
<td>0</td>
<td></td>
<td></td>
</tr>
<tr>
<td>(n = 2)</td>
<td>0</td>
<td>0</td>
<td></td>
<td></td>
</tr>
<tr>
<td>severe burn</td>
<td>0</td>
<td>0</td>
<td></td>
<td></td>
</tr>
<tr>
<td>(n = 2)</td>
<td>0</td>
<td>0</td>
<td></td>
<td></td>
</tr>
<tr>
<td>no data</td>
<td>0</td>
<td>0</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

A study of different-aged burns in western hemlock-Douglas-fir forests in the Olympic Mountains of Washington revealed stickywilly’s preference for recently disturbed forests. The author described past fires as “catastrophic,” but no additional
information regarding fire season or severity was given. The percent frequency of stickywilly is shown below (Huff 1984):

<table>
<thead>
<tr>
<th>Time since fire (years)</th>
<th>2</th>
<th>3</th>
<th>19</th>
<th>110</th>
<th>515</th>
</tr>
</thead>
<tbody>
<tr>
<td>Percent frequency</td>
<td>0.04</td>
<td>0.19</td>
<td>0.10</td>
<td>0</td>
<td>0</td>
</tr>
</tbody>
</table>

Deciduous forests: Reestablishment of stickywilly following fires in deciduous woodlands is quick. In a red alder woodland in the Oregon Coast Range, sites were clearcut in early spring (March-April), treated with herbicide in June, and burned in early August. The pre-fire frequency of stickywilly was 15%. Two months following treatments frequency of stickywilly was 0%, and 4 months later stickywilly frequency was 1% (Roberts 1975).

“Moderately disturbed” upland slippery elm-dominated forests of northern Illinois burned during the 1992 dormant and growing seasons. The dormant season fire burned in March when temperatures averaged 16.7° C, relative humidity was 70%, and the 8 days prior received no precipitation. Approximately 75% to 80% of the unit burned, flame heights measured between 15-100 cm, and fire spread was 1.3 m/minute. The growing season fire burned in May when temperatures averaged 25.6° C, relative humidity was 29%, and the 9 days prior received no precipitation. Approximately 75%-80% of the unit burned, flame heights were between 10-75 cm, and fire spread was 1.7 m/minute. The density of stickywilly decreased on all burned and unburned sites in 1992 and 1993. Stickywilly had not recovered on either burned site by the third post-fire year. The pre-fire and post-fire stem densities (per m²) of stickywilly on dormant season burns, growing season burns, and unburned plots are provided below (Schwartz and Heim 1996).

<table>
<thead>
<tr>
<th>Season</th>
<th>dormant (March)</th>
<th>growing (May)</th>
<th>unburned</th>
</tr>
</thead>
<tbody>
<tr>
<td>Stickywilly stem density (per m²)</td>
<td>4.6</td>
<td>0.1</td>
<td>1.2</td>
</tr>
</tbody>
</table>

Shrublands/grasslands: In shrubland and grassland fires, stickywilly was commonly present in the first post-fire community. Following a July wildfire in the chaparral riparian zone of Ventura County, California, stickywilly was present 1, 2, and 3 years following fire (Davis et al. 1989). In west-central Utah, a fire burned big sagebrush and Colorado pinyon-Utah juniper (Pinus edulis-Juniperus osteosperma) ecosystems. Stickywilly occurred on 2 plots in the first post-fire season but was not encountered in the second or third post-fire years. The frequency of stickywilly on nearby unburned sites was 0 for all 3 years of post-fire sampling (Ott et al. 2001).
A late July fire in southern California’s foothill chaparral vegetation produced surface temperatures of 354°C and soil temperatures of 69°C 5 cm below the soil surface. In the preburn community, stickywilly occupied 11 m²; in the first year post-fire stickywilly occupied 32 m². Researchers indicate that annual forbs were replaced by increasingly dense grasses in the second, third, and fourth post-fire years (Lawrence 1966).

An “intense wildfire” burned Gambel oak and big sagebrush/bluebunch wheatgrass communities in the Wasatch Mountains of Utah in August of 1990. Coverage and frequency of stickywilly were greater on burned sites compared to unburned areas. The coverage and frequency (percent of quadrats in which species occurred) of stickywilly on burned and unburned plots is given below (Poreda and Wullstein 1994):

<table>
<thead>
<tr>
<th>Community type</th>
<th>Gambel oak</th>
<th>big sagebrush/bluebunch wheatgrass</th>
</tr>
</thead>
<tbody>
<tr>
<td>Burn status</td>
<td>unburned</td>
<td>burned</td>
</tr>
<tr>
<td>Frequency (%)</td>
<td>4.5</td>
<td>24.6</td>
</tr>
<tr>
<td>Cover (%)</td>
<td>0.02</td>
<td>0.52</td>
</tr>
</tbody>
</table>

In northeastern Oregon, fires burned in 2 grazing exclosures (1 excluding livestock and game animals, 1 excluding just livestock) within a common snowberry-rose (Rosa spp.) community. The fire was moderately severe: it consumed the litter and woody debris, blackened shrub stems, and charred and partially burned tree trunks. Stickywilly coverage in the fifth post-fire year surpassed pre-fire coverages in both exclosures. Pre-fire and post-fire percent coverage for stickywilly is provided below (Johnson 1998):

<table>
<thead>
<tr>
<th>Burn severity</th>
<th>moderate burn/no game or livestock post-fire disturbance (n = 1)</th>
<th>moderate burn/no livestock post-fire disturbance (n = 1)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Time since fire</td>
<td>pre-fire 1 year 5 yrs</td>
<td>pre-fire 1 year 5 yrs</td>
</tr>
<tr>
<td>Cover (%)</td>
<td>3 3 12</td>
<td>0 0 trace</td>
</tr>
</tbody>
</table>

While stickywilly is common in the first post-fire year in shrub or grassland communities, some studies did not detect stickywilly the first season following fire. Following a November, 1994 fire in southern California’s chaparral vegetation, stickywilly was not present the first post-fire growing season. Stickywilly did occur in the second, third, and fourth post-fire years (Guo 2001). In a rough fescue-dominated grassland near Missoula, Montana, a late June fire burned in 1977. The fire, pushed by gusty winds, consumed virtually all above ground vegetation. The following fall (August and September) received above normal precipitation. Researchers compared burned and nearby unburned sites in the fall, spring, and summer immediately following the fire. Stickywilly was not present on burned sites by the next summer (Antos et al. 1983).
The following study presents more long-term fire effects information by comparing burned and unburned Gambel oak communities in central and northern Utah. On unburned sites, the average frequency of stickywilly was 36.8; on burned sites, stickywilly frequency was 33.1. A majority of the burned sites experienced fires 8 years prior, while others burned less than 30 years before initiating the study. Researchers provided no data regarding fire severity or season (Kunzler et al. 1981).

Repeated fires: Stickywilly’s probability of recovery from fires seems to decrease as fire frequency increases. In a mixed mesophytic forest of northern Kentucky, sites burned repeatedly. For 2 and 3 consecutive fall seasons, prescription fires with flame heights of up to 15 cm burned. The importance of stickywilly was significantly (p < 0.05) greater on unburned sites than on sites repeatedly burned (Kunzler et al. 1981). Spring fires (late March-early April) burned annually, biennially, and at 4-, 10-, and 20-year intervals in tallgrass prairie wetlands of northeastern Kansas. The relative importance of stickywilly decreased with increased fire frequency. The relative importance values (%) are provided below. Data are means and 1 standard deviation (Johnson and Knapp 1995).

<table>
<thead>
<tr>
<th>Fire frequency</th>
<th>10 and 20</th>
<th>2 and 4</th>
<th>annual fires</th>
</tr>
</thead>
<tbody>
<tr>
<td>Relative importance value</td>
<td>19.7±3.2</td>
<td>3.5±3.1</td>
<td>0</td>
</tr>
</tbody>
</table>

The Research Project Summary Vegetation response to restoration treatments in ponderosa pine-Douglas-fir forests provides information on prescribed fire and post-fire response of plant community species including stickywilly.

FIRE MANAGEMENT CONSIDERATIONS

It seems likely that annual fires could control stickywilly if this were a management objective. However, if managing to promote stickywilly, fire is likely unnecessary.

LITERATURE CITED


*Galium boreale*
Bedstraw

**FIRE ECOLOGY OR ADAPTATIONS**

Fire adaptations: Fire severity likely dictates which bedstraw recolonization strategy prevails. When burned in low-severity fires, bedstraw likely sprouts from rhizomes. However, high-severity fires may damage sweetscented bedstraw’s more delicate rhizomes (Stickney and Campbell 2000), and recolonization of the area would likely be from on- or off-site seed sources. Northern bedstraw’s more robust rhizomes are capable of withstanding more severe fires, but on- or off-site seed sources may still colonize disturbed sites.
Fire regimes: A diversity of communities provide bedstraw habitat, and since fire regimes are dictated by the overstory community, bedstraw experiences a wide range of fire regimes. Davis and others (1980), in a review, classify the spruce/sweetscented bedstraw and subalpine fir/sweetscented bedstraw habitat types as having “infrequent, severe fires with long-lasting effects.” The same habitat types described on the Lolo National Forest, Montana, occupy moist environments that burn infrequently. The estimated fire return interval for these sites is 24 to 140 years (Losensky 1987). Both bedstraw species occupy northern spruce-fir forests that are typically maintained under moist conditions, and the fire return interval in these forests ranges from 35 to more than 200 years (Duchesne and Hawkes 2000). Much shorter fire return intervals are reportedly tolerated by bedstraw as well. Northern bedstraw is typical in fescue-oatgrass mountain grasslands that are characterized by a fire return interval of less than 35 years (Paysen et al. 2000). Sweetscented bedstraw occupies southern California walnut woodlands of southern California that burn annually due to an increased presence of annual grasses (Quinn 1990).

The following table provides fire return intervals for plant communities and ecosystems where bedstraw is important. For further information, see the FEIS review of the dominant species listed below. This list may not be inclusive for all plant communities in which bedstraw occurs.

<table>
<thead>
<tr>
<th>Community or Ecosystem</th>
<th>Dominant Species</th>
<th>Fire Return Interval Range (years)</th>
</tr>
</thead>
<tbody>
<tr>
<td>grand fir</td>
<td><em>Abies grandis</em></td>
<td>35-200 [1]</td>
</tr>
<tr>
<td>maple-beech-birch</td>
<td><em>Acer-Fagus-Betula</em> spp.</td>
<td>&gt; 1,000</td>
</tr>
<tr>
<td>silver maple-American elm</td>
<td><em>Acer saccharinum-Ulmus americana</em></td>
<td>&lt; 35 to 200</td>
</tr>
<tr>
<td>sugar maple</td>
<td><em>Acer saccharum</em></td>
<td>&gt; 1,000</td>
</tr>
<tr>
<td>sugar maple-basswood</td>
<td><em>Acer saccharum-Tilia americana</em></td>
<td>&gt; 1,000 [2]</td>
</tr>
<tr>
<td>bluestem prairie</td>
<td><em>Andropogon gerardii var. gerardii-Schizachyrium scoparium</em></td>
<td>&lt; 10 [3,4]</td>
</tr>
<tr>
<td>sagebrush steppe</td>
<td><em>Artemisia tridentata/Pseudoroegneria spicata</em></td>
<td>20-70 [4]</td>
</tr>
<tr>
<td>basin big sagebrush</td>
<td><em>Artemisia tridentata var. tridentata</em></td>
<td>12-43 [5]</td>
</tr>
<tr>
<td>mountain big sagebrush</td>
<td><em>Artemisia tridentata var. vaseyana</em></td>
<td>15-40 [6,7,8]</td>
</tr>
<tr>
<td>Wyoming big sagebrush</td>
<td><em>Artemisia tridentata var. wyomingensis</em></td>
<td>10-70 (40**) [9,10]</td>
</tr>
<tr>
<td>plains grasslands</td>
<td><em>Bouteloua</em> spp.</td>
<td>&lt; 35 [4,11]</td>
</tr>
<tr>
<td>blue grama-needle-and-thread grass-western wheatgrass</td>
<td><em>Bouteloua gracilis-Hesperostipa comata-Pascopyrum smithii</em></td>
<td>&lt; 35 [4,12,11]</td>
</tr>
<tr>
<td>cheatgrass</td>
<td><em>Bromus tectorum</em></td>
<td>&lt; 10 [13,14]</td>
</tr>
<tr>
<td>Common Name</td>
<td>Scientific Name</td>
<td>Range</td>
</tr>
<tr>
<td>-----------------------------------</td>
<td>----------------------------------------</td>
<td>---------------------</td>
</tr>
<tr>
<td>sugarberry-America elm-green ash</td>
<td><em>Celtis laevigata</em>-Ul<em>mus americana</em>-Fraxinus pennsylvanica*</td>
<td>&lt; 35 to 200</td>
</tr>
<tr>
<td>Atlantic white-cedar</td>
<td><em>Chamaecyparis thyoides</em></td>
<td>35 to &gt; 200</td>
</tr>
<tr>
<td>beech-sugar maple</td>
<td><em>Fagus</em> spp.-Acer saccharum*</td>
<td>&gt; 1,000</td>
</tr>
<tr>
<td>black ash</td>
<td><em>Fraxinus nigra</em></td>
<td>&lt; 35 to 200 [2]</td>
</tr>
<tr>
<td>western juniper</td>
<td><em>Juniperus occidentalis</em></td>
<td>20-70</td>
</tr>
<tr>
<td>Rocky Mountain juniper</td>
<td><em>Juniperus scopulorum</em></td>
<td>&lt; 35 [4]</td>
</tr>
<tr>
<td>western larch</td>
<td><em>Larix occidentalis</em></td>
<td>25-350 [15,16,17]</td>
</tr>
<tr>
<td>yellow-poplar</td>
<td><em>Liriodendron tulipifera</em></td>
<td>&lt; 35 [2]</td>
</tr>
<tr>
<td>Great Lakes spruce-fir</td>
<td><em>Picea</em>-Abies spp.</td>
<td>35 to &gt; 200</td>
</tr>
<tr>
<td>northeastern spruce-fir</td>
<td><em>Picea</em>-Abies spp.</td>
<td>35-200 [18]</td>
</tr>
<tr>
<td>southeastern spruce-fir</td>
<td><em>Picea</em>-Abies spp.</td>
<td>35 to &gt; 200 [2]</td>
</tr>
<tr>
<td>Engelmann spruce-subalpine fir</td>
<td><em>Picea engelmannii</em>-Abies lasiocarpa</td>
<td>35 to &gt; 200 [1]</td>
</tr>
<tr>
<td>black spruce</td>
<td><em>Picea mariana</em></td>
<td>35-200</td>
</tr>
<tr>
<td>conifer bog*</td>
<td><em>Picea mariana</em>-Larix laricina</td>
<td>35-200 [18]</td>
</tr>
<tr>
<td>blue spruce*</td>
<td><em>Picea pungens</em></td>
<td>35-200 [1]</td>
</tr>
<tr>
<td>red spruce*</td>
<td><em>Picea rubens</em></td>
<td>35-200 [18]</td>
</tr>
<tr>
<td>pinyon-juniper</td>
<td><em>Pinus-Juniperus</em> spp.</td>
<td>&lt; 35 [4]</td>
</tr>
<tr>
<td>whitebark pine*</td>
<td><em>Pinus albicaulis</em></td>
<td>50-200 [19,20]</td>
</tr>
<tr>
<td>jack pine</td>
<td><em>Pinus banksiana</em></td>
<td>&lt; 35 to 200 [18]</td>
</tr>
<tr>
<td>Rocky Mountain lodgepole pine*</td>
<td><em>Pinus contorta</em> var. latifolia</td>
<td>25-340 [21,16,22]</td>
</tr>
<tr>
<td>Sierra lodgepole pine*</td>
<td><em>Pinus contorta</em> var. murrayana</td>
<td>35-200</td>
</tr>
<tr>
<td>Pacific ponderosa pine*</td>
<td><em>Pinus ponderosa</em> var. ponderosa</td>
<td>1-47 [1]</td>
</tr>
<tr>
<td>interior ponderosa pine*</td>
<td><em>Pinus ponderosa</em> var. scopulorum</td>
<td>2-30 [1,23,24]</td>
</tr>
<tr>
<td>Arizona pine</td>
<td><em>Pinus ponderosa</em> var. arizonica</td>
<td>2-15 [23,25,26]</td>
</tr>
<tr>
<td>red pine (Great Lakes)</td>
<td><em>Pinus resinosa</em></td>
<td>10-200 (10**) [18,27]</td>
</tr>
<tr>
<td>red-white-jack pine*</td>
<td><em>Pinus resinosa</em>-P. strobus*-P. banksiana*</td>
<td>10-300 [18,28]</td>
</tr>
<tr>
<td>eastern white pine</td>
<td><em>Pinus strobus</em></td>
<td>35-200</td>
</tr>
<tr>
<td>eastern white pine-eastern hemlock</td>
<td><em>Pinus strobus</em>-Tsuga canadensis</td>
<td>35-200</td>
</tr>
<tr>
<td>eastern white pine-northern red oak-red maple</td>
<td><em>Pinus strobus</em>-Quercus rubra-Acer rubrum</td>
<td>35-200</td>
</tr>
<tr>
<td>sycamore-sweetgum-American elm</td>
<td><em>Platanus occidentalis</em>-Liquidambar styraciflua-Ul<em>mus americana</em></td>
<td>&lt; 35 to 200 [2]</td>
</tr>
<tr>
<td>Plant Community</td>
<td>Species Description</td>
<td>Area Covered</td>
</tr>
<tr>
<td>---------------------------------------</td>
<td>----------------------------------------------------------</td>
<td>---------------</td>
</tr>
<tr>
<td>eastern cottonwood</td>
<td><em>Populus deltoides</em></td>
<td>&lt; 35 to 200 [4]</td>
</tr>
<tr>
<td>aspen-birch</td>
<td><em>Populus tremuloides-Betula papyrifera</em></td>
<td>35-200 [18,2]</td>
</tr>
<tr>
<td>quaking aspen (west of the Great Plains)</td>
<td><em>Populus tremuloides</em></td>
<td>7-120 [1,29,30]</td>
</tr>
<tr>
<td>black cherry-sugar maple</td>
<td><em>Prunus serotina-Acer saccharum</em></td>
<td>&gt; 1,000 [2]</td>
</tr>
<tr>
<td>mountain grasslands</td>
<td><em>Pseudoroegneria spicata</em></td>
<td>3-40 (10**) [31,1]</td>
</tr>
<tr>
<td>Rocky Mountain Douglas-fir*</td>
<td><em>Pseudotsuga menziesii var. glauca</em></td>
<td>25-100 [1,6,32]</td>
</tr>
<tr>
<td>coastal Douglas-fir*</td>
<td><em>Pseudotsuga menziesii var. menziesii</em></td>
<td>40-240 [1,33,34]</td>
</tr>
<tr>
<td>California mixed evergreen</td>
<td><em>Pseudotsuga menziesii var. menziesii-Lithocarpus densiflorus-Arbutus menziesii</em></td>
<td>&lt; 35</td>
</tr>
<tr>
<td>California oakwoods</td>
<td><em>Quercus</em> spp.</td>
<td>&lt; 35 [1]</td>
</tr>
<tr>
<td>oak-hickory</td>
<td><em>Quercus-Carya</em> spp.</td>
<td>&lt; 35</td>
</tr>
<tr>
<td>northeastern oak-pine</td>
<td><em>Quercus</em> spp.</td>
<td>10 to &lt; 35 [2]</td>
</tr>
<tr>
<td>oak-gum-cypress</td>
<td><em>Quercus-Nyssa-spp.-Taxodium distichum</em></td>
<td>35 to &gt; 200 [35]</td>
</tr>
<tr>
<td>southeastern oak-pine</td>
<td><em>Quercus-Pinus</em> spp.</td>
<td>&lt; 10 [2]</td>
</tr>
<tr>
<td>coast live oak</td>
<td><em>Quercus agrifolia</em></td>
<td>2-75 [36]</td>
</tr>
<tr>
<td>white oak-black oak-northern red oak</td>
<td><em>Quercus alba-Q. velutina-Q. rubra</em></td>
<td>&lt; 35 [2]</td>
</tr>
<tr>
<td>canyon live oak</td>
<td><em>Quercus chrysolepis</em></td>
<td>&lt; 35 to 200 [1]</td>
</tr>
<tr>
<td>northern pin oak</td>
<td><em>Quercus ellipsoidalis</em></td>
<td>&lt; 35 [2]</td>
</tr>
<tr>
<td>Oregon white oak</td>
<td><em>Quercus garryana</em></td>
<td>&lt; 35 [1]</td>
</tr>
<tr>
<td>California black oak</td>
<td><em>Quercus kelloggi</em></td>
<td>5-30 [4]</td>
</tr>
<tr>
<td>bur oak</td>
<td><em>Quercus macrocarpa</em></td>
<td>&lt; 10 [2]</td>
</tr>
<tr>
<td>oak savanna</td>
<td><em>Quercus macrocarpa/Andropogon gerardii-Schizachyrium scoparium</em></td>
<td>2-14 [4,2]</td>
</tr>
<tr>
<td>chestnut oak</td>
<td><em>Quercus prinus</em></td>
<td>3-8</td>
</tr>
<tr>
<td>northern red oak</td>
<td><em>Quercus rubra</em></td>
<td>10 to &lt; 35</td>
</tr>
<tr>
<td>black oak</td>
<td><em>Quercus velutina</em></td>
<td>&lt; 35</td>
</tr>
<tr>
<td>live oak</td>
<td><em>Quercus virginiana</em></td>
<td>10 to &lt; 100 [2]</td>
</tr>
<tr>
<td>little bluestem-grama prairie</td>
<td><em>Schizachyrium scoparium-Bouteloua spp.</em></td>
<td>&lt; 35 [4]</td>
</tr>
<tr>
<td>redwood</td>
<td><em>Sequoia sempervirens</em></td>
<td>5-200 [1,37,38]</td>
</tr>
<tr>
<td>western redcedar-western hemlock</td>
<td><em>Thuja plicata-Tsuga heterophylla</em></td>
<td>&gt; 200 [1]</td>
</tr>
<tr>
<td>eastern hemlock-yellow</td>
<td><em>Tsuga canadensis-Betula</em></td>
<td>&gt; 200 [2]</td>
</tr>
<tr>
<td>Plant Type</td>
<td>Scientific Name</td>
<td>Reference</td>
</tr>
<tr>
<td>----------------------------------</td>
<td>-----------------------------------------------</td>
<td>--------------------</td>
</tr>
<tr>
<td>birch</td>
<td>alleghaniensis</td>
<td></td>
</tr>
<tr>
<td>western hemlock-Sitka spruce</td>
<td>Tsuga heterophylla-Picea sitchensis</td>
<td>&gt; 200</td>
</tr>
<tr>
<td>mountain hemlock*</td>
<td>Tsuga mertensiana</td>
<td>35 to &gt; 200 [1]</td>
</tr>
<tr>
<td>elm-ash-cottonwood</td>
<td>Ulmus-Fraxinus-Populus spp.</td>
<td>&lt; 35 to 200 [18,2]</td>
</tr>
</tbody>
</table>


POST-FIRE REGENERATION STRATEGY (Stickney 1989):

Rhizomatous herb, rhizome in soil; ground residual colonizer (on-site, initial community); initial off-site colonizer (off-site, initial community); secondary colonizer (on-site or off-site seed sources).

IMMEDIATE FIRE EFFECT ON PLANT

Bedstraw can be killed by fire (Bradley et al. 1992, Smith and Fischer 1997), but underground structures often survive low-severity fires. Likely fire severity and/or seasonality dictate the survival of bedstraw.
DISCUSSION AND QUALIFICATION OF FIRE EFFECT

Powell (1994) indicates that northern bedstraw has better chances of surviving fire than sweetscented bedstraw. Sweetscented bedstraw has less than 35% chance of 50% population survival (Kovalchik et al. 1988, Powell 1994) while northern bedstraw’s chances are 35% to 64% for 50% of the species population to survive fires with average flame lengths of 30.5 cm (Powell 1994). Stickney and Campbell (Stickney and Campbell 2000) consider sweetscented bedstraw a nonsurvivor because of its delicate rhizome; this classification was tentative as the researchers observed few post-fire responses. However, Edgerton (1987) described sweetscented bedstraw as a “surviving forb” following clearcutting and broadcast burning of a mixed conifer forest in the Umatilla National Forest of Oregon.

PLANT RESPONSE TO FIRE:

Bedstraw regenerates from rhizomes or seeds (Bradley et al. 1992, Kovalchik et al. 1988, Powell 1994) and likely regains pre-fire frequency or coverage 5 to 10 years after fire (Powell 1994). Stickney (1986) classifies sweetscented bedstraw as a residual colonizer, coming from an on-site ground source.

DISCUSSION AND QUALIFICATION OF PLANT RESPONSE

Severe fires can cause large decreases in bedstraw; however, bedstraw can remain unchanged or increase following low-severity understory fires or cool season spring and fall fires (Bradley et al. 1992, Kruse and Higgins 1998, Powell 1994, Smith and Fischer 1997).

Northern and sweetscented bedstraw: The post-fire response is not always the same for northern and sweetscented bedstraw in burned areas where both occur together.

Fire effects related to seasonality/severity: In the early 1960s, 17 wildfires burned in south-central New York. All but 1 of the fires burned in the spring, and sampling occurred 10 to 26 months following fire. Northern and sweetscented bedstraw frequencies were significantly higher (p = 0.01) on burned sites. Northern bedstraw averaged 2% frequency on unburned sites and 33% on burned sites within goldenrod (Solidago spp.-poverty oatgrass habitats. Sweetscented bedstraw averaged 2% frequency on unburned sites and 29% on burned sites in hardwood and mixed oak forests (Swan 1970).

The post-fire responses for northern and sweetscented bedstraw were opposite in quaking aspen boreal forests of northeastern Alberta. Following a lightning-ignited spring wildfire, Lee (2004) compared the immediate post-fire seed banks and second year post-fire vegetation of unburned, “lightly” burned, and severely burned sites. Severely burned sites had all downed wood (≥ 20 cm) and the top 6-10 cm of organic material oxidized. Light burns partially oxidized small and mid-sized downed wood and just the top 2 cm of organic matter. Seed density estimates came from seedling and vegetative emergence techniques. Sweetscented bedstraw rhizomes likely did
not survive the fire while seed did, and the reverse was true for northern bedstraw. These findings may reflect different rhizome and seed heat tolerances for the 2 species or may indicate the occupation of different microsites where fire effects were different. Data are summarized below (Lee 2004):

<table>
<thead>
<tr>
<th>Post-fire characteristics</th>
<th>Mean seed density (seeds/m²)</th>
<th>Second post-fire year coverage (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Fire type</td>
<td>UB</td>
<td>L</td>
</tr>
<tr>
<td>Northern bedstraw</td>
<td>1.9</td>
<td>0</td>
</tr>
<tr>
<td>Sweetscented bedstraw</td>
<td>24.5</td>
<td>30.1</td>
</tr>
</tbody>
</table>

UB-unburned, L-light, S-severe

Repeated fires: Sites in east-central Minnesota’s oak savannas burned at frequencies of 0 to 19 years in a 20-year period. The coverage of sweetscented bedstraw decreased with increased fire frequency; however, the r value for this relationship was -0.52 (significant at p < 0.10). Northern bedstraw coverage was not significantly changed by fire (Tester 1996).

Northern bedstraw: Northern bedstraw recovers quickly following fire. Severe or growing season fires may result in decreased northern bedstraw coverage and/or frequency, but typically decreases are short lived.

Northern bedstraw is often mentioned as an important species in postburn communities of Canada. In the Selkirk Mountains of British Columbia, Shaw (1916) lists northern bedstraw as a prominent herb in the early post-fire reforestation of western hemlock, lodgepole pine, and quaking aspen forests. Seip and Bunnell (1985) describe northern bedstraw in mountain grasslands resulting from stand-replacing fires in subalpine spruce forests of northern British Columbia. In coniferous forests of Alberta’s eastern Rockies, northern bedstraw is most frequent in recently burned areas (10 to 20 years since fire) (Cormack 1953).

In interior Alaska’s white spruce forests, Foote (1983) visited sites burned between 6 months and 200 or more years ago. Northern bedstraw frequency was greatest but coverage was lowest on sites burned 6 months prior in a surface fire that scorched stems and killed some trees (Foote 1983). One of the fires that was included in the previous post-fire recovery chronosequence was the 1950 Porcupine River fire. Foote (1993) investigated the post-fire vegetation recovery 1, 4, 7, 10, 23, and 30 years following the fire. Northern bedstraw frequency and coverage were greatest in the tenth post-fire year (Foote 1993).

Fire effects related mainly to severity: Generally, northern bedstraw increases following low-severity fires. The post-fire response of northern bedstraw to high-severity fires is less predictable. Researchers burned ponderosa pine-dominated forests in the fall on the Coeur d’Alene, Idaho, Indian Reservation. Different fire severity levels resulted. High-severity fires consumed 80% of the duff layer, and low-severity fires removed 40%.
Comparisons between unburned and burned sites revealed northern bedstraw frequency and coverage were greatest on sites burned in low-severity fires and lowest on sites burned in high-severity fires. However, treatment differences were not significant (p < 0.1) (Armour et al. 1984).

In mixed oak forests of Eastford, Connecticut, researchers burned 2 sites in April. The first site burned in 1984, and the second site burned in 1985. Fires burned under similar conditions; fuel moistures were between 18% and 28%, fire spread was slow (1m/min), and flame lengths were 30 cm or less. Within each site, portions burned more severely than others resulting in high mortality of the overstory. On the severely burned portion of site 1, 70% of the density and 60% of the basal area were removed. On the severely burned part of site 2, 95% of the density and basal area were removed. Northern bedstraw occurred only on burned sites. The density and frequency of northern bedstraw 7 to 8 years following these fires are shown below (Ducey et al. 1996).

<table>
<thead>
<tr>
<th>Site</th>
<th>Dominants</th>
<th>Density (stems/ha)</th>
<th>Frequency (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>intact</td>
<td>no</td>
</tr>
<tr>
<td></td>
<td>eastern white pine black oak white oak</td>
<td>0</td>
<td>40 600</td>
</tr>
<tr>
<td></td>
<td>northern red oak white oak sweet birch</td>
<td>5600</td>
<td>18 400</td>
</tr>
</tbody>
</table>

Pre-fire vegetation was compared to lightly (1%-20% of litter and duff consumed and 0-few trees killed), moderately (21%-80% of litter and duff consumed and < 90% of trees killed), and heavily (81%-100% of litter and duff consumed and > 90% of trees killed) burned vegetation following a late August prescription fire in quaking aspen-dominated communities of northwestern Wyoming. Northern bedstraw produced less biomass before the fire than 3 years following the fire on lightly and heavily burned sites. Northern bedstraw productivity was less 12 years following the fire than before the fire (Bartos and Mueggler 1981, Bartos et al. 1994). See the Research Project Summary Vegetation recovery following a mixed-severity fire in aspen groves of western Wyoming for an extended report on this fire study.

<table>
<thead>
<tr>
<th>Fire</th>
<th>Pre-fire</th>
<th>Production (kg/ha)</th>
<th>Post-fire year</th>
<th>Light</th>
<th>Moderate</th>
<th>Heavy</th>
<th>Light</th>
<th>Moderate</th>
<th>Heavy</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td></td>
<td>NA</td>
<td>3</td>
<td>3</td>
<td>12</td>
<td>33</td>
<td>42</td>
<td>14</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>
Fall prescription fires burned quaking aspen communities of Colorado’s Front Range. Fire severity was greater on plots with an understory of common juniper than on plots with an herbaceous understory. Northern bedstraw densities were significantly (p = 0.05) greater 1 year following fire. Increases were greater on less severely burned plots.

The differences for pre- and postburn northern bedstraw coverage and density are given below (Smith et al. 1985). See the Research Project Summary Vegetation changes following prescription fires in quaking aspen stands of Colorado’s Front Range for an extended report on this fire study.

<table>
<thead>
<tr>
<th>Community</th>
<th>Herbaceous understory (low severity)</th>
<th>Juniper understory (high severity)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Density (number of stems/0.1m)</td>
<td>1.2</td>
<td>3.1</td>
</tr>
<tr>
<td>Coverage (%)</td>
<td>0.7</td>
<td>0.5</td>
</tr>
</tbody>
</table>

Fire effects related mainly to seasonality: Dormant season fires (early spring or late fall) rarely cause decreases in northern bedstraw frequency and/or cover, but growing season (summer) fires may initially decrease northern bedstraw. In a central Saskatchewan rough fescue grassland, researchers compared the post-fire recovery of northern bedstraw following spring (May 6), summer (June 26), and fall (October 8) prescription fires. In the second post-fire season, northern bedstraw density was lower for spring and summer burns than for unburned and fall burned sites (Archibold et al. 2003). See the Research Project Summary Seasonal fires in Saskatchewan rough fescue prairie for an extended report on this fire study.

In central Alberta in 1972, almost pure, semimature quaking aspen stands burned in spring and fall prescription fires. Northern bedstraw coverage and frequency were greater on burned sites regardless of fire season or number of fires. Northern bedstraw coverage and frequency on burned sites, reburned sites, and unburned sites as of August 1978 are given below (Quintilio et al. 1991). See the Research Project Summary Understory recovery after burning and reburning quaking aspen stands in central Alberta for an extended report on this fire study.

<table>
<thead>
<tr>
<th>Site</th>
<th>Burned (October 1972)</th>
<th>Reburned (May 1978)</th>
<th>Unburned</th>
</tr>
</thead>
<tbody>
<tr>
<td>Frequency (%)</td>
<td>20</td>
<td>20</td>
<td>8</td>
</tr>
<tr>
<td>Cover (%)</td>
<td>8</td>
<td>3.2</td>
<td>1.2</td>
</tr>
<tr>
<td>Prominence value*</td>
<td>35.8</td>
<td>14.3</td>
<td>3.4</td>
</tr>
</tbody>
</table>

* Prominence value = % cover * (\sqrt{frequency})
Western snowberry-dominated communities southeast of Edmonton, Alberta, burned in spring prescription fires. Fires burned in early May of 1970 and 1971. The coverage of northern bedstraw was significantly greater on burned plots \((p < 0.05)\) 3 months following the fire. Researchers monitored vegetation for the next 2 growing seasons as well. Post-fire results are below (Anderson and Bailey 1979):

<table>
<thead>
<tr>
<th>Burn status</th>
<th>Unburned (n = 125)</th>
<th>Burned (n = 125)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Time since fire</td>
<td>0</td>
<td>3 months</td>
</tr>
<tr>
<td>Cover (%)</td>
<td>&lt; 0.05</td>
<td>4</td>
</tr>
<tr>
<td>Frequency (%)</td>
<td>7</td>
<td>22</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Burn status</th>
<th>Unburned (n = 23)</th>
<th>Burned (n = 28)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Cover (%)</td>
<td>1</td>
<td>&lt; 0.05</td>
</tr>
<tr>
<td>Frequency (%)</td>
<td>17</td>
<td>22</td>
</tr>
</tbody>
</table>

On the Namekagon River barrens of northern Wisconsin where jack pine and bur oak codominate, spring prescription fires burned. Northern bedstraw was common on both burned and unburned sites. On unburned sites, northern bedstraw averaged 89% frequency. On burned sites, frequency averaged 76%. An increased frequency of grasses following the fire may explain the slightly lower northern bedstraw frequency on burned sites (Vogl 1971).

McGee (1977) compared early spring and late summer prescription fires in northwestern Wyoming’s mountain big sagebrush communities. Two years after the fires, northern bedstraw coverage and frequency were greatest the second post-fire season on sites burned in the late summer. The coverage and frequency on unburned and spring burned sites were very similar (McGee 1977).

In a fescue-oatgrass community of southern Alberta, researchers compared burned and unburned vegetation following a mid-December wildfire in 1997. The Granum wildfire burned when temperatures averaged 13 °C, relative humidity was 17%, and winds were 30-40 km/h with gusts of 70 km/h. Pre-fire fuel loads were unavailable, but nearby unburned sites had 900 kg/ha litter loadings. Post-fire growing season precipitation was 46% greater than the long-term average. Northern bedstraw coverage was similar on interior burned plots and unburned plots 2 years following the fire. However, coverage was almost double on perimeter burn sites (those on the blackened side of fire line) when compared to unburned sites the first post-fire year (Bork et al. 2002).
An early-spring prescription fire (May 2, 1972) stimulated northern bedstraw flowering on burned undisturbed mesic, highly-disturbed mesic, and on highly-disturbed wet to mesic prairie sites of northeastern Minnesota. Disturbances on the sites included grazing, sod production, and hay production but were discontinued approximately 15 years prior to the study. The fire occurred during periods of high humidity, virtually no wind, and wet to damp soils (Pemble et al. 1981).

Repeated fires: The following studies report mixed post-fire responses of northern bedstraw following multiple fires. Some report a tolerance of annual fires while others suggest that multiple fires followed by multiple years of rest are favored by northern bedstraw. Higgins and others (1989) in a review suggest that northern bedstraw does not change or slightly decreases following periodic spring fires in the Northern Great Plains. In north-central South Dakota, northern bedstraw density was significantly greater (p < 0.05) on northern mixed prairie plots burned annually for 3 consecutive years in the fall (October 5-17) than on unburned control plots (Biondini et al. 1989).

In oak woodlands of east-central Minnesota's Cedar Creek Natural History Area, White (1986) compared several burning schedules. All fires burned in the spring. However, particular overstory densities and soil series of the different sites were significantly (p≤0.05) correlated with northern bedstraw and could not be reliably related to burned sites (White 1986).

Sweetscented bedstraw: The following information suggests that sweetscented bedstraw is not as fire tolerant as northern bedstraw. Fewer studies report increases in sweetscented bedstraw following low-severity and dormant season fires than were reported for northern bedstraw. In north-central Idaho western hemlock-western redcedar habitats, sweetscented bedstraw was absent 3 years following fire. Fire timing or severity are unknown. Steele and Geier-Hayes (1995) consider sweetscented bedstraw a major late-seral species in central Idaho’s Douglas-fir/ninebark habitat type that decreases following logging and wildfires.

As time since fire increases however, the presence of sweetscented bedstraw can decrease as well. In 1955 and 1956, Neiland (1958) compared northwestern Oregon’s mature (~300 years) western hemlock and Douglas-fir stands to sites that burned severely in 1933, 1939, and 1945. Sweetscented bedstraw was absent from unburned forests but averaged 3% frequency on burned sites. In forests codominated by balsam fir, black spruce, and paper birch around Lake Duparquet, Quebec, sweetscented bedstraw coverage was 1.9% on sites burned 26 years ago. On other sites that burned between 46 and 230 years ago, coverage of bedstraw varied from 0.1% to 0.5% (DeGrandpre et al. 1993).

The following studies highlight fire effects that are likely a result of fire severity or seasonality. These fire characteristics are difficult to consider singly; studies listed in this section highlight severity or seasonality.
Fire effects related mainly to severity: Sweetscented bedstraw can survive low- and high-severity fires, but typically unburned frequencies or coverages are greater than those of burned sites. Following a “holocaustic” fire that killed all above ground vegetation, consumed all litter, and left bare mineral soil in the Pack River Valley of northern Idaho, sweetscented bedstraw occurred on 5 of 18 sites and averaged 2% frequency (Stickney 1986).

Following severe fires in 270-year-old red pine stands of northeastern Minnesota, sweetscented bedstraw occurred at 40% frequency on burned sites and 93% frequency in unburned stands (Ahlgren 1979). See Seed banking for more information on this study.

In the Priest River Experimental Forest of northern Idaho, researchers compared the post-fire regeneration following dry and moist prescription fires. Douglas-fir, western redcedar, and grand fir mixed forests were harvested and burned. The moist season burn occurred on June 1, 1989, when air temperatures were 21-24° C, relative humidity was 43% to 50%, and winds were 1 to 12 km/h. The dry season fire burned on September 13 and 14, 1989, when air temperatures were 12-25° C, relative humidity was 39% to 66%, and winds were 1 to 8 km/h. Coverage of sweetscented bedstraw decreased on both the moist and dry burn sites. There was no statistical analysis of the data. However, decreases were greater on dry burn sites. Coverage increased on unburned sites (Simmerman et al. 1991):

<table>
<thead>
<tr>
<th>Fire type</th>
<th>Pre-fire cover (%)</th>
<th>Post-fire cover (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Unburned</td>
<td>1.4</td>
<td>3.2</td>
</tr>
<tr>
<td>Moist burn</td>
<td>0.9</td>
<td>0.7</td>
</tr>
<tr>
<td>Dry burn</td>
<td>1.6</td>
<td>0.8</td>
</tr>
</tbody>
</table>

Two forest sites within the Engelmann spruce-subalpine fir zone of central British Columbia were clearcut in the winter. One site burned in a low-severity prescribed fire the following fall. The coverage of sweetscented bedstraw on the burned sites had not regained pre-fire levels by 11 years post-fire. On logged unburned sites, increased sweetscented bedstraw coverage lasted for 5 years following the disturbance (Hamilton and Peterson 2003). See the Research Project Summary Revegetation in a subalpine forest after logging and fire in central British Columbia for an extended report on this study. Cox (1970) compared the recovery of sweetscented bedstraw in clearcut and clearcut and burned Douglas-fir forests of Oregon’s Coast Range. The slash burn produced a moderately severe fire (litter, duff, and woody debris consumed, but mineral soil color unchanged). No pre-fire data were available. Differences between burned and unburned plots 1 and 2 years following fire were negligible (Cox 1970).
While decreases in sweetscented bedstraw coverage and frequency following fire predominate, the frequency of sweetscented bedstraw increased following low-severity, spring prescription fires in quaking aspen woodlands of southern Ontario. The frequency of sweetscented bedstraw on unburned sites was 4%. The frequency 4 months post-fire was 21% and a little over 1 year post-fire was 12.5% (Smith and James 1978). Following a mid-July crown fire near Missoula, Montana, sweetscented bedstraw frequency had doubled from the first to the second post-fire year (Crane et al. 1983).

Fire effects related mainly to seasonality: Many of the following studies suggest that spring and fall fires may increase the frequency of sweetscented bedstraw, while summer fires may decrease its frequency. Following spring fires in American beech-sugar maple and black oak-red maple forests in south-central New York, burned and unburned sites were compared. Sweetscented bedstraw frequency was significantly higher (p = 0.01) on burned sites; frequency on unburned sites was 2.4% and on burned sites was 28.6% (Swan 1966).

In mixed conifer-hardwood forests of northeastern Minnesota, researchers assessed vegetation recovery in burned areas. Two sites dominated by black spruce, jack pine, and paper birch burned, one in late April and the other in mid-July. The late April fire occurred during high winds, leaving small unburned patches. Sweetscented bedstraw frequency of occurrence on burned sites was over double that of unburned sites 3 years following the spring fire. The data collected on burned and unburned sites are summarized below (Krefting and Ahlgren 1974):

<table>
<thead>
<tr>
<th>Fire</th>
<th>Unburned</th>
<th>Spring fire</th>
<th>Summer fire</th>
</tr>
</thead>
<tbody>
<tr>
<td>Years post-fire</td>
<td>0</td>
<td>3 5 14</td>
<td>2 5 11</td>
</tr>
<tr>
<td>Occurrence (%)</td>
<td>3 3 7</td>
<td>3 7 0 3 3 3</td>
<td></td>
</tr>
</tbody>
</table>

In white pine forests of Strafford County, New Hampshire, fall (1976) and spring (1977) prescription fires burned. The fires produced flame lengths of 7.6 - 61 cm and scorched trees at heights of 0.6 - 2.4 m. Sweetscented bedstraw was not on control plots and was not present on plots before the fire. However, it did occur following the fall and spring fires on white pine-dominated forests and following the spring fires in white pine mixed forests. Sweetscented bedstraw plants on the burned plots resulted from seed germination. Plants on the spring-burned plots matured by late July and produced seed by the end of the growing season. The survival and/or development of plants on fall burned plots is unknown (Chapman and Crow 1981, Ross 1978).

Sweetscented bedstraw frequency decreased, but coverage was unchanged following a prescription fire in beetle-damaged white spruce forests of southern Alaska’s Chuguch National Forest. The fire top-killed all overstory and understory vegetation in June of 1984. Pre-fire (1980) coverage of sweetscented bedstraw was 2%, and frequency of occurrence was 24%. Seven years following the fire coverage remained 2% and the frequency of occurrence was 12% (Holsten et al. 1995).
A prescription head fire within the Grand fir-Oregon boxwood (Paxistima myrsinites) habitat type of north-central Idaho also decreased the frequency of sweetscented bedstraw. The fire burned mid-May of 1975 when temperatures were 28° C, relative humidity was 25%, and winds were negligible. Decreases in frequency were greater for sites that were grass seeded than unseeded sites following the fire. Statistical significance of the results was not addressed. Sweetscented bedstraw frequency of occurrence is provided below (Leege and Godbolt 1985):

<table>
<thead>
<tr>
<th>Time since fire</th>
<th>Pre-fire</th>
<th>3 months</th>
<th>1 year</th>
<th>2 years</th>
</tr>
</thead>
<tbody>
<tr>
<td>Burned &amp; seeded (# of occurrences/10 plots)</td>
<td>6</td>
<td>0</td>
<td>0</td>
<td>2</td>
</tr>
<tr>
<td>Burned (# of occurrences/10 plots)</td>
<td>3</td>
<td>2</td>
<td>1</td>
<td>1</td>
</tr>
</tbody>
</table>

Repeated fires: The only study reporting sweetscented bedstraw recovery following multiple fires indicates a tolerance of annual fires for up to 3 years. In southern Ohio hardwood forests, prescription fires burned some sites once and burned other sites for 3 consecutive years in March and April. Flame lengths were less than 50 cm, and fire severity was low. The frequency of sweetscented bedstraw increased by more than 10% on burned plots (Hutchinson and Sutherland 2000).

FIRE MANAGEMENT CONSIDERATIONS

Fire management decisions are likely unaffected by the presence of bedstraw in the understory. Bedstraw recovers quickly, remains unchanged, or increases following fire. Special consideration of bedstraw when developing a fire management plan is unnecessary in most cases.

However, burning bedstraw may increase its forage value as indicated by the following study. A tall grass prairie in eastern North Dakota burned in early May of 1966. Frequencies of northern bedstraw were the same on burned and unburned sites, but herbage production was much greater on burned sites. Statistical comparisons were not made. The results of this study are summarized below (Hadley 1970):

<table>
<thead>
<tr>
<th>Site condition</th>
<th>Herbage weight (dry g/m²)</th>
<th>Calories/g (ash-free)</th>
<th>Calories/m²</th>
<th>% total (calories/m²)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Unburned</td>
<td>2.8</td>
<td>4,690</td>
<td>12,991</td>
<td>0.85</td>
</tr>
<tr>
<td>Burned</td>
<td>8.8</td>
<td>4,654</td>
<td>40,723</td>
<td>2.07</td>
</tr>
</tbody>
</table>
LITERATURE CITED


Hypericum perforatum
St. Johnswort

FIRE ECOLOGY OR ADAPTATIONS

Fire adaptations: Mature St. Johnswort plants have deep, extensive perennial root systems, and reproduce vegetatively from lateral roots and root crowns. Vegetative propagation in St. Johnswort seems to be stimulated by grazing, fire, and defoliation (Clark 1953, Tisdale et al. 1959) (see Asexual regeneration). This adaptation allows St. Johnswort to survive even severe fire, depending on site conditions. A protective tissue of suberized cells called polyderm has been described on St. Johnswort roots (Esau 1960, as cited by Crompton et al 1988). It is unclear whether this tissue might provide protection from heat.

St. Johnswort also establishes from seed, and St. Johnswort seed is commonly found in soil seed banks. Estimates of 6 to 30 years or more have been suggested for longevity of viable St. Johnswort seed in soil seed banks (see Seed banking). Additionally, heat seems to stimulate germination in St. Johnswort seed, and researchers have observed flushes of St. Johnswort seedlings following fire (Briese 1996, Sampson and Parker 1930, Walker 2000).

Fire regimes: St. Johnswort occurs in a wide variety of ecosystems in North America which represent a wide range of historic fire regimes. In many areas where St. Johnswort occurs, historic fire regimes have been dramatically altered due to fire exclusion and to massive disturbances associated with human settlement. The historic fire regimes of native communities in which St. Johnswort sometimes occurs range from low frequency, high-severity stand replacing fires in wet forest types; to high frequency, high-severity fires in prairie grasslands; to high frequency, low-severity fires in open ponderosa pine forests. St. Johnswort did not occur in these communities at the time in which historic fire regimes were functioning, but has established since fire exclusion and habitat alteration began. It is unclear how historic fire regimes might affect St. Johnswort populations.

St. Johnswort also occurs in areas where annual grasses such as cheatgrass are dominant. Fire regime change due to invasion of annual grasses is well documented (Brooks et al. 2004, D’Antonio 2000). Cheatgrass expansion has dramatically changed fire regimes and plant communities over vast areas of western rangelands by changing the fuel properties of invaded communities (sensu Brooks et al. 2004) and thus creating an environment where fires are easily ignited, spread rapidly, cover large areas, and occur frequently (Young and Evans 1981). Short fire return intervals in cheatgrass-dominated communities (< 10 years [Peters and Bunting 1994, Whisenant 1990]) may favor St. Johnswort, with its large root system, ability to sprout after injury, and seed germination stimulated by heat. More research and field observations are needed to understand how St. Johnswort responds to the current fire ecology of these areas.
It is also unclear how the presence of St. Johnswort might affect fire regimes in invaded communities. In general, in ecosystems where St. Johnswort replaces plants similar to itself (in terms of fuel characteristics), St. Johnswort may alter fire intensity or slightly modify an existing fire regime. However, if St. Johnswort is qualitatively unique to the invaded ecosystem, it has the potential to completely alter the fire regime (sensu Brooks et al. 2004, D’Antonio 2000). Two authors suggest that presence of dry senescent stems of St. Johnswort create a fire hazard in forest areas in California (Sampson and Parker 1930) and Australia (Parsons and Cuthbertson 1992). It is unclear whether these assertions are based on conjecture or on observations made by the authors. No examples of historic fire regimes altered by St. Johnswort invasion are described in the available literature.

The following list provides fire return intervals for plant communities and ecosystems where St. Johnswort may be important.

<table>
<thead>
<tr>
<th>Community or Ecosystem</th>
<th>Dominant Species</th>
<th>Fire Return Interval Range (years)</th>
</tr>
</thead>
<tbody>
<tr>
<td>silver fir-Douglas-fir</td>
<td><em>Abies amabilis-Pseudotsuga menziesii var. menziesii</em></td>
<td>&gt; 200</td>
</tr>
<tr>
<td>grand fir</td>
<td><em>Abies grandis</em></td>
<td>35-200 [1]</td>
</tr>
<tr>
<td>maple-beech-birch</td>
<td><em>Acer-Fagus-Betula</em></td>
<td>&gt; 1,000</td>
</tr>
<tr>
<td>silver maple-American elm</td>
<td><em>Acer saccharinum-Ulmus americana</em></td>
<td>&lt; 35 to 200</td>
</tr>
<tr>
<td>sugar maple</td>
<td><em>Acer saccharum</em></td>
<td>&gt; 1,000</td>
</tr>
<tr>
<td>sugar maple-basswood</td>
<td><em>Acer saccharum-Tilia americana</em></td>
<td>&gt; 1,000 [2]</td>
</tr>
<tr>
<td>California chaparral</td>
<td><em>Adenostoma and/or Arctostaphylos spp.</em></td>
<td>&lt; 35 to &lt; 100 [3]</td>
</tr>
<tr>
<td>bluesem prairie</td>
<td><em>Andropogon gerardii var. gerardii-Schizachyrium scoparium</em></td>
<td>&lt; 10 [4,3]</td>
</tr>
<tr>
<td>silver sagebrush steppe</td>
<td><em>Artemisia cana</em></td>
<td>5-45 [5,6,7]</td>
</tr>
<tr>
<td>sagebrush steppe</td>
<td><em>Artemisia tridentata/Pseudoroegneria spicata</em></td>
<td>20-70 [3]</td>
</tr>
<tr>
<td>basin big sagebrush</td>
<td><em>Artemisia tridentata var. tridentata</em></td>
<td>12-43 [8]</td>
</tr>
<tr>
<td>mountain big sagebrush</td>
<td><em>Artemisia tridentata var. vaseyana</em></td>
<td>15-40 [9,10,11]</td>
</tr>
<tr>
<td>Wyoming big sagebrush</td>
<td><em>Artemisia tridentata var. wyomingensis</em></td>
<td>10-70 (40**) [12,13]</td>
</tr>
<tr>
<td>coastal sagebrush</td>
<td><em>Artemisia californica</em></td>
<td>&lt; 35 to &lt; 100 [3]</td>
</tr>
<tr>
<td>plains grasslands</td>
<td><em>Bouteloua spp.</em></td>
<td>&lt; 35 [3,7]</td>
</tr>
<tr>
<td>cheatgrass</td>
<td><em>Bromus tectorum</em></td>
<td>&lt; 10 [14,15]</td>
</tr>
<tr>
<td>California montane chaparral</td>
<td><em>Ceanothus and/or Arctostaphylos spp.</em></td>
<td>50-100 [3]</td>
</tr>
<tr>
<td>sugarberry-America elm-green ash</td>
<td><em>Celtis laevigata-Ulmus americana-Fraxinus pennsylvanica</em></td>
<td>&lt; 35 to 200 [2]</td>
</tr>
<tr>
<td>Ecological Community</td>
<td>Species Description</td>
<td>Dominance Range</td>
</tr>
<tr>
<td>---------------------------------------------</td>
<td>---------------------------------------------</td>
<td>-----------------</td>
</tr>
<tr>
<td>curlleaf mountain-mahogany*</td>
<td>Cercocarpus ledifolius</td>
<td>13-1,000 [16,17]</td>
</tr>
<tr>
<td>northern cordgrass prairie</td>
<td>Distichlis spicata-Spartina spp.</td>
<td>1-3 [3]</td>
</tr>
<tr>
<td>beech-sugar maple</td>
<td>Fagus spp.-Acer saccharum</td>
<td>&gt; 1,000 [2]</td>
</tr>
<tr>
<td>California steppe</td>
<td>Festuca-Danthonia spp.</td>
<td>&lt; 35 [3,18]</td>
</tr>
<tr>
<td>black ash</td>
<td>Fraxinus nigra</td>
<td>&lt; 35 to 200 [2]</td>
</tr>
<tr>
<td>juniper-oak savanna</td>
<td>Juniperus ashei-Quercus virginiana</td>
<td>&lt; 35</td>
</tr>
<tr>
<td>western juniper</td>
<td>Juniperus occidentalis</td>
<td>20-70</td>
</tr>
<tr>
<td>Rocky Mountain juniper</td>
<td>Juniperus scopulorum</td>
<td>&lt; 35 [3]</td>
</tr>
<tr>
<td>cedar glades</td>
<td>Juniperus virginiana</td>
<td>3-22 [19,3]</td>
</tr>
<tr>
<td>western larch</td>
<td>Larix occidentalis</td>
<td>25-350 [20,21,22]</td>
</tr>
<tr>
<td>wheatgrass plains grasslands</td>
<td>Pascopyrum smithii</td>
<td>&lt; 5-47+ [3,6,7]</td>
</tr>
<tr>
<td>Great Lakes spruce-fir</td>
<td>Picea-Abies spp.</td>
<td>35 to &gt; 200 [23]</td>
</tr>
<tr>
<td>pine-cypress forest</td>
<td>Pinus-Cupressus spp.</td>
<td>&lt; 35 to 200 [1]</td>
</tr>
<tr>
<td>pinyon-juniper</td>
<td>Pinus-Juniperus spp.</td>
<td>&lt; 35 [3]</td>
</tr>
<tr>
<td>Rocky Mountain lodgepole pine*</td>
<td>Pinus contorta var. latifolia</td>
<td>25-340 [24,21,25]</td>
</tr>
<tr>
<td>Sierra lodgepole pine*</td>
<td>Pinus contorta var. murrayana</td>
<td>35-200</td>
</tr>
<tr>
<td>western white pine*</td>
<td>Pinus monticola</td>
<td>50-200</td>
</tr>
<tr>
<td>Pacific ponderosa pine*</td>
<td>Pinus ponderosa var. ponderosa</td>
<td>1-47 [1]</td>
</tr>
<tr>
<td>interior ponderosa pine*</td>
<td>Pinus ponderosa var. scopulorum</td>
<td>2-30 [1,26,27]</td>
</tr>
<tr>
<td>red pine (Great Lakes region)</td>
<td>Pinus resinosa</td>
<td>10-200 (10**) [23,28]</td>
</tr>
<tr>
<td>red-white-jack pine*</td>
<td>Pinus resinosa-P. strobus-P. banksiana</td>
<td>10-300 [23,29]</td>
</tr>
<tr>
<td>eastern white pine</td>
<td>Pinus strobus</td>
<td>35-200</td>
</tr>
<tr>
<td>eastern white pine-eastern hemlock</td>
<td>Pinus strobus-Tsuga canadensis</td>
<td>35-200</td>
</tr>
<tr>
<td>eastern white pine-northern red oak</td>
<td>Pinus strobus-Quercus rubra-Acer rubrum</td>
<td>35-200</td>
</tr>
<tr>
<td>red maple</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Virginia pine</td>
<td>Pinus virginiana</td>
<td>10 to &lt; 35</td>
</tr>
<tr>
<td>Virginia pine-oak</td>
<td>Pinus virginiana-Quercus spp.</td>
<td>10 to &lt; 35</td>
</tr>
<tr>
<td>sycamore-sweetgum-American elm</td>
<td>Platanus occidentalis-Liquidambar styraciflua-Ulmus americana</td>
<td>&lt; 35 to 200 [2]</td>
</tr>
<tr>
<td>eastern cottonwood</td>
<td>Populus deltoides</td>
<td>&lt; 35 to 200 [3]</td>
</tr>
<tr>
<td>aspen-birch</td>
<td>Populus tremuloides-Betula papyfera</td>
<td>35-200 [23,2]</td>
</tr>
<tr>
<td>quaking aspen (west of the Great Plains)</td>
<td>Populus tremuloides</td>
<td>7-120 [1,30,31]</td>
</tr>
<tr>
<td>black cherry-sugar maple</td>
<td>Prunus serotina-Acer saccharum</td>
<td>&gt; 1,000 [2]</td>
</tr>
<tr>
<td>mountain grasslands</td>
<td>Pseudoroegneria spicata</td>
<td>3-40 (10**) [32,1]</td>
</tr>
<tr>
<td>Rocky Mountain Douglas-fir*</td>
<td>Pseudotsuga menziesii var. glauca</td>
<td>25-100 [1,9,33]</td>
</tr>
<tr>
<td>coastal Douglas-fir*</td>
<td>Pseudotsuga menziesii var.</td>
<td>40-240 [1,34,35]</td>
</tr>
<tr>
<td>Ecosystem Type</td>
<td>Species Description</td>
<td>Fire Interval</td>
</tr>
<tr>
<td>-------------------------------------------</td>
<td>----------------------------------------------------------</td>
<td>---------------</td>
</tr>
<tr>
<td>California mixed evergreen</td>
<td><em>Pseudotsuga menziesii</em> var. <em>menziesii-Lithocarpus densiflorus-Arbutus menziesii</em></td>
<td>&lt; 35</td>
</tr>
<tr>
<td>California oakwoods</td>
<td><em>Quercus</em> spp.</td>
<td>&lt; 35 [1]</td>
</tr>
<tr>
<td>oak-hickory</td>
<td><em>Quercus-Carya</em> spp.</td>
<td>&lt; 35</td>
</tr>
<tr>
<td>northeastern oak-pine</td>
<td><em>Quercus</em>-<em>Pinus</em> spp.</td>
<td>10 to &lt; 35 [2]</td>
</tr>
<tr>
<td>coast live oak</td>
<td><em>Quercus</em> agrifolia</td>
<td>2-75 [36]</td>
</tr>
<tr>
<td>white oak-black-oak-northern red oak</td>
<td><em>Quercus alba-Q. velutina-Q. rubra</em></td>
<td>&lt; 35 [2]</td>
</tr>
<tr>
<td>canyon live oak</td>
<td><em>Quercus chrysolepis</em></td>
<td>&lt; 35 to 200</td>
</tr>
<tr>
<td>blue oak-foothills pine</td>
<td><em>Quercus douglasii-P. sabiniana</em></td>
<td>&lt; 35 [1]</td>
</tr>
<tr>
<td>northern pin oak</td>
<td><em>Quercus ellipsoideal</em></td>
<td>&lt; 35 [2]</td>
</tr>
<tr>
<td>Oregon white oak</td>
<td><em>Quercus garryana</em></td>
<td>&lt; 35 [1]</td>
</tr>
<tr>
<td>California black oak</td>
<td><em>Quercus kelloggii</em></td>
<td>5-30 [3]</td>
</tr>
<tr>
<td>oak savanna</td>
<td><em>Quercus macrocarpa/Andropogon gerardii-Schizachyrium scoparium</em></td>
<td>2-14 [3,2]</td>
</tr>
<tr>
<td>northern red oak</td>
<td><em>Quercus rubra</em></td>
<td>10 to &lt; 35</td>
</tr>
<tr>
<td>post oak-blackjack oak</td>
<td><em>Quercus stellata-Q. marilandica</em></td>
<td>&lt; 10</td>
</tr>
<tr>
<td>black oak</td>
<td><em>Quercus velutina</em></td>
<td>&lt; 35</td>
</tr>
<tr>
<td>live oak</td>
<td><em>Quercus virginiana</em></td>
<td>10 to &lt; 100 [2]</td>
</tr>
<tr>
<td>interior live oak</td>
<td><em>Quercus wislizenii</em></td>
<td>&lt; 35 [1]</td>
</tr>
<tr>
<td>little bluestem-grama prairie</td>
<td><em>Schizachyrium scoparium-Bouteloua</em> spp.</td>
<td>&lt; 35 [3]</td>
</tr>
<tr>
<td>western redcedar-western hemlock</td>
<td><em>Thuja plicata-Tsuga heterophylla</em></td>
<td>&gt; 200</td>
</tr>
<tr>
<td>western hemlock-Sitka spruce</td>
<td><em>Tsuga heterophylla-Picea sitchensis</em></td>
<td>&gt; 200</td>
</tr>
<tr>
<td>mountain hemlock*</td>
<td><em>Tsuga mertensiana</em></td>
<td>35 to &gt; 200 [1]</td>
</tr>
<tr>
<td>elm-ash-cottonwood</td>
<td><em>Ulmus-Fraxinus-Populus</em> spp.</td>
<td>&lt; 35 to 200 [23,2]</td>
</tr>
</tbody>
</table>

*Fire return interval varies widely; trends in variation are noted in the species review

**Mean**


POST-FIRE REGENERATION STRATEGY (Stickney 1989):

Caudex/herbaceous root crown, growing points in soil; geophyte, growing points deep in soil; ground residual colonizer (on-site, initial community); initial off-site colonizer (off-site, initial community); secondary colonizer (on-site or off-site seed sources).

IMMEDIATE FIRE EFFECT ON PLANT

Little information is available regarding the immediate effects of fire on St. Johnswort stems, roots, and seeds. Fire is likely to top-kill St. Johnswort; however, fire may or may not damage St. Johnswort root crowns and lateral roots. One author reports that St. Johnswort lateral roots occur 1-8 cm below the soil surface (Clark 1953), a depth at which they may be damaged by severe fire. Evidence presented by Sampson and Parker (1930) and field observations by other researchers (Briese 1996, Walker 2000) suggest that fire may stimulate sprouting from undamaged St. Johnswort roots and root crowns, and germination in St. Johnswort seeds. According to observations presented by Agee (1994), even high-severity fire may stimulate sprouting and/or seed germination in St. Johnswort.

DISCUSSION AND QUALIFICATION OF FIRE EFFECT

Field observations (Briese 1996, Sampson and Parker 1930, Walker 2000) and laboratory tests (Sampson and Parker 1930) suggest that fire stimulates germination in St. Johnswort seed, although it is unclear how fire severity and season of burning may affect germination response in St. Johnswort.

In fall 1996, more than 400 ha of land planted to hard fescue (Festuca trachyphylla), tall wheatgrass (Thinopyrum ponticum), and alfalfa (Medicago sativa) were burned under prescription at the Kilby Ranch in Oregon. Before the burn, isolated patches of St. Johnswort occurred around the perimeter of the ranch, with the largest patches south of the burn area. Within 7 months of the fire, immature St. Johnswort plants...
(presumably seedlings, although this is not clearly stated) established across the burn area. The following season (1998), St. Johnswort plants had matured and the St. Johnswort infestation was very dense. By the third post-fire year (1999), St. Johnswort had spread beyond the boundary of the burn area. The author suggests that an extensive St. Johnswort seed bank was present and stimulated to germinate by the fire (Walker 2000).

Similarly, a rangeland site was burned twice under prescription in California in October of 1926 and 1927 in an effort to control St. Johnswort. The fire completely consumed St. Johnswort crowns, and apparently stimulated germination in St. Johnswort seed. The authors describe large numbers of St. Johnswort seedlings where the fire had been “very hot.” Timing of seedling emergence relative to time of burning is not given, nor do the authors indicate how it was determined that St. Johnswort plants were seedlings and not root sprouts. The authors do indicate that a temperature of 127° C was recorded at 0.6 cm below the soil surface in selected localities during the field burning operation, but they do not describe how this was measured, or any other temperatures recorded during the burn.

Based on their observations, the authors conducted laboratory experiments to test the effects of various heat treatments on St. Johnswort seed. An oven was used to expose St. Johnswort seeds to 100° C for 5, 15, 30, and 60 minutes; or to 127° C for periods of 1 to 5 minutes. Seeds were then placed in sterilized sand and germinated in the greenhouse. The unheated control seed lots had the lowest germination rate at 44%, while seed lots exposed to 100° C for 5, 15, 30, and 60 minutes had germination rates of 52%, 63%, 81%, and 75%, respectively. The germination rates of seed exposed to 127° C were not given, although the authors state, “even at this temperature the seed gave a distinctly higher percentage of germination than did the untreated seeds” (Sampson and Parker 1930).

It is unclear whether germination of St. Johnswort seed after fire is a function of heat stimulation of germination or of reduction in plant cover that allows for seedling emergence. For example, Greiling and Kichanan (2002) found that St. Johnswort seedling emergence was 100 times higher (p < 0.01) when plant neighbors (little bluestem and old field species) were removed.

According to Agee (1994), severe burning associated with log corridors in disturbed Oregon white oak woodlands provides favorable sites for many nonnative species such as St. Johnswort, common velvetgrass (Holcus lanatus), and tansy ragwort. It is unclear whether the author refers to seedlings or to sprouts from established roots or root crowns, and what this observation is based on.
PLANT RESPONSE TO FIRE

It is generally purported that fire encourages establishment, vegetative spread, and increased density of St. Johnswort patches (Campbell and Delfosse 1984, Clark 1953) by stimulating germination of St. Johnswort seed (see Discussion and Qualification of Fire Effect) and sprouting in surviving St. Johnswort roots and root crowns (Briese 1996, Sampson and Parker 1930, Tisdale et al. 1959). Several references indicate that St. Johnswort often occurs in previously burned areas, especially forests (e.g. Campbell and Delfosse 1984, Catling and Brownell 1998, Catling et al. 2001, Cholewa 1977, Clark 1953, Everett 1997, Lavendar 1958, Streatfield and Frenkel 1997, Tisdale et al. 1959).

Accounts in the literature of St. Johnswort’s response to fire are varied, from no response; to immediate increases in cover and/or density; to immediate decreases in cover and/or density, followed by increases several post-fire years later. Because most information available in the literature on St. Johnswort’s response to fire comes from studies in which the response of St. Johnswort to fire was not the primary objective of the study, and because all variables and details of fires are not consistently reported, it is unclear why results differ among reports.

DISCUSSION AND QUALIFICATION OF PLANT RESPONSE

Variation in St. Johnswort response to fire may be affected by plant community type, fire size, fire severity and/or season of burning. A study from New South Wales, Australia, suggests that fire severity, as influenced by plant community type and associated fuel loads, may affect St. Johnswort’s response to fire (Briese 1996). During the course of monitoring populations of biological control insects in a native forest in southeastern New South Wales, a monitoring site was burned under prescription in an effort to reduce risk of major wildfires. Briese (1996) examined the effects of these fuel reduction burns on St. Johnswort populations and populations of associated biological control insects at this site.

The study site included areas with 2 distinct plant communities: an area of open woodland dominated by eucalyptus species (Eucalyptus pauciflora and E. stellulata) and an area in an adjacent clearing. Before burning, ground cover consisted of perennial grasses and various forbs including St. Johnswort. In the open area total ground cover was 50% to 70%, with 12% to 36% St. Johnswort. In the timbered area, total ground cover was 22% to 44% total with 12% to 24% St. Johnswort. The site was burned in spring (September) 1982. Details of the fire were not given; however, fire intensity (severity) was estimated from the percentage of ground cover consumed, mortality of marked St. Johnswort plants, and height of crown scorch in the timbered area. In the open area, ground cover was reduced 59% ± 11%, with 9% mortality of St. Johnswort crowns.
St. Johnswort cover returned to pre-fire levels (12% to 36%) rapidly in the open area, with a slight increase in crown density within months of the fire. In the timbered area, ground cover was reduced 100%, with 64% St. Johnswort crown mortality. St. Johnswort recovered more slowly in the timbered area than in the open area, although it recovered more rapidly than associated vegetation, mainly by growth from surviving roots. By summer (January) 1984, St. Johnswort had reached 65% cover in the timbered area, mainly due to enhanced growth of individual plants rather than an increase in crown density. A similar, but less extreme increase (to about 45% cover) of St. Johnswort was seen in the open plots. The net result was a very large increase in the production of St. Johnswort flowering stems and seed in both areas in the summer of 1983/1984 (Briese 1996).

Germination of St. Johnswort seeds in the first post-fire season was higher than average, but contributed little to plant recovery. Post-fire growth of St. Johnswort was mainly from surviving roots, whereas associated grasses and herbs reestablished from seeds that did not germinate until the following autumn. A regression of vegetative regeneration against the proportion of original crown surviving the fire suggests that fire can stimulate regrowth in surviving rootstocks. When damage is light, as in the open plots, this can lead to regeneration that is greater than the replacement rate, resulting in an increase in the proportion of crowns originating from roots after the fire (from 78.3% ± 2.8% to 91.6% ± 2.1% in the open area and from 82.7% ± 4.8% to 93.8% ± 2.4% in the timbered area, p < 0.05 in both cases). In the severely burned timbered plots, regeneration did not equal replacement, though growth of individual plants was greatly enhanced. The response of St. Johnswort and of introduced biocontrol insects to increased soil fertility following fire is important to post-fire population dynamics of both (see Fire interactions with other control methods) (Briese 1996).

Results presented by Hooker and Tisdale (1974) following prescribed burning on a seral brush community in the Lochsa River area in northern Idaho suggest a somewhat different relationship of St. Johnswort recovery relative to fire severity. The authors state that St. Johnswort “increased following low intensity burning but did not benefit when the treatment was more intense.” However, plant recovery was only measured for 1 season following burning, and the data upon which this conclusion was based were not provided. Surface soil temperatures were measured in this study using pyrometers, and sites with highest recorded temperatures supported dense stand of old brush (various shrubs) and bracken fern. Bracken fern recovered rapidly following fire and grew in dense stands, possibly explaining why St. Johnswort did not recover as well on high severity sites (Hooker and Tisdale 1974).
Season of burning may also affect St. Johnswort response. A second prescribed fire was conducted in the New South Wales study area (described above) in fall (March) 1986. In the open area, ground cover was reduced 96% + 2%, while ground cover was reduced 85% + 9% in the timbered area by the fall burn. There was massive germination of herbs and grasses, including St. Johnswort, following the fall burn. Because St. Johnswort seedlings are poor competitors, the native grass and forb component remained dominant in both areas in 1987, despite some recovery of mature St. Johnswort plants (Briese 1996).

Other fire studies where St. Johnswort occurred in the plant community but was not the focus of the study provide no clear picture of St. Johnswort’s response to fire. Results from studies in Oregon, Washington, and Idaho are presented to demonstrate the variability of this response. A native wetland prairie site in Willamette Valley, Oregon, dominated by tufted hairgrass and invaded by several woody species, was treated to remove woody species by burning, hand-removal, or mowing. St. Johnswort was present before treatments were imposed. Where plots were burned or woody plants removed by hand, St. Johnswort cover was also reduced. The authors speculate that this reduction may have been due to increased abundance of insect herbivores with increased light (Clark and Wilson 2001). A moderate-severity fire in a snowberry-rose association in northeastern Oregon had little effect on St. Johnswort cover. Pre-fire cover of St. Johnswort averaged 1% in a stock exclosure and 10% in a game exclosure. No St. Johnswort was found 1 year after fire, and St. Johnswort cover was 3% in the stock exclosure and 9% in the game exclosure 5 years after fire (Johnson 1998).

Thickets of scotchbroom in prairies and oak woodlands in Ft. Lewis, Washington, were burned under prescription in fall (September) 1994 to try to reduce frequency and density of scotchbroom. Average pre-fire cover of St. Johnswort was 1.1%, and post-fire cover, recorded in May 1995, was 1.9%. These results indicate no significant ($p < 0.05$) change in St. Johnswort cover; however, the duration of the study is insufficient to be conclusive. Additionally, no data are given for St. Johnswort response to spring burning in the same study (some fire details are given) (Tveten and Fonda 1999). Brush covered slopes in northern Idaho were burned in May 1975, and seeded with several nonnative herbaceous species in May 1975 in an effort to improve winter/spring forage for elk. Plant frequency and green weight production were measured for 4 growing seasons following burning. St. Johnswort “occurred on all 3 study areas and did not show any obvious changes following any treatment” (some fire details are given) (Leege and Godbolt 1985).
FIRE MANAGEMENT CONSIDERATIONS

Fire as a control agent: While experimental evidence is inconclusive regarding St. Johnswort’s response to fire, much of the available literature suggests that fire increases frequency and density of St. Johnswort (Briese 1996, Campbell and Delfosse 1984, Clark 1953, Sampson and Parker 1930, Tisdale et al. 1959). Therefore, burning is not indicated as a potentially effective method for controlling St. Johnswort. Other evidence suggests that, in some cases, burning may provide effective control for St. Johnswort. Very little literature examines the effects of prescribed burning intended to control St. Johnswort.

According to Sampson and Parker (1930), some stockmen contend that burning infestations of St. Johnswort when the tops are dry in autumn will kill the present cover and destroy the accumulated seed. By burning 2 or 3 years in succession it was suggested that grass may invade the area and crowd out any remaining St. Johnswort plants. To test these suggestions, 120 ha near Blocksburg, California was burned “closely” in October of 1926 and 1927. The fire completely consumed St. Johnswort tops, and “carried” to all isolated patches, leaving no tops unburned. Density of St. Johnswort was measured before the first burn and after the second burn in “representative” areas. However, these data are not reported, nor do the authors indicate how long after burning density measurements were taken. The authors summarize results by stating “instead of the weed cover being killed or subsequently thinned out by invading grasses, the stand of St. Johnswort was denser and seemingly more vigorous than before.” They further suggest that not only does fire stimulate germination of St. Johnswort seed, but that repeated burning may deplete the soil of organic material and thus favor St. Johnswort and other undesirable plants (Sampson and Parker 1930). Similarly, at Dye Creek and Vina Plains Preserves in California, St. Johnswort is said to be encouraged by burning, and the preserves' manager recommends against using prescribed fire in St. Johnswort-infested areas (Rice and Randall 2004).

Conversely, preserve managers for The Nature Conservancy in Michigan and Ohio indicate that fire suppression encourages invasion of St. Johnswort, while burning and restoration treatments discourage invasion of St. Johnswort. At Kitty Todd, Ohio, where St. Johnswort is found in areas that were previously farmed and grazed and around old homesites, burning “seems to be somewhat effective” as a control method for St. Johnswort. Unfortunately, further details on how and when burning was conducted are not available (Rice and Randall 2004).

According to Jack McGowan-Stinski, Fire Manager for Michigan sites of The Nature Conservancy, St. Johnswort has been reduced or eradicated with both prescribed fire and spot-burning (using propane torches), with and without additional control methods. Control, using either prescribed or spot-burning, is most successful in dry sand prairies and oak barrens, possibly due to nutrient-poor soil conditions.
Prescribed burning on sites where there is a diversity of native prairie grasses (little bluestem, big bluestem, prairie dropseed (*Sporobolus heterolepis*), and sideoats grama (*Bouteloua curtipendula*)) and native forbs eliminates St. Johnswort when burning is conducted during the growing season or early fall (June, July, August), and repeated for 2 to 3 consecutive years. Native plants on these sites are adapted to fires during this season and out-compete St. Johnswort in the post-fire environment. In areas with dense St. Johnswort populations, repeated spot-burning has successfully reduced or eliminated St. Johnswort. This is also likely due to native perennial prairie plants’ tolerance of repeated burning (in the same year) in late summer or early fall. Prescribed or spot burning followed by hand removal of St. Johnswort sprouts and seedlings by volunteers throughout the season is also successful. The advantages of burning followed by hand removal include fewer restrictions by weather, staff time, and equipment, and lower associated costs (McGowan-Stinski 2005).

Fire interactions with other control methods: Fire managers may need to include the existence of biological control agents in their decision making process and fire management plans. Fire may adversely affect populations of biological control agents and thus lead to increases in host plant populations. In the Awatere Valley of New Zealand, the only reported resurgence of St. Johnswort following successful control by *Chrysolina hyperici* occurred following a fire (Syrett 1989, as cited by Briese 1996). Intensity, frequency, and season, plus scale of individual fires are important for both the host weed and the biological control agent. The final outcome depends on how a particular weed or biological control agent responds to these components of the fire regime (Briese 1996).

One study in Australia examines effects of fuel reduction burns on biological control of St. Johnswort. Details of the study site and St. Johnswort response to burning are given above (see Discussion and Qualification of Plant Response). St. Johnswort on this site supported populations of *C. quadrigemina*, a chrysomelid beetle introduced to Australia for biological control of St. Johnswort (Briese 1996).

The immediate effect of the spring 1982 prescribed fire on *C. quadrigemina* was the virtual disappearance of the insect during 1983, with only a few eggs observed during that period. In autumn 1984, massive egg-laying and subsequent larval defoliation of rosettes were observed, most likely due to re-invasion by adult beetles from neighboring unburned plots. Large-scale destruction of St. Johnswort by beetles was repeated in 1985. This resulted in a substantial decrease in St. Johnswort ground cover, crown density, and production of flowering stems, and an increase in dominance of the grass and forb component in both open and timbered areas.
Briese (1996) suggests that a short term post-fire increase in soil nutrients ("the fertilizer effect") may have contributed indirectly to the impact of the biological control agent on St. Johnswort. Nutrient analyses of St. Johnswort plants collected the season following the fire showed a 25% increase (p < 0.05) in nitrogen levels in plants from burned areas relative to plant from adjacent unburned areas. Burned plants were also larger than average, which may have been particularly attractive to beetles and/or favor survival and development of hatching larvae, thus triggering a population build-up of the biological control agents (Briese 1996).

Following the fall 1986 fire, St. Johnswort populations and associated grasses and forbs responded much differently, with massive seed germination of several species, including St. Johnswort. Most St. Johnswort seedlings did not survive and persist, and associated grasses and forbs dominated the post-fire environment. Large populations of *C. quadrigemina* were not observed on the site until 1990 (Briese 1996).

Frequency of fire is a key factor for effects on biocontrol agents, and the cycle of their population build-up needs to be understood and respected to enable them to have the desired impact on the target weed. The frequency of prescribed burning may need adjusting according to the reproductive capacity and life history of the control agent. Reproductive strategy and mobility of control agents are important factors to consider. If one considers both fire management and biological control as long-term protection strategies, it is necessary to look at the interaction of weed biology, control agent biology, fire regime, and whether they can be manipulated to help realize the potential of the control agent, or at least to not hinder it (Briese 1996).

Post-fire colonization/spread potential: Several sources suggest that St. Johnswort has high potential for post-fire colonization in some areas. Several references indicate that St. Johnswort often occurs in previously burned areas, especially forests (e.g. Campbell and Delfosse 1984, Cholewa 1977, Clark 1953, Lavender 1958, Monleon *et al.* 1999, Streitfield and Frenkel 1997, Tisdale *et al.* 1959). The source (seeds vs. sprouts) of St. Johnswort establishment in these references is unclear. St. Johnswort may occur in the initial post-fire community by establishing from root crowns, roots, or seeds in the soil seed bank, or it may occur in burned areas as a secondary colonizer, establishing from off-site sources some years after fire.

Where St. Johnswort occurs as mature plants, it is likely to occur in the initial post-fire community (e.g. Briese 1996, Clark and Wilson 2001, Leege and Godbolt 1985, Tveten and Fonda 1999), although it may or may not persist (See Successional Status). Several studies in which the soil seed bank was sampled and germinated indicate the presence of St. Johnswort seed in areas where mature plants are rare, do not occur, or occur only at some distance from the sampled sites (Halpern *et al.* 1999, Laughlin 2003, Leckie *et al.* 2000, Livingston and Allessio 1968, Wilson *et al.* 1991). Harris and Gill (1997) suggest that when a pine plantation (or forest) reaches stand closure, St. Johnswort may disappear from aboveground vegetation, but remain (as seed) in the soil seed bank (see Seed banking for details).
When fire occurs in such sites, St. Johnswort may establish from seed as part of the initial post-fire community. Rapid post-fire infestation of St. Johnswort following prescribed burning on an Oregon range site suggests that an extensive seed bank was present on the site before the fire (Walker 2000) (see Discussion and Qualification of Fire Effect). Evidence of St. Johnswort establishment following disturbances from timber harvest or thinning (e.g. Del Tredici 1977, Thysell and Carey 2001) further support the possibility of St. Johnswort establishment after a fire that reduces or removes canopy cover and/or disturbs the soil.

Although there is no clear evidence in the literature, managers should be aware that St. Johnswort may also establish as an initial off-site colonizer in areas where St. Johnswort populations are in the vicinity of a burn site and seed may be transported by wild or domestic animals, or by vehicles.

St. Johnswort may also establish as a secondary off-site colonizer that establishes after post-fire year 1 (Stickney 1986). Evidence to support this is provided from several forested habitats in Idaho (Habeck 1985, Stickney 1986, Stickney and Campbell 2000), where St. Johnswort established several years after fire and did not persist as canopies established. St. Johnswort may, however, persist in closed-canopy Douglas-fir forests (e.g. Ruggiero et al. 1991), as well as in more open-canopied forests such as some eucalyptus (Briese 1997, Buckley et al. 2003) and ponderosa pine forests (Tisdale et al. 178) (see Successional Status).

Several authors suggest that spread of St. Johnswort is also encouraged by fire in some areas and under some conditions (Briese 1996, Rice and Randall 2004, Sampson and Parker 1930, Walker 2000). According to Campbell and Delfosse (1984) when stands of St. Johnswort are burned, density of the stand increases. Thus, control of rangeland fires may be important to the control of St. Johnswort (Crompton et al. 1988).


Preventing invasive plants from establishing in weed-free burned areas is the most effective and least costly control method. This can be accomplished through careful monitoring, early detection and eradication, and limiting invasive plant seed dispersal into burned areas by (Goodwin et al. 2002, USDA Forest Service 2001):

- re-establishing vegetation on bare ground as soon as possible
- using only certified weed-free seed mixes when revegetation is necessary
- cleaning equipment and vehicles prior to entering burned areas
• regulating or preventing human and livestock entry into burned areas until desirable site vegetation has recovered sufficiently to resist invasion by undesirable vegetation

• detecting weeds early and eradicating before vegetative spread and/or seed dispersal

• eradicating small patches and containing or controlling large infestations within or adjacent to the burned area

In general, early detection is critical for preventing establishment of large populations of invasive plants. Monitoring in spring, summer, and fall is imperative. Managers should eradicate established St. Johnswort plants and small patches adjacent to burned areas to prevent or limit dispersal into the site (Goodwin et al. 2002, USDA Forest Service 2001).

The need for revegetation after fire can be based on the degree of desirable vegetation displaced by invasive plants prior to burning and on post-fire survival of desirable vegetation. Revegetation necessity can also be related to invasive plant survival as viable seeds, root crowns, or root fragments capable of reproduction. In general, post-fire revegetation should be considered when desirable vegetation cover is less than about 30% (Goodwin et al. 2002).

When pre-fire cover of St. Johnswort is absent to low, and pre-fire cover of desirable vegetation is high, revegetation is probably not necessary after low- and medium-severity burns. After a high-severity burn on a site in this condition, revegetation may be necessary (depending on post-fire survival of desirable species), and intensive monitoring for invasive plant establishment is necessary to detect and eradicate newly established invasives before they spread (Goodwin et al. 2002).

When pre-fire cover of St. Johnswort is moderate (20%-79%) to high (80%-100%), revegetation may be necessary after fire of any severity if cover of desired vegetation is less than about 30%. Intensive weed management is also recommended, especially after fires of moderate to high severity (Goodwin et al. 2002).

Fall dormant broadcast seeding into ash will cover and retain seeds. If there is insufficient ash, seedbed preparation may be necessary. A seed mix should contain quick-establishing grasses and forbs (exclude forbs if broadleaf herbicides are anticipated) that can effectively occupy available niches. Managers can enhance the success of revegetation (natural or artificial) by excluding livestock until vegetation is well established (at least 2 growing seasons) (Goodwin et al. 2002). See Integrated Noxious Weed Management after Wildfires for more information.
When planning a prescribed burn, managers should preinventory the project area and evaluate cover and phenology of any St. Johnswort and other invasive plants present on or adjacent to the site, and avoid ignition and burning in areas at high risk for St. Johnswort establishment or spread due to fire effects. Managers should also avoid creating soil conditions that promote weed germination and establishment. Weed status and risks must be discussed in burn rehabilitation plans. Also, wildfire managers might consider including weed prevention education and providing weed identification aids during fire training; avoiding known weed infestations when locating fire lines; monitoring camps, staging areas, helibases, etc., to be sure they are kept weed free; taking care that equipment is weed free; incorporating weed prevention into fire rehabilitation plans; and acquiring restoration funding. Additional guidelines and specific recommendations and requirements are available (USDA Forest Service 2001).

LITERATURE CITED


FIRE ECOLOGY OR ADAPTATIONS

Fire adaptations: Baltic rush is fire tolerant when dormant (USDA Natural Resources Conservation Service 2006) and top-killed by fire during the growing season (Clark 1991, Wehking 2002, Young 1986). It establishes after fire through seed and/or lateral spread by rhizomes. Baltic rush fruits lack an awn; therefore, initial seed dispersal onto burned sites is primarily effected by wind.

Fire regimes: Many diverse communities provide Baltic rush habitat. Baltic rush experiences extreme ranges in fire frequency. In the northern and southern cordgrass prairies where Baltic rush is found fire may occur every 1 to 3 years (Paysen et al. 2000). Conversely, Baltic rush also occurs in sugar maple communities where the fire return interval is 1,000 years or more (Wade et al. 2000).

Baltic rush is a dominant species in Afton Canyon located in the lower Mojave River drainage of the western Mojave Desert, California (Lovich et al. 1994). Historically, the fire interval in this mesquite-saltbush (Prosopis spp.-Atriplex confertifolia) dominated environment was < 35 to < 100 years (McPherson 1995, Paysen et al. 2000). However, the infestation of the drought-deciduous saltcedar (Tamarix ramosissima) has altered the historic fire regime, promoting a fire interval of about 10 to 20 years (Lovich et al. 1994). Baltic rush grows on rough fescue-Parry’s oatgrass dominated grasslands in Banff and Jasper national parks, Alberta. Prior to national park designation, fires were “frequent” and “extensive” especially in the main valleys. For the last 80 years, however, a fire control policy has been put in place, and while fires are still frequent, they are small and mostly in grassland, scrub, or forest litter areas close to human activity (Stringer 1973).

The following table provides fire return intervals for plant communities and ecosystems where Baltic rush is important. For further information, see the FEIS review of the dominant species listed below.

<table>
<thead>
<tr>
<th>Community or Ecosystem</th>
<th>Dominant Species</th>
<th>Fire Return Interval Range (years)</th>
</tr>
</thead>
<tbody>
<tr>
<td>silver fir-Douglas-fir</td>
<td>Abies amabilis-Pseudotsuga menziesii var. menziesii</td>
<td>&gt; 200</td>
</tr>
<tr>
<td>grand fir</td>
<td>Abies grandis</td>
<td>35-200 [1]</td>
</tr>
<tr>
<td>sugar maple</td>
<td>Acer saccharum</td>
<td>&gt; 1,000 [2]</td>
</tr>
<tr>
<td>bluestem prairie</td>
<td>Andropogon gerardii var. gerardii-Schizachyrium scoparium</td>
<td>&lt; 10 [3,4]</td>
</tr>
<tr>
<td>Nebraska sandhills prairie</td>
<td>Andropogon gerardii var. paucipilus-Schizachyrium</td>
<td>&lt; 10</td>
</tr>
<tr>
<td>Habitat/Community</td>
<td>Dominant Species</td>
<td>Cover (%)</td>
</tr>
<tr>
<td>-------------------</td>
<td>----------------------------------------------------------------------------------</td>
<td>-----------</td>
</tr>
<tr>
<td>bluestem-Sacahuista prairie</td>
<td>Andropogon littoralis-Spartina spartinae</td>
<td>&lt; 10</td>
</tr>
<tr>
<td>sagebrush steppe</td>
<td>Artemisia tridentata/Pseudoroegneria spicata</td>
<td>20-70</td>
</tr>
<tr>
<td>basin big sagebrush</td>
<td>Artemisia tridentata var. tridentata</td>
<td>12-43</td>
</tr>
<tr>
<td>mountain big sagebrush</td>
<td>Artemisia tridentata var. vaseyana</td>
<td>15-40</td>
</tr>
<tr>
<td>Wyoming big sagebrush</td>
<td>Artemisia tridentata var. wyomingensis</td>
<td>10-70 (µ = 40)</td>
</tr>
<tr>
<td>saltbush-greasewood</td>
<td>Atriplex confertifolia-Sarcobatus vermiculatus</td>
<td>&lt; 35 to 100</td>
</tr>
<tr>
<td>desert grasslands</td>
<td>Bouteloua eriopoda and/or Pleuraphis mutica</td>
<td>&lt; 35 to 100</td>
</tr>
<tr>
<td>plains grasslands</td>
<td>Bouteloua spp.</td>
<td>&lt; 35</td>
</tr>
<tr>
<td>blue grama-buffalo grass</td>
<td>Bouteloua gracilis-Buchloe dactyloides</td>
<td>&lt; 35 [4,12]</td>
</tr>
<tr>
<td>grama-galleta steppe</td>
<td>Bouteloua gracilis-Pleuraphis jamesii</td>
<td>&lt; 35 to 100</td>
</tr>
<tr>
<td>paloverde-cactus shrub</td>
<td>Parkinsonia spp./Opuntia spp.</td>
<td>&lt; 35 to 100</td>
</tr>
<tr>
<td>blackbrush</td>
<td>Coleogyne ramosissima</td>
<td>&lt; 35 to 100</td>
</tr>
<tr>
<td>Arizona cypress</td>
<td>Cupressus arizonica</td>
<td>&lt; 35 to 200</td>
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<tr>
<td>northern cordgrass prairie</td>
<td>Distichlis spicata-Spartina spp.</td>
<td>1-3 [4]</td>
</tr>
<tr>
<td>California steppe</td>
<td>Festuca-Danthonia spp.</td>
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<td>western juniper</td>
<td>Juniperus occidentalis</td>
<td>20-70</td>
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<tr>
<td>Rocky Mountain juniper</td>
<td>Juniperus scopulorum</td>
<td>&lt; 35</td>
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<tr>
<td>tamarack</td>
<td>Larix laricina</td>
<td>35-200 [4]</td>
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<tr>
<td>western larch</td>
<td>Larix occidentalis</td>
<td>25-350 [14,15,16]</td>
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<tr>
<td>creosotebush</td>
<td>Larrea tridentata</td>
<td>&lt; 35 to 100 [4]</td>
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<td>yellow-poplar</td>
<td>Liriodendron tulipifera</td>
<td>&lt; 35 [2]</td>
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<tr>
<td>Great Lakes spruce-fir</td>
<td>Picea-Abies spp.</td>
<td>35 to &gt; 200</td>
</tr>
<tr>
<td>northeastern spruce-fir</td>
<td>Picea-Abies spp.</td>
<td>35-200 [17]</td>
</tr>
<tr>
<td>Engelmann spruce-subalpine fir</td>
<td>Picea engelmannii-Abies lasiocarpa</td>
<td>35 to &gt; 200 [1]</td>
</tr>
<tr>
<td>black spruce</td>
<td>Picea mariana</td>
<td>35-200</td>
</tr>
<tr>
<td>conifer bog*</td>
<td>Picea mariana-Larix laricina</td>
<td>35-200 [17]</td>
</tr>
<tr>
<td>pinyon-juniper</td>
<td>Pinus-Juniperus spp.</td>
<td>&lt; 35 [4]</td>
</tr>
<tr>
<td>Rocky Mountain bristlecone pine</td>
<td>P. aristata</td>
<td>9-55 [18,19]</td>
</tr>
<tr>
<td>whitebark pine*</td>
<td>Pinus albicaulis</td>
<td>50-200 [20,21]</td>
</tr>
<tr>
<td>Rocky Mountain lodgepole pine*</td>
<td>Pinus contorta var. latifolia</td>
<td>25-340 [22,15,23]</td>
</tr>
<tr>
<td>Species/Morphotype</td>
<td>Scientific Name</td>
<td>Densities</td>
</tr>
<tr>
<td>--------------------</td>
<td>----------------</td>
<td>-----------</td>
</tr>
<tr>
<td>Sierra lodgepole pine*</td>
<td><em>Pinus contorta var. murrayana</em></td>
<td>35-200 [1]</td>
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<td>Colorado pinyon</td>
<td><em>Pinus edulis</em></td>
<td>10-400+ [24,25,26,4]</td>
</tr>
<tr>
<td>Sand pine</td>
<td><em>Pinus elliottii var. elliottii</em></td>
<td>25-45 [2]</td>
</tr>
<tr>
<td>Jeffrey pine</td>
<td><em>Pinus jeffreyi</em></td>
<td>5-30</td>
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<tr>
<td>Western white pine*</td>
<td><em>Pinus monticola</em></td>
<td>50-200</td>
</tr>
<tr>
<td>Pacific ponderosa pine*</td>
<td><em>Pinus ponderosa var. ponderosa</em></td>
<td>1-47 [1]</td>
</tr>
<tr>
<td>Interior ponderosa pine*</td>
<td><em>Pinus ponderosa var. scopulorum</em></td>
<td>2-30 [1,27,28]</td>
</tr>
<tr>
<td>Arizona pine</td>
<td><em>Pinus ponderosa var. arizonica</em></td>
<td>2-15 [27,29,30]</td>
</tr>
<tr>
<td>Galleta-threeawn shrubsteppe</td>
<td>Pleuraphis jamesii-Aristida purpurea</td>
<td>&lt; 35 to &lt; 100</td>
</tr>
<tr>
<td>Eastern cottonwood</td>
<td><em>Populus deltoides</em></td>
<td>&lt; 35 to 200 [4]</td>
</tr>
<tr>
<td>Aspen-birch</td>
<td><em>Populus tremuloides-Betula papyrifera</em></td>
<td>35-200 [17,2]</td>
</tr>
<tr>
<td>Quaking aspen (west of the Great Plains)</td>
<td><em>Populus tremuloides</em></td>
<td>7-120 [1,31,32]</td>
</tr>
<tr>
<td>Mesquite</td>
<td><em>Prosopis glandulosa</em></td>
<td>&lt; 35 to &lt; 100 [33,4]</td>
</tr>
<tr>
<td>Mountain grasslands</td>
<td><em>Pseudoroegneria spicata</em></td>
<td>3-40 (µ = 10) [34,1]</td>
</tr>
<tr>
<td>Rocky Mountain Douglas-fir*</td>
<td><em>Pseudotsuga menziesii var. glauca</em></td>
<td>25-100</td>
</tr>
<tr>
<td>Coastal Douglas-fir*</td>
<td><em>Pseudotsuga menziesii var. menziesii</em></td>
<td>40-240 [1,35,36]</td>
</tr>
<tr>
<td>California mixed evergreen</td>
<td><em>Pseudotsuga menziesii var. menziesii-Lithocarpus densiflorus-Arbutus menziesii</em></td>
<td>&lt; 35</td>
</tr>
<tr>
<td>California oakwoods</td>
<td><em>Quercus spp.</em></td>
<td>&lt; 35 [1]</td>
</tr>
<tr>
<td>Oak-hickory</td>
<td><em>Quercus-Carya spp.</em></td>
<td>&lt; 35 [2]</td>
</tr>
<tr>
<td>Oak-juniper woodland (Southwest)</td>
<td><em>Quercus-Juniperus spp.</em></td>
<td>&lt; 35 to &lt; 200 [4]</td>
</tr>
<tr>
<td>Canyon live oak</td>
<td><em>Quercus chrysolepis</em></td>
<td>&lt; 35 to 200 [1]</td>
</tr>
<tr>
<td>Northern pin oak</td>
<td><em>Quercus ellipsoidealis</em></td>
<td>&lt; 35 [2]</td>
</tr>
<tr>
<td>Oregon white oak</td>
<td><em>Quercus garryana</em></td>
<td>&lt; 35 [1]</td>
</tr>
<tr>
<td>California black oak</td>
<td><em>Quercus kelloggii</em></td>
<td>5-30 [4]</td>
</tr>
<tr>
<td>Bur oak</td>
<td><em>Quercus macrocarpa</em></td>
<td>&lt; 10 [2]</td>
</tr>
<tr>
<td>Oak savanna</td>
<td><em>Quercus macrocarpa/Andropogon gerardii-Schizachyrium scoparium</em></td>
<td>2-14 [4,2]</td>
</tr>
<tr>
<td>Black oak</td>
<td><em>Quercus velutina</em></td>
<td>&lt; 35</td>
</tr>
<tr>
<td>Live oak</td>
<td><em>Quercus virginiana</em></td>
<td>10 to&lt; 100 [2]</td>
</tr>
<tr>
<td>Little bluestem-grama prairie</td>
<td><em>Schizachyrium scoparium-</em></td>
<td>&lt; 35</td>
</tr>
<tr>
<td>Habitat Type</td>
<td>Species</td>
<td>Fire Return Interval</td>
</tr>
<tr>
<td>----------------------------------</td>
<td>------------------------</td>
<td>----------------------</td>
</tr>
<tr>
<td>tule marshes</td>
<td><em>Bouteloua</em> spp.</td>
<td>&lt; 35</td>
</tr>
<tr>
<td>southern cordgrass prairie</td>
<td><em>Scirpus</em> and/or <em>Typha</em> spp.</td>
<td>1-3 [4]</td>
</tr>
<tr>
<td>mountain hemlock*</td>
<td><em>Spartina alterniflora</em></td>
<td>35 to &gt; 200 [1]</td>
</tr>
</tbody>
</table>

*fire return interval varies widely; trends in variation are noted in the species review


POST-FIRE REGENERATION STRATEGY (Stickney 1989):

Rhizomatous herb, rhizome in soil; initial off-site colonizer (off-site, initial community); secondary colonizer (on-site or off-site seed sources).

IMMEDIATE FIRE EFFECT ON PLANT

Baltic rush is top-killed by fire, with rhizomes and seeds protected by insulating soil (Clark 1991, Wehking 2002, Young 1986).

DISCUSSION AND QUALIFICATION OF FIRE EFFECT

Baltic rush seed banks survive fire, but the number of seeds is less in burned areas compared to unburned areas (Clark 1991, Wehking 1997). Within the tufted hairgrass-sedge community-type seed bank of Yellowstone National Park, Baltic rush seeds occurred in the seed bank in densities of 1667 seeds/m² on unburned sites and 493 seeds/m² on sites burned during the Yellowstone fire of 1988 (Clark 1991).
In October, 1996, 2 water tables (dry and wet) where Baltic rush was present were identified in the Humboldt-Toiyabe National Forest, Nevada. The dominant species in the area is basin big sagebrush. One half of each site was subjected to a controlled burn while the other half was left unburned and used as a control. In October, 1997 and 1998, seed bank counts (mean number of seeds/m² ± s x) were conducted on the dry and wet sites in undershrub and interspace areas. Fewer Baltic rush seeds were found in the seed bank of burned sites. Results of the study (--- indicates no data) are presented below (Wehking 2002):

<table>
<thead>
<tr>
<th>Site</th>
<th>Burned</th>
<th>Not burned (control)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Undershrub</td>
<td>Interspace</td>
</tr>
<tr>
<td>Dry</td>
<td>18±18</td>
<td>12±12</td>
</tr>
<tr>
<td>Wet</td>
<td>---</td>
<td>6±6</td>
</tr>
</tbody>
</table>

Further data were collected on the depth of Baltic rush seeds stored in the seed bank on the burned and unburned sites. At all sites and depths, except for the dry site at 5-15 cm, more seeds were found at varying depths on unburned sites. Seed bank counts (mean number of seeds/m² ± s x) were collected in October, 1997, and are presented below (Wehking 2002):

<table>
<thead>
<tr>
<th>Seed depth</th>
<th>Dry site</th>
<th>Wet site</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Burned</td>
<td>Unburned</td>
</tr>
<tr>
<td></td>
<td>Under shrub</td>
<td>Interspace</td>
</tr>
<tr>
<td>Litter - 1 cm</td>
<td>---</td>
<td>---</td>
</tr>
<tr>
<td>1 cm - 5 cm</td>
<td>18±18</td>
<td>---</td>
</tr>
<tr>
<td>5 cm - 15 cm</td>
<td>---</td>
<td>21±21</td>
</tr>
</tbody>
</table>

**PLANT RESPONSE TO FIRE**

Baltic rush has a high tolerance to fire (USDA Natural Resources Conservation Service 2006). Baltic rush recovers from fire by rhizomatous spread and/or establishing by seed. Fire has a positive effect on Baltic rush frequency and coverage (Bailey and Anderson 1979, Hargis and McCarthy 1986, Quinlan et al. 2003, Walhof 1997, Wright and Chambers 2002, Wright 2001, Young 1986). Some research suggests that post-fire Baltic rush plants have a faster growth rate and attain a greater mature height than plants growing on unburned sites (Young 1986).
DISCUSSION AND QUALIFICATION OF PLANT RESPONSE

Prescribed fire was used to control an invasive mixed poplar-willow forest on Baltic rush-Kentucky bluegrass grasslands in central Alberta along the shore of Beaverhill Lake. Burning occurred on 18 May, 1971, on four 0.25 ha plots. Five years after the controlled burn, Baltic rush canopy cover was significantly (p < 0.05) higher on burned plots (60%) than unburned plots (35%) (Bailey and Anderson 1979).

In the fall of 1981, a 26-ha plot of Great Basin meadow at Inyo National Forest, California, underwent a prescribed burn to restore brush-encroached meadows. Baltic rush’s average coverage significantly (p < 0.01) increased on burned dry meadow plots and increased as well on burned wet meadow plots but not significantly (Hargis and McCarthy 1986):

<table>
<thead>
<tr>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td>Dry meadow average coverage (%)</td>
<td>2.0</td>
<td>3.0</td>
<td>4.0</td>
<td>4.0</td>
</tr>
<tr>
<td>Wet meadow average coverage (%)</td>
<td>12.0</td>
<td>13.0</td>
<td>12.0</td>
<td>14.0</td>
</tr>
</tbody>
</table>

To control basin big sagebrush (*Artemisia tridentata* ssp. *tridentata*) in the Toiyabe Mountain Range of central Nevada, 2 plots measuring 740 m² and 900 m² on a wet site were burned and plots of equal size were left unburned and used as a control. The prescribed burn occurred from 19 to 21 October, 1996, top-killing all shrubs and herbaceous vegetation. After 3 years, Baltic rush mean biomass (grams/m²) was greater on the burned undershrub and interspace sites than on the unburned undershrub and interspace sites (Wright and Chambers 2002, Wright 2001):

<table>
<thead>
<tr>
<th>Site</th>
<th>Burned</th>
<th>Unburned</th>
</tr>
</thead>
<tbody>
<tr>
<td>Undershrub</td>
<td>4.2</td>
<td>1.5</td>
</tr>
<tr>
<td>Interspace</td>
<td>5.2</td>
<td>2.0</td>
</tr>
</tbody>
</table>

Martha Lake Field is located within the Malheur National Wildlife Refuge in southeastern Oregon. On 20 October, 1981, a prescribed fire was set in a Baltic rush dominated wetland. In 1982, April through August, mean shoot height (cm ± s) and growth rate (cm/day) were recorded in the burn and control (unburned) sites. Baltic rush plants in the burned area were found to have a greater height and growth rate than those in the control site (Young 1986):
<table>
<thead>
<tr>
<th>Date</th>
<th>Period</th>
<th>Burned</th>
<th>Control</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>(days)</td>
<td>Height</td>
<td>Growth rate</td>
</tr>
<tr>
<td></td>
<td></td>
<td>(cm ± s)</td>
<td>(cm/day)</td>
</tr>
<tr>
<td>22 April 1982</td>
<td>---</td>
<td>7±5</td>
<td>---</td>
</tr>
<tr>
<td>29 May 1982</td>
<td>37</td>
<td>17±6</td>
<td>0.3</td>
</tr>
<tr>
<td>3 July 1982</td>
<td>35</td>
<td>68±23</td>
<td>1.5</td>
</tr>
<tr>
<td>29 July 1982</td>
<td>26</td>
<td>80±26</td>
<td>0.5</td>
</tr>
<tr>
<td>19 August 1982</td>
<td>21</td>
<td>82±24</td>
<td>0.1</td>
</tr>
</tbody>
</table>

In the spring of 1992, 1993, and 1995, meadows in the Slave River Lowlands, Northwest Territories, were subjected to prescribed burn treatments. Baltic rush was strongly associated with meadows burned 3 times (Quinlan et al. 2003). On 26 October, 1987, a controlled burn was conducted on a northern stretch of the Wise River in southwestern Montana. The burn site sits at an elevation of 2000 m, with an aspect and slope of 330° and 6%, respectively. The plant community is dominated by mountain big sagebrush with Wyoming big sagebrush a minor shrub component. In June, 1995, the burned site was sampled as was an adjacent unburned control site. On the burned site Baltic rush had a canopy cover and frequency of 5.5% and 37.0%, compared to 0.3% and 20% on the unburned sites (Wade et al. 2000).

**FIRE MANAGEMENT CONSIDERATIONS**

Since Baltic rush re-establishes quickly after fire (Bailey and Anderson 1979, Hargis and McCarthy 1986, Quinlan et al. 2003, Walhof 1997, Wright and Chambers 2002, Wright 2001, Young 1986), prescribed fire can be used as a management tool to promote its growth. Some research does suggest, however, that while fire does not completely destroy Baltic rush seed banks, it does have a detrimental effect on them (Clark 1991, Wehking 2002). If fire is chosen as a management tool for Baltic rush, managers should be cognizant of potential negative effects on associated or surrounding vegetation. For instance, Baltic rush is a dominant species in Afton Canyon located in the lower Mojave River drainage of the western Mojave Desert, California. In the past several decades the area has been infested by the drought-deciduous saltcedar which is highly fire tolerant and may expand after disturbances such as fire and severely reduce native plant coverage (Lovich et al. 1994).

**LITERATURE CITED**


Koeleria macrantha
Prairie Junegrass

FIRE ECOLOGY OR ADAPTATIONS

Prairie Junegrass is reported as showing little or no damage (Young 1983) to moderate damage (Wright et al. 1979) from fire. Perennial grasses possess growing points insulated near or below the soil surface (Young 1983). The small stature of prairie Junegrass and coarse textured foliage aid in protection of these meristematic tissue areas (Young 1983). Possessing coarsely textured foliage and a small clump size also limits the potential for fire damage (Young 1983). Coarse grasses like prairie Junegrass burn quickly, transferring little heat below the soil surface (Volland and Dell 1981). As a member of eastern Oregon grasslands, prairie Junegrass is considered a superior fire-resistant perennial bunch grass (Wright et al. 1979).

Fire survival strategy for prairie Junegrass is based upon seed germination and residual plant survival (Bradley et al. 1992). The extent of damage or benefit imposed by fire is highly variable. Response can vary according to fire severity, physiological state of plant, soil moisture, and season of burn (Bradley et al. 1992, Smith and Busby 1981).

Fire regimes for plant communities in which prairie Junegrass occurs are summarized below.

<table>
<thead>
<tr>
<th>Community or Ecosystem</th>
<th>Dominant Species</th>
<th>Fire Return Interval Range in Years (mean)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Pacific ponderosa pine*</td>
<td>Pinus ponderosa var. ponderosa</td>
<td>1-47 [1]</td>
</tr>
<tr>
<td>Plant Type</td>
<td>Scientific Name</td>
<td>Size</td>
</tr>
<tr>
<td>----------------------------------</td>
<td>------------------------------------------------------</td>
<td>--------</td>
</tr>
<tr>
<td>interior ponderosa pine*</td>
<td><em>P. ponderosa var. scopulorum</em></td>
<td>2-200 [1]</td>
</tr>
<tr>
<td>Colorado pinyon</td>
<td><em>P. edulis</em></td>
<td>10-49 [1]</td>
</tr>
<tr>
<td>Mexican pinyon</td>
<td><em>P. cembroides</em></td>
<td>20-70 [1]</td>
</tr>
<tr>
<td>Rocky Mountain Douglas-fir*</td>
<td><em>Pseudotsuga menziesii var. glauca</em></td>
<td>25-100 [1]</td>
</tr>
<tr>
<td>coastal Douglas-fir*</td>
<td><em>Pseudotsuga menziesii var. menziesii</em></td>
<td>40-240 [1,4,5]</td>
</tr>
<tr>
<td>quaking aspen (west of the Great Plains)</td>
<td><em>Populus tremuloides</em></td>
<td>7-100 [6,7]</td>
</tr>
<tr>
<td>oak-hickory</td>
<td><em>Quercus-Carya spp.</em></td>
<td>50-100 [8]</td>
</tr>
<tr>
<td>Texas savanna</td>
<td><em>Prosopis glandulosa var. glandulosa</em></td>
<td>&lt; 10 [1]</td>
</tr>
<tr>
<td>California montane chaparral</td>
<td><em>Ceanothus and/or Arctostaphylos spp.</em></td>
<td>50-100 [1]</td>
</tr>
<tr>
<td>basin big sagebrush</td>
<td><em>Artemisia tridentata var. tridentata</em></td>
<td>12-43 [9]</td>
</tr>
<tr>
<td>mountain big sagebrush</td>
<td><em>Artemisia tridentata var. vaseyana</em></td>
<td>5-15 [10]</td>
</tr>
<tr>
<td>Wyoming big sagebrush</td>
<td><em>Artemisia tridentata var. wyomingensis</em></td>
<td>10-70 (40) [11,10]</td>
</tr>
<tr>
<td>mountain grasslands</td>
<td><em>Pseudoroegneria spicata</em></td>
<td>3-40 (10) [12]</td>
</tr>
<tr>
<td>plains grasslands</td>
<td><em>Bouteloua gracilis and/or Buchloe dactyloides</em></td>
<td>20-40 [1]</td>
</tr>
<tr>
<td>prairie</td>
<td><em>Andropogon gerardii var. gerardii</em></td>
<td>1-6 [13]</td>
</tr>
</tbody>
</table>


POST-FIRE REGENERATION STRATEGY

Tussock graminoid; caudex, growing points in soil; secondary colonizer - on-site seed.

IMMEDIATE FIRE EFFECT ON PLANT

Prairie Junegrass is usually top-killed or killed by fire. Fast-moving, low-intensity fires will consume above ground vegetation without damaging the plant's crown (Smith and Busby 1981). In general, late-spring burns are more damaging to prairie Junegrass than early-spring, late-summer, fall, or winter burns (Towne and Owensby 1984).

DISCUSSION AND QUALIFICATION OF FIRE EFFECT

The root crown of prairie Junegrass may sustain immediate damage depending upon amount of heat transferred through the soil (Dion and Davis 2000). Meristematic tissue of most grasses is found at the ground surface in the root crown (Smith and Busby 1981). A study (Dion and Davis 2000) in the southwest United States evaluated patterns of plant growth in relation to soil heating from wildfire in chaparral systems. Prairie Junegrass was found to have a strong negative correlation between heat load and sprouts.

PLANT RESPONSE TO FIRE

Positive post-fire vegetational responses are common for prairie Junegrass. Increased seedhead presence and height-of-inflorescence have been documented (Ehrenreich and Aikman 1963). Annual burning of a native grassland in the aspen parkland of central Alberta caused a 40% increase in seedhead presence compared to unburned areas (Anderson and Bailey 1980).

Prairie Junegrass's response to fire is related to season of burn, fire intensity, and post-fire water availability (Habeck 1980). Several studies evaluating the effect of fire on following season vigor report positive correlations (Adams 1980, Aikman 1955, Aldous 1934, Antos et al. 1983, Arnold et al. 1964). Time required to acquire the approximate preburn frequency or coverage, is rapid, averaging 2 to 5 years (Volland and Dell 1981).
DISCUSSION AND QUALIFICATION OF PLANT RESPONSE

Prairie Junegrass shows positive responses to fire. Fall and early-spring burns produce better responses than summer burns (Aldous 1934, Kartesz 1994). A summation on the effect of burn seasonality in eastern Oregon is given below (Wright et al. 1979):

<table>
<thead>
<tr>
<th>Burn Season</th>
<th>Change in Basal Area</th>
<th>Mortality</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mid-May</td>
<td>-32%</td>
<td>20%</td>
</tr>
<tr>
<td>Mid-June (Post-seed-out)</td>
<td>-18%</td>
<td>0%</td>
</tr>
<tr>
<td>Mid-October</td>
<td>approx. -18%</td>
<td>0%</td>
</tr>
</tbody>
</table>

Observations of prairie Junegrass populations two years after a spring burn were conducted in Galena Gulch within the Deer Lodge National Forest in western Montana. Burned sites experienced a 40% increase in prairie Junegrass occurrence (Antos et al. 1983). Percent coverage of prairie Junegrass was evaluated in burned and unburned sites after a June fire in western Montana. Coverage was greater in the burned areas two years after the burn populations decreased, but maintained levels above the control stands (Antos et al. 1983).

<table>
<thead>
<tr>
<th>Autumn 1977</th>
<th>Spring 1978</th>
<th>Summer 1978</th>
</tr>
</thead>
<tbody>
<tr>
<td>Unburned</td>
<td>Burned</td>
<td>Unburned</td>
</tr>
<tr>
<td>1.6</td>
<td>1.5</td>
<td>1.4</td>
</tr>
</tbody>
</table>

An experiment was initiated in 1926 on a bluestem pasture in eastern Kansas to record effects of fire upon several ecological parameters. Fires were set the same times each year beginning in 1926: early-spring (March 20), medium-spring (April 10), late-spring (May 5) and late-fall (December 1). The response of prairie Junegrass to the burns from late June, 1928, to early July, 1933, are summarized below (Aldous 1934):

Number of plants/year within 2 rods square - observed late June and early July

<table>
<thead>
<tr>
<th>Year Observed</th>
<th>Burn date</th>
<th>1928</th>
<th>1929</th>
<th>1930</th>
<th>1931</th>
<th>1932</th>
<th>1933</th>
<th>Mean</th>
</tr>
</thead>
<tbody>
<tr>
<td>Fall</td>
<td>1,210</td>
<td>1,864</td>
<td>2,114</td>
<td>2,283</td>
<td>1,783</td>
<td>1,080</td>
<td>1,722</td>
<td></td>
</tr>
<tr>
<td>Early-spring</td>
<td>1,296</td>
<td>1,402</td>
<td>1,465</td>
<td>1,340</td>
<td>1,002</td>
<td>482</td>
<td>1,165</td>
<td></td>
</tr>
<tr>
<td>Medium-spring</td>
<td>574</td>
<td>555</td>
<td>756</td>
<td>1,031</td>
<td>767</td>
<td>523</td>
<td>701</td>
<td></td>
</tr>
<tr>
<td>Late-spring</td>
<td>1,058</td>
<td>1,052</td>
<td>1,571</td>
<td>1,818</td>
<td>1,068</td>
<td>447</td>
<td>1,169</td>
<td></td>
</tr>
<tr>
<td>Check plot</td>
<td>619</td>
<td>1,084</td>
<td>1,052</td>
<td>992</td>
<td>627</td>
<td>181</td>
<td>759</td>
<td></td>
</tr>
</tbody>
</table>

Prairie Junegrass density on all burn treatments except medium-spring, exceeded density on the 1928 check plot until 1933, when densities were low on early-spring and late-spring treatment sites as well (Aldous 1934).
An August burn in northeastern Oregon resulted in elevated postburn coverage for prairie Junegrass at 1 and 5 years after moderate and low severity burns. Moderate burns maintained 5% prairie Junegrass cover, near the preburn 3% coverage. The low severity burn showed an increase the first year from 2% to 9%, dropping to 4% the fifth year (Johnson 1998).

Average % composition for prairie Junegrass was observed under different burning regimes from 1928 to 1982 on a tall-grass prairie in Kansas. Winter (December 1) and early spring (March 20) burns highly favored prairie Junegrass (p < 0.05) (Towne and Owensby 1984). Burns at different times show degenerative effects (Blankespoor 1987).

Twelve to 15 years after prescribed burns in eastern Idaho, prairie Junegrass produced more herbage on burned than unburned sites. Production on “heavy” burns (main stem of sagebrush consumed by fire) was less than on “light” (only leaves consumed) and “moderate” (leaves and small branches consumed) burns. Results in pounds/acre are (Blaisdell 1953):

<table>
<thead>
<tr>
<th>Fire Season</th>
<th>Years post-fire</th>
<th>Unburned</th>
<th>Light burn</th>
<th>Moderate burn</th>
<th>Heavy burn</th>
</tr>
</thead>
<tbody>
<tr>
<td>August</td>
<td>12</td>
<td>9.4</td>
<td>14.2</td>
<td>13.3</td>
<td>10.7</td>
</tr>
<tr>
<td>September</td>
<td>15</td>
<td>32.8</td>
<td>43.6</td>
<td>53.4</td>
<td>36.8</td>
</tr>
</tbody>
</table>

FIRE MANAGEMENT CONSIDERATIONS

Burning, in general, has a positive influence on prairie Junegrass populations when conducted in early-spring and fall and followed by mean or above average annual precipitation (Towne and Owensby 1984). Water availability after a burn is important for maintaining healthy post-fire populations of prairie Junegrass. A fall burn and above average precipitation increased post-fire prairie Junegrass density in a southwestern ponderosa pine (Pinus ponderosa) forest (Oswald and Covington 1984). The opposite was seen at a fall burn in North Dakota, but burned areas possessed a lower moisture content than unburned areas (Barnes and Harrison 1982).

A study comparing fire tolerance and burn season found prairie Junegrass tolerant to burns conducted in May, June and November on a grassland in eastern Oregon (Britton et al. 1979). Evaluations were made 1 and 2 years after the burn.

LITERATURE CITED


Linaria dalmatica
Toadflax, Dalmatian toadflax

FIRE ECOLOGY OR ADAPTATIONS

Toadflax has a deep and extensive perennial, sprouting root system that is likely to allow it to survive even severe fire. Toadflax is also capable of establishing either from on-site seed, or seed dispersed into a burned area. Seed may be dispersed by animals into recently burned areas where it is adapted to establish under conditions of reduced competition. It is unclear what the effects of fire are on toadflax seed.

Dalmatian toadflax occurs in ecosystems with historic fire regimes of varied frequency and severity; from frequent, low-severity fires in ponderosa pine ecosystems, to less frequent and more severe fires in bunchgrass and sagebrush ecosystems, to frequent and severe fires in plains and prairie grassland ecosystems. Yellow toadflax occurs primarily in agricultural communities in the western U.S. and throughout Canada, and in disturbed areas in the north-central and northeastern U.S. A variety of native plant communities once dominated these areas, and historic fire regimes have been dramatically altered.

Toadflax was not widespread in these communities when historic fire regimes were functioning, but has established since habitat alteration and fire exclusion began. It is unclear how historic fire regimes might affect toadflax populations, and it is unclear how the presence of toadflax in native ecosystems might affect fire regimes.

In general, in ecosystems where toadflax replaces plants similar to itself (in terms of fuel characteristics), it may alter fire intensity or slightly modify an existing fire regime. However, if toadflax is qualitatively unique to the invaded ecosystem, it has the potential to completely alter the fire regime (D’Antonio 2000). No examples of fire regimes altered by toadflax invasion are described in the available literature.

The following table provides fire return intervals for plant communities and ecosystems in which toadflax may be found.

<table>
<thead>
<tr>
<th>Community or Ecosystem</th>
<th>Dominant Species</th>
<th>Fire Return Interval Range (years)</th>
</tr>
</thead>
<tbody>
<tr>
<td>bluestem prairie</td>
<td>Andropogon gerardii var. gerardii-</td>
<td>&lt; 10 [1,2]</td>
</tr>
<tr>
<td></td>
<td>Schizachyrium scoparium</td>
<td></td>
</tr>
<tr>
<td>Nebraska sandhills prairie</td>
<td>A. g. var. paucipilus-S. s.</td>
<td>&lt; 10</td>
</tr>
<tr>
<td>bluestem-Sacahuista prairie</td>
<td>Andropogon littoralis-Spartina spartinae</td>
<td>&lt; 10</td>
</tr>
<tr>
<td>sagebrush steppe</td>
<td>Artemisia tridentata/Pseudoroegneria spicata</td>
<td>20-70 [2]</td>
</tr>
<tr>
<td>plains grasslands</td>
<td>Bouteloua spp.</td>
<td>&lt; 35 [2,3]</td>
</tr>
<tr>
<td>Vegetation Type</td>
<td>Species Name</td>
<td>Height Range</td>
</tr>
<tr>
<td>---------------------------------------</td>
<td>-------------------------------------------</td>
<td>--------------</td>
</tr>
<tr>
<td>Cheatgrass</td>
<td><em>Bromus tectorum</em></td>
<td>&lt; 10 [4,5]</td>
</tr>
<tr>
<td>Curlleaf Mountain-mahogany*</td>
<td><em>Cercocarpus ledifolius</em></td>
<td>13-1,000 [6,7]</td>
</tr>
<tr>
<td>Mountain-mahogany-Gambel Oak Scrub</td>
<td><em>C. l.-Quercus gambelii</em></td>
<td>&lt; 35 to &lt; 100</td>
</tr>
<tr>
<td>Arizona Cypress</td>
<td><em>Cupressus arizonica</em></td>
<td>&lt; 35 to 200 [2]</td>
</tr>
<tr>
<td>California Steppe</td>
<td><em>Festuca-Danthonia spp.</em></td>
<td>&lt; 35 [2,8]</td>
</tr>
<tr>
<td>Juniper-oak Savanna</td>
<td><em>Juniperus ashei-Quercus virginiana</em></td>
<td>&lt; 35</td>
</tr>
<tr>
<td>Western Juniper</td>
<td><em>J. occidentalis</em></td>
<td>20-70</td>
</tr>
<tr>
<td>Rocky Mountain Juniper</td>
<td><em>J. scopulorum</em></td>
<td>&lt; 35</td>
</tr>
<tr>
<td>Tamarack</td>
<td><em>Larix laricina</em></td>
<td>35-200 [2]</td>
</tr>
<tr>
<td>Wheatgrass Plains Grasslands</td>
<td><em>Pascopyrum smithii</em></td>
<td>&lt; 5-47+ [2,9,3]</td>
</tr>
<tr>
<td>Pinyon-juniper</td>
<td><em>Pinus-Juniperus spp.</em></td>
<td>&lt; 35 [2]</td>
</tr>
<tr>
<td>Jack Pine</td>
<td><em>P. banksiana</em></td>
<td>&lt; 35 to 200 [10]</td>
</tr>
<tr>
<td>Rocky Mountain Lodgepole Pine*</td>
<td><em>P. contorta var. latifolia</em></td>
<td>25-300+ [11,12,13]</td>
</tr>
<tr>
<td>Sierra Lodgepole Pine*</td>
<td><em>P. c. var. murrayana</em></td>
<td>35-200 [12]</td>
</tr>
<tr>
<td>Colorado Pinyon</td>
<td><em>P. edulis</em></td>
<td>10-400+ [14,15,16,2]</td>
</tr>
<tr>
<td>Jeffrey Pine</td>
<td><em>P. jeffreyi</em></td>
<td>5-30</td>
</tr>
<tr>
<td>Western White Pine*</td>
<td><em>P. monticola</em></td>
<td>50-200</td>
</tr>
<tr>
<td>Pacific Ponderosa Pine*</td>
<td><em>P. ponderosa var. ponderosa</em></td>
<td>1-47 [12]</td>
</tr>
<tr>
<td>Interior Ponderosa Pine*</td>
<td><em>P. p. var. scopulorum</em></td>
<td>2-30 [12,17,18]</td>
</tr>
<tr>
<td>Arizona Pine</td>
<td><em>P. p. var. arizonica</em></td>
<td>2-15 [17,19,20]</td>
</tr>
<tr>
<td>Red Pine (Great Lakes Region)</td>
<td><em>P. resinosa</em></td>
<td>10-200 (10**) [10,21]</td>
</tr>
<tr>
<td>Red-white-jack Pine*</td>
<td><em>P. r.-P. strobus-P. banksiana</em></td>
<td>10-300 [10,22]</td>
</tr>
<tr>
<td>Eastern Cottonwood</td>
<td><em>Populus deltoides</em></td>
<td>&lt; 35 to 200 [2]</td>
</tr>
<tr>
<td>Aspen-birch</td>
<td><em>P. tremuloides-Betula papyrifera</em></td>
<td>35-200 [10,23]</td>
</tr>
<tr>
<td>Quaking Aspen (West of the Great Plains)</td>
<td><em>P. tremuloides</em></td>
<td>7-120 [12,24,25]</td>
</tr>
<tr>
<td>Mountain Grasslands</td>
<td><em>Pseudoroegneria spicata</em></td>
<td>3-40 (10**) [11,12]</td>
</tr>
<tr>
<td>California Oakwoods</td>
<td><em>Quercus spp.</em></td>
<td>&lt; 35 [12]</td>
</tr>
<tr>
<td>Oak-Hickory</td>
<td><em>Q.-Carya spp.</em></td>
<td>&lt; 35 [23]</td>
</tr>
<tr>
<td>Oak-Juniper Woodland (Southwest)</td>
<td><em>Q.-Juniperus spp.</em></td>
<td>&lt; 35 to &lt; 200 [2]</td>
</tr>
<tr>
<td>Northeastern Oak-Pine</td>
<td><em>Q.-Pinus spp.</em></td>
<td>10 to &lt; 35</td>
</tr>
<tr>
<td>White Oak-Black Oak-Northern Red Oak</td>
<td><em>Q. alba-Q. velutina-Q. rubra</em></td>
<td>&lt; 35 [23]</td>
</tr>
<tr>
<td>Canyon Live Oak</td>
<td><em>Q. chrysolepis</em></td>
<td>&lt; 35 to 200</td>
</tr>
<tr>
<td>Blue Oak-Foothills Pine</td>
<td><em>Q. douglasii-P. sabiniana</em></td>
<td>&lt; 35 [12]</td>
</tr>
<tr>
<td>Northern Pin Oak</td>
<td><em>Q. ellipsoidalis</em></td>
<td>&lt; 35 [23]</td>
</tr>
<tr>
<td>Oregon White Oak</td>
<td><em>Q. garryana</em></td>
<td>&lt; 35 [12]</td>
</tr>
</tbody>
</table>
California black oak  
*Q. kelloggii*  
5-30 [2]

bur oak  
*Q. macrocarpa*  
< 10 [23]

oak savanna  
*Q. m./Andropogon gerardii-Schizachyrium scoparium*  
2-14 [2,23]

black oak  
*Q. velutina*  
< 35 [23]

little bluestem-grama prairie  
*S. scoparium-Bouteloua spp.*  
< 35 [2]

Elm-ash-cottonwood  
*Ulmus-Fraxinus-Populus spp.*  
< 35 to 200 [10,23]

*fire return interval varies widely; trends in variation are noted in the species summary

**mean**


POST-FIRE REGENERATION STRATEGY (Stickney 1989)

Geophyte, growing points deep in soil; ground residual colonizer (on-site, initial community); initial off-site colonizer (off-site, initial community); and secondary colonizer (on-site or off-site seed sources).

IMMEDIATE FIRE EFFECT ON PLANT

Toadflax is likely to be top killed by fire, however its deep, extensive root system is likely to survive even severe fire and allow reestablishment of the population from vegetative buds on roots. Many root-sprouting plants, including toadflax, have high fire survival rates, regardless of burn severity. This is because even the most severe fires typically damage roots only to 10 cm below the soil (Goodwin and Sheley 2001), and toadflax roots typically penetrate the soil to a depth of nearly a metre.

There is little information available regarding the direct effects of fire on toadflax plants, and no information available regarding heat or fire effects on toadflax seed.

PLANT RESPONSE TO FIRE

Toadflax is able to recover after fire and may even be promoted by fire, especially if other species are reduced. The post-fire environment is well suited to toadflax establishment by seed.

Three sites in big sagebrush-bluebunch wheatgrass communities in western Montana were burned under prescription in mid-March to reduce shrubs and trees. Dalmatian toadflax density, cover, biomass per square meter, per plant biomass and per plant seed production were measured on burned plots and immediately adjacent unburned plots 6 months after burning. Dalmatian toadflax density and cover were not different between burned and unburned plots. Biomass of Dalmatian toadflax per square meter varied depending on site, while biomass per plant was significantly higher ($p > 0.05$) on all burned plots compared with unburned plots. Burning also significantly increased ($p > 0.05$) Dalmatian toadflax seed production per plant at all 3 sites. Seed production ranged from 7 to 79 seeds per plant on unburned plots, and from 158 to 1328 seeds per plant on burned plots (Jacobs and Sheley 2003a).

DISCUSSION AND QUALIFICATION OF PLANT RESPONSE

Response of toadflax to fire may depend on site characteristics and the fire adaptations of other species in the plant community. Most reviews suggest that toadflax is likely to increase or to be unaffected by fire (e.g. Carpenter and Murray 1998, Lajeunesse 1999, Saner et al. 1995, Vujnovic and Wein 1997). Several studies provide examples of toadflax establishment following fire (Kyle 2000, Phillips and Crisp 2001, Sackett and Haase 1998, Sackett et al. 1993, Sirois 1995).
A study in south-central New York designed to investigate the effects of accidental spring fires on the vegetation and soils of native plant communities found that yellow toadflax cover in quaking aspen groves was not significantly different (p < 0.05) between burned and unburned stands. The authors suggested that toadflax is a neutral species with regard to fire (Swan 1966).

Other studies suggest that toadflax is negatively affected by fire (e.g. Nernberg 1995), although experimental evidence is lacking. Percent cover of yellow toadflax decreased 2 growing seasons after spring prescribed fire on 2 study sites in Buena Vista Marsh, Wisconsin. Percent cover was recorded as follows (Halvorsen and Anderson 1983):

<table>
<thead>
<tr>
<th>Study site 1</th>
<th>Study site 2</th>
</tr>
</thead>
<tbody>
<tr>
<td>Burn</td>
<td>Control</td>
</tr>
<tr>
<td>pre-fire</td>
<td>post-fire</td>
</tr>
<tr>
<td>15.0</td>
<td>8.3</td>
</tr>
<tr>
<td>35.7</td>
<td>16.0</td>
</tr>
</tbody>
</table>

FIRE MANAGEMENT CONSIDERATIONS

Fire as a control agent: Burning is not usually a recommended or effective control method for toadflax, because root buds and buried seeds are unaffected by fire, and burning may increase competitiveness of toadflax by removing desirable plants (Carpenter and Murray 1998, Lajeunesse 1999, Rice and Randall 2003, Saner et al. 1995). Removal of toadflax top-growth may even stimulate production of vegetative shoots (Knight 2003, Lajeunesse 1999). Nonetheless, Nernberg (1995) describes a mixed-grass prairie restoration program in Saskatchewan in which “appropriately timed” prescribed burning is used to control growth and eliminate seed production in yellow toadflax and other nonnative invasive species; although no specific details or results are provided.

Scorching of floral stalks using propane burners can help prevent toadflax seed production (Lajeunesse 1999).

Prescribed fire can be used as a management tool on some sites in an effort to restore historic fire regimes and promote desirable species. The disturbance created by fire may, however, favor many invasive species. At the time of this writing, there are ongoing studies in western Montana designed to test the effects of prescribed fire combined with herbicide application to control Dalmatian toadflax and other invasive species (Rice and Flynn 2000). Herbicides may prevent domination by invasive species after fire in the short term; however, they also affect nontarget forbs that can compete with invasive species (Jacobs and Sheley 2003b).
On a big sagebrush-bluebunch wheatgrass site in western Montana, Jacobs and Sheley (2003b) tested the effects of prescribed burning with and without herbicide application on density, cover, and biomass of Dalmatian toadflax and native forbs as well as species richness, diversity, and evenness. Herbicides (picloram at 0.56 kg a.i./ha, and chlorsulfuron at 0.075 kg a.i./ha) were applied in October (prior to burning) and April (after burning). Burning was conducted in April. They found that fire effectively killed nearly all trees on the test plots (primary objective of prescribed burning), but did not affect species richness, diversity, or evenness. One season after burning, the biomass and cover of toadflax, but not its density, were 2 times greater on burned plots than on unburned plots. All herbicide treatments reduced biomass, cover, and density of Dalmatian toadflax to 90% of the control.

Chlorsulfuron decreased overall forb biomass to 50% of the control, while picloram decreased forb biomass to nearly zero. Both herbicides reduced richness and diversity, but not evenness. Timing of herbicide application made no difference in toadflax control or diversity indices, although spring application resulted in higher grass production. The authors conclude that using an herbicide such as chlorsulfuron may provide short-term control of Dalmatian toadflax. However, the combination of open niches left by forbs killed by herbicides, trees and shrubs killed by fire, and pressure on grasses by wildlife leaves sites susceptible to reestablishment of Dalmatian toadflax from the soil seed bank (Jacobs and Sheley 2003b).

Post-fire colonization potential: Because of its propensity to establish in dry, open areas with little plant competition, toadflax has high potential for establishing after fire (when competition from other vegetation is removed or reduced) either by seed imported to the site or by soil-stored seed. Several examples follow where toadflax established following fire. It is not clear in any of these examples whether toadflax plants or seeds were on-site prior to burning.

Two fire case studies near Flagstaff, Arizona, (Chimney Spring and Limestone Flat) were established to determine a burning interval that would adequately manipulate fuels and stocking of a ponderosa pine stand so that it could survive wildfire. The initial objective was to reduce fuel loads by reintroducing fire in areas where it had long been suppressed and fuels had accumulated. The fire at Chimney Spring reduced forest floor fuels by 63%, compared to a 42% reduction of forest floor fuel at Limestone Flat. In both places, several invasive species were abundant after fire, including Dalmatian toadflax, common mullein (Verbascum thapsus), and thistle (Cirsium pulchellum). Common mullein and Dalmatian toadflax were dominant on heavily burned sites around large, old-growth trees that have died since the initial burns (Sackett and Haase 1998, Sackett et al. 1993). Similarly, Kyle (2000) observed several sites in northern Arizona where large-scale, high-severity fires have burned over the past 4 decades and noted that many of these areas are dominated by nonnative plants such as cheatgrass, smooth brome (Bromus inermis), sweetclover (Melilotus spp.), orchardgrass (Dactylis glomerata), perennial ryegrass (Lolium perenne), and Dalmatian toadflax.
In ponderosa pine forest sites in northern Arizona in 1989, prescribed burning was conducted to investigate its effects on the rare plant, Flagstaff pennyroyal (*Hedeoma diffusum*). Dalmatian toadflax was not noted prior to burning or during counts from 1989 to 1995; but in 2000 it had invaded all of the spring burn plots, the control plot (which was adjacent to the spring burn plots), and none of the fall burn plots. It is unclear whether it was present prior to the initiation of the study, or if it was introduced during fire line construction, or following burning; however, it had not yet invaded the area surrounding the burn study area where grasses were very dense. The authors urge caution and awareness of Dalmatian toadflax and other weed species when prescribing fire treatments (Phillips and Crisp 2001).

Yellow toadflax was found in a jack pine-lichen woodland of the upper boreal forest in northern Quebec, 2 years after wildfire (Sirois 1995).

Toadflax invasion after fire may also be related to soil disturbances brought about through fire suppression activities. For example, after a wildfire and suppression activities in Glacier National Park, Montana, Dalmatian toadflax was found in trace amounts in bulldozed areas, but was not present in either burned or undisturbed areas (Benson and Kurth 1995).

Preventing post-fire establishment and spread: The USDA Forest Service’s “Guide to Noxious Weed Prevention Practices” (USDA Forest Service 2001) provides several fire management considerations for preventing establishment of invasive species that apply to toadflax.

Preventing invasive plants from establishing in weed-free burned areas is the most effective and least costly management method. This can be accomplished through early detection and eradication by careful monitoring, and by limiting invasive plant seed dispersal into the burned area by (Goodwin and Sheley 2001, USDA Forest Service 2001):

- re-establishing vegetation on bare ground as soon as possible
- using only certified invasive plant-free seed mixes when revegetation is necessary
- cleaning equipment and vehicles prior to entering the burned area
- regulating or preventing human and livestock entry into burned areas until desirable site vegetation has recovered sufficiently to resist invasion by undesirable vegetation
- detecting weeds early and eradicating before vegetative spread and/or seed dispersal
- eradicating small patches and containing or controlling large infestations within or adjacent to the burned area

Early detection is key, and monitoring in spring, summer, and fall is imperative. Eradicate newly established toadflax plants and small patches adjacent to burned areas to prevent or limit seed dispersal into the site (Goodwin and Sheley 2001, USDA Forest Service 2001).
Revegetation necessity can be based on the degree of desirable vegetation displaced by invasive plants prior to burning and on post-fire survival of desirable vegetation. Revegetation necessity can also be related to invasive plant survival as viable seeds, root crowns, or rhizomes capable of reproduction. In general, post-fire revegetation should be considered when desirable vegetation cover is less than about 30% (Goodwin and Sheley 2001).

When pre-fire cover of toadflax is absent to low (and has been so for 10 or more years- i.e. no seeds in the seed bank), and pre-fire cover of desirable vegetation is high, revegetation is probably not necessary after low- and medium-severity burns. After a high-severity burn on a site in this condition, revegetation may be necessary (depending on post-fire survival of desirable species), and intensive monitoring for invasive plant establishment is necessary to detect and eradicate newly established invasives before they spread (Goodwin and Sheley 2001).

When pre-fire cover of toadflax is moderate (20 to 79%) to high (80-100%), revegetation may be necessary after fire of any severity if desired vegetation cover is less than about 30%. Toadflax plants are likely to survive even severe fires, so intense weed management is also recommended, especially after fires of moderate to high severity (Goodwin and Sheley 2001).

Fall dormant broadcast seeding into ash will cover and retain seeds. If there is insufficient ash, seedbed preparation may be necessary. A seed mix should contain quick-establishing grasses and forbs (exclude forbs if broadleaf herbicides are anticipated) that can effectively occupy available niches. Managers can enhance the success of revegetation (natural or artificial) by excluding livestock until vegetation is well established (at least 2 growing seasons) (Goodwin and Sheley 2001).

When planning a prescribed burn, managers should preinventory the project area and evaluate cover and phenology of any toadflax present on or adjacent to the site, and avoid ignition and burning in areas at high risk for toadflax establishment or spread due to fire effects. Avoid creating soil conditions that promote weed germination and establishment. Discuss weed status and risks in burn rehabilitation plans. Also, wildfire managers might consider including weed prevention education and providing weed identification aids during fire training; avoiding known weed infestations when locating fire lines; monitoring camps, staging areas, helibases, etc., to be sure they are kept weed free; taking care that equipment is weed free; incorporating weed prevention into fire rehabilitation plans; and acquiring restoration funding. Additional guidelines and specific recommendations and requirements are available (USDA Forest Service 2001).
LITERATURE CITED


*Lolium multiflorum* (= *L. persicum*)

Italian ryegrass

**FIRE ECOLOGY OR ADAPTATIONS**

Natural fires in Italian ryegrass stands are most likely to occur during the dry season when Italian ryegrass has already produced seed and dried out. Dense stands of dry Italian ryegrass burn readily (Griffin 1982, Nadkarni and Odion 1986).

**POST-FIRE REGENERATION STRATEGY**

Tussock graminoid; ground residual colonizer (on-site, initial community).

**IMMEDIATE FIRE EFFECT ON PLANT**

Fire probably kills Italian ryegrass. Seeds may survive fire.
PLANT RESPONSE TO FIRE

Italian ryegrass generally decreases after fire. A California annual grassland site on the coastal foothills near Berkeley burned in July. Forbs generally increased and grasses decreased in the first growing season after the fire. Burned sites produced lower Italian ryegrass yields than unburned sites. Italian ryegrass height was significantly lower the year after the fire on burned sites than unburned sites (Hervey 1949):

<table>
<thead>
<tr>
<th></th>
<th>Dec 1</th>
<th>Feb 20</th>
<th>May 1</th>
</tr>
</thead>
<tbody>
<tr>
<td>Burned</td>
<td>6.1</td>
<td>8.9</td>
<td>34.8</td>
</tr>
<tr>
<td>Unburned</td>
<td>9.4</td>
<td>17.5</td>
<td>40.9</td>
</tr>
</tbody>
</table>

A March 5 fire on a Georgia old field resulted in a significant (p < 0.05) decrease in Italian ryegrass yield. The control produced 98.4 grams per square meter and the burned area produced 47.4 grams per square meter in the summer following the fire (Odum et al. 1986).

Italian ryegrass was present (0.1% cover) after a July fire in San Bernardino County, California. The pre-fire community, dominated by brittle bush (*Encelia farinosa*), had not burned for 21 years and did not contain Italian ryegrass (Westman et al. 1981). The seed source for the post-fire population of Italian ryegrass was not described.

FIRE MANAGEMENT CONSIDERATIONS

For years Italian ryegrass has been the most commonly used species for controlling erosion on burned chaparral sites, especially in northern California and coastal areas. Seed is usually broadcast aerially in the fall following the fire but preceding the winter rains. Italian ryegrass has reliable germination, rapid early growth, a short life span, abundant fibrous roots, inexpensive seed, and broad site adaptability (Taskey et al. 1989). However, the wisdom of ryegrass seeding has been questioned for decades, and recent literature indicates that post-fire seeding of Italian ryegrass may cause more harm than good. The purpose of the seeding is to control erosion which is often severe during winter rains on steep slopes in California. However, studies have shown that the seeding is not effective at controlling erosion the first year and may even increase erosion in succeeding years. In addition, Italian ryegrass outcompetes the recovering native vegetation and may increase the fire hazard. The controversy is reviewed by Barro and Conard (Barro and Conard 1987) and Gautier (1983), and is summarized here.

Several studies have shown that erosion was not controlled, and even increased, with Italian ryegrass seeding (Griffin 1982, Taskey et al. 1989). Erosion was greater on seeded sites than on unseeded sites of the 1985 Las Plititas Fire in the Santa Lucia Range, Monterey County, California. Researchers found that pocket gopher activity was greater on seeded sites and was the cause of the increased erosion (Taskey et al. 1989). Nadkami and Odion (1986) suggest that as Italian ryegrass
declines, vegetative cover on seeded sites may actually be less than cover on unseeded sites, and thus erosion may be greater. After the Marble-Cone Fire in the Santa Lucia Range, heavy rains in January washed 3-8 cm of surface soil from slopes greater than 20 percent. The erosion occurred before seeded Italian ryegrass had formed an effective cover (Griffin 1982). Winter rains are often not sufficient for Italian ryegrass germination until December or January when the daily temperature is too cool for adequate growth. A wetting agent applied to the soil surface during the seeding may encourage earlier germination (DeBano and Conrad 1974).

Seeding Italian ryegrass may have long-term detrimental effects on chaparral communities because Italian ryegrass interferes with native species regeneration. On sites seeded with Italian ryegrass, the seedbank becomes depleted of fire-following species because they may germinate but do not establish (Nadkarni and Odion 1986). After Italian ryegrass dies out it often leaves behind a thinned out chaparral with considerably fewer nonsprouting species such as wedgeleaf ceanothus (*Ceanothus cuneatus*) than in areas without Italian ryegrass seeding (Biswell 1974). In seeded plots in burned chaparral in the Santa Ynez Mountains, California, there was a 40 percent reduction in species diversity compared with unseeded plots.

The predominant native Amador rushrose (*Helianthemum scoparium*) was less dense in the seeded treatment, and two other species usually found (hoaryleaf ceanothus [*C. crassifolius*] and common turricula [*Turricula parryi*]) were absent from the seeded plots. The fire-annual yellow whisperingbells (*Emmenanthe penduliflora*) had over 50 percent less cover on seeded plots than unseeded plots at one site in southern California (Beyers et al. 1993). The first year following the Las Pilitas Fire in the Santa Lucia Mountains, Italian ryegrass interfered with the regeneration of lupine (*Lupinus* spp.), lotus (*Lotus* spp.), and chamise (*Adenostoma fasciculatum*) (Taskey et al. 1989). One year after fire in chaparral in the Santa Monica Mountains, California, Italian ryegrass cover was negatively correlated with herbaceous species and with island ceanothus (*Ceanothus megacarpus*) (Conard et al. 1991). Because of the ability of Italian ryegrass to compete well with woody species, it is recommended for seeding of fire breaks within chaparral communities (Green 1977).

Italian ryegrass restricts tree regeneration where seeded on burned forested sites. Italian ryegrass interfered with the regeneration of sugar pine (*Pinus lambertiana*) and Coulter pine (*P. coulteri*) seedlings after the Marble-Cone Fire (Griffin 1982). Two years after fire on the Stanislaus National Forest, California, post-fire regeneration of ponderosa pine (*P. ponderosa*) was absent where Italian ryegrass cover was greater than 40 percent (Conard et al. 1991).

An additional detrimental effect of post-fire seeding is that dense stands of Italian ryegrass burn readily, and early recurring fire is destructive to regenerating shrubs (Zedler et al. 1983). The natural fire interval in chaparral is about 10 to 100 years (Westman et al. 1981). An August 1979 fire on Otay Mountain, San Diego County, California, was seeded with Italian ryegrass. The year had near-record precipitation so Italian ryegrass growth was exceptional. In July another fire occurred in the areas
seeded with Italian ryegrass. This second fire killed nearly all seedlings of explorer's bush (C. oliganthus), and chamise was reduced by up to 97 percent. Mission manzanita (Xylococcus bicolor), a post-fire sprouter, suffered substantial mortality. It is believed that such an early return fire causes drastic shifts in species composition (Zedler et al. 1983). In the Santa Ynez Mountains a July fire burned into seeded areas but not unseeded areas 2 years after the original fire (Nadkarni and Odion 1986).

After fire in grasslands containing Italian ryegrass, grazing should be delayed or reduced to allow Italian ryegrass to recover (Hervey 1949).

LITERATURE CITED


Lolium perenne
Perennial ryegrass

FIRE ECOLOGY OR ADAPTATIONS

As with most perennial grasses, perennial ryegrass is well adapted to fire. It is top-killed and will sprout quickly from the rhizome. Fire is beneficial to grass swards; by removing litter, it allows more light to penetrate to the leaf bases and new tillers (Wright and Bailey 1982).

POST-FIRE REGENERATION STRATEGY

Tussock graminoid; ground residual colonizer (on-site, initial community); secondary colonizer - off-site seed.

IMMEDIATE FIRE EFFECT ON PLANT

Fire top-kills perennial ryegrass; high-severity fire can kill the rhizomes (Hafenrichter et al. 1968, Hardison 1980).

PLANT RESPONSE TO FIRE

Fire stimulates the production of reproductive tillers in perennial ryegrass (Frakes 1973). When field burning was initiated on seed fields in Oregon, seed yields tripled (Hardison 1980).

FIRE MANAGEMENT CONSIDERATIONS

Fields of perennial ryegrass grown for seed are usually burned to control blind-seed disease, remove crop residues, improve herbicide effectiveness, and stimulate reproductive tiller initiation. For effective burning, it is important to ensure that crop residues are fairly evenly distributed, since hot spots can be lethal to underground rhizomes (Frakes 1973, Hafenrichter et al. 1968).

LITERATURE CITED


Maianthemum canadense
Wild lily-of-the-valley

FIRE ECOLOGY OR ADAPTATIONS

Wild lily-of-the-valley sprouts following fire; very few plants come in as seed (Ahlgren 1966, Brumelis and Carleton 1989, McRae 1979). Wild lily-of-the-valley recovery may be affected by the season of burning due to the amount of nutrient reserves in its roots and rhizomes. It had reduced recovery after spring burning, apparently due to reserves depleted during leafing out (Flinn 1980, Flinn et al. 1983). However, it has been rated as an increaser after fire, including after spring burning (Flinn 1980, Swan 1970). Wild lily-of-the-valley survives fire because its meristems grow in the damp litter and ground (Beasleigh and Yarranton 1974). Wild lily-of-the-valley located in damp depressions survived a wildfire on Isle Royale, Michigan, and had 1.2 stems/m² (Cooper 1928). Its rhizomes can withstand low- to moderate-severity fires (Flinn 1980, Flinn and Wein 1977). After fire has opened forest canopies, wild lily-of-the-valley can cover large areas where it was previously sparse under the closed canopy (Cooper 1913). In the upland boreal mixed woods that wild lily-of-the-valley is a part of, the natural fire return intervals are between 20 and 340 years (Despain and Romme 1991, Thomas and Wein 1985).

Wild lily-of-the-valley rhizomes can tolerate brief exposures to high temperatures. Its rhizomes were collected spring, summer, and fall and subjected to wet heat treatments. Maximum shoot growth and number of stems occurred after spring-collected rhizomes were placed at 55⁰ C for 5 minutes. Growth also continued after 60⁰ deg C treatments; however, summer- and autumn-collected rhizomes died after this high temperature treatment (Flinn and Pringle 1983).

POST-FIRE REGENERATION STRATEGY

Rhizomatous herb, rhizome in soil; secondary colonizer - off-site seed.
IMMEDIATE FIRE EFFECT ON PLANT

Fire top-kills wild lily-of-the-valley. Surviving rhizomes grow, and flowers are initiated the first growing season following a fire (Cooper 1913). Wild lily-of-the-valley flowered in June following low-severity fires (i.e., surface litter layer was consumed) during October and April. Flowering began 22 days earlier on the fall-burned than on the spring-burned plants, and fruit developed on fall-burned but not on the spring-burned plants (Chapman and Crow 1981). Wild lily-of-the-valley sprouted within 2 weeks after a prescribed fire and was common in all stands; however, cover had decreased (Sidhu 1973a). In stump-prairies of northeastern Wisconsin that were burned in the spring, it sprouted by summer and increased in frequency (Vogl 1964). Seven soil core samples were collected 1 week following a low- to moderate-severity ground wildfire in April in a boreal mixed conifer-hardwood forest. In the soil samples, most wild lily-of-the-valley developed from surviving rhizomes; however, five seeds germinated from three of the samples (Archibold 1979).

PLANT RESPONSE TO FIRE

Wild lily-of-the-valley recovers slowly after fire (Chapman and Crow 1981, Krefting and Ahlgren 1974). Its recovery rate may be variable due to the severity of burning or to successive annual fires. Wild lily-of-the-valley can be one of the first species reported on a fresh burn (Wright and Bailey 1982). In a boreal mixed wood in New Brunswick, it had widely variable responses during different seasons of burning (spring, summer, or fall); therefore, averaged responses among seasons was similar (Flinn and Wein 1988). In western Maine following a severe fire where the organic soil was consumed, surviving wild lily-of-the-valley sprouted after 1 month (Gilley 1982). However, it occurred infrequently 2 years after a severe summer burn in which all the litter and humus were destroyed and the mineral soil was exposed (Martin 1955). In a jack pine stand in northeastern Minnesota under various silvicultural and prescribed burning treatments, there was a 20 percent decrease in wild lily-of-the-valley 1 year following prescribed burning when temperatures were less than 482º C. Its frequency decreased by 70 percent where temperatures mostly exceeded 482º C (Ahlgren 1966).

There was a significant (p < 0.05) decrease in wild lily-of-the-valley biomass 2 years after a winter clearcut and summer prescribed burning in northern Minnesota (Outcalt and White 1981).

Wild lily-of-the-valley took 4 to 10 years to seed in from nearby areas following prescribed fires on clearcuts seeded with jack pine (Chrosciewicz 1983a, Chrosciewicz 1983b). Soil samples were taken from burned and unburned areas 3 years after a fire in an old-growth red pine (Pinus resinosa) stand. Wild lily-of-the-valley germinated only in soil from the unburned area (Ahlgren 1979). It was less frequent in open, burned areas than in unburned areas in oak-pine woods (Blewett 1978).
Following two successive annual, low-severity fires where the duff was not consumed, wild lily-of-the-valley remained a dominant species with increases in relative densities or frequencies at post-fire years 1 to 3 (Martin 1955, Methven 1973, Ohmann and Grigal 1966, Sidhu 1973b). It decreased in frequency from pretreatment levels of 42 percent down to 1 percent following logging with 2 successive years of prescribed burning (Hall 1955).

In cutover areas aged 2 to 40 years since fire, it had only 0 to 2 percent cover (Martin 1956). Following prescribed spring fires in boreal mixed woods, wild lily-of-the-valley frequency declined from 40 to 16 percent. Its frequency further declined to 8 percent following another fire 6 years later on this area (Loope 1991).

Wild lily-of-the-valley had lower frequencies (53 and 57 percent) than the control (97 percent) 11 and 14 years after fires in mixed conifer-hardwoods in northeastern Minnesota (Krefting and Ahlgren 1974). It was one of the most abundant species present 13 years following a severe wildfire in mixed conifer stand in Minnesota and Ontario (Ahlgren 1976). In different burns aged 9 to 50 years in Ontario, wild lily-of-the-valley had the highest density on burns aged 25, 29, or 50 years (Scheiner and Teeri 1981, Smith 1966). At post-fire year 33, it had similar frequencies (25 to 33 percent) and cover (1 to 2 percent) in four different forest communities of aspen-birch, birch, jack pine-birch, and jack pine (Ohmann et al. 1973). In moist mixed woods in North Dakota, its relative cover 80 years following fire was not different from unburned areas (Potter and Moir 1961). Fuel loadings were variable in fire-prone forest stands in Michigan where wild lily-of-the-valley was a typical understory species; it was present in low frequency (0.3 percent) 84 years following fire (Loope 1991).

There was no significant ($p > 0.05$) difference in the occurrence of wild lily-of-the-valley under five shade treatments (0 to 100 percent shade) following a low-severity prescribed spring fire (Hoefs and Shay 1981).

**FIRE MANAGEMENT CONSIDERATIONS**

Measurements on wild lily-of-the-valley were used to develop regression equations for predicting changes in forest floor moisture in upland pine communities (Chrosciewicz 1989). There were no seasonal trends in change of moisture content for wild lily-of-the-valley during a study to assess understory flammability in Great Lakes coniferous forests for use in the National Fire Danger Rating System (Loomis et al. 1979). Spring burning may be the most effective control for wild lily-of-the-valley during site preparation because carbohydrate reserves are lowest, potentially reducing plant vigor (Flinn et al. 1983).
REFERENCES


Hall, I.V. 1955. Floristic changes following the cutting and burning of a woodlot for blueberry production. Canadian Journal of Agricultural Science 35:143-152.


Medicago sativa
Alfalfa

FIRE ECOLOGY OR ADAPTATIONS

As a perennial with a narrow root crown, alfalfa will survive most fires by sprouting after being top-killed. Alfalfa hard seeds may be scarified by moderate-severity fires (Yoakum 1978, Richardson 1985).

POST-FIRE REGENERATION STRATEGY

Rhizomatous herb, rhizome in soil; ground residual colonizer (on-site, initial community); secondary colonizer - off-site seed.

IMMEDIATE FIRE EFFECT ON PLANT

Moderately severe fires will top-kill alfalfa shoots, and severe fires may cause damage to or kill the root crown, killing the plant (Olson 1975).
PLANT RESPONSE TO FIRE

Alfalfa fields that were burned to control insect pests were monitored for soil changes and plant response. The root systems of the plants were not adversely affected by the fire, and subsequent crops were similar in appearance and productivity to that of unburned control plots. Soil preburn conditions (organic matter and nitrogen) were attained within 160 days (Dormaar and Schaber 1985). Canopy coverage of alfalfa increased by the end of the first growing season following a prescribed fire in May but showed no significant difference from unburned controls in the second growing season (which may be attributed to the low precipitation that year) (Olson 1975). Mixtures of cool-season grasses and alfalfa and/or sweetclover respond best (in productivity) to prescribed fires from March to June. The lowest response by alfalfa is to late summer-early fall fires (Higgins et al. 1989, Kruse and Higgins 1990, Olson 1975).

In a study of individual plant responses to a spring fire in a tallgrass prairie stand, Pemble and others (Pemble et al. 1981) found that a moderate-severity fire resulted in a slight decrease in the amount of flowering (flowers per plant and plants in flower) in alfalfa.

FIRE MANAGEMENT CONSIDERATIONS

The fire susceptibility of rangeland vegetation depends on the reduction of fuel loads through animal use or drought which reduces standing crop size (Heady 1988). Seeded alfalfa fields are often burned prior to growth initiation in the spring to reduce insect pests. This treatment results in destruction of insect eggs and adults, and reduces debris from the previous growing season that encourages insect population growth. Since soil preburn conditions are attained within 160 days of the fire, it is unlikely that a 3-year interval between fires would be detrimental to the soil (Dormaar and Schaber 1985).

Under current evaluation is a method of presuppression fire management called “greenstrip management.” This involves the production of a vegetative fuelbreak of green plants that are less flammable than the surrounding native vegetation. Alfalfa is the most commonly used forb for this purpose (Pellant 1990).

LITERATURE CITED


Melilotus albus (= M. alba)
White sweetclover

FIRE ECOLOGY OR ADAPTATIONS

White sweetclover seeds have hard, impermeable seed coats, and may remain dormant in soil seed banks for years. The heat from a fire breaks the seed coat, allowing the seed to germinate. Where soil-stored seed is present, burning is stimulatory, resulting in abundant seed germination and seedling establishment (Heitlinger 1975, Kline 1986, Olson 1975).

Second-year plants present as buds on the caudex may survive dormant-season burns, as they are located about 5 cm below the ground surface (Turkington et al. 1978). On the Curtis Prairie in Wisconsin, second-year white sweetclover was abundant in the spring following dormant season fall burns (Kline 1986). Actively growing second-year plants, however, are easily killed by fire.

POST-FIRE REGENERATION STRATEGY

Caudex, growing points in soil; ground residual colonizer (on-site, initial community).

IMMEDIATE FIRE EFFECT ON PLANT

Once the crown buds of second-year plants have expanded, growth originates from branch tips or branch axils. Thus once shoot growth has begun, fire kills second-year plants simply by removing or scorching the growing points, or on large individuals by charring the stem base (Heitlinger 1975, Kline 1986). Numerous studies have shown that spring or summer burning in prairies and old fields effectively kills most second-year plants.

DISCUSSION AND QUALIFICATION OF FIRE EFFECT

Burning of a remnant tallgrass prairie in Minnesota on May 1, when second-year white sweetclover plants were 5-15 cm tall, resulted in the virtual elimination of second-year plants from the site. July burning on another portion of the same prairie similarly resulted in killing all first- and second-year plants. Although some large second-year plants were not completely consumed by fire, none sprouted by early September (Heitlinger 1975).

In the summer following early or mid-May burning on the Curtis Prairie in Wisconsin, frequency of second-year plants ranged from 5 to 26 percent on burned plots and 93 to 100 percent on unburned plots (Kline 1986).

In grass-dominated old fields in eastern North Dakota, late June burning “completely eradicated” second-year white sweetclover (Olson 1975).
PLANT RESPONSE TO FIRE

Numerous studies have documented that fire stimulates germination of white sweetclover seed (Glenn-Lewin et al. 1990, Heitlinger 1975, Kline 1986, Olson 1975, Rice 1932, Schwegman and McClain 1986). Seedling establishment generally occurs as follows after burning at different times of the year:

Spring burning - rapid seed germination and abundant seedling establishment occur shortly after burning. Additional seeds germinate in the summer of the first postburn growing season, but few late-germinating seedlings survive the winter.

Summer burning - poor to fair seed germination can occur after summer burning, but few of these late-germinating seedlings survive the winter. Additional seeds germinate in post-fire year 2.

Fall burning - abundant germination occurs in the spring following fall burning.

Following spring burning of grasslands, frequency and cover of first-year white sweetclover is much higher on burned than unburned areas during the first post-fire growing season. During post-fire year 2, first-year plants are rare, while second-year plants are abundant. White sweetclover frequency declines after post-fire year 3.

FIRE MANAGEMENT CONSIDERATIONS

White sweetclover thrives under a management program of periodic spring burning on a 2-year or longer cycle, which has been a common practice in many managed grasslands. Under this regime, soil-stored white sweetclover seed is scarified, resulting in abundant seedling establishment. The plants then overwinter, produce abundant seed in their second year, and replenish the soil seed bank. Because of these life history attributes, the use of fire to suppress white sweetclover is possible, but several successive annual or biennial burns are probably required to exhaust the seed supply. Dormant-season burns, whether early spring or late fall, are not recommended because they do not kill overwintering second-year plants.

Heitlinger (1975) recommended the following strategies to suppress white sweetclover and reduce seed supplies in tallgrass prairie: (1) burn annually about early May (for Minnesota) when second-year shoots are clearly visible, (2) burn every second year in early July before seed of second-year plants ripens, or (3) burn annually in early September near the beginning of the critical growth period (see Management Considerations and Seasonal Development for more information on the critical growth period).

In Wisconsin, a combination of an April burn followed the next year by a May burn was more successful in reducing white sweetclover than other burning combinations. Heavily infested parrie stands where this burning combination was conducted twice, separated by 2 years without burning, became almost completely free of white sweet clover (Kline 1986).
LITERATURE CITED


Phleum pratense
Timothy

FIRE ECOLOGY OR ADAPTATIONS

As with most perennial grasses, timothy is well adapted to fire. Susceptibility of pasture or range vegetation to fire depends on specific fire adaptations of the species and phenological stage when burned. Timothy has underground regenerative organs that are not harmed by moderately severe fires. Timothy is harmed if burned when actively growing in the spring and summer but is fairly fire tolerant when dormant (Wasser 1982).
In Yellowstone National Park after the fires of 1988, timothy sprouted from the roots after being top-killed (Anderson and Romme 1991). Timothy can occur on extremely cold sites; these sites seldom burn (Wasser 1982).

**POST-FIRE REGENERATION STRATEGY**

Tussock graminoid; ground residual colonizer (on-site, initial community); secondary colonizer - off-site seed.

**IMMEDIATE FIRE EFFECT ON PLANT**

Moderately severe fires will top-kill timothy, and severe fires may cause damage to or kill the root crown, killing the plant (Anderson and Romme 1991).

**PLANT RESPONSE TO FIRE**

Fire stimulates the production of reproductive tillers in timothy. In Illinois, a prescribed burn in August was beneficial for rejuvenation of timothy sods. Seed production increased following fire, and there was an increased success of timothy 2 to 4 years after the burn (Westemeier 1973).

In Oregon in early November, fire increased the vegetative yield and maximum height of timothy (Cornely et al. 1983). Following a prescribed prairie fire in Iowa carried out after snowmelt but while the soil was still frozen, timothy started growth 2 to 3 weeks earlier in the spring and matured earlier on burned areas than on adjacent unburned areas (Ehrenreich and Aikman 1963).

Several forage species were tested for performance on recently burned lodgepole pine sites in northeastern Washington. After 4 years, timothy was considered adequate in vigor and density. On a northeastern slope, timothy was more successful because of better soil and moisture conditions (Evanko 1953).

On a game farm in Pennsylvania, 2 ha were burned on April 22, 1983 to determine vegetation response. Timothy production decreased after the spring burn. There was no change in percent composition of timothy between 4 and 16 months after the fire. Early spring burning temporarily reduced perennial grasses and increased forbs. Grassy cover improved by post-fire year 2 (Hughes 1985).

Total herbaceous production of timothy following the 1983 burn (Hughes 1985):

<table>
<thead>
<tr>
<th>months after burn</th>
<th>control (%)</th>
<th>burn (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>28</td>
<td>3</td>
</tr>
<tr>
<td>2</td>
<td>41</td>
<td>10</td>
</tr>
<tr>
<td>3</td>
<td>45</td>
<td>7</td>
</tr>
<tr>
<td>4</td>
<td>55</td>
<td>0</td>
</tr>
<tr>
<td>6</td>
<td>48</td>
<td>6</td>
</tr>
</tbody>
</table>
FIRE MANAGEMENT CONSIDERATIONS

Timothy is often used to stabilize soil against erosion and to provide cover for wildlife in clearcut areas that have been burned (Anderson and Elliott 1957). In the midwestern states, prairie fires are often prescribed and timothy seeded to provide nesting cover for prairie chickens and waterfowl (Anderson and Elliott 1957).

In Montana, timothy was aerially seeded on a lodgepole pine clearcut that had been burned. It was monitored for 12 years, from 1962 to 1973. Timothy was a strong competitor in the early years postburn, but eventual dominance by native grasses was suggested by the decline of timothy from 3.0 percent in 1964 to 0.7 percent in 1973 (Lyon 1976).

In Oregon on a clearcut burned in 1969, timothy was seeded with a mixture of other grasses and legumes at a rate of 6.8 kg/ha. In 1973, timothy was abundant. By 1984, timothy declined drastically in numbers, partially because of heavy grazing pressure (Miller et al. 1986).

In Deadwood, South Dakota in 1959, an intense forest fire burned 1800 ha of land. Artificial seeding on 1604 ha at 12.4 kg/ha of a mixture containing timothy was completed. The mixture consisted of 3.4 kg/ha of timothy, 3.4 kg/ha of smooth brome, 2.25 kg/ha of Kentucky bluegrass, 2.25 kg/ha of yellow sweet clover, and 1.125 kg/ha of hairy vetch. Two sites were seeded. Site one was on stoney-loam soil at 1620 m and site two was on a finer textured soil at 1470 m. Timothy established quickly and persisted in dominance on site one. At site two, timothy was codominant with other species (Orr 1970).

In northern Alberta, timothy was used to reseed burned-over land after a fire in 1950. The organic matter was destroyed and the depth of ash was 2.5-7.6 cm. Seedings were done in the fall on 7.6-15 cm of snow and in April at the same depth with no snow or frost. Productivity was not influenced by the time of seeding. Timothy seeds established where moisture was adequate. Stands of timothy declined with age (Anderson and Elliott 1957).

LITERATURE CITED


FIRE ECOLOGY OR ADAPTATIONS

During grassland fires, the fire front passes quickly and temperatures 2.5 cm below the soil surface rise very little (Daubenmire 1968). Located about 5 cm below the soil surface, Canada bluegrass rhizomes survive and initiate new growth after aboveground plant portions are consumed by fire. Although the plant survives because of soil-insulated rhizomes, post-fire plant vigor and density are greatly affected by phenological stage at time of burning.

Information regarding the importance that seedling establishment plays in Canada bluegrass immediate post-fire recovery was not found in the literature. Post-fire growth is assumed to be primarily due to rhizome survival.

POST-FIRE REGENERATION STRATEGY

Rhizomatous herb, rhizome in soil

IMMEDIATE FIRE EFFECT ON PLANT

Plant phenological stage at time of burning greatly influences fire damage to herbaceous plants. In general, as new foliage of perennial grasses reaches full development major food reserves have been depleted, so that plants are injured most from fires occurring at this time (Daubenmire 1968). Late spring fires, after plants have been growing for about a month or more, appear to be the most damaging to Canada bluegrass.

PLANT RESPONSE TO FIRE

Season of burning and frequency of burning greatly influence Canada bluegrass post-fire recovery. Dormant-season fires favor Canada bluegrass, and biomass and density may increase during post-fire year 1. Late spring burning, when plants are actively growing, reduces biomass and density during post-fire year 1, but biomass and density may return to preburn levels within 1 or 2 years. Thus Canada bluegrass often recovers within 1 or 2 years after a single late spring fire, but density and biomass are progressively reduced if burned annually or biennially in late spring.

In abandoned fields in southern Wisconsin, Canada bluegrass flowering stem density was reduced 50 percent when burned annually in May for 5 years. Conversely, flowering stem density increased 170 and 440 percent following 5 years of annual burning in March or October, respectively (Curtis and Partch 1948). A similar study in southern Wisconsin found that 3 years of annual burning in mid-May reduced Canada and Kentucky bluegrass flowering stem density by 70 percent, while late March or early April burning had little affect on flowering (Henderson et al. 1983).
In a reconstructed tallgrass prairie in Illinois, bluegrass (*Poa compressa* and *P. pratensis* combined) percent relative biomass decreased as fire frequency increased in two communities as follows (Hadley and Kieckhefer 1963):

<table>
<thead>
<tr>
<th>Burning Treatment*</th>
<th>not burned</th>
<th>burned twice</th>
<th>burned 3 times</th>
<th>burned 4 times</th>
</tr>
</thead>
<tbody>
<tr>
<td>Community type</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>big bluestem</td>
<td>23.4**</td>
<td>18.3</td>
<td>4.6</td>
<td>0</td>
</tr>
<tr>
<td>indiangrass</td>
<td>18.6</td>
<td>15.9</td>
<td>3.3</td>
<td>0</td>
</tr>
</tbody>
</table>

*not burned = unburned for 19 years
burned twice = burned Feb. 28, 1952 and April 16, 1959
burned three times = burned Feb. 28, 1952; April 16, 1959; and May 2, 1961
burned four times = burned Feb. 28, 1952; April 16, 1959; May 2, 1961; and May 10, 1962

**sampled at the end of the 1962 growing season

In oak (*Quercus* spp.) woods and goldenrod (*Solidago* spp.) fields accidently burned between April 6 and May 2 in south-central New York, Canada bluegrass frequency increased from 6 to 17 percent and 56 to 81 percent, respectively, 10 to 26 months after burning (Swan 1970).

After early May prescribed burning in seral brushfields in northern Idaho, Canada bluegrass recovered rapidly on lightly burned plots. During the first post-fire growing season, it produced the bulk of grass biomass on lightly burned plots, which was 151 kg/ha. In comparison, grass production on heavily burned and control plots averaged only 0.8 and 11.4 kg/ha, respectively (Hooker and Tisdale 1974).

**FIRE MANAGEMENT CONSIDERATIONS

Annual or biennial late spring burning can be used to control Canada bluegrass and promote the growth of warm-season grasses in the Midwest. The timing of burning is critical and should take place just prior to the resumption of warm-season grass growth. Such burning favors warm-season grasses because they are dormant at the time of burning. Conversely, cool-season species like Canada bluegrass are harmed by late spring fire because they resume growth in the early spring and are thus actively growing at the time of burning (Higgins *et al.* 1989).

**LITERATURE CITED


Poa pratensis
Kentucky bluegrass

FIRE ECOLOGY OR ADAPTATIONS

During grassland fires, the fire front passes quickly and temperatures 2.5 cm below the soil surface rise very little (Daubenmire 1968). During a late April prescribed fire in an oak savanna in Minnesota, where Kentucky bluegrass formed an almost complete sod between bunches of native tallgrasses, temperatures immediately below the soil surface rarely exceeded 51° C (Tester 1965). Located about 5 cm below the soil surface, Kentucky bluegrass rhizomes survive and initiate new growth after aboveground plant portions are consumed by fire. Although the plant survives because of soil-insulated rhizomes, post-fire plant vigor and density are greatly affected by phenological stage at time of burning.

Seedling establishment is unimportant in immediate post-fire recovery. However, burning may enhance seed germination of Kentucky bluegrass during the second post-fire growing season. On an Iowa prairie codominated by big bluestem (Andropogon gerardii var. gerardii), indiangrass (Sorghastrum nutans), and Kentucky bluegrass, Kentucky bluegrass seedlings were more abundant in 1986 on plots burned in May, June, August, or November of 1985 than on unburned plots (Johnson 1987).
POST-FIRE REGENERATION STRATEGY

Rhizomatous herb, rhizome in soil.

IMMEDIATE FIRE EFFECT ON PLANT

Plant phenological stage at time of burning greatly influences fire damage to herbaceous plants. In general, as new foliage of perennial grasses reaches full development major food reserves have been depleted, so that plants are injured most from fires occurring at this time (Daubenmire 1968, Risser et al. 1981). Because Kentucky bluegrass is a cool-season grass, active in the spring and fall, it is most susceptible to fire damage at those times. Late spring fires, after plants have been growing for about a month or more, are the most damaging to Kentucky bluegrass. Sampling at the end of the first growing season after late spring burning shows that Kentucky bluegrass basal cover and tiller density are typically much lower in burned areas than in nearby unburned areas (Blankespoor 1987, Diboll 1986, Engle and Bultsma 1984, Hadley 1970, Nagal 1983, Old 1969, Olson 1975, Schacht and Stubbendieck 1985, Svedarsky and Buckley 1975).

Cool fires conducted when plants are dormant have little effect on Kentucky bluegrass (Kovalchik 1987).

PLANT RESPONSE TO FIRE

Kentucky bluegrass’s fire response varies greatly depending on season of burning, fire frequency, and post-fire precipitation and soil moisture.

Season of burning: Kentucky bluegrass post-fire cover, biomass, and flower stalk density are often greatly reduced during the first post-fire growing season by a single late spring fire. Three examples are presented to demonstrate rather typical first-year responses to late spring burning: (1) in mixed-grass prairie unburned for several years in north-central Nebraska, a single prescribed fire in mid-April or mid-May greatly reduced Kentucky bluegrass basal cover in October, with cover on burned plots only half that found on unburned plots (Nagal 1983), (2) after a single mid-April fire on a tallgrass prairie site unburned for several years in Iowa, Kentucky bluegrass relative biomass decreased from 80 percent to 25 percent during the first post-fire growing season (Hill and Platt 1975), and (Abrams and Hulbert 1987) in the mountains of western Montana, Kentucky bluegrass frequency was reduced 27.5 percent by a single late May fire in a sagebrush/bunchgrass habitat type (Bushey 1985).

Kentucky bluegrass biomass production and density may be unaffected or increase after burning at other times of the year, such as early spring, summer, or fall. It consistently recovers more quickly from burning at these times of year than from burning in late spring.
In fields dominated by cool-season grasses in Wisconsin, Kentucky bluegrass was reduced to one-fifth of its original density after 6 years of annual burning in May; annual burning in March or October did not affect Kentucky bluegrass density (Curtis and Partch 1948). A different study in Wisconsin showed that flower stalk density was reduced 70 percent by three annual mid-May prescribed fires but was slightly increased by annual burning in late March or early April (Henderson et al. 1983). Although summer grass fires can be relatively intense, Kentucky bluegrass is dormant at this time. It may not be harmed by summer burning, and if precipitation is favorable, it may even increase. In mixed-grass prairie in north-central South Dakota, Kentucky bluegrass frequency increased or remained unchanged on uplands burned in early August followed by a wet spring, but decreased on uplands burned in summer following a dry spring (Steuter 1986a and 1986b).

Kentucky bluegrass’s density tripled 1 year after late October and early November low-intensity prescribed fires in aspen stands in Colorado (Smith et al. 1983). In ponderosa pine habitat types in British Columbia, Kentucky bluegrass biomass was unchanged by an October prescribed fire (Thomson 1988).

Fire frequency: Even after late spring burning, unless burned a second time, Kentucky bluegrass density and cover often return to preburn levels within 1 to 3 years. For example, burning in May or June in Wind Cave National Park, South Dakota, consistently reduced Kentucky bluegrass canopy coverage, height, shoot density, flower stalk density, and biomass during the first post-fire growing season but not during post-fire years 2 and 3 (Olson 1975). In fact, biomass and density were often greater on burned plots than on control plots during post-fire year 2. Other studies in mixed-grass prairie have shown Kentucky bluegrass cover can be reduced for 2 or 3 years by a single late spring fire (Forde 1983, Nagal 1983, Schacht and Stubbendieck 1985).

Kentucky bluegrass cannot withstand frequent spring burning. In the tallgrass prairie, its density decreases with increased fire frequency, and it may be eliminated from sites that are burned annually for several years (Abrams 1988, Anderson et al. 1970, Dokken and Hulbert 1978, Hadley and Kieckhefer 1963, Launchbaugh and Owensby 1978, McMurphy and Anderson 1965). In the Flint Hills of northeastern Kansas, Kentucky bluegrass canopy coverage under different burning regimes was 30.3 percent on an area unburned for 11 years, 7.0 percent on an area burned 1 and 5 years before sampling, and 0 percent on an area burned annually for 5 years (Abrams 1988). A similar response was observed on a reconstructed tallgrass prairie in Illinois subjected to the following burning treatments (Hadley and Kieckhefer 1963):

not burned = unburned for 19 years
burned twice = burned Feb. 28, 1952 and April 16, 1959
burned three times = burned Feb. 28, 1952; April 16, 1959; and May 2, 1961
burned four times = burned Feb. 28, 1952; April 16, 1959; May 2, 1961; and May 10, 1962
Sampling at the end of the 1962 growing season showed the relative percentage of bluegrass (*P. compressa* and *P. pratensis*) shoot biomass decreased with increased burning frequency in two community types as follows:

<table>
<thead>
<tr>
<th>Community type</th>
<th>Burning Treatment*</th>
<th>not burned</th>
<th>burned twice</th>
<th>burned 3 times</th>
<th>burned 4 times</th>
</tr>
</thead>
<tbody>
<tr>
<td>big bluestem</td>
<td>23.4**</td>
<td>18.3</td>
<td>4.6</td>
<td>0</td>
<td></td>
</tr>
<tr>
<td>indiangrass</td>
<td>18.6</td>
<td>15.9</td>
<td>3.3</td>
<td>0</td>
<td></td>
</tr>
</tbody>
</table>

**sampled at the end of the 1962 growing season

Vogl (1971) sampled several pine barrens in northern Wisconsin and reported that Kentucky bluegrass frequency either increased or decreased within 1 year of a single spring fire but that Kentucky bluegrass was eliminated on sites spring burned more than once every few years.

Influence of post-fire moisture: Kentucky bluegrass is more susceptible to fire damage on ridge sites than in depressions, especially in dry years (Higgins et al. 1989b). In fact, in swales and low prairie sites that receive upslope moisture, Kentucky bluegrass often increases after spring burning. In bluegrass fields in Wisconsin, Kentucky bluegrass density and biomass increased in depressions but decreased or remained unchanged on ridgetops after two successive mid-April fires (Zedler and Loucks 1969). In eastern South Dakota, Kentucky bluegrass recovered well from early May burning if irrigated.

On burned but unirrigated plots, however, biomass decreased sharply (Blankespoor and Bich 1991). In eastern North Dakota, lowland and upland prairies were burned on May 8, 1966. Post-fire data on August 4, 1966 showed that Kentucky bluegrass frequency increased on lowlands but remained unchanged on uplands. Biomass on both uplands and lowlands decreased, but the decrease was much greater on uplands (Hadley 1970). When post-fire growing season precipitation was “considerably below normal” in Wind Cave National Park, South Dakota, Kentucky bluegrass biomass on burned areas was less than half that found on unburned areas whether burned on September 18, February 13, or April 10 (Gartner 1975).

In a sagebrush/rough fescue habitat type in Montana, Kentucky bluegrass biomass increased the first summer after a mid-May prescribed fire (Schwecke and Hann 1989). This increase was unexpected because bluegrass should be susceptible to burning at this time. This increase may be due to the high moisture availability in surface soils at this site due to concave slope shape. In contrast, another study in western Montana found Kentucky bluegrass decreased after a prescribed fire on May 24 in a sagebrush/fescue habitat type (Bushey 1985).
DISCUSSION AND QUALIFICATION OF PLANT RESPONSE

In the Mountain West, Kentucky bluegrass is often more abundant in recently burned areas than in nearby unburned areas. Sampling 2- to 36-year-old burns in sagebrush/grassland habitat types in southeastern Idaho, Humphrey (1984) found that Kentucky bluegrass was more abundant in recent than in old burns. McKell (1950) compared four different-aged burns in the Gambel oak (Quercus gambelii) zone of north-central Utah. Kentucky bluegrass cover and density were higher 1 year after a November fire and 2 years after a January fire, but on 9- and 18-year-old burns cover and density were the same as on nearby unburned areas.

In the Klamath Mountains of southern Oregon, Kentucky bluegrass was a codominant grass in open ponderosa pine stands that were burned annually in the spring for 16 years (Weaver 1958).

FIRE MANAGEMENT CONSIDERATIONS

Burning for bluegrass control: Frequent (annual or biennial) late spring burning can be used to control Kentucky bluegrass and promote the growth of warm-season grasses in the Midwest. The timing of burning is critical and should take place just prior to the resumption of warm-season grass growth. Such burning favors warm-season grasses because they are dormant at the time of burning. Conversely, cool-season species like Kentucky bluegrass are harmed by late spring fire because they resume growth in the early spring and are thus actively growing at the time of burning. In mixed-grass prairie, mid-May has proven to be the most effective time to burn for Kentucky bluegrass control and has resulted in concomitant increases in warm-season grasses (Engle and Bultsma 1984, Nagal 1983).

In native bluestem prairie in eastern Kansas, Kentucky bluegrass has been nearly eliminated from sites annually spring burned for decades (Towne and Owensby 1984). In aspen parkland in northwestern Minnesota, 13 years of annual spring burning in late April, when bluegrass was 10-15 cm high, reduced Kentucky bluegrass to about half its original percent composition (Svedarsky et al. 1986). After 10 years of biennial spring burning on the Curtis Prairie on the University of Wisconsin Arboretum, Kentucky bluegrass frequency decreased from 60 to 13 percent (Anderson 1973).

Burning to promote bluegrass growth: When using prescribed fire to promote the growth of cool-season species in the Northern Great Plains, Kentucky bluegrass will probably respond best to very early spring (March-April) or late summer (August-September) fires (Higgins et al. 1989a).

Disease control: In Kentucky bluegrass commercial seed fields, burning after harvest successfully controls several diseases. It is effective in controlling ergot (Claviceps purpurea); silver top, caused by the fungus Fusarium tricinctum; and the mite,Sitopsis cerealium. Burning also helps control leaf rust (Puccinia poae-nemoralis) and other fungi harbored in crop residue (Hardison 1976).
Wildlife considerations: Succulent new grass shoots arising from burned mountain grasslands are highly palatable to wildlife. On the Front Range in Colorado, mule deer and bighorn sheep ate considerably more Kentucky bluegrass on areas burned in late September than on nearby unburned areas (Spowart and Hobbs 1985).

Following late October and early November fires in aspen stands in Colorado, Kentucky bluegrass cover increased and thus provided more forage to wildlife (Smith et al. 1983). Where Kentucky bluegrass is desired for providing ruffed grouse drumming ground cover, it can be burned when the soil is damp and plants are dormant (Wasser 1982).

Burning under aspen: Powell (1988) reported that in south-central Colorado, aspen/Kentucky bluegrass communities have only a moderate probability of carrying a prescribed fire and only if livestock grazing is deferred for at least one season. For fall prescribed burning, the likelihood of a relatively uniform burning treatment may be increased by burning after aspen leaf fall (Smith et al. 1983).

LITERATURE CITED


Poa secunda
Sandberg bluegrass

FIRE ECOLOGY OR ADAPTATIONS

Sandberg bluegrass is generally unharmed by fire. It produces little litter, and its small bunch size and sparse litter reduces the amount of heat transferred to perennating buds in the soil (Kellogg 1985). Its rapid maturation in the spring also reduces fire damage, since it is dormant when most fires occur (Kearney et al. 1960). Sandberg bluegrass cover often increases when interference from other species is reduced by fire (Blackburn et al. 1971).

POST-FIRE REGENERATION STRATEGY

Tussock graminoid.

IMMEDIATE FIRE EFFECT ON PLANT

Sandberg bluegrass is usually unharmed or only slightly damaged by fire (Pechanec et al. 1954, Wright et al. 1979). In a big sagebrush-Thurber needlegrass (Stipa thurberiana) community near Boise, Idaho, Wright and Klemmedson (1965) observed no size reduction of dormant Sandberg bluegrass 2.5-7.6 cm in basal area after either June, July, or August fires. Fire may cause damage if litter has accumulated at the base of the plant, and/or if plants are old and pedestaled (Wright et al. 1979). Large bunches are more susceptible to damage than small ones, probably because of greater litter buildup (Wright and Klemmedson 1965) and/or because the growing points of the elevated plants are no longer insulated by soil (Clifton 1981, Tisdale and Hironaka 1981, Wright et al. 1979). Tisdale (1959) reported some damage to pedestaled Sandberg bluegrass in sagebrush with 7 to 14 percent big sagebrush cover (Range et al. 1982).

Seed mortality and post-fire seedling emergence: Fire effects on the Sandberg bluegrass seedbank are not well documented, but fire may kill some seed in the upper layer of soil. In one study, Sandberg bluegrass seedling emergence was significantly reduced by both “cool” and “hot” prescribed fires. In a burning chamber, used onsite in a mountain big sagebrush community in eastern Oregon, soil surface temperatures reached a maximum of 104º C after 30 seconds with prescribed cool fire and a maximum of 416º C after 60 seconds with hot fire. After the fires, soil samples were collected from the burn sites from two depths (0 - 1 cm and 1 - 2 cm), samples from the two depths were mixed, and the mixed-depth samples were used for greenhouse emergence trials.
Number of emerging Sandberg bluegrass seedlings follows. Means followed by different letter differ at the 5 percent significance level; means followed by an asterisk also differed at the 1 percent significance level (Champlin 1982).

<table>
<thead>
<tr>
<th></th>
<th>Control</th>
<th>Cool Fire</th>
<th>Hot Fire</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>8.5a*</td>
<td>2.8b</td>
<td>0.3b*</td>
</tr>
</tbody>
</table>

PLANT RESPONSE TO FIRE

Sandberg bluegrass generally increases after fire (Daubenmire 1975, Wright and Klemmedson 1965, Wright et al. 1979). Some variability in response has been reported, however. Conditions that may produce variability such as site differences, pre-fire plant condition, and post-fire weather are not well documented.

Variability in fire effects is reported for Sandberg bluegrass on big sagebrush-bunchgrass sites on the Snake River Plain of Idaho. The sites were prescription burned in 1936, protected from grazing for 1 year, then lightly grazed in spring and fall by domestic sheep. At post-fire year 15, Sandberg bluegrass on severely burned plots was producing less than plants on less severely burned plots. At a different location in the same study, there was no difference in Sandberg bluegrass production on plots of different burn severity after 12 years (Blaisdell 1958). After 30 years, all burned plots were producing more Sandberg bluegrass than unburned plots, and the differences in Sandberg bluegrass production attributed to fire severity were negligible. Annual production of Sandberg bluegrass (lb/acre, air-dry) on unburned (UB) and burned (B) plots was as follows (Harniss and Murray 1973):

<table>
<thead>
<tr>
<th>Year</th>
<th>UB</th>
<th>B</th>
<th>UB</th>
<th>B</th>
<th>UB</th>
<th>B</th>
<th>UB</th>
<th>B</th>
<th>UB</th>
<th>B</th>
</tr>
</thead>
<tbody>
<tr>
<td>1916</td>
<td>7</td>
<td>10</td>
<td>12</td>
<td>12</td>
<td>33</td>
<td>39</td>
<td>58</td>
<td>88</td>
<td>8</td>
<td>15</td>
</tr>
</tbody>
</table>

Increases: Fire generally favors production of Sandberg bluegrass and other bluegrasses (Poa spp.) over bluebunch wheatgrass when bluegrasses and bluebunch wheatgrass occur together. Bluegrasses may also compete successfully with cheatgrass as a result of the tillering that occurs following the reduction of litter and improved insolation caused by fire (Daubenmire 1975). But these post-fire gains last only a few years, after which cheatgrass resumes pre-fire dominance.

After a mid-July fire in western Montana, an increase in Sandberg bluegrass cover was noted the first post-fire year. Additionally, the percentage of Sandberg bluegrass plants bearing flowering stalks was 73 percent on burned plots compared to 44 percent on unburned control plots (Mitchell 1957).
Sandberg bluegrass cover increased significantly ($p < 0.05$) on burned plots compared to unburned control plots following September and October (1978) prescribed burning in Wyoming big sagebrush (*Artemisia tridentata* ssp. *wyomingensis*)-bluebunch wheatgrass near Boise, Idaho. In post-fire year 1 (1979), precipitation was below normal in spring and near normal for the rest of the year. In 1980, precipitation was two times above normal. Percent cover of Sandberg bluegrass was (Clifton 1981):

<table>
<thead>
<tr>
<th></th>
<th>Control</th>
<th>Burned</th>
</tr>
</thead>
<tbody>
<tr>
<td>1979</td>
<td>7.03</td>
<td>1.33</td>
</tr>
<tr>
<td>1980</td>
<td>6.69</td>
<td>2.65</td>
</tr>
</tbody>
</table>

Four years after August wildfire in a big sagebrush-bunchgrass community in southeastern Oregon, Sandberg bluegrass and other bunchgrasses dominated burned sites. Big sagebrush and forbs dominated adjacent unburned sites (Acker 1992).

Decline: Sandberg bluegrass cover was less on burned plots relative to unburned plots 2 years after spring or fall prescribed burning in Wind Cave National Park, South Dakota (Bock and Bock n.d.).

**LITERATURE CITED**


Polygonum spp.
Smartweed, knotweed

FIRE ECOLOGY OR ADAPTATIONS

Smartweed can reproduce by seed or sprout from rhizomes following fire (Ahlgren 1960).

POST-FIRE REGENERATION STRATEGY

Survivor species; on-site surviving rhizomes; off-site colonizer; seed carried by animals or water; post-fire years 1 & 2.

IMMEDIATE FIRE EFFECT ON PLANT

Fire top-kills smartweed.

PLANT RESPONSE TO FIRE

Smartweeds usually sprout from seeds following fire. They tend to reproduce more after severe burns than after light burns (Ahlgren 1960, Bushey and Kilgore 1985).

DISCUSSION AND QUALIFICATION OF PLANT RESPONSE

P. cilinode, an annual, has been known to colonize severely burned sites in the forests of northeastern Minnesota (Ahlgren 1960). August wildfires in the pine forests of northern Minnesota resulted in an increase of P. cilinode from zero percent cover on the unburned sites to 36 percent ground cover on the burned sites (Apfelbaum and Haney 1981). P. bistortoides was present following wildfires in krummholz and alpine meadows of the central Rocky Mountains (Billings 1969). P. convolvulus sprouted from seed following a May prescribed burn in shrub communities of central Alberta. However, this species was only prevalent for the first post-fire year (Anderson and Bailey 1979).

Fire was simulated in northern Alberta wetlands to monitor its potential effects on the plant community. Here, P. amphibium was almost equal in percent cover for the three treatments: no burn, “light burn,” and “deep” burn in willow (Salix spp.) savanna zones (Hogenbirk and Wein 1991). (“Deep burns” had the first 5-10 cm of soil removed and the new surface burned with a propane torch for 1 minute; “light burns” had some soil removed and the new surface lightly burned with a propane torch). In the pinegrass (Calamagrostis spp.) zones of these same Alberta wetlands, P. amphibium was found in trace amounts on the unburned and lightly burned sites, and not at all on the “deeply” burned sites.
Prescribed fires on sagebrush (Artemisia spp.)-grass (Poaceae) ranges in northern Idaho showed significant differences in P. douglasii cover between the light to moderately burned and severely burned sites (Blaisdell 1953). Only 7.83 kg/ha of this species was found on unburned sites, while 29 kg/ha were found on severely burned sites. Smaller differences in cover were shown for light and moderately burned areas compared to unburned sites.

FIRE MANAGEMENT CONSIDERATIONS

Where smartweed is a desired waterfowl food fire can be used to stimulate the growth of smartweeds while reducing competition from sedge (Carex spp.), cattail (Typha spp.), and giant reed (Phragmites spp.) (Kantrud 1990).

LITERATURE CITED


Polystichum munitum
Western sword fern

FIRE ECOLOGY OR ADAPTATIONS

Western sword fern has two post-fire regeneration strategies. It sprouts from its stout, woody rhizomes (Halpern 1988). Additionally, a single western sword fern frond may produce millions of light wind-borne spores each year, enabling the species to colonize burn sites (Haeussler and Coates 1986, Page 1979).

Sites where western sword fern is a major understory species are generally resistant to the effects of fire (Barnett 1984 unpub., cited in Hemstrom and Logan 1986).

POST-FIRE REGENERATION STRATEGY

Rhizomatous herb, rhizome in soil; initial-offsite colonizer (off-site, initial community); secondary colonizer - off-site seed.

IMMEDIATE FIRE EFFECT ON PLANT

Fire top-kills western sword fern. It can survive intense fire (Haeussler and Coates 1986, Halpern 1988), but aboveground structures may lacking for several years afterwards (Halpern 1988).

PLANT RESPONSE TO FIRE

Recovery depends upon degree of fire severity. Morris (1970) found that western sword fern was equally abundant on burned and unburned sites following slash burning in Washington and Oregon. A study in the Tillamook Burn of northwestern Oregon, however, found frequency in burned areas was about 2 percent, while frequency was 4 percent in the unburned forest (Neiland 1958). On severely burned sites, western sword fern is greatly reduced and recovers slowly over a period of 15 or more years (Halpern 1988).

FIRE MANAGEMENT CONSIDERATIONS

On highly productive sites where competition from western sword fern delays tree seedling establishment, slash burning and prompt forest regeneration allow tree seedlings to establish before western sword fern recovers (Wright and Bailey 1982). On sites where shrubs such as salmonberry are dominant, clearcutting followed by slashburning, conifer planting, and release treatments may convert the understory to western sword fern (Hemstrom and Logan 1986).
LITERATURE CITED


*Potentilla glandulosa*
Sticky cinquefoil

FIRE ECOLOGY OR ADAPTATIONS

Very little is known about the adaptations of sticky cinquefoil to fire. Fire is infrequent on many of the more mesic alpine sites occupied by sticky cinquefoil but probably occurs at least occasionally on some of the more xeric montane coniferous sites. Some varieties or populations of sticky cinquefoil have short to well-developed rhizomes (Great Plains Flora Association 1986), while others are nonrhizomatous (Kramer 1984), suggesting the possibility of variable responses to fire. Plants with rhizomes could potentially sprout after fire. Nonrhizomatous individuals would probably be killed by fire.

Sticky cinquefoil is a seedbanker in parts of central Idaho and presumably elsewhere (Kramer 1984, Kramer and Johnson 1987). Large numbers of seed which are stored in the soil germinate after mechanical- or fire-induced scarification.
POST-FIRE REGENERATION STRATEGY

Rhizomatous herb, rhizome in soil; ground residual colonizer (on-site, initial community).

IMMEDIATE FIRE EFFECT ON PLANT

Nonrhizomatous populations of shrubby cinquefoil are probably killed by most fires.

PLANT RESPONSE TO FIRE

Vegetative regeneration has not been documented for this species. However, the potential for some individuals, populations, or varieties to sprout after fire is strongly suggested by the presence of “short to well-developed” rhizomes (Great Plains Flora Association 1986). Sticky cinquefoil typically produces numerous seeds (Welsh et al. 1987), and reestablishment through seed stored in seed banks has been well documented (Kramer 1984, Kramer and Johnson 1987). Seed is typically derived from individuals which are reproducing on-site (Kramer 1984). Long-distance dispersal mechanisms have not been reported for this species (Kramer 1984, Strickler and Edgerton 1976). The length of time buried seed remains viable is unknown.

Various forms of mechanical scarification are known to promote germination of sticky cinquefoil seed (Steele and Geier-Hayes 1987). Fire may also stimulate seed to germinate. Seedlings of this early seral, shade-intolerant species often grow rapidly following fire or other types of disturbance.

Recovery time for sticky cinquefoil has not been well studied. After a fire in a Gambel oak (Quercus gambelii) community in northern Utah, the frequency and total number of sticky cinquefoil shoots on burned sites exceeded those on unburned control plots 2, 9, and 18 years after the fire. Total shoot numbers, cover, and frequency were reduced on burned sites 1 year after the burn (McKell 1950). Factors such as fire intensity and severity, site characteristics, and possible ecotypic variation influence how sticky cinquefoil responds to fire.

DISCUSSION AND QUALIFICATION OF PLANT RESPONSE

Frequency, cover, and total numbers of shoots of sticky cinquefoil following fire in a Gambel oak community in northern Utah were as follows (McKell 1950):
<table>
<thead>
<tr>
<th>Frequency (#)</th>
<th>Total Shoots (#)</th>
<th>Total Cover (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>1st Year Burn:</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Burned</td>
<td>1</td>
<td>2</td>
</tr>
<tr>
<td>Unburned</td>
<td>4</td>
<td>9</td>
</tr>
<tr>
<td><strong>2nd Year Burn:</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Burned</td>
<td>2</td>
<td>6</td>
</tr>
<tr>
<td>Unburned</td>
<td>1</td>
<td>1</td>
</tr>
<tr>
<td><strong>9th Year Burn:</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Burned</td>
<td>6</td>
<td>12</td>
</tr>
<tr>
<td>Unburned</td>
<td>2</td>
<td>5</td>
</tr>
<tr>
<td><strong>18th Year Burn:</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Burned</td>
<td>4</td>
<td>14</td>
</tr>
<tr>
<td>Unburned</td>
<td>2</td>
<td>4</td>
</tr>
</tbody>
</table>

(50 burned and unburned plots)

Sticky cinquefoil remained well-represented on many central Idaho burn sites for as long as 14 years after fire (Steele and Geier-Hayes 1987).

**LITERATURE CITED**


Western bracken fern is considered a fire-adapted species throughout the world (Page 1986). It is not only well adapted to fire, it promotes fire by producing a highly flammable layer of dried fronds every fall (Agee and Huff 1987, Frye 1956, Isaac 1940, US Department of Agriculture Forest Service 1937). In the Pacific Northwest western bracken fern fronds grow to 2 m, resulting in several tonnes of flashy fuel per hectare (McCulloch 1942) and western bracken fern adds to the high fuel loads in northern Idaho brushfields (Habeck 1972). Repeated fires favor western bracken fern (Agee and Huff 1987, 127, Isaac 1940, Sharik et al. 1989).

Most sources agree that western bracken fern’s primary fire adaptation is its deeply buried rhizomes which sprout vigorously following fires before most competing vegetation is established (Ahlgren 1974, Chapman and Crow 1981, Page 1986, Skutch 1929, Stickney 1985, Stickney 1986, Taylor 1986). Western bracken fern’s windborne spores may disperse over long distances. Fire removes competition and creates the alkaline soil conditions suitable for its establishment from spores (Page 1986).

POST-FIRE REGENERATION STRATEGY

Survivor species; on-site surviving rhizomes; off-site colonizer; spores carried by wind; post-fire years one and two.

IMMEDIATE FIRE EFFECT ON PLANT

Western bracken fern is a survivor (Stickney 1985, Stickney 1986). The fronds of plants are generally killed by fire, but some rhizomes survive (A.D. Revill Associates 1978, Flinn and Pringle 1983, Flinn and Wein 1977). The rhizomes are sensitive to elevated temperatures. Except in the spring, sprouting is less vigorous when rhizomes are exposed to temperatures of 45º C), and they die when exposed to temperatures above 55º C (Flinn and Pringle 1983). During fires the rhizome system is insulated by mineral soil (Flinn and Pringle 1983, Flinn and Wein 1977). Depth of the main rhizome system is normally between 8 and 30 cm; short rhizomes may be within 3.7 cm of the surface and some rhizomes may be as deep as 1 m (Conway 1949, Evers 1988, Flinn and Pringle 1983, Flinn and Wein 1977, Frye 1956, Gliessman and Muller 1978, Hellum and Zahner 1966).
PLANT RESPONSE TO FIRE

Western bracken fern is well known as a post-fire colonizer in western coniferous forests and eastern pine and oak forests (Boerner 1981, Lyon and Stickney 1976). Fire benefits western bracken fern by removing its competition while it sprouts profusely from surviving rhizomes (Haeussler and Coates 1986, Page 1986, Tiedemann and Klock 1976). New sprouts are more vigorous following fire, and western bracken fern becomes more fertile, producing far more spores than it does in the shade (Page 1982). Sprouting is slower following summer burns than following spring and fall burns (Flinn and Wein 1988).

Western bracken fern spores germinate well on alkaline soils, allowing them to establish in the basic conditions created by fire (Gliessman 1978, Page 1982, Page 1986). In a moist tropical habitat in Costa Rica, western bracken fern gametophyte plants were observed covering the burned surface of bare ground and ash, but no plants were observed on unburned sites (Gliessman 1978). In North America establishment of new plants from spores on recently burned areas appears to be most likely in the moister conditions near either coastline (Hallisey and Wood 1976, Isaac 1940).

DISCUSSION AND QUALIFICATION OF PLANT RESPONSE

All varieties of western bracken fern are well adapted to fire, but there are differences in rhizome growth rates and their response to disturbance (Fletcher and Kirkwood 1979, Page 1976, Page 1986, Tryon 1941). Among the most important North American varieties, *P. a. var. latiusculum* and *P. a. var. pseudocaudatum* are slower growing and considered less weedy (Tryon 1941, Webster and Steeves 1958). This along with factors such as season, fire severity and intensity, and site characteristics may explain some reported differences in response following fire.

*P. a. var. pubescens*: Western bracken fern invades recently logged and burned areas in the Oregon Cascades, sometimes in the first year and sometimes after several years (Halpern 1989, Morris 1970, Steen 1966, Yerkes 1960). Repeated fires or burns that are delayed following logging favor a rapid increase in cover and encroachment of western bracken fern (Garrison and Smith 1974). Along the Pacific coast western bracken fern invades recent burns by windborne spores and also spreads from its buried rhizome (Isaac 1940). After spring fires in northern Idaho, bracken fern production dropped somewhat in the first year and then increased in the second and third years (Leege and Godbolt 1985). Western bracken fern increased following single or multiple broadcast fires in northern Idaho (Mueggler 1965). After logging or fire in Arizona ponderosa pine communities, western bracken fern may cover up to 30 percent of the area for 10 or more years (Campbell et al. 1977, Oswald and Covington 1983, Oswald and Covington 1984).
P. a. var. latiusculum: It is generally agreed that the bracken-grasslands (Curtis 1959) of Wisconsin originated as a result of fires (Vogl 1964). However, following early spring prescribed fires in these areas, western bracken fern’s relative frequency decreased the year after the fire (Vogl 1964). In New York oak woods, Swan (1970) also found a decrease in frequency following spring fires; however, western bracken fern increased in abundance at the same time. He suggested that existing clumps became denser. Studies in Great Lakes area jack pine forests show that western bracken fern sprouts, and its cover and biomass usually remain fairly stable, either decreasing or increasing slightly after burning (Ahlgren 1966, Ahlgren 1970, McRae 1979, Ohmann and Grigal 1977, Ohmann and Grigal 1979). In red and white pine (Pinus resinosa and P. strobus) forests of Ontario, western bracken fern decreased slightly after logging without burning but increased strongly following logging and early summer burning (Sidhu 1973a, Sidhu 1973b). Increased bracken fern following a spring fire in a Pennsylvania scrub oak community was attributable to both spore germination and rhizome sprouts (Hallisey and Wood 1976). In northeastern hardwood stands western bracken fern sprouts rapidly following fire and repeated fires may lead to its domination (Little 1974, Skutch 1929). In oak-pine forests of the Pine Barrens region of New Jersey, western bracken fern thrives following severe fires (Boerner 1981, Matlack and Good 1989). It increases moderately in canopy gaps in these forests following surface fires.

P. a. var. pseudocaudatum: Western bracken fern is well adapted to fires and increases its cover greatly when it is burned repeatedly in longleaf pine and slash pine forests (Komarek 1973). After two successive wintertime prescribed underburns, western bracken fern increased its frequency from 16.7 to 20.6 percent and doubled its biomass in a Florida slash and longleaf pine forest (Moore et al. 1982). Western bracken fern is common following fire in the pocosins of the Southern Coastal Plain (Christensen 1981). Its regrowth following a severe July wildfire in mixed pine (Pinus taeda or P. palustris) and oak (Quercus virginiana and Q. laurifolia) was vigorous, and cover increased each of the first 2 years (Davison and Bratton 1988).

In South Carolina loblolly pine stands that have been repeatedly burned for 20 years, western bracken fern is found only in areas burned during the summer and not on winter-burned areas (Little 1974). In the southeastern United States, prescribed fire has been used extensively since 1960, favoring western bracken fern and allowing it to dominate other understory species, including wiregrass (Aristida stricta) which had been prominent (Taylor 1986).

FIRE MANAGEMENT CONSIDERATIONS

A fire at this time can reduce western bracken fern for up to 2 years (Preest 1975). Although more fronds may be produced, total frond weight and rhizome starch are greatly reduced (Preest and Cranswick 1978). If a prescribed fire at this time is followed with a second treatment, the rhizome system will be further depleted and fewer dormant buds may sprout. Since there are more fronds, a herbicide would have more entry points to the rhizome system (Preest and Cranswick 1978).

Fine fuel loading in areas dominated by western bracken fern can be quite high (Agee and Huff 1987, Isaac 1940, Habeck 1972, McCulloch 1942, US Department of Agriculture Forest Service 1937). Brown and Marsden (1976) have developed a formula to estimate fuel loading using the relationship between fuel loading and the ground cover and height of western bracken fern.

LITERATURE CITED


*Rumex acetosella*
Sheep sorrel, Sour weed

FIRE ECOLOGY OR ADAPTATIONS

Sheep sorrel probably survives fire by sprouting from rhizomes and roots (Keown 1978, Radford *et al.* 1968). It probably regenerates from on-site buried seed.

POST-FIRE REGENERATION STRATEGY

Rhizomatous herb, rhizome in soil; ground residual colonizer (on-site, initial community); secondary colonizer - off-site seed.
IMMEDIATE FIRE EFFECT ON PLANT

Sheep sorrel is probably top-killed by fire.

PLANT RESPONSE TO FIRE

Sheep sorrel probably sprouts from rhizomes following fire and establishes from on-site seed (Del Tredici 1977, Fyles 1989, Granstrom and Schimmel 1993). Several studies describe establishment or increase of sheep sorrel after fire. Very severe fire may kill sheep sorrel.

In New Brunswick a woodlot was clearcut in the fall of 1949 and prescribed burned in April 1951. The number of stems of sheep sorrel per area present in June 1949, 1950, 1951, and 1952 were 0, 0, 18, and 28, respectively (Hall 1955). In New Brunswick understory layers of 11 mixed hardwood stands representing an age sequence of 7 to 57 post-fire years were examined. Sampling occurred in July and August 1973 and 1974.

Sheep sorrel was found in stands 7, 10, 13, 17, and 25 years old. It did not occur in some 7-year-old stands, or in stands 18, 20, 29, and 37 years old (MacLean and Wein 1977).

In Idaho seral brushfields in a grand fir/pachistima habitat type were prescribed burned on May 14, 1975, and a portion was seeded on May 15, 1975. Sheep sorrel was present on the burn-only area, but did not occur on the burn-and-seed site. Frequency (out of 10 possible plots) of sheep sorrel was as follows (Leege and Godbolt 1985):

<table>
<thead>
<tr>
<th>Pre-fire</th>
<th>Post-fire year</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>July 3, 1974</td>
</tr>
<tr>
<td>control</td>
<td>1</td>
</tr>
<tr>
<td>burn only</td>
<td>0</td>
</tr>
<tr>
<td>burn and seed</td>
<td>0</td>
</tr>
</tbody>
</table>

In Idaho a wildfire burned a ponderosa pine (*Pinus ponderosa*) forest and adjacent montane grassland on August 10, 1973 for 43 days. Fourteen sites were examined in June 1974 and June 1976. Percent cover and frequency of sheep sorrel on burned and unburned sites were as follows (Merrill *et al.* 1980):

<table>
<thead>
<tr>
<th></th>
<th>1974</th>
<th>1976</th>
</tr>
</thead>
<tbody>
<tr>
<td>cover</td>
<td>burned +/- 1</td>
<td>burned +/- 3</td>
</tr>
<tr>
<td>frequency</td>
<td>unburned - t</td>
<td>unburned - t</td>
</tr>
<tr>
<td></td>
<td></td>
<td>2 +/- 4</td>
</tr>
<tr>
<td></td>
<td></td>
<td>1 +/- 2</td>
</tr>
</tbody>
</table>

* t = trace
In Washington on the Mount Adams huckleberry (Vaccinium spp.) fields, an experimental area was prescribed burned from October 3-7, 1972. Average understory cover (%) of sheep sorrel from 1972 to 1977 was as follows (Minore et al. 1979):

<table>
<thead>
<tr>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td>unburned, uncut</td>
<td>0.2</td>
<td>0.2</td>
<td>0.3</td>
<td>0.3</td>
<td>0.2</td>
</tr>
<tr>
<td>thin, underburn</td>
<td>0.2</td>
<td>0.6</td>
<td>0.9</td>
<td>1.2</td>
<td>1.5</td>
</tr>
<tr>
<td>clearcut and burn</td>
<td>0.2</td>
<td>0.9</td>
<td>1.9</td>
<td>1.0</td>
<td>1.6</td>
</tr>
</tbody>
</table>


DISCUSSION AND QUALIFICATION OF PLANT RESPONSE

Some research describes no change in cover or frequency in sheep sorrel after fire. In California the effects of a late fall burn on a mountain meadow in Grover Hot Springs State Park were evaluated. Both wet and dry meadow plots were prescribed burned by a low- to moderate-intensity fire in mid-November 1987. Sheep sorrel was found only on dry plots before burning and did not increase following fire (Boyd et al. 1993).

In Connecticut experimental tracts were set up in a little bluestem grassland in 1967. Tract A was prescribed burned annually from 1968-1976, and in 1978, 1980, 1983, and 1985. Tract B was prescribed burned annually from 1968-1975, and in 1978, 1980, 1983, and 1985. Sheep sorrel percent cover and frequency in two burns and 2 control plots on each tract were as follows (Niering and Dreyer 1989):

<table>
<thead>
<tr>
<th></th>
<th>Tract A</th>
<th></th>
<th>Tract B</th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>cover</td>
<td>freq</td>
<td>cover</td>
<td>freq</td>
</tr>
<tr>
<td>Burn</td>
<td>&lt; 1</td>
<td>9</td>
<td>&lt; 1</td>
<td>29</td>
</tr>
<tr>
<td>Control</td>
<td>&lt; 1</td>
<td>22</td>
<td>0</td>
<td>0</td>
</tr>
</tbody>
</table>

LITERATURE CITED


Hall, I.V. 1955. Floristic changes following the cutting and burning of a woodlot for blueberry production. Canadian Journal of Agricultural Science 35:143-152.


Taraxacum officinale
Common dandelion

FIRE ECOLOGY OR ADAPTATIONS

Common dandelion is a component of diverse ecosystems in boreal and temperate regions with variable fire regimes. Common dandelion is primarily adapted to fire through its prolific production of wind-dispersed seed (Toth 1991). Site colonization after fires occurs in many forested areas because of common dandelion's persistent, viable seedbank (Ahlgren 1979).

POST-FIRE REGENERATION STRATEGY

Ground residual colonizer (on-site, initial community); initial-offsite colonizer (off-site, initial community); caudex, growing points in soil.

PLANT RESPONSE TO FIRE

Common dandelion generally establishes during the first or second post-fire year. It usually increases in frequency after fire (Bushey 1985, Diboll 1986, Eichhorn and Watts 1984). One year after a spring burn (May 24, 1983) in Galena Gulch, Montana, common dandelion showed a 50 percent increase in frequency, but by the second year showed only a 47.5 percent increase over the preburn level (Bushey 1985).

Common dandelion increased in frequency following a fire in 1974 in a Scotch pine forest in Scotland, but by post-fire year 4, frequency started to decrease. Maximum frequency occurred at 3 years after fire (Sykes and Horrill 1981). Common dandelion frequency was greater in burned than in unburned oak communities in Utah (Kunzler et al. 1981).

Following a prescribed fire in a Douglas-fir stand in south-central Idaho, common dandelion frequency increased significantly by post-fire year 2. Pre-fire frequency was 8 percent; at post-fire year 1 frequency was 4 percent; and at post-fire year 2 frequency was 24 percent (Lyon 1966).

In the Hedges Mountain area of the Helena National Forest, Montana, a sagebrush/rough fescue habitat type was burned in spring (May) and fall (September). Preburn and postburn community types, as named by the dominant species, were compared. Following the spring burn, bluegrass and common dandelion were the dominant species during both post-fire years 1 and 2. Following the fall burn, the dominant species during post-fire year 1 were bluegrass, mountain brome (Bromus marginatus), and common dandelion. By post-fire year 2, common dandelion was no longer a dominant; the site was dominated by bluegrass, Wood’s rose, and common snowberry (Schwecke and Hann 1989).
A fire on June 28, 1977 in Montana in a rough fescue community minimally disrupted reproduction and carbohydrate production of common dandelion. Its frequency increased slightly on burned sites by the summer of 1978 (Antos et al. 1983).

In the timbered breaks along the Missouri River in central Montana, common dandelion was favored by big game animals every post-fire year except year 28. At post-fire year 17 common dandelion was found at high frequencies. First peak in frequency occurred at post-fire year 4 (Eichhorn and Watts 1984).

FIRE MANAGEMENT CONSIDERATIONS

Late spring burning in the tallgrass prairies of Kansas reduced common dandelion cover compared with burning at earlier dates. In shortgrass prairies of western Kansas, common dandelion was less affected by dormant season (fall and winter) burns than by spring burns (Bragg 1991). Burning to decrease cover of common dandelion on rangelands should be done in the spring after growth initiation. Annual burning in March or November in Nebraska resulted in the highest total cover of common dandelion. Burning in April decreased cover (Gibson 1989).

Following logging, bulldozing, and slash burning, common dandelion will establish in the open spots (Bedunah et al. 1988).

Common dandelion competes with tree seedlings on burned sites. Grasses aerially seeded on burns may compete with and displace common dandelion. After 4 to 5 years of grass seeding on sites in Montana common dandelion populations eventually decreased (Bedunah et al. 1988).

LITERATURE CITED


Information regarding white clover survival following fire is lacking in the literature. White clover is probably a decreaser following fire since most of its growing parts are above ground and fire would quickly defoliate these aboveground parts (Anderson 1972). White clover probably regenerates following fire via soil-stored seed. It may also sprout from the taproot and/or caudex (Johnson 1987, Livingston and Allessio 1968).

POST-FIRE REGENERATION STRATEGY

Ground residual colonizer (on-site, initial community); surface rhizome/chamaephytic root crown; caudex, growing points in soil.

IMMEDIATE FIRE EFFECT ON PLANT

The stolons of white clover are killed by fire. If fire occurs in a young population, where taproots are still vigorous, plants may resprout, although probably with reduced vigor.

DISCUSSION AND QUALIFICATION OF FIRE EFFECT

Where fire enhances grass cover, the increase competition may reduce cover of white clover further.

PLANT RESPONSE TO FIRE

Very little information about how white clover responds to fire was given in the literature. Following mid-May prescribed burning of a Wisconsin oldfield being reclaimed to bluestem (Andropogon spp.) prairie, white clover frequency was 4 percent. Frequency was 1 percent on control and 6 percent on mowed plots (Diboll 1986). Johnson (1987)] reported that white clover seeds germinated on both burned
and unburned plots in central Iowa. After white clover was planted on the Sleeping Child Burn in western Montana, it was present in post-fire year 3 but was not present in successional years (Lyon 1976).

LITERATURE CITED


_Vicia americana_
American vetch

FIRE ECOLOGY OR ADAPTATIONS

American vetch is rated as moderately resistant to fire (McKell 1950). It typically increases following fire (Olson 1975). The fibrous roots and rhizomes are 1.5 to 5 cm below the soil surface and sprout following light- to moderate-severity fires (McKell 1950). American vetch also revegetates burned sites via soil-stored seed (Ahlgren 1966, Ahlgren 1979)

POST-FIRE REGENERATION STRATEGY

Rhizomatous herb, rhizome in soil; ground residual colonizer (on-site, initial community).
IMMEDIATE FIRE EFFECT ON PLANT

Fire probably top-kills American vetch (Quintilo et al. 1991, Young 1986).

PLANT RESPONSE TO FIRE

American vetch typically increases after low- to moderate-severity fires (Anderson and Bailey 1979, Armour et al. 1984, Wright and Bailey 1982). In a study of plant succession in the Gambel oak (Quercus gambelii) brush zone in Utah, American vetch showed a higher average number of plants on burned areas than on unburned areas, even after 9 years (McKell 1950). In northeastern North Dakota American vetch canopy cover was greater on some sites burned 1-3 years before the plant survey than on unburned sites (Olson 1975). In a Douglas-fir habitat type in Idaho, American vetch cover and frequency on sites burned by low-severity fires were greater than on unburned or severely burned sites. This effect was greatest in post-fire year 2 (Armour et al. 1984).

On a prescribed burn in northeastern Minnesota, the frequency of American vetch increased greatly on the burned areas during post-fire year 1 (Ahlgren 1966).

LITERATURE CITED


Vulpia microstachys
Small fescue

FIRE ECOLOGY OR ADAPTATIONS

There is little specific information on the adaptations of small fescue to fire; other annual fescues generally increase in abundance after fire (Cave and Patten 1984). Small fescue reoccupies a site through seed. In many areas, annual fescues mature early and drop their seeds before most wildfires occur (Young et al. 1972). Seeds are apparently undamaged by fire when buried in the soil, and late season fires probably have very little effect on small fescue.

Recovery is rapid where viable annual fescue seeds remain buried in the soil, with an abundance of seedlings growing to maturity during the first year after the burn (Cline et al. 1977). Small fescue generally increases in response to heavy grazing and other types of disturbance (Daubenmire 1970, Franklin and Dyrness 1973), and it is likely that fire creates an environment favorable to the germination and growth of this species.

POST-FIRE REGENERATION STRATEGY

Ground residual colonizer (on-site, initial community).

IMMEDIATE FIRE EFFECT ON PLANT

Little is known about the specific effects of fire on small fescue. The dry foliage of annual grasses is typically consumed and the plant killed by fire.
PLANT RESPONSE TO FIRE

Reestablishment of a site is through seed. Seeds buried in the soil can probably survive most fires. Where seeds do survive fire, recovery is rapid, with numerous seedlings germinating during the year after the burn. The relationship between fire intensity and severity, and seed survivorship has apparently not been examined. Potential differences in plant response according to season of burn are also unknown.

LITERATURE CITED


Vulpia myuros
Foxtail fescue, Rattail fescue

FIRE ECOLOGY OR ADAPTATIONS

Reestablishment of a burned site is through seed, which is produced in abundance even in relatively poor years (Sampson et al. 1951). Foxtail fescue cures early and drops seed prior to the onset of the main fire season in many areas (Young et al. 1972). Seed buried in the soil or litter is capable of surviving most fires. Fire generally creates conditions that are favorable for the germination and growth of foxtail fescue.
POST-FIRE REGENERATION STRATEGY

Ground residual colonizer (on-site, initial community).

IMMEDIATE FIRE EFFECT ON PLANT

Little is known about the specific effects of fire on foxtail fescue. The dry foliage is typically consumed by fire; seeds usually remain undamaged in the soil or litter. Late season fires presumably have very little effect on this species.

PLANT RESPONSE TO FIRE

Foxtail fescue generally remains unchanged or increases in response to fire (Hironaka et al. 1977). This species typically becomes abundant on dry, disturbed sites such as those created by fire (Cronquist et al. 1977, Hironaka et al. 1977). It is a common constituent of many recently burned chaparral communities in California (Sampson et al. 1951).

Limited research indicates that season of burn may significantly influence the fire response of this species. Wildfires often occur after the seeds of annual grasses such as foxtail fescue have dropped to the ground (Young et al. 1972). Seed already buried in the litter or soil is usually undamaged by fire. Late season fires probably have little effect on foxtail fescue, while creating an environment favorable for seedling germination and growth. When undamaged seed is present on a site, recovery is generally rapid and presumably occurs by the following growing season. Little information exists on the specific response to various fire intensities and severities.

FIRE MANAGEMENT CONSIDERATIONS

Foxtail fescue has replaced original bunchgrass vegetation as a result of overgrazing and fire in foothills big sagebrush (Artemisia tridentata ssp. vaseyana var. xericensis)- bluebunch wheatgrass (Pseudoroegneria spicata) habitat types in southern Idaho (Hironaka et al. 1977).

LITERATURE CITED


Zygadenus venenosus
Meadow deathcamas

FIRE ECOLOGY OR ADAPTATIONS

Meadow deathcamas survives fires by regeneration from deep underground bulbs (Volland and Dell 1981).

POST-FIRE REGENERATION STRATEGY

Geophyte, growing points deep in soil.

IMMEDIATE FIRE EFFECT ON PLANT

Fire top-kills meadow deathcamas (Humphrey and Weaver 1915).

PLANT RESPONSE TO FIRE

Fire research on this species is limited. It is known, however, that geophytic lilies such as meadow deathcamas stop producing new leaves once the flower stalk is formed in early spring. Meadow deathcamas, therefore, cannot produce new growth at post-fire year 1 except when fire occurs in very early spring, before the flower stalk has developed. Experiments performed on a closely related species, foothill deathcamas (Zigadenus paniculatus), showed that plants were unable to produce a second crop of leaves following defoliation (Tepedino 1982).

Meadow deathcamas will emerge again at post-fire year 2, but flowering may be delayed until post-fire year 3. Zigadenus spp. seedling survival is enhanced following fire due to reduced competition (Wright and Bailey 1982). Peak population density is reached 2 to 5 years following fire, and then declines preburn density (Kozlowski and Ahlgren 1974).
FIRE MANAGEMENT CONSIDERATIONS

Repeated annual burning from mid-spring to mid-summer greatly reduces or eliminates meadow deathcamas populations. Carbohydrates stored in the bulb from the previous year are metabolized in early spring to produce new leaves. Because yearly burning destroys the photosynthetic surface responsible for new carbohydrate production, the plant may have insufficient carbohydrate stores for next year’s growth. The result is plant death (Tepedino 1982). Fire at other seasons probably has little effect on meadow deathcamas.

LITERATURE CITED


