

Fire effects on selected shrubs, trees, amphibians,
birds, and mammals 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 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
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*Fuchs, Marilyn A. 2001. Towards a Recovery Strategy for Garry Oak and Associated Ecosystems in Canada: Ecological Assessment and Literature Review. Technical Report GBEI/EC-00-030. Environment Canada, Canadian Wildlife Service, Pacific and Yukon Region. xi + 106 p.

Shrub Layer

<i>Acer circinatum</i>	1
<i>Amelanchier alnifolia</i>	3
<i>Arctostaphylos columbiana</i>	11
<i>Betula occidentalis</i>	13
<i>Cornus stolonifera</i> = <i>Cornus sericea</i>	14
<i>Corylus cornuta</i> var. <i>california</i>	17
<i>Crataegus douglasii</i>	18
<i>Gaultheria shallon</i>	19
<i>Holodiscus discolor</i> = <i>Holodiscus dumosus</i>	25
<i>Juniperus communis</i>	26
<i>Juniperus scopulorum</i>	30
<i>Ligustrum vulgare</i>	38
<i>Mahonia nervosa</i> = <i>Berberis nervosa</i>	43
<i>Pachystima myrsinites</i>	47
<i>Philadelphus lewisii</i>	49
<i>Prunus virginiana</i>	52
<i>Rosa gymnocarpa</i>	59
<i>Rosa nutkana</i>	61
<i>Rubus laciniatus</i>	62
<i>Rubus parviflorus</i>	64
<i>Rubus ursinus</i>	71
<i>Salix scouleriana</i>	76
<i>Sambucus caerulea</i> = <i>S. nigra</i> ssp. <i>cerulea</i>	92
<i>Sambucus racemosa</i>	95
<i>Shepherdia canadensis</i>	98
<i>Sorbus sitchensis</i>	100
<i>Spiraea douglasii</i>	101
<i>Symphoricarpos albus</i>	103
<i>Symphoricarpos mollis</i>	108
<i>Taxus brevifolia</i>	109
<i>Toxicodendron diversilobum</i>	112
<i>Vaccinium ovatum</i>	116

Tree Layer

<i>Abies grandis</i>	119
<i>Acer glabrum</i>	133
<i>Acer macrophyllum</i>	146
<i>Arbutus menziesii</i>	148
<i>Betula papyrifera</i>	152
<i>Cornus nuttallii</i>	158
<i>Picea sitchensis</i>	165
<i>Pinus contorta</i> var. <i>contorta</i>	167
<i>Pinus ponderosa</i>	169
<i>Populus balsamifera</i> ssp. <i>trichocarpa</i>	174
<i>Populus tremuloides</i>	182
<i>Prunus emarginata</i>	196

<i>Pseudotsuga menziesii</i> var. <i>menziesii</i>	203
<i>Quercus garryana</i>	208
<i>Rhamnus purshiana</i>	211
<i>Thuja plicata</i>	213
Amphibians	
<i>Ambystoma macrodactylum</i>	216
<i>Bufo boreas</i>	217
Birds	
<i>Accipiter striatus</i>	220
<i>Anas platyrhynchos</i>	221
<i>Buteo jamaicensis</i>	223
<i>Falco sparverius</i>	226
<i>Haliaeetus leucocephalus</i>	229
<i>Molothrus ater</i>	231
<i>Pandion haliaetus</i>	233
<i>Phasianus colchicus</i>	233
<i>Sialia currucoides</i>	236
<i>Sialia mexicana</i>	240
<i>Zenaida macroura</i>	242
Mammals	
<i>Canis lupus</i>	244
<i>Castor canadensis</i>	245
<i>Martes americana</i>	246
<i>Mustela vison</i>	247
<i>Odocoileus hemionus</i>	248
<i>Peromyscus maniculatus</i>	252
<i>Procyon lotor</i>	262
<i>Sylvilagus floridanus</i>	264
<i>Tamiasciurus hudsonicus</i>	268
<i>Ursus americanus</i>	270

Shrub Layer

Acer circinatum
Vine maple

FIRE ECOLOGY OR ADAPTATIONS

Vine maple is well adapted to fire. Following aerial crown kill or destruction by fire, root crowns often produce numerous sprouts (Hubbard 1950, Russel 1974, Volland and Dell 1981). This long-lived, seral species often persists in the understory of late seral or climax coniferous stands. Its sprouting ability allows it to become part of the immediate post-fire community when the conifer overstorey is removed or killed (Hubbard 1950, Russel 1974).

POSTFIRE REGENERATION STRATEGY

Survivor species; on-site surviving root crown or caudex.

IMMEDIATE FIRE EFFECT ON PLANT

Most fires top-kill vine maple; however, plants normally resprout from the root crown (Hubbard 1950, Halpern 1989, Russel 1974, Volland and Dell 1981). Successional trends in the western Cascades of Oregon show that vine maple cover and frequency are dramatically reduced following fire (Dyrness 1973, Halpern 1989). The amount of this initial reduction may be related to fire severity. A study of broadcast-burned clearcuts in western Oregon found that vine maple was abundant on lightly burned plots (surface litter charred but not completely removed) but very scarce on heavily burned plots (surface litter completely consumed by intense fire) (Dyrness 1973, Halpern 1988). Similarly, observation of fire effects on Rocky Mountain maple (*Acer glabrum*) plants in Montana show that hot fires (those which effectively transfer heat below the mineral soil surface) damage root crowns and thus prevent sprouting of some plants (Russel 1974).

PLANT RESPONSE TO FIRE

Vine maple produces numerous root crown sprouts the first growing season following burning (Miller and Miller 1976, Russel 1974, Volland and Dell 1981). Frequency and cover of vine maple drop dramatically following fire. Pre-burn levels may be reached as quickly as 2 to 5 years after fire (Dyrness 1973, Kovalchik *et al.* 1988) but may take up to 25 years (Halpern 1989, Russel 1974).

Following wildfire or logging, vine maple/sword fern is a common seral community during the tall shrub stage of succession (Hubbard 1950, Franklin and Dyrness 1973).

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Amelanchier alnifolia
Saskatoon serviceberry

FIRE ECOLOGY OR ADAPTATIONS

Fire adaptations: Saskatoon serviceberry sprouts from the root crown and/or rhizomes after fire (Arno *et al.* 1986, Bradley 1984, Hemmer 1975, Stickney 1991). Bradley (1984) concluded that because Saskatoon serviceberry sprouts from existing plants, fire is not likely to alter its frequency unless plants were in poor condition before fire. After light- to moderate-severity fire, it usually sprouts from the root crown or from shallowly-buried rhizomes (Bradley 1984). However, deeply-buried rhizomes enable Saskatoon serviceberry to sprout even after the most intense wildfire. For example, the Sundance Fire on the Kaniksu National Forest of northern Idaho was an intense, running crown fire that reached firestorm proportions in the Pack River Valley. (The fire broke out on Aug. 23, 1967.) Saskatoon serviceberry sprouts were a principle component of Pack River Valley vegetation at post-fire year 1, with 12 percent frequency and 4 percent cover (Stickney 1986).

Seedling establishment is apparently not an important post-fire regeneration strategy. After wildfire in trembling aspen-paper birch in northern Saskatchewan, a single Saskatoon serviceberry seedling was found at post-fire year 2 on one of seven plots (Archibold 1980). Leege (1978) found an occasional Saskatoon serviceberry seedling after prescribed burning on the Clearwater National Forest of northern Idaho, but the seedlings survived for only a few post-fire years. Stickney (1986) found that on 21 plots on the Sundance Burn, 100 percent of Saskatoon serviceberry regeneration resulted from sprouting of burned plants.

Fire ecology: Forests - Saskatoon serviceberry in forests is fire-dependent and declines with fire exclusion (Arno and Ottmar 1994, Gruell 1983). It may persist in the understorey for decades, but eventually dies out with canopy closure. Through time-series photographs, Gruell (1983) has documented decline of Saskatoon serviceberry in ponderosa pine habitat types in the Northern Rocky Mountains due to canopy closure with fire exclusion.

Sagebrush (*Artemisia* spp.) - In sagebrush steppe in southeastern Idaho, Saskatoon serviceberry was prominent on burn sites of all ages. Field sampling was conducted on 2- to 36-year-old burns (Humphrey 1984).

Fire frequency: Forests - Saskatoon serviceberry occurs in forests with fire regimes varying from frequent, low-severity fire to infrequent, severe fire. In low-elevation forests, where Saskatoon serviceberry is most common, the historical regime was frequent, low-severity fire (Agee 1996, Agee *et al.* 1990, Arno and Ottmar 1994). Wright (1978) compiled historical fire frequencies of ponderosa pine communities in which Saskatoon serviceberry occurs:

State(s)	Fire Frequency
Arizona and New Mexico	4.8 - 11.9 yrs
California and eastern Oregon	8 - 10 yrs
Colorado and Wyoming	12 - 25 yrs
western Montana	2 - 48 yrs
South Dakota (Black Hills)	15 - 20 yrs
eastern Washington	6 - 47 yrs

On the west slope of the Cascade Range of Washington, mean historical fire return intervals in forests with Saskatoon serviceberry were (Agee *et al.* 1990):

ponderosa pine-Douglas-fir	52 years
lodgepole pine-Douglas-fir	76 years
Douglas-fir-grand fir	93 years

POSTFIRE REGENERATION STRATEGY

Tall shrub, adventitious-bud root crown; geophyte, growing points deep in soil.

IMMEDIATE FIRE EFFECT ON PLANT

Saskatoon serviceberry is top-killed by moderate to severe fire. Larger branches may survive light-severity fire (Bradley 1984, Noste *et al.* 1989, Stickney 1991).

PLANT RESPONSE TO FIRE

Saskatoon serviceberry sprouts after top-kill by fire (Arno *et al.* 1986, Bradley 1984, Stickney 1991). Bradley (1984) found that on burn sites in western Montana, Saskatoon serviceberry sprouted mostly from upper portions of the root crown.

When the root crown was killed by fire, Saskatoon serviceberry sprouted from rhizomes further beneath the soil surface. Seed production may resume soon after fire: Saskatoon serviceberry sprouts produced fruits the second summer after a July 1977 wildfire in Pattee Canyon near Missoula, Montana (Keller 1980).

Saskatoon serviceberry cover usually increases (Arno *et al.* 1986) or is unaffected (Arno *et al.* 1986, Thomson 1988) by fire. Even when there is little change between pre- and post-fire cover, fire usually makes Saskatoon serviceberry more accessible as wildlife browse by lowering shrub height (Thomson 1988). Arno and others (1986) found that in western Montana, Saskatoon serviceberry cover generally increased after wildland or prescribed fires in Douglas-fir/ninebark habitat types. It sometimes took 10 or more years before the increase occurred, however. The authors suggested that slow recovery in some areas may be due to big game browsing pressure after fire.

Current-year annual twig production is usually greater after fire in the absence of heavy browsing pressure (Arno *et al.* 1986, Cook *et al.* 1994). In a mountain brush community in Wyoming, Saskatoon serviceberry mortality was 12 percent, 15 percent, and 15 percent, 1, 2, and 3 years after fall wildfire, respectively. Mortality after spring prescribed burning on a nearby site was 1, 2, and 2 percent at post-fire years 1, 2, and 3. Post-fire browsing pressure was not heavy, but wildfire- and prescription-burned areas were browsed more than unburned areas. Despite this, current-year twig production was significantly greater on burned sites than on unburned sites in post-fire years 1 to 3. Current-year annual twig production was greater on the wildfire-burned site than on the spring prescribed-burned site (37 vs. 15 g/plant) (Cook *et al.* 1994).

Fire season: In a western Montana study contrasting the ability of spring vs. fall prescribed fire to improve wildlife habitat, severe fall fire killed 15 percent of Saskatoon serviceberry plants on the site, while a less severe spring treatment killed only 5 percent. Sprouting response in the first two post-fire years was greater on the spring burn (Noste 1982).

Fire in various habitat/plant community types: In a western redcedar (*Thuja plicata*)/ninebark habitat type of central Idaho, Saskatoon serviceberry sprouted from the root crown and grew rapidly after prescribed burning. Height growth of sprouts follows (Asherin 1975). (Pre-fire height not available.)

	Height (m)	
	Avery Site	Lochsa Site
post-fire year 1	0.9	1.2
post-fire year 2	1.5	1.3
post-fire year 3	1.2	3.0
unburned control	2.3	3.2

In Douglas-fir/blue huckleberry (*Vaccinium membranaceum*) habitat types of western Montana, prescribed fire had little effect on Saskatoon serviceberry cover (Asherin 1975).

Near Ketchum, Idaho, a prescribed fire was conducted on August 1, 1963, to reduce dwarf-mistletoe (*Arceuthobium douglasii*) infestation in Douglas-fir and to promote sprouting of browse, which was above the browseline. The fire was successful in both respects. Saskatoon serviceberry recovered from the fire as follows (Lyon 1966, Lyon 1971):

	Plants*/1,000 sq ft	Percent Canopy Cover
pre-fire	0.2	0.25
post-fire yr 1	0.1	0.03
post-fire yr 2	0.1	0.05
post-fire yr 3	0.2	0.06
post-fire yr 4	0.1	0.06
post-fire yr 5	0.1	0.09
post-fire yr 6	0.3	0.12
post-fire yr 7	0.2	0.12

*only plants over 18 inches in height were included in density measurements

After prescribed fire in Garry oak (*Quercus garryana*) woodlands in western Washington, Saskatoon serviceberry sprouts were most common on sites that were treated with low-severity fire and had no pre-fire mechanical disturbance. Saskatoon serviceberry sprouts usually co-occurred with Garry oak sprouts on such sites. Neither Saskatoon serviceberry sprouts, Saskatoon serviceberry seedlings, nor Garry oak sprouts occurred on microsites that were heavily disturbed before fire. After prescribed fire, those microsites were colonized by herbs, especially exotic herbs, and Garry oak seedlings (Agee 1996).

Saskatoon serviceberry appears to be slow to recover from prescribed burning in the sub-boreal spruce-fir (*Picea-Abies* spp.) zone in British Columbia (Hamilton 1988).

Response to very frequent fire: Saskatoon serviceberry response to repeated burning is unclear. In a trembling aspen-rough fescue (*Festuca scabrella*) ecotone in Alberta, Saskatoon serviceberry was one of the few woody shrubs that was not harmed by low-severity annual spring prescribed fire. Frequency was 8 percent on unburned sites and 16 percent on annually burned sites. Canopy cover was not significantly different between the two areas (4 and 1.4 percent, respectively) (Anderson and Bailey 1980).

In the Willamette Valley of Oregon, Kalapuyan Indians apparently controlled Saskatoon serviceberry with frequent fire in order to promote acorn production by Garry oak. Open oak savannas were noted by early travellers, but in the absence of aboriginal burning, Saskatoon serviceberry has formed a closed subcanopy in Garry oak woodlands (Boyd 1986).

FIRE MANAGEMENT CONSIDERATIONS

Saskatoon serviceberry is most vigorous in seral plant communities (Arno *et al.* 1986, Hemmer 1975, Hickman 1993), and prescribed fire can be used to maintain and/or promote seral communities. On big game rangelands, prescribed fire can improve condition of Saskatoon serviceberry and other shrubs by reducing shrub height, promoting growth of new twigs, and increasing nutritional content of browse (Arno *et al.* 1986, Lyon 1966, Miller *et al.* 1988). Sites where prescribed burning may harm

Saskatoon serviceberry in the long term include harsh (especially very dry) sites with low Saskatoon serviceberry density (Hemmer 1975), and very cold sites where post-fire growth would be limited by temperature (Hamilton 1988).

Fire stimulates production of Saskatoon serviceberry by killing understory conifers, removing old Saskatoon serviceberry topgrowth, and promoting sprouting (Arno *et al.* 1986, Miller *et al.* 1988). On Douglas-fir/ninebark winter Elk range on the Lolo National Forest, Montana, Makela (1990) found that after spring prescribed fire, biomass production of new Saskatoon serviceberry twigs was significantly greater ($p < 0.1$) on burned sites than on unburned sites the first two growing seasons after fire.

Ponderosa pine: Saskatoon serviceberry usually occurs in the moister, cooler ponderosa pine habitat types. Average loading of downed and dead woody fuels is slightly higher than in drier ponderosa pine types. Fire hazard is further increased by the tendency of this type to form subcanopies and dog-hair thickets of conifer saplings. Wildfire hazard is particularly high in this type during drought. Common management objectives are to eliminate large areas of overstocking and create a two-storied stand rather than a multilayered one. Periodic prescribed surface fire in early spring or late fall is recommended. Fuels management includes treatment of slash following logging and thinning, and controlling stocking levels. Scattered thickets of Saskatoon serviceberry and other shrubs can be left for wildlife (Fischer and Clayton 1983).

Trembling aspen: Light fuels and grazing can inhibit fire spread in trembling aspen. Brown and Simmerman (1986) assigned probabilities of successful prescribed burning in trembling aspen/Saskatoon serviceberry habitat types as follows:

Grazing	Woody Fuel	Fuel Type Aspen/ serviceberry	Mixed aspen-conifer/ serviceberry
ungrazed	light	high	high
ungrazed	heavy	high	high
grazed	light	moderate	moderate
grazed	heavy	high	high

FIRE MANAGEMENT IMPLICATIONS

Neither spring nor fall prescribed burning had much short-term effect on Saskatoon serviceberry cover or production in one study. This is consistent with other short-term studies in southeastern British Columbia. Davidson (1983) found a slight decrease in Saskatoon serviceberry two years after prescribed burning. Demarchi and Lofts (1985) found production of current-annual twigs was greater, but leaf production less, the first two years after prescribed burning.

This is not to say that the prescribed fires were unsuccessful. Shrubs were reduced an average of 39 percent by spring prescribed burning, making Saskatoon serviceberry and other browse more accessible to ungulates. Comparing percent cover in short-term studies can be deceiving because unburned areas may contain a few tall, inaccessible shrubs while burned areas contain a number of small shoots. Data from the long term may show that the number of sprouts on burned areas exceeds sprouts on unburned areas.

Thomson (1988) suggested that total grass biomass prior to burning is important in determining success of burning on sparsely-treed Douglas-fir habitat types in the extreme northern Rocky Mountains. Moisture content of grass and other herbaceous fuels was not reported in this study, but is also an important factor in success of burning.

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Arctostaphylos columbiana
Hairy manzanita

FIRE ECOLOGY OR ADAPTATIONS

Fire activates hairy manzanita seed by scarifying the seedcoat. Oligosaccharides in charred wood leachate may further enhance rates of post-fire germination (Adams 1940, Kruckeberg 1977).

POSTFIRE REGENERATION STRATEGY

Shrub without adventitious-bud root crown; ground residual colonizer (on-site, initial community); initial-offsite colonizer (off-site, initial community); and secondary colonizer - off-site seed.

IMMEDIATE FIRE EFFECT ON PLANT

Fire kills hairy manzanita (Adams 1940, Gratkowski 1978, Jepson 1925, Yerkes 1960).

PLANT RESPONSE TO FIRE

Current information concerning hairy manzanita's response to fire is limited to studies of its regeneration following clearcutting and slash burning in Douglas-fir (*Pseudotsuga menziesii*) forest types. It was reported as establishing from seed after such forest management near Blue River, Oregon (Yerkes 1960). Further details were not available. Schoonmaker and McKee (1988) reported post-fire hairy manzanita seedling recruitment on the H.J. Andrews Experimental Forest, Oregon. Sites were broadcast burned and planted with Douglas-fir seedlings. No hairy manzanita seedlings were found on transects at post-fire year 2, but mean transect cover value of hairy manzanita seedlings was 0.04 percent at post-fire year 5. Mean transect cover value was greatest (0.15%) at post-fire year 15.

A comparison of hairy manzanita cover on burned versus adjacent unburned plots was made near Oakridge, Oregon. Fire treatment was broadcast burning of slash. At post-fire year 9, hairy manzanita cover was 3.2 percent on burned plots and 0.3 percent on unburned plots (Steen 1966).

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Betula occidentalis
Water birch

FIRE ECOLOGY OR ADAPTATIONS

Water birch is generally restricted to streamside communities where fires occur infrequently (Crane 1982). When fires do occur, water birch's above-ground plant parts are easily killed by fire; however, plants normally resprout from basal buds (Hansen *et al.* 1989). Wind-transported seed from off-site plants may also be important in establishing new individuals following fire (Hansen *et al.* 1988).

POSTFIRE REGENERATION STRATEGY

Survivor species; on-site surviving root crown or caudex; off-site colonizer; seed carried by wind; post-fire years 1 and 2.

IMMEDIATE FIRE EFFECT ON PLANT

Above-ground plant parts of water birch are easily destroyed by fire because its bark is both thin and flammable (Hansen *et al.* 1989). However, plants will normally resprout from uninjured basal buds (Hansen *et al.* 1988, Hansen *et al.* 1989).

PLANT RESPONSE TO FIRE

Water birch often sprouts from uninjured basal buds following fire (Hansen *et al.* 1988, Hansen *et al.* 1989). Fires may expose mineral soils which are favourable seedbeds for this species' light, wind-dispersed seed (Hansen *et al.* 1988).

FIRE MANAGEMENT CONSIDERATIONS

Water birch is normally restricted to riparian areas which may act as natural fire breaks (Crane 1982).

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Cornus stolonifera = *Cornus sericea*
Red-osier dogwood

IMMEDIATE FIRE EFFECT ON PLANT

Above-ground foliage of red-osier dogwood is usually killed by fire. However, the roots will survive all but the most severe fires which remove the duff and heat the upper soil for extended periods (Fischer and Bradley 1987).

PLANT RESPONSE TO FIRE

Red-osier dogwood generally increases following fire (Aksamit and Irving 1984, Chrosciewicz 1976, Fischer and Clayton 1983, Perala 1974), and it may invade a recently burned area from adjacent unburned areas (Fischer and Bradley 1987). It may take some time before resprouting red-osier dogwood regains its former cover and volume. A Montana study in aspen found that two years after prescribed burning red-osier dogwood resprouts had attained 72 percent of their pre-fire cover and 54 percent of their pre-fire volume, while density was back to pre-fire levels (Gordon 1976).

In moist forests of British Columbia, red-osier dogwood appears to increase in abundance following logging and burning (Eis 1981, Hamilton 1988). It established in a logged and burned Manitoba black spruce (*Picea mariana*) stand by the fifth post-fire year (Chrosciewicz 1976). On Minnesota black spruce sites most fires stimulate sprouting of red-osier dogwood, although severe fires favour tree seedlings (Aksamit and Irving 1984). A study in the cedar-hemlock (*Thuja* spp.-*Tsuga* spp.) zone of northern Idaho found no red-osier dogwood in closed stands. It established with very slight frequency (1%) in logged stands without fire; somewhat higher frequency (5%) in areas with both single and multiple broadcast burns; and highest frequency (12%) in areas that were piled and burned (Mueggler 1965). In a northwestern Montana subalpine fir/queencup beadlily (*Abies lasiocarpa/Clintonia uniflora*) habitat type, red-osier dogwood cover was highest (15%) in stands that had burned 35 to 70 years ago and very low in clearcuts (0-1.4%) whether dozer piled or not (Zager 1980, Zager *et al.* 1983).

In Wisconsin shrub-carrs (wet ground tall-shrub communities) light to medium fires cause resprouting in red-osier dogwood and serve to maintain the shrub-carr (White 1965). Following prescribed burning in central Wisconsin shrub-invaded sedge meadows, red-osier dogwood resprouted and was favoured over other shrubs (Warners 1987).

FIRE MANAGEMENT CONSIDERATIONS

In the northern Rockies red-osier dogwood is a common member of the seral brushfields which occur following fire and compete with tree seedlings (Orme and Leege 1976, Pojar *et al.* 1984). In order to reduce brushfields on sites that are away from streams and floodplains, the use of logging methods which cause a relatively high amount of site disturbance followed by a fire which removes most of the soil organic horizons is recommended (Pojar *et al.* 1984). In northwestern Montana, clearcutting alone apparently dislodged enough roots of red-osier dogwood so that few plants were able to sprout (Zager 1980, Zager *et al.* 1983).

Post-fire sprouts in the early stages of growth are the most valuable for Beaver (Patric and Webb 1953, Wright and Bailey 1982). Following fire in Minnesota, red-osier dogwood became more important to Moose and White-tailed Deer, even though it grew only near streams and was not abundant within the burn (Irwin 1985).

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Corylus cornuta var. californica
California hazelnut

FIRE ECOLOGY OR ADAPTATIONS

California hazelnut sprouts from the root crown after the above-ground portion of the plant has been destroyed (USDA Forest Service 1937).

POSTFIRE REGENERATION STRATEGY

Tall shrub, adventitious-bud root crown; initial-offsite colonizer (off-site, initial community).

IMMEDIATE FIRE EFFECT ON PLANT

The above-ground parts of California hazelnut are easily killed by fire, even by light spring fires (Buckman 1964, Haeussler and Coates 1986).

Because of their moisture requirements, seeds probably do not survive the high temperatures and drying associated with fire.

PLANT RESPONSE TO FIRE

California hazelnut responds to spring fires by vigorous and abundant sprouting; sprouting after summer fires is less vigorous (Buckman 1964). Pre-fire stature and density are regained in only a few years (Haeussler and Coates 1986).

FIRE MANAGEMENT CONSIDERATIONS

Fire would destroy California hazelnut only under the most extreme fuel and weather conditions. Repeated spring burning conducted when the soil is moist simply increases the number of stems present (Haeussler and Coates 1986). However, repeated summer fires destroy hazel's ability to sprout by exposing and destroying underground root systems and exhausting stored food reserves (Buckman 1964).

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Crataegus douglasii Douglas hawthorn

FIRE ECOLOGY OR ADAPTATIONS

Douglas hawthorn is fire tolerant (Daubenmire 1970). This tree has a shallow and diffuse root structure that allows for sprouting and sucker-rooting following the destruction of above-ground parts (Hansen *et al.* 1988).

POSTFIRE REGENERATION STRATEGY

Survivor species; on-site surviving root crown or caudex; survivor species; on site surviving deep underground stems; off-site colonizer; seed carried by animals or water; post-fire year 1 and 2.

IMMEDIATE FIRE EFFECT ON PLANT

Both high- and low-severity fires will consume the above-ground parts of Douglas hawthorn.

DISCUSSION AND QUALIFICATION OF FIRE EFFECT

The structural configuration of Douglas hawthorn limbs makes it highly flammable due to the sheltering of dry grasses and twigs. These fuels may create a "ladder" for fire to be carried up to the crown, destroying the entire thicket.

PLANT RESPONSE TO FIRE

The range of Douglas hawthorn is limited by fire. Removal of the plant may require years of growth for full reestablishment. Frequent fires may confine Douglas hawthorn plants to dense thickets (Daubenmire 1970).

DISCUSSION AND QUALIFICATION OF PLANT RESPONSE

Daubenmire (1970) hypothesized that the expanded range of Douglas hawthorn stands in eastern Washington was the result of improved agricultural cropping practices which exclude stubble burning. Douglas hawthorn thickets have redeveloped from stump sprouts as the number and size of fires have decreased (Daubenmire 1970, Mack 1988).

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Gaultheria shallon Salal

FIRE ECOLOGY OR ADAPTATIONS

The shade-tolerant salal appears well able to persist under a regime of relatively infrequent fires. Long fire-free intervals are common in any climax coastal coniferous forests of the Pacific Northwest (Hemstrom *et al.* 1982). Fire occurs infrequently in most coastal western hemlock forests due to marine climatic influences (Atzet and Wheeler 1982). Western hemlock-Douglas-fir forests codominated by salal and dwarf Oregon grape commonly burn at approximately 320-year intervals (Ossinger 1983). Fire intervals in tanoak-salal/dwarf Oregon grape communities of the western Siskiyou have been estimated at 60 years (Atzet and Wheeler 1982). While inland redwood forests burn every 26 to 52 years, coastal redwood forests experience fires at 50 to 500-year intervals (Stuart 1987). In western Oregon, Douglas-fir/oceanspray-salal communities are common on sites which have been lightly burned during the past 200 years. Salal, because of its prolific sprouting ability, can also survive shorter fire-free intervals. In western Oregon, bracken fern-salal communities commonly develop on frequently burned sites (Bailey 1966).

Salal generally sprouts from the roots, rhizomes, or stem base after above-ground vegetation is damaged or consumed by fire. Birds and mammals may disperse some seed from off-site. Limited reestablishment through seed may occur, although vegetative regeneration is apparently the dominant mode of reestablishment (Halpern 1989).

POSTFIRE REGENERATION STRATEGY

Tall shrub, adventitious-bud root crown; rhizomatous shrub, rhizome in soil.

IMMEDIATE FIRE EFFECT ON PLANT

Salal is described as a woody survivor (Yerkes 1960). Underground portions of the plant commonly survive even when above-ground vegetation is consumed by fire (Bunnell and McCann 1990, Hawkes *et al.* 1990). Portions of the stem base also survive many low severity fires (Hawkes *et al.* 1990). Hot burns on dry, shallow soil can result in lethal heat penetration to underground regenerative structures (Haeussler and Coates 1986). "Moderate damage" has been reported after light burns (Atzet and Wheeler 1982).

PLANT RESPONSE TO FIRE

Vegetative response: Salal typically sprouts readily from the roots, rhizomes, or stem base after light to moderate fires (Atzet and Wheeler 1982, Bunnell and McCann 1990, Klinka *et al.* 1989). Fires of light to moderate intensity stimulate sprouting, but more intense fires can damage underground regenerative structure and reduce or eliminate sprouting (Haeussler and Coates 1986).

Seed: Post-fire reestablishment through seed appears to be relatively unimportant in salal (Halpern 1989).

Post-fire recovery: Recovery of salal varies according to fire intensity and severity (Halpern and Franklin 1989). Rhizome expansion can be rapid (Weetman *et al.* 1989) or relatively slow depending on the amount of damage received (Hawkes *et al.* 1990, Yerkes 1960). Plants are often observed soon after fire (Hooven 1969, Kienholz 1929, Lafferty 1972) but may only develop slightly during the first year (Hooven 1969). Following a moderate burn in British Columbia, salal was present during the first growing season and increased in abundance by the third growing season (Lafferty 1972). However, few plants were observed during the first growing season after an intense fire in the same area (Lafferty 1972). By the third year after this fire, only small, scattered colonies of salal were present (Lafferty 1972). Bailey (1966) observed increases in cover by the eighth year after logging and fire in western Oregon. Salal can become dominant within 10 years after fire in parts of British Columbia (Nuszdorfer 1989). Salal can reach 0.6-0.9 m in height by the tenth growing season after fire (Ruth 1957). Recovery was documented as follows after logging and fire in the Oregon Coast Range (Stewart 1978):

	before burn		1 year after burn	
	orig. stems	seedlings (# per acre)	orig. stems + sprouts	seedlings
N aspect	250	0	640	0
S aspect	2,840	0	15,960	0

Following fire in British Columbia, cover reached 18 percent after two years and had increased to 55 percent with 8 years (McDonald 1990). However, four years after intense summer wildfires in the North Cascades of Washington, cover of salal on two sites ranged from 0.7 to 1 percent (Miller and Miller 1976).

FIRE MANAGEMENT CONSIDERATIONS

Timber harvest: Evidence indicates that post-fire recovery of salal on some harvested sites may be delayed by slashburning (Miller *et al.* 1974, Vihnanek and Ballard 1988). Recovery may be particularly slow after hot slash burns on dry sites with shallow soil (Haeussler and Coates 1986). Fire can thus be used to control salal on dry sites but is often ineffective on wet sites (Hawkes *et al.* 1990). Slash burning in Douglas-fir plantations of eastern Vancouver Island reduced the height and cover of salal while improving the nutrient status of Douglas-fir (Vihnanek and Ballard 1988). Often the moderate fires that reduce salal produce a positive response in conifer seedlings. Where slash burning is contemplated, plots should be burned immediately after timber harvest for best results. Because slash burns delay but do not eliminate salal, it is important that sites are planted within two years after logging and fire (Bunnell and McCann 1990).

While slash burns often aid conifer regeneration, in some locations salal cover is not significantly reduced and competition remains a considerable problem. Factors such as site characteristics, community composition, and fire intensity and severity are all important influences. In old growth Douglas-fir forests of the western Cascades, salal may triple in cover during the first five years after logging and slash burning as shown below (Dyrness 1965):

	1962 before logging	1963 1st yr. after logging	1964 1st yr. after slash burn	1965	1966	1967	1968
% cover	5.9	1.1	0.5	1.3	1.6	2.2	3.0
% freq.	20.2	5.8	4.0	5.8	6.4	7.7	9.5

Response of salal after timber harvest and subsequent slash burns has been examined by a number of researchers (Dyrness 1965, Dyrness 1973, Ingram 1931, Isaac 1940, Morris 1970, Steen 1966, Stewart 1978, Vihnanek and Ballard 1988, Yerkes 1960).

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Holodiscus discolor = *Holodiscus dumosus*
Bush oceanspray

FIRE ECOLOGY OR ADAPTATIONS

The fire ecology and fire adaptations of bush oceanspray are poorly documented. Fire is reported to be the principal factor initiating succession on many of the high-elevation sites occupied by this shrub in the Southwest (Hanks 1966), and bush oceanspray is apparently able to maintain itself into late successional stage on many of these rocky, forested sites (Alexander *et al.* 1984).

It is currently unknown whether post-fire reestablishment is from sprouting on surviving plants, or from on-site or off-site seed sources, or both. In the White Mountains of New Mexico, bush oceanspray was a conspicuous component of the shrub layer 2 to 3 years after fire (Hanks and Dick-Peddie 1974).

POSTFIRE REGENERATION STRATEGY

Initial off-site colonizer (off-site, initial community); small shrub, adventitious-bud root crown; ground residual colonizer (on-site, initial community).

IMMEDIATE FIRE EFFECT ON PLANT

The effect of fire on bush oceanspray is poorly documented. The intricately branched nature of this shrub suggests that most plants are readily top-killed by relatively low-intensity fires.

PLANT RESPONSE TO FIRE

Fire response of bush oceanspray is poorly documented. A similar species, oceanspray (*H. discolor*), increases by prolific sprouting.

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Juniperus communis
Common juniper

FIRE ECOLOGY OR ADAPTATIONS

Common juniper is generally described as “susceptible” to fire (Crane 1982, Hoffman and Alexander 1987). It is described as being “not equipped with fire-surviving regeneration properties” (Mallik 1995). Foliage is resinous and very flammable (Diotte and Bergeron 1989). The degree of damage received increases with progressively greater fire severity. In eastern Canada, older common juniper often survives fires of low severity. Some fire regimes allow common juniper to survive several fires (Diotte and Bergeron 1989).

Where common juniper is killed by fire, some seeds may survive in the soil on-site and germinate when conditions become favourable. Other seed is brought to the site by bird (Crane 1982) or, less commonly, mammal dispersers. These factors contribute to slow post-fire reestablishment on many sites.

For information on fire regimes in forest and woodland communities where common juniper occurs, see the FEIS species summaries on dominant tree species including:

Species	Fire return interval
interior ponderosa pine (<i>P. ponderosa</i> var. <i>scopulorum</i>)	2 - 45 years
Rocky Mountain juniper (<i>J. scopulorum</i>)	
lodgepole pine (<i>P. contorta</i>)	25 - 300 years
trembling aspen (<i>Populus tremuloides</i>)	
big sagebrush (<i>Artemisia tridentata</i>)	

POSTFIRE REGENERATION STRATEGY

Tree without adventitious bud/root crown; shrub without adventitious bud/root crown; secondary colonizer - off-site seed.

IMMEDIATE FIRE EFFECT ON PLANT

Common juniper is generally killed or seriously damaged by fire (Crane and Fischer 1986, Crane 1982). Patchy fires may allow individual plants to survive in protected areas such as on rocky cliffs. More rarely, portions of a lightly-burned plant may survive. The amount of damage this species incurs increases with increasing fire severity (Stark and Steele 1977).

In the boreal forest of Quebec, at least 37% of common junipers survived fire. Although the dominant fire regime here is crown fires or "important surface fires covering large areas," common juniper often survives on sites made up of exposed bedrock or where protected by lakes and island complexes. Survival can occur if fire affects only part of an area or where fires are of low intensity. In some cases, fires of low intensity "can allow sections of the plant to survive and reproduce vegetatively" (Diotte and Bergeron 1989).

DISCUSSION AND QUALIFICATION OF FIRE EFFECT

In a Scottish study, common juniper was killed by 800° C heat treatment when heath was burned. Plants made only "feeble regrowth" when burned at 600° C. However, following treatment at 400° C, new shoots were produced (Mallik and Gimingham 1983).

PLANT RESPONSE TO FIRE

Common juniper does not sprout after disturbance. Surviving individuals serve as sources of seed for adjacent areas. Post-fire regeneration is more frequent in proximity to existing populations of common junipers (Diotte and Bergeron 1989). Regrowth can generally take place after fire if some of the basal branches remain alive (Mallik and Gimingham 1983), which only occurs in fires of low severity or where spread is patchy.

Common juniper also reestablishes after fire through off-site seed dispersed by birds or mammals. Poor seed dispersal from existing stands along with low germination rates can explain why some favourable sites are not readily occupied by common juniper (Diotte and Bergeron 1989).

It is possible that seed protected by overlying soil can survive at least some fires. After low-severity fires, some seed may germinate. However, Mallik and Gimingham (1983) observed that high temperatures did not increase germination in common juniper seed and little seed germinated after fire.

DISCUSSION AND QUALIFICATION OF PLANT RESPONSE

Following "light" (less than 50% litter reduction) fires in forested areas of western Montana, burned sites often exhibit minimal shrub damage and have at least some surviving common juniper (Stark and Steele 1977). Common juniper averaged 6.7% cover three years after a light burn in Montana (Stark and Steele 1977). Laboratory heating experiments on common juniper plants from Scottish heath showed that growth could take place after heating only if some of the basal green branches remain alive (Mallik and Gimingham 1983):

Effect of temperature* on vegetative regrowth after heating

	400° C	600° C	800° C
mean # sprouts per plant 3 months after treatment	5	4	0
height of veg. regrowth (cm) 17 months after treatment	8	6	0
oven-dry biomass per plant of veg. regrowth (grams) 17 months after treatment	3	0.8	0

* temperature maintained for about 2 minutes

Most fires kill common juniper (Crane 1982), leading to the slow post-fire recovery typical of this species. In northern Canada, common juniper is generally absent from burned areas, but may grow in small refugia within burned areas (Kelsall 1957).

Post-fire recovery of common juniper is generally slow. The following table gives the density and frequency of occurrence for common juniper in stands of different ages in two Colorado forest types (Clagg 1975):

Stand age after fire	Spruce -fir density	Freq.	Stand age after fire	Lodgepole pine density	Freq.
1	---	---	1	---	---
2	---	---	2	---	---
8	---	---	8	---	---
8	0.2	20	8	---	---
18	0.2	20	18	0.2	20
74	2.7	70	18	0.8	20
200	0.2	8	18	0.4	20
280	0.6	40	45	0.8	40
290	0.4	20	85	2.0	100
			85	0.8	60
			108	0.6	60
			115	0.4	40
			190	1.4	60
			248	0.8	20
			251	1.3	30
			257	3.4	85

FIRE MANAGEMENT CONSIDERATIONS

In a north-central Colorado study of fire behaviour in trembling aspen stands, common juniper patches burned more intensely and released more heat than adjacent herbaceous areas. A caloric analysis of five foliage samples yielded an average low heat content of 5064 kcal/kg. Common juniper fuels tended to be deeper and heavier than herbaceous fuels and flames were longer and deeper in common juniper patches. Fire removed almost all litter, standing herbs, and common juniper foliage, leaving only bare branches. The moisture of green common juniper foliage averaged 112% of oven dry weight on two burns (Smith *et al.* 1993). Fuel loading for common juniper can be estimated as follows:

$$\text{branch load (kg/m}^2\text{)} = 0.000191 * \text{crown height above duff (cm)} ** 2.135$$

$$\text{foliage load (kg)} = 6.456 * \text{crown volume (m}^3\text{)} ** 1.93$$

In xeric red pine (*Pinus resinosa*) communities of northern Canada, an understory of low sweet blueberry (*Vaccinium angustifolium*), lichen, and common juniper creates a low and discontinuous fuel load. Fires in these communities tend to have an irregular pattern of intensity that is largely dependent on the distribution of fine fuels. Intense crown fires are unlikely here (Bergeron and Brisson 1990).

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Juniperus scopulorum
Rocky Mountain juniper

FIRE ECOLOGY OR ADAPTATIONS

Fire adaptations: Due to its thin bark and compact crown, Rocky Mountain juniper trees up to 0.9-1.2 m tall are easily killed by fire. Since the species grows slowly, trees are especially susceptible to fire for their first 20 years or more (Crane 1982, Fischer and Clayton 1983, Hansen and Hoffman 1988, Mitchell 1984, Mueggler 1976, Stanton 1974). Large Rocky Mountain junipers, however, have survived at least 4 - 6 fires (Crane 1982, Fischer and Clayton 1983, Mitchell 1984). As trees mature, they develop thicker bark and a more open crown, allowing them to survive surface fires if the low, spreading branches do not carry fire to the crown (Crane 1982, Fischer and Clayton 1983, Hansen and Hoffman 1988, Mitchell 1984). A severe fire, however, may damage or kill such trees (Fischer and Clayton 1983). High volatile oil content, especially in the lower branches, also makes the trees more flammable (Noble 1990, Sieg 1997).

Fire regimes: Fire return intervals vary for habitats where Rocky Mountain juniper occurs. For example, in pinyon-juniper habitat (including Rocky Mountain juniper) of the Sacramento Mountains of New Mexico, the mean fire interval was 28 years with a range of 10-49 years, and fires that covered more than 10 hectares occurred at 15 - 20-year intervals in other areas of New Mexico. Research in the Walnut Canyon National Monument in Arizona reported surface fire intervals of 20-30 years for pinyon-juniper habitat where Rocky Mountain juniper occurs (Paysen *et al.* 2000).

A fire history study in Mesa Verde National Park estimated the historic interval between stand-replacing fires for pinyon-juniper habitat, where Rocky Mountain juniper was a dominant, at approximately 400 years, and large fires may have not occurred for more than 600 years in some areas. In contrast, fire intervals for chaparral communities in the park were estimated at 100 years. It appears that, in this area, pinyon-juniper habitat that was burned severely was replaced by chaparral species, which are more fire tolerant. As a result, pinyon-juniper habitat is found mostly in the southern part of the park, where cliffs and sparsely vegetated slopes form a barrier to fire. Though this habitat type may support heavy fuel loads, horizontal fuel continuity remains low, so crown fires are usually confined to relatively small areas unless high winds and extreme drought occur (Floyd *et al.* 2000).

Little information is available regarding fire regimes specific to Rocky Mountain juniper. However, fire regimes for plant communities and ecosystems in which Rocky Mountain juniper occurs are summarized below.

Community or Ecosystem	Dominant Species	Fire Return Interval Range (years)
sagebrush steppe	<i>Artemisia tridentata/Pseudoroegneria spicata</i>	20-70 [1]
basin big sagebrush	<i>Artemisia tridentata var. tridentata</i>	12-43 [2]
mountain big sagebrush	<i>Artemisia tridentata var. vaseyana</i>	15-40 [3,4,5]
Wyoming big sagebrush	<i>Artemisia tridentata var. wyomingensis</i>	10-70 (40**) [6,7]
saltbush-greasewood	<i>Atriplex confertifolia-Sarcobatus vermiculatus</i>	< 35 to < 100
desert grasslands	<i>Bouteloua eriopoda</i> and/or <i>Pleuraphis mutica</i>	5-100
plains grasslands	<i>Bouteloua</i> spp.	< 35 [1]
curlleaf mountain-mahogany*	<i>Cercocarpus ledifolius</i>	13-1000 [8,9]
mountain-mahogany-Gambel oak scrub	<i>Cercocarpus ledifolius-Quercus gambelii</i>	< 35 to < 100
western juniper	<i>Juniperus occidentalis</i>	20-70
Rocky Mountain juniper	<i>Juniperus scopulorum</i>	< 35 [1]
western larch	<i>Larix occidentalis</i>	25-100
blue spruce*	<i>Picea pungens</i>	35-200 [10]

pinyon-juniper	<i>Pinus-Juniperus</i> spp.	< 35 [1]
whitebark pine*	<i>Pinus albicaulis</i>	50-200 [10]
Rocky Mountain lodgepole pine*	<i>Pinus contorta</i> var. <i>latifolia</i>	25-300+ [10,11,12]
Colorado pinyon	<i>Pinus edulis</i>	10-49 [1]
interior ponderosa pine*	<i>Pinus ponderosa</i> var. <i>scopulorum</i>	2-30 [10,13,14]
aspen-birch	<i>Populus tremuloides-Betula papyrifera</i>	35-200 [15,16]
mountain grasslands	<i>Pseudoroegneria spicata</i>	3-40 (10**) [11,10]
Rocky Mountain Douglas-fir*	<i>Pseudotsuga menziesii</i> var. <i>glauca</i>	25-100 [10,3,17]
oak-juniper woodland (Southwest)	<i>Quercus-Juniperus</i> spp.	< 35 to < 200 [1]
Garry oak	<i>Quercus garryana</i>	< 35 [10]
bur oak	<i>Quercus macrocarpa</i>	< 10 [16]
oak savanna	<i>Quercus macrocarpa/Andropogon gerardii-Schizachyrium scoparium</i>	2-14 [1,16]
elm-ash-cottonwood	<i>Ulmus-Fraxinus-Populus</i> spp.	< 35 to 200 [15,16]

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

**mean

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POSTFIRE REGENERATION STRATEGY (Stickney 1989)

Tree without adventitious bud/root crown, shrub without adventitious bud/root crown; initial off-site colonizer (off-site, initial community); secondary colonizer (on-site or off-site seed sources).

IMMEDIATE FIRE EFFECT ON PLANT

Rocky Mountain juniper is readily killed by ground fires or by low intensity surface burns (Arno and Hammerly 1977, Floyd *et al.* 2000). Larger trees may survive low intensity fires due to their thicker bark and more open crown.

DISCUSSION AND QUALIFICATION OF FIRE EFFECT

The effects of fire on Rocky Mountain juniper vary according to site characteristics. The species often occurs in dry, subhumid areas with sparse undergrowth; in these habitats where surface fuels are limited, fire damage is often minor (Fischer and Clayton 1983).

PLANT RESPONSE TO FIRE

Rocky Mountain juniper does not resprout after top-kill (Hansen and Hoffman 1988, Sieg 1997, Stanton 1974, Wright 1972). Post-fire re-establishment is solely by seed (Floyd *et al.* 2000), and animal transport of seeds is an important factor (Paysen *et al.* 2000). Numerous seedlings often germinate after burning of old trees (Stanton 1974, Wright 1972).

Fire is a major factor controlling the distribution of Rocky Mountain juniper (Stanton 1974, Wright 1972). Reduced fire frequency, along with climate change and introduction of grazing, accounts for the expansion of juniper woodlands into meadows, grasslands, sagebrush communities, and aspen groves that began in the late 1800s. Prior to this time, more frequent fires probably maintained low density in woodlands and often restricted junipers to rocky sites (Miller and Wigand 1994). In general, the species grows in areas that do not burn frequently or intensely.

Frequent fires in the pinyon-juniper type can maintain a grassland setting, and the absence of fire will allow conversion to woodlands (Gruell 1986). Wildfire eliminated Rocky Mountain juniper for 28 years in the Missouri, Judith, and Musselshell river breaks of central Montana (Eichhorn and Watts 1984). In many areas where Rocky Mountain juniper grows, lack of heavy fuels may limit fire activity to surface fires of low intensity, allowing the species to persist (Roberts and Sibbersen 1979). It is often found in ponderosa pine forests where fire has been absent for long periods (Oswald *et al.* 2000, Roberts and Sibbersen 1979), and the resurgence of Rocky Mountain juniper in Idaho grasslands is due to fire cessation (Kucera 1981). Severe fires in Douglas fir-Rocky Mountain juniper habitats in Montana appear limited to local areas where fire is carried into the crowns of widely-spaced trees (Roberts and Sibbersen 1979).

After fire in pinyon-juniper habitat, junipers will usually invade the area first, followed by pinyon, which may eventually replace juniper on higher sites (Holland 1990). The following stages have been outlined for post-fire succession in southwestern Colorado climax pinyon-juniper forest (including Rocky Mountain juniper): 1) skeleton forest and bare soil, 2) annual stage, 3) perennial grass-forb stage, 4) shrub stage, 5) shrub-open tree stage, 6) climax pinyon-juniper forest (Evans 1988, Paysen *et al.* 2000). It takes approximately 300 years to reach climax (Paysen *et al.* 2000).

Post-fire succession in western Utah juniper woodland (including Rocky Mountain juniper) takes approximately 85-90 years: 1) skeleton forest and bare soil, 2) annual stage, 3) perennial grass-forb stage, 4) perennial grass-forb-shrub stage, 5) perennial grass-forb-shrub-young juniper stage, 6) shrub-juniper stage, 7) juniper woodland (Evans 1988, Paysen *et al.* 2000).

FIRE MANAGEMENT CONSIDERATIONS

Fire has long been recognized as a control mechanism for juniper (Fischer and Clayton 1983). In the 1950s and 1960s some pinyon-juniper elimination operations were conducted by mechanical methods and slash was piled and burned. Some areas where these large fuel loads were burned remained free of vegetation 20 years later (Paysen *et al.* 2000).

In areas where Rocky Mountain juniper is not desirable, young trees have been killed mechanically by scorching the crown and stems (Fischer and Clayton 1983). Tree-by-tree burning and wildfire both control Rocky Mountain juniper effectively in juniper and sagebrush-grass types in Wyoming (Fisser 1981). In central Oregon, one juniper control technique is to conduct prescribed fires several years after harvesting trees, when herbaceous vegetation will be present to provide fuel to carry fire to juniper seedlings (Paysen *et al.* 2000). In general, control of Rocky Mountain juniper by fire has been more effective in the southern part of its range (Noble 1990).

Thinning undergrowth in pinyon-juniper woodlands favours Rocky Mountain juniper by reducing the number and intensity of fires and reducing competition for moisture (Mueggler 1976).

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Ligustrum vulgare
Chinese privet

FIRE ECOLOGY OR ADAPTATIONS

Fire adaptations: Chinese privet survives fire by sprouting from the root crown in response to damage of above-ground tissue (Faulkner *et al.* 1989, Louisiana State University 2001). It is likely, though speculative, that privets generally respond to fire damage by sprouting from the root crown, and/or by root suckering (see the Fire Effects section of this summary). More research is needed on the fire ecology of privets in North America.

Fire regimes: Chinese privet was present in a longleaf pine (*Pinus palustris*) forest in southern Alabama, prior to and following three annual prescribed burns where fire had previously been excluded for > 45 years. No further information is available about Chinese privet at this particular site, but we may presume from this report that it has some ability to persist (at least in the short term) in frequent, low-severity fire regimes characteristic of longleaf pine ecosystems in the Southeast (Varner *et al.* 2000).

As of this writing (2003), there are no other accounts in the literature of interactions between privets and specific fire regimes.

The following table lists fire return intervals for communities or ecosystems throughout North America where privet may occur. Amur privet has not been included in this list because information about North American distribution and occurrence is lacking for this species. This list is presented as a guideline to illustrate historic fire regimes and is not to be interpreted as a strict description of fire regimes for privets.

Privet spp.*	Community or Ecosystem	Dominant Species	Fire Return Interval Range (years)
CE	maple-beech-birch	<i>Acer-Fagus-Betula</i>	> 1000
E	silver maple-American elm	<i>A. saccharinum-Ulmus americana</i>	< 35 to 200
C E	sugar maple	<i>A. saccharum</i>	> 1000
E	sugar maple-basswood	<i>A. saccharum-Tilia americana</i>	> 1000
JCE	Atlantic white-cedar	<i>Chamaecyparis thyoides</i>	35 to > 200
CE	beech-sugar maple	<i>Fagus spp.-Acer saccharum</i>	> 1000 [1]

CE	cedar glades	<i>Juniperus virginiana</i>	3-7 [2]
JCE	yellow-poplar	<i>Liriodendron tulipifera</i>	< 35 [1]
C	Everglades	<i>Mariscus jamaicensis</i>	< 10
C	melaleuca	<i>Melaleuca quinquenervia</i>	< 35 to 200 [3]
E	northeastern spruce-fir	<i>Picea-Abies</i> spp.	35-200
E	black spruce	<i>Picea mariana</i>	35-200
E	conifer bog**	<i>Picea mariana-Larix laricina</i>	35-200
E	red spruce**	<i>P. rubens</i>	35-200
E	jack pine	<i>Pinus banksiana</i>	<35 to 200 [4]
CE	shortleaf pine	<i>P. echinata</i>	2-15
CE	shortleaf pine-oak	<i>P. echinata-Quercus</i> spp.	< 10
JC	slash pine	<i>P. elliotii</i>	3-8
JC	slash pine-hardwood	<i>P. elliotii</i> -variable	< 35 [1]
C	South Florida slash pine	<i>P. elliotii</i> var. <i>densa</i>	1-5 [3, 1]
JC	longleaf-slash pine	<i>P. palustris-P. elliotii</i>	1-4 [3, 1]
JC	longleaf pine-scrub oak	<i>P. palustris-Quercus</i> spp.	6-10 [1]
E	red pine (Great Lakes region)	<i>P. resinosa</i>	10-200 (10***) [4, 5]
E	red-white-jack pine**	<i>P. resinosa-P. strobus-P. banksiana</i>	10-300 [4, 6]
CE	pitch pine	<i>P. rigida</i>	6-25 [7, 8]
JC	pocosin	<i>P. serotina</i>	3-8
CE	eastern white pine	<i>P. strobus</i>	35-200
CE	eastern white pine-eastern hemlock	<i>P. strobus-Tsuga canadensis</i>	35-200
E	eastern white pine-northern red oak-red maple	<i>P. strobus-Q. rubra-Acer rubrum</i>	35-200
JCE	loblolly pine	<i>P. taeda</i>	3-8
JCE	loblolly-shortleaf pine	<i>P. taeda-P. echinata</i>	10 to < 35
CE	Virginia pine	<i>P. virginiana</i>	10 to < 35
CE	Virginia pine-oak	<i>P. virginiana-Quercus</i> spp.	10 to < 35
JCE	sycamore-sweetgum-American elm	<i>Platanus occidentalis-Liquidambar styraciflua-Ulmus americana</i>	< 35 to 200 [1]
JCE	eastern cottonwood	<i>Populus deltoides</i>	< 35 to 200 [2]
E	aspen-birch	<i>P. tremuloides-Betula papyrifera</i>	35-200 [4, 1]
C	mesquite	<i>Prosopis glandulosa</i>	< 35 to < 100 [9, 2]
CE	black cherry-sugar maple	<i>Prunus serotina-Acer saccharum</i>	> 1000
CE	oak-hickory	<i>Quercus-Carya</i> spp.	< 35
E	northeastern oak-pine	<i>Quercus-Pinus</i> spp.	10 to < 35 [1]
JCE	oak-gum-cypress	<i>Quercus-Nyssa</i> -spp.- <i>Taxodium distichum</i>	35 to > 200 [3]

CE	southeastern oak-pine	<i>Quercus-Pinus</i> spp.	< 10
E	white oak-black oak-northern red oak	<i>Q. alba-Q. velutina-Q. rubra</i>	< 35
E	northern pin oak	<i>Q. ellipsoidalis</i>	< 35
CE	bear oak	<i>Q. ilicifolia</i>	< 35
E	bur oak	<i>Q. macrocarpa</i>	< 10 [1]
E	oak savanna	<i>Q. macrocarpa/Andropogon gerardii-Schizachyrium scoparium</i>	2-14 [2, 1]
CE	chestnut oak	<i>Q. prinus</i>	3-8
CE	northern red oak	<i>Q. rubra</i>	10 to < 35
JCE	post oak-blackjack oak	<i>Q. stellata-Q. marilandica</i>	< 10
CE	black oak	<i>Q. velutina</i>	< 35
JC	live oak	<i>Q. virginiana</i>	10 to < 100 [1]
JC	cabbage palmetto-slash pine	<i>Sabal palmetto-Pinus elliotii</i>	< 10 [3, 1]
C	southern cordgrass prairie	<i>Spartina alterniflora</i>	1-3 [2]
CE	eastern hemlock-yellow birch	<i>Tsuga canadensis-Betula alleghaniensis</i>	> 200 [1]
JCE	elm-ash-cottonwood	<i>Ulmus-Fraxinus-Populus</i> spp.	< 35 to 200 [4, 1]

* J = Japanese privet, C = Chinese privet, E = European privet

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

*** mean

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POSTFIRE REGENERATION STRATEGY (Stickney 1989)

Tall shrub, adventitious bud/root crown; small shrub, adventitious bud/root crown; 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

Fire can kill above-ground portions of Chinese privet and European privet (Batcher 2000, Faulkner *et al.* 1989). Although documentation is lacking, it is likely that fire also top-kills Amur privet and Japanese privet.

PLANT RESPONSE TO FIRE

Japanese privet can “resprout following fire,” although further details are lacking (Louisiana State University 2001). Chinese privet responds to above-ground damage from fire by vigorously sprouting from the root crown (Faulkner *et al.* 1989). It is also likely that European privet and Amur privet respond to fire damage by sprouting, although as of this writing (2003), documentation is lacking.

DISCUSSION AND QUALIFICATION OF PLANT RESPONSE

As of this writing (2003), there is no mention in the literature of fire-induced root suckering in privet, although such a response seems likely. Chinese privet is known to produce suckers from its extensive, shallow root system (Louisiana State University 2001, Urbatsch 2000), usually following damage to shallow roots (Swarbrick *et al.* 1999).

FIRE MANAGEMENT CONSIDERATIONS

Effectiveness of prescribed fire to control privet may vary. Prescribed burning to control Chinese privet in northwestern Georgia resulted in a mixed-severity fire, with some above-ground mortality and survival of substantial numbers of mature stems. This result was attributed to the erratic, spotty nature of the fire. This may have been due to Chinese privet's affinity for moist, low-lying habitat (Faulkner *et al.* 1989). More research is needed that documents the effects of fire on privet and subsequent plant responses.

Due to the ability of privet to sprout following damage from fire, persistent annual burning will likely be required for local eradication. Nature Conservancy preserve managers in Alabama and Florida have reported that repeated annual prescribed burning, when conducted during periods of low fuel moisture, can eventually eliminate Chinese privet and European privet from invaded sites. Burning is not likely to be effective on perpetually moist sites (Batcher 2000).

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Mahonia nervosa = *Berberis nervosa*
Dwarf Oregon-grape

FIRE ECOLOGY OR ADAPTATIONS

Dwarf Oregon-grape persists in closed forest stands with long fire-free intervals. However, as a residual survivor, it is also well-adapted to a regime of “relatively frequent surface fires” such as those common in certain Douglas-fir-western hemlock/dwarf Oregon-grape types of Oregon (Hemstrom *et al.* 1987). Fire intervals in Douglas-fir-western hemlock forest inhabited by dwarf Oregon-grape commonly range from 137 to 320 years (Agee *et al.* 1990, Ossinger 1983). Fire intervals in other forest types occupied by dwarf Oregon-grape have been estimated as follows in Desolation Peaks, Washington (Agee *et al.* 1990):

ponderosa pine-Douglas-fir	52 years
lodgepole pine-Douglas-fir	76 years
Douglas-fir-grand fir	93 years
Douglas-fir-Pacific silver fir	108 years

Fire can produce gaps in old-growth redwood forests which are conducive to dwarf Oregon-grape growth (Lenihan 1990).

Dwarf Oregon-grape commonly sprouts and grows vigorously after fire (Roberts 1975). Reestablishment through seed may occur, although vegetative regeneration is the dominant mode of post-fire establishment (Halpern 1989).

POSTFIRE REGENERATION STRATEGY

Rhizomatous shrub, rhizome in soil.

IMMEDIATE FIRE EFFECT ON PLANT

Dwarf Oregon-grape is moderately damaged by light- to moderate-severity fires (Atzet and Wheeler 1982). Underground regenerative structures often survive even if above-ground portions are consumed by fire (Roberts 1975, Yerkes 1960).

PLANT RESPONSE TO FIRE

Dwarf Oregon-grape often sprouts from underground rhizomes after above-ground portions of the plant are killed (Roberts 1975, Yerkes 1960). However, response varies with fire intensity, severity (Halpern and Franklin 1989, Lafferty 1972), and season. Atzet and Wheeler (1982) noted sprouts after light-severity fires but did not observe sprouting after moderate-severity fires. Seedling establishment after fire has not been documented (Halpern 1989) and may be insignificant.

Post-fire recovery: Post-fire reestablishment and growth of dwarf Oregon-grape is often rapid (Roberts 1975). In western Washington, sprouts are commonly observed soon after fire (Kienholz 1929). Under some circumstances cover may equal or exceed that of pre-fire levels within several years (Miller and Miller 1976).

Dwarf Oregon-grape cover 9 years after slash burning near Oakridge, Oregon, surpassed that of adjacent unburned plots (Steen 1966).

Dwarf Oregon-grape abundance may not peak until mid- to late-seral stages, particularly after hot fires (Halpern 1989). Recovery can be slow after moderate to hot fires that damage or kill portions of underground rhizomes. Few dwarf Oregon-grape were present by the third growing season after a moderate fire in coastal British Columbia (Lafferty 1972).

DISCUSSION AND QUALIFICATION OF PLANT RESPONSE

Recovery of dwarf Oregon-grape after July, 1970 wildfires in North Cascades National Park was as follows (Miller and Miller 1976):

	1971		1972		1974	
	freq.	cover	freq.	cover	freq.	cover
site 1	44	--	40	0.1	32	0.6
site 2	82.6	1.6	82.6	2.3	82.6	3.4
site 3	90.3	0.16	83.9	2.2	83.9	4.9

FIRE MANAGEMENT CONSIDERATIONS

Timber harvest: Dwarf Oregon-grape commonly exhibits dramatic reductions soon after timber harvest and subsequent slash fires in western hemlock-western redcedar-Douglas-fir forests of the Cascade Ranges, but then often undergoes a dramatic recovery (Halpern 1989). In some areas, dwarf Oregon-grape cover has tripled during the first five years after logging and slash fires (Dyrness 1973). However, initial recovery may be fairly slow on some sites (Yerkes 1960). Post-treatment cover is presumably related to a number of factors including fire intensity and severity, season of fire, and site characteristics. Dwarf Oregon-grape commonly reaches greatest abundance during secondary succession (Ingram 1931). Abundance peaked at 30 to 40 years after clearcutting, broadcast burning, and

planting in western hemlock-Douglas-fir forests of the western Cascades (Schoonmaker and McKee 1988). Post-treatment recovery was as follows (Schoonmaker and McKee 1988):

years since treatment (percent cover)							
2	5	10	15	20	30	40	undisturbed old growth
1.88	5.04	4.22	9.48	6.98	22.18	20.97	11.52

Post-treatment response of dwarf Oregon-grape has been documented in a number of other studies (Bailey 1966, Dyrness 1965, Dyrness 1973, Gashwiler 1970, Schoonmaker and McKee 1988, Yerkes 1960).

Fuels: Many dwarf Oregon-grape communities are characterized by low to medium fuel levels (Atzet and Wheeler 1982).

Prescribed fire: Prescribed fire in Pacific rhododendron-dwarf Oregon-grape communities can greatly increase herb and shrub production (Garrison and Smith 1974).

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Pachystima myrsinites
Oregon boxwood

FIRE ECOLOGY OR ADAPTATIONS

Following fire, Oregon boxwood can sprout from buds on the taproot or from the root crown (Crane and Fischer 1986, McLean 1968, Noste and Bushey 1987). Some seedling establishment via short-term viability seed stored on-site may also occur (Stickney 1986).

POSTFIRE REGENERATION STRATEGY (Lyon and Stickney 1976)

Small shrub, adventitious bud/root crown.

IMMEDIATE FIRE EFFECT ON PLANT

Oregon boxwood can survive low- to moderate-severity fires that do not consume the duff or raise the soil temperature too high (Crane and Fischer 1986). It can, however, be killed by severe fires (Canon *et al.* 1987).

PLANT RESPONSE TO FIRE

Oregon boxwood usually sprouts from its root crown or from buds on its taproot following low- to moderate-severity fires (Crane and Fischer 1986, Noste and Bushey 1987).

DISCUSSION AND QUALIFICATION OF PLANT RESPONSE

The development of Oregon boxwood cover following wildfire and clearcutting/broadcast burning has been recorded for western larch (*Larix occidentalis*) and Douglas-fir forests in Montana (Stickney 1980). Oregon boxwood appears to have a varied response to both wildfire and broadcast burning, depending on site (Brown and Simmerman 1986, Edgerton 1987, Stickney 1980). Stickney tracked first decade post-fire succession following a severe fire in western hemlock/ Oregon boxwood

habitat type. Oregon boxwood exhibited a steady-state frequency pattern throughout the decade, with little expansion or reduction in distribution within the study site (Stickney 1985, Stickney 1986). Some have classified Oregon boxwood as “neutral” in its resistance to fire, meaning that it has less than a 12.5 percent frequency increase or decrease when compared to average frequencies of those shrubs in unburned areas (Vogl and Calvin 1969).

FIRE MANAGEMENT CONSIDERATIONS:

Nalley (1982) developed models for predicting fuel loading in western redcedar/Oregon boxwood types in northern Idaho. Brown (1981) lists bulk densities of some Montana and Idaho habitat types (in which Oregon boxwood is an indicator) for determining fuel depth. Fuel loadings and fire ratings for trembling aspen/Oregon boxwood community types have also been listed (Brown and DeByle 1989).

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Philadelphus lewisii
Lewis' mockorange

FIRE ECOLOGY OR ADAPTATIONS

Lewis' mockorange occurs in both forested environments which frequently experience fire and on rocky scree slopes which may not burn at all. In western Montana, Lewis' mockorange occurs in Douglas-fir habitat types which had historical fire frequencies of 5 to 45 years (Fischer and Bradley 1987). The ability of Lewis' mockorange to sprout after top-kill by fire enables it to persist in these forests.

POSTFIRE REGENERATION STRATEGY

Tall shrub, adventitious-bud root crown; ground residual colonizer (on-site, initial community).

IMMEDIATE FIRE EFFECT ON PLANT

Lewis' mockorange is top-killed by fire, but the root crown usually survives and produces sprouts (Fischer and Bradley 1987, Leege 1979, Leege and Hickey 1971).

PLANT RESPONSE TO FIRE

Lewis' mockorange sprouts vigorously from the root crown following fire (Fischer and Bradley 1987, Leege 1979, Leege and Hickey 1971). After an April fire on a south-facing slope in north-central Idaho, Lewis' mockorange increased to pre-fire densities by the third post-fire growing season (Leege 1978).

The following two studies have investigated the sprouting response of Lewis' mockorange to fire.

Seral brushfields within the grand fir (*Abies grandis*)/pachistima (*Pachistima myrsinites*) habitat type in northern Idaho were burned in either spring (late March - early April) or fall (October). Temperatures during the fires ranged from 19-26° C, and relative humidities ranged from 25 to 48%.

Lewis' mockorange on sites burned in October did not sprout until the following April. Lewis' mockorange on the spring-burned sites sprouted 4 to 8 weeks after the fire. Twenty completely top-killed Lewis' mockorange (eleven on the fall burned sites and nine on the spring burned sites) were measured at the end of the first post-fire growing season (Leege and Hickey 1971):

	Fall Fires	Spring Fires
Average crown diameter (cm)		
pre-fire	143	143
post-fire	73	67
Average crown height (cm)		
pre-fire	268	293
post-fire	125	113
No. basal sprouts per plant		
pre-fire	1.5	0.6
post-fire	38.0	28.9
Avg post-fire sprout ht. (cm)	73	64

A multiple regression equation is presented which relates the number of post-fire basal sprouts to pre-fire crown height, crown diameter, and crown volume (Leege and Hickey 1971).

In another study in north-central Idaho, a brushfield was burned three times at 5-year intervals (31 March 1965, 3 May 1970, and 14 May 1975). Maximum air temperatures during the fires were 25, 27, and 31° C, and relative humidities at 4:00 pm were 35, 16, and 27%, respectively. Leaves on shrubs and succulent herbaceous growth depressed the fire in 1975. A single Lewis' mockorange was followed during the study. The plant was dormant during the first fire, but leaves were beginning to emerge at the time of the second fire and were completely emerged at the time of the third fire.

With each successive fire, average sprout height decreased. Reduced growth following the second and third fires may have resulted from the advanced phenological stage at the time of those fires. Sprout height and number were measured the first growing season following each fire. Crown height and diameter were measured during the second post-fire growing season (Leege 1979).

	No. basal sprouts	Average sprout height (cm)
1965	14	61
1970	19	46
1975	16	30

	Crown height (cm)	Crown diameter (cm)
pre-fire	213	46
1966	122	61
1971	76	61
1976	76	46

FIRE MANAGEMENT CONSIDERATIONS

Lewis' mockorange palatability increases following fire. After a spring fire in northern Idaho, utilization of Lewis' mockorange by Rocky Mountain Elk was significantly ($p < 0.05$) greater on recently burned sites than on adjacent unburned sites (Leege 1969).

	% of available twigs browsed	
	1st season	2nd season
Control	1.3	0.6
Burn	36.3	30.0

Allometric equations, which can be used to estimate fuel quantities, have been developed for Lewis' mockorange. The equations use basal stem diameter to estimate foliage biomass and total biomass (Brown 1976).

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Prunus virginiana
Chokecherry

FIRE ECOLOGY OR ADAPTATIONS

Chokecherry is well adapted to disturbance by fire (Anderson and Bailey 1980, Brockway and Lewis 1997, Gartner and Thompson 1973, Leege and Hickey 1966, McKell 1950, Wright and Bailey 1982, Young 1983). Although susceptible to top-kill by fire, it resprouts rapidly and prolifically from surviving root crowns and rhizomes (Gartner 1975, Leege and Hickey 1966, McKell 1950, Volland and Dell 1981, Wasser 1982). Several studies reporting chokecherry recovery by sprouting are discussed in the Fire Effects section of this report. Seed germination improves with heat treatment, suggesting scarification by fire is an important adaptation (Sampson 1944). Post-fire regeneration probably also involves the germination of off-site seed dispersed by mammals and birds (Volland and Dell 1981).

No data were found for natural intervals of fire in stands that consist mainly of chokecherry. Gartner and Thompson (1973) provides a description of pre- and post-settlement accounts of fire in the grasslands and ponderosa forests of the Black Hills of South Dakota. The historical information is detailed, but fire return intervals are

not given. Hansen and Kurmis (1972) provides limited historical and fire interval information for Minnesota forests in Isle Royale National Park, Itasca State Park, and the Boundary Waters Canoe Area. Twenty-six lightning fires were recorded on Isle Royale from 1965 to 1949. In Itasca State Park the incidence of fires caused by lightning or set by Indians before 1859 averaged about one fire every 12 years. In the Boundary Waters Canoe Area, major fires recurred at 5- to 50-year intervals from 1600 to 1920.

Fire regimes for other plant communities in which chokecherry occurs are summarized below.

Community or Ecosystem Scientific Name of Dominant Species Mean Fire Return

Community or Ecosystem	Scientific Name of Dominant Species	Mean Fire Return Interval
Pacific ponderosa pine*	<i>Pinus ponderosa</i> var. <i>ponderosa</i>	1-47 yrs
Rocky Mountain ponderosa pine*	<i>P. ponderosa</i> var. <i>scopulorum</i>	2-10 yrs [1]
Rocky Mountain lodgepole pine*	<i>P. contorta</i> var. <i>latifolia</i>	50-300+ yrs [2, 3]
Colorado pinyon	<i>P. edulis</i>	10-49 yrs
Rocky Mountain Douglas-fir*	<i>Pseudotsuga menziesii</i> var. <i>glauca</i>	40-140 yrs [1]
Wyoming big sagebrush	<i>Artemisia tridentata</i> var. <i>wyomingensis</i>	10-70 yrs [4, 5]
mountain big sagebrush	<i>A. tridentata</i> var. <i>vaseyana</i>	5-15 yrs
curlleaf mountain-mahogany*	<i>Cercocarpus ledifolius</i>	13-1,350 yrs [6, 7]
trembling aspen (west of the Great Plains)	<i>Populus tremuloides</i>	7-100 yrs [8, 9]

*fire return intervals vary widely; trends in variation are noted in the FEIS species' summary

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POSTFIRE REGENERATION STRATEGY

Tree with adventitious-bud root crown/soboliferous species root sucker; tall shrub, adventitious-bud root crown; small shrub, adventitious-bud root crown; ground residual colonizer (on-site, initial community); initial-offsite colonizer (off-site, initial community).

IMMEDIATE FIRE EFFECT ON PLANT

Fire often kills above-ground chokecherry stems and foliage, but it quickly sprouts, either the same year following a spring burn, or by the next growing season (Leege and Hickey 1966, McKell 1950, Volland and Dell 1981, Wasser 1982, Young 1983). In the South Dakota Black Hills chokecherry sprouts were double the pre-burn numbers within two months of an early May burn (Gartner 1975). Conversely, in an early May prescribed burn in central Alberta trembling aspen parkland, chokecherry shrubs did not sprout within the first three months following burning (Hansen and Kurmis 1972). Fire intensity was not described for either study.

A prescribed fire study was conducted in northern Idaho to test the effect of spring versus fall burning on Elk browse. Measurements were made of crown diameter, crown height, number of basal sprouts, and sprout height before and after each burn. Post-fire measurements were made on 11 shrubs the first growing season after the fall burns. Seasonal fire effects were similar for chokecherry crown diameter, crown height, and sprout height. Though not statistically significant (at $p = 0.05$), the number of chokecherry basal sprouts was somewhat higher after the spring burn, suggesting that spring burning may be more conducive to the rapid recovery of chokecherry than fall burning (Leege and Hickey 1966).

PLANT RESPONSE TO FIRE

Most studies report either an increase in chokecherry in the years following fire, or an increase followed by a return to pre-fire numbers. After wildfires in the oakbrush zone in Utah, McKell (1950) reported twice as many chokecherry stems sprouting from root crowns on 1-year-old burns than on adjacent unburned sites. A reduction to pre-fire densities occurred within 18 years.

Following wildfires in Rocky Mountain Douglas-fir/Rocky Mountain juniper/Wyoming big sagebrush associations in the Missouri Breaks area of central Montana, chokecherry canopy cover increased consistently for 13 years, then stabilized (Eichhorn and Watts 1984).

Bock and Bock (1984) compared data from prescribed October burns in 1974 and 1979 in the South Dakota Black Hills. The 1974 burn escaped and became a crown fire, killing ponderosa pines of all sizes. The 1979 fire remained a controlled understory fire. When measured in 1981, the 1974 burn site supported higher densities of all woody taxa except chokecherry and western poison-ivy (*Toxicodendron rydbergii*). There was no significant difference ($p = 0.71$) in numbers of chokecherry plants between the two burn sites. Measurements taken within the 1979 surface fire site (pre-fire, post-fire year 1, and post-fire year 2) showed that chokecherry stems were not significantly ($p = 0.75$) reduced by the fire.

Geier-Hayes (1989) included chokecherry in a study of vegetation response to helicopter logging and broadcast burning in an Idaho Douglas-fir forest. Data were collected in three cutting units prior to burning and 1, 2, 5 and 10 years after. Fire severity was higher in two of the units and altered the vegetation from the original. Fires classed at a severity level of 2M (Ryan and Noste 1985) were less severe and had little or no impact on chokecherry percent cover and root frequency. 2M fires are characterized as having a flame length of 0.6-1.2 m and a corresponding crown scorch height of 2.7-7.3 m, with moderate ground charring. The units that burned hotter, having a severity rating of 3M had markedly less chokecherry cover and root frequency during all post-burn years measured. 3M fires have flame lengths of 2.4-3.7 m, corresponding crown scorch to 20 m, with moderate ground charring.

Following a September prescribed burn in a trembling aspen stand in Idaho, chokecherry biomass exceeded pre-burn biomass within two seasons and was double pre-burn biomass after five seasons (Brown and DeByle 1989). Biomass was computed using weight versus stem diameter relationships (Brown 1976). Details of fuel conditions are provided: litter and woody material moisture content was 8 to 9% and herbaceous vegetation was 40 to 50% cured. Fire severity was rated as moderate to high (Ryan and Noste 1985).

After 24 years of annual early spring burning in trembling aspen parklands in Alberta, chokecherry percent cover had decreased but the number of stems increased in density from 6% to 15% (Anderson and Bailey 1980).

In a 20-year study of the effects of fire frequency on Minnesota oak savanna herbs and shrubs, Tester (1989, 1996) determined that increased fire frequency tended to increase the density of native, true prairie shrubs and decrease the density of native, non-prairie shrubs (including chokecherry). Chokecherry cover estimates were negatively correlated with burn frequency ($r = -0.51$, $p = 0.09$).

FIRE MANAGEMENT CONSIDERATIONS

Chokecherry is a component of persistent, fire-maintained seral shrubfields on steep slopes in Northern Idaho. Fuels in shrubfields differ in quantity and distribution from those on forested sites. Herbaceous and large woody fuels are relatively light. Live and dead shrub biomass, which includes chokecherry, can reach nearly 20 tons per acre. After fires, which are severe during summer drought conditions, dense shrub cover regenerates within 10 years. Trees regenerate slowly or not at all on these dry sites, because of erosion, depleted soil organic matter, high soil temperatures, and lack of seed (Smith and Fischer 1997).

Arno (1985) hypothesized that relatively frequent fires set by Native Americans in western grassland and sagebrush communities, where chokecherry occurs, favoured expansion of grasslands into adjacent shrub or tree communities. In recent times shrub and tree communities have developed in former grasslands due to fire exclusion and grazing. Arno argues that baseline information on Native American use of fire will aid land managers in predicting vegetative development under different fire regimes.

Morber and Miyanishi (1995) studied fire as a tool for controlling chokecherry and black cherry in Ontario oak savanna. A controlled spring burn had no effect on chokecherry seedlings. Post-fire seedling emergence was concluded to be largely dependent on post-fire seed production or seed influx from adjacent unburned areas, because there was no viable soil seedbank.

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Rosa gymnocarpa
Baldhip rose

FIRE ECOLOGY OR ADAPTATIONS

Baldhip rose is well adapted to low- to medium-severity fires. It sprouts from both root crowns and rhizomes. It is also an off-site colonizer (Hooker and Tisdale 1974, Hungerford 1986, Morgan and Neuenschwander 1988).

POSTFIRE REGENERATION STRATEGY

Small shrub, adventitious-bud root crown; rhizomatous shrub, rhizome in soil; secondary colonizer - off-site seed.

IMMEDIATE FIRE EFFECT ON PLANT

Above-ground portions of the plant are killed by fire. Root crowns and underground rhizomes typically survive low- to moderate-severity fires (Hooker and Tisdale 1974, Hungerford 1986, Morgan and Neuenschwander 1988, Mueggler 1965, Neuenschwander [n.d.], Stickney 1986). Severe fires can cause damage to root crowns, decreasing potential regrowth (Haeussler *et al.* 1990).

PLANT RESPONSE TO FIRE

Baldhip rose rapidly recovers following low- to medium-severity fires. Top-killed plants typically sprout vigorously from the root crown or rhizomes (Hooker and Tisdale 1974, Hungerford 1986, Morgan 1984, Mueggler 1965, Neuenschwander [n.d.], Stickney 1986).

Seedlings are rarely observed in a burn area (Morgan 1984, Neuenschwander [n.d.]). No seedlings were present on clearcut 2-year-old burns in a western redcedar/queencup beadlily (*Thuja plicata/Clintonia uniflora*) habitat type in northern Idaho (Morgan and Neuenschwander 1988).

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Rosa nutkana
Nootka rose

IMMEDIATE FIRE EFFECT ON PLANT

Nootka rose is top-killed by fire. Shallowly buried root crowns and rhizomes of *Rosa* species, including Nootka rose, may be killed by severe fire (Haeussler *et al.* 1990).

PLANT RESPONSE TO FIRE

Nootka rose may increase or decrease following fire. *Rosa* species of British Columbia, including Nootka rose, usually initially decrease in cover after wildfire. Cover gradually increases, and then declines again as the canopy closes (Haeussler *et al.* 1990). In a Douglas-fir/ninebark habitat type in Idaho, Nootka rose cover was greater on burned sites or sites logged and then burned than on undisturbed sites.

Cover was less, however, than on logged or grazed sites (Cholewa and Johnson 1983). In aspen and aspen-mixed conifer stands in Idaho and Wyoming, prescribed fires of low to high severity caused a decrease in cover of Nootka and Wood's rose (*Rosa woodsii*) in post-fire year 1. Rose biomass in post-fire year 2 was near pre-fire levels (Brown and DeByle 1989). In Washington and Oregon, burning had little effect on abundance of Nootka rose. Multiple fires can significantly reduce cover of rose species (Haeussler *et al.* 1990).

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Rubus laciniatus
Evergreen blackberry

FIRE ECOLOGY OR ADAPTATIONS

Evergreen blackberry is frequently observed on recently burned sites (Alaback and Herman 1988, Stickney 1989). Most species of blackberry sprout prolifically from rootstocks, roots, or rhizomes, even when above-ground foliage is totally consumed by fire. Evergreen blackberry can root from the nodes of above-ground stems (Ingram 1931), and rapid spread is likely where portions of the stem remain undamaged.

Evergreen blackberry is described as a seedbanking species which can readily reoccupy disturbed sites through seed stored on-site (Kellman 1970). Seed can apparently remain viable for long periods of time when stored in the soil or duff (Brinkman 1974) and germinate in large numbers after fire. The large, sweet, succulent fruit of blackberries amply "reward" animal dispersers (Janzen 1984), and post-fire establishment of some evergreen blackberry seed from off-site is probable (Alaback and Herman 1988).

POSTFIRE REGENERATION STRATEGY

Tall shrub, adventitious-bud root crown; rhizomatous shrub, rhizome in soil; geophyte, growing points deep in soil; ground residual colonizer (on-site, initial community); initial-offsite colonizer (off-site, initial community).

IMMEDIATE FIRE EFFECT ON PLANT

Although evergreen blackberry plants may be top-killed, actual mortality appears to be uncommon due to the prolific sprouting ability of this shrub.

Most evergreen blackberry seeds stored on-site in the soil or duff are probably unharmed by fire.

PLANT RESPONSE TO FIRE

Vegetative regeneration: Most blackberries readily regenerate from roots, rhizomes, or rootstocks when aboveground foliage is consumed by fire (Ferguson 1983, Great Plains Flora Association 1986). Roots are generally well protected from the direct effects of heat by overlying layers of soil. The evergreen blackberry is known to produce adventitious root suckers (Hall *et al.* 1986) and presumably sprouts when aboveground vegetation is totally consumed by fire.

This shrub is capable of regenerating by means of horizontal aboveground stems, which root at the nodes in soil or duff (Ingram 1931), even in the absence of disturbance. Rapid spread is probable where portions of the aboveground stem remain undamaged by fire.

Post-fire recovery: The evergreen blackberry quickly assumes prominence on many types of burned or disturbed sites (Steen 1966) and is often well represented on waste ground (Radford *et al.* 1968). Its role as a vigorous invader suggests the potential for rapid post-fire recovery in many areas.

Seedling establishment: Seedbanking may be an important regenerative strategy in the evergreen blackberry (Kellman 1970). Some seed may also be transported from off-site by birds or mammals (Brinkman 1974).

FIRE MANAGEMENT CONSIDERATIONS

Wildlife: Species which consume large amounts of blackberries are often benefited by fire (Kramp *et al.* 1983).

Competition: Many blackberries are favoured by fire and can aggressively compete with conifer seedlings in some post-fire communities.

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Rubus parviflorus
Thimbleberry

FIRE ECOLOGY OR ADAPTATIONS

Thimbleberry is resistant to and generally enhanced by fire (Hamilton 1988, Hamilton and Yearsley 1988, Volland and Dell 1981). In areas of rigorous fire suppression, thimbleberry fruit production and plant vigour has declined (Lepofsky *et al.* 1985). This species is well-adapted to vigorously invade many types of burned sites through rhizomes or seed (Halpern and Franklin 1989, Mueggler 1965, Yerkes 1960). Rhizome sprouting is an important post-fire strategy which enables rapid reestablishment and spread (Crane *et al.* 1983, Lyon and Stickney 1976, Noste and Bushey 1987, Stickney 1986). Thimbleberry, a seed banker (Kellman 1970, Kramer 1984, Kramer and Johnson 1987), also reestablishes through viable seed stored in the soil or duff (Crane *et al.* 1983). Birds and mammals add to seedling establishment by transporting seed to the site (Crane *et al.* 1983, Keller 1980).

Thimbleberry typically becomes abundant within the first few years after fire and remains prominent during the early post-fire decades (Crane *et al.* 1983, Morgan and Neuenschwander 1988, Stickney 1986). Hamilton and Yearsley (1988) note that thimbleberry "may be well-adapted to the high nutrient availability and low competition from other species found immediately after burning, but is less successful once other species have reestablished." This fire-adapted species typically declines as the overstorey develops in post-fire communities.

POSTFIRE REGENERATION STRATEGY

Tall shrub, adventitious-bud root crown; rhizomatous shrub, rhizome in soil; ground residual colonizer (on-site, initial community); initial-offsite colonizer (off-site, initial community).

IMMEDIATE FIRE EFFECT ON PLANT

Thimbleberry is described as resistant to fire (Hamilton and Yearsley 1988, Volland and Dell 1981). Although often top-killed, underground rhizomes generally survive. High severity fires which damage belowground regenerative structures may be most damaging to rhizomes (Morgan and Neuenschwander 1988). The aerial portions of relatively few plants actually survive fire, except for those on unusual microsites (Halpern 1989). Most seed stored on-site is probably unharmed by fire.

PLANT RESPONSE TO FIRE

Thimbleberry cover and vigour is generally enhanced by fire (Crane and Fischer 1986, Hamilton and Yearsley 1988, Mueggler 1965).

Vegetative response: Thimbleberry is capable of rapid, vigorous post-fire spread through an extensive network of often deeply-buried rhizomes (Crane *et al.* 1983, Keller 1980, Steele and Geier-Hayes 1989). Sprouting through surviving rootcrowns is also possible, although rhizome sprouting probably represents the primary mode of post-fire regeneration (Noste and Bushey 1987). Multiple sprouts often replace the single stems observed in pre-burn communities (Crane and Fischer 1986).

Limited evidence suggests that sprouting of rhizomes is favoured by fires of low intensity and severity (Crane and Fischer 1986, Morgan and Neuenschwander 1988). Hot fires presumably offer greater potential for damaging underground regenerative structures. In western redcedar habitat types of northern Idaho, thimbleberry cover was estimated at 10.9 percent two years after low severity fire. However, following a high severity fire, thimbleberry cover reached only 4.9 percent (Morgan and Neuenschwander 1988).

Seedling establishment: Seed banking is an important post-fire regenerative strategy in thimbleberry, a prolific seed producer (Hamilton and Yearsley 1988). Birds and mammals also transport some viable seed to burned sites (Crane *et al.* 1983, Keller 1980). Some seedlings have been observed to germinate from rodent caches present on-site (Morris 1958). Thimbleberry seed is stimulated by fire (Halpern 1989, Stewart 1978) and subsequently germinates in great numbers (Steele and Geier-Hayes 1989, Stewart 1978). Researchers have observed up to 7.5 times more seedlings on plots which have been burned than on adjacent untreated plots (Stewart 1978). Most seedling establishment occurs immediately after fire (Hamilton and Yearsley 1988).

Some researchers have suggested that fires of high severity and intensity can expose mineral soil, which serves as a favourable seedbed for thimbleberry (Morgan and Neuenschwander 1988, Noste and Bushey 1987). However, Morgan and Neuenschwander (1988) observed greatest seedling establishment on low severity burns where duff reduction was incomplete. The capacity to regenerate through seed may be reduced by extremely hot, duff-reducing burns (Morris 1958, Van Dersal 1938).

Rate of post-fire recovery: Both sprouting (Hamilton 1988) and seedling establishment occur soon after fire. Growth is rapid and some plants occasionally bear fruit during the first post-fire year (Crane *et al.* 1983). Thimbleberry generally reaches greatest abundance during the first years after fire and decreases as the overstorey canopy closes (Hamilton and Yearsley 1988). Dwindling soil nutrients may also contribute to its decline (Klinka *et al.* 1985). On coastal sites in British Columbia, thimbleberry typically attains greatest height growth during the first few years after fire and remains abundant for approximately five years (Hamilton and Yearsley 1988).

Rhizome sprouting may produce dense cover up to 1 m tall within three years after disturbance on some particularly moist sites (Comeau *et al.* 1989). Thimbleberry reportedly dominates the understorey of many Northwestern Douglas-fir forests within just five years after logging and slash disposal (Mitchell 1983). In Douglas-fir forests of western Montana, thimbleberry was well-represented on many sites during the first 2 to 3 years after fire (Crane and Habeck 1982, Keller 1980). Maximum cover values have been observed within 3 to 5 years after fire in western redcedar habitat types of Idaho (Morgan and Neuenschwander 1988). In many areas, this shrub remains widely distributed during the first several decades after fire (Stickney 1986). Dense early growth of aggressive species such as fireweed can slow thimbleberry establishment and growth (Stickney 1981).

DISCUSSION AND QUALIFICATION OF PLANT RESPONSE

Rate of post-fire recovery may be significantly related to fire severity and intensity. However, contradictory findings have been reported. Hooker and Tisdale (1974) observed rapid growth rates on severely burned sites (0.6 cm/day) but slower growth on less severely burned plots (Hooker and Tisdale 1974). Slower growth may be expected where cover (and thus increased intraspecific competition) is greatest, and these results do not necessarily indicate that actual cover was greatest on these severely burned plots. Other research has suggested that thimbleberry cover develops most rapidly after fires of low intensity (Comeau *et al.* 1989).

Some researchers suggest that, in general, rhizome sprouting may be favoured by fires of low intensity and severity, whereas seedling establishment may be more effectively promoted by fires of relatively high severity (Morgan and Neuenschwander 1988, Noste and Bushey 1987). However, researchers in northern Idaho have occasionally observed greater seedling establishment after fires of low severity (Morris 1958).

Factors other than fire severity and intensity can also contribute to the rate of post-fire recovery. Differences in the season of burn, plant density and vigour in the pre-burn community, site differences, and climatic factors all contribute to post-fire recovery. Fire severity and intensity are not consistently defined in the literature, which compounds the problems of interpreting plant response. The rate of post-fire recovery is significantly related to climatic and topographic factors. In coastal Oregon, 21 382 seedlings per ha were present one year after fire on north aspects where 10 988 per ha had been observed prior to the fire. On south aspects, increases were even more dramatic, with 24 640 per ha counted one year after fire where none had been observed prior to the fire (Stewart 1978).

Mean frequency of thimbleberry by burn age and severity was documented as follows after fire in a western redcedar/queencup beadle habitat type of northern Idaho (Morgan and Neuenschwander 1988):

fire severity	years					
	1	2	3	4	5	15
low	83	80	80	59	56	98
high	61	76	78	75	53	100

FIRE MANAGEMENT CONSIDERATIONS

Timber harvest: Thimbleberry typically increases dramatically after timber harvest and subsequent slash burns (Gratkowski 1978, Laursen 1984). In many areas, it may provide formidable competition for regenerating conifer seedlings (Hamilton and Yearsley 1988). Mueggler (1965) reports that thimbleberry often assumes dominance on broadcast burns in western redcedar and western hemlock habitat types of northern Idaho. Response may be rapid. Maximum cover values were reached within five years after fire in western hemlock and western redcedar habitat types of northern Idaho (Stickney 1985).

This shrub can establish as early as the first growing season after clearcutting and broadcast burns in grand fir/pachistima habitat types of north-central Idaho. More serious disturbance may enhance the growth and recovery of thimbleberry more than lighter disturbance. In the Intermountain West, Laursen (1984) reported greater increases in ground cover in stands that were clearcut and burned than in unburned shelterwood or selection cuts. Maximum cover was typically reached within 20 to 25 years after clearcutting and burning (Laursen 1984). Thimbleberry commonly assumes importance soon after timber harvest and slash burning in Douglas-fir forests of the Oregon Cascades. Frequency of occurrence by years after treatment was documented as follows (Yerkes 1960):

site #	1	4	5	(years)
1	15	32	36	
2	4	14	14	

Wildlife: Fire generally benefits animals that eat the fruits of *Rubus* (Kramp *et al.* 1983).

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Rubus ursinus
Trailing blackberry

FIRE ECOLOGY OR ADAPTATIONS

Trailing blackberry is a common invader on recently burned sites in the Pacific Northwest (Dyrness 1973, Isaac 1940, Morris 1958). Populations are capable of dramatic and rapid expansion on disturbed sites (Halpern 1989, Yerkes 1960) through sprouting or seedling establishment (Conrad 1987, Dyrness 1973). Belowground regenerative structures are generally well protected from the harmful effects of heat and permit rapid recovery where trailing blackberry plants were present in the pre-burn community. Seedbanking is also an important post-fire regenerative strategy (Morgan and Neuenschwander 1988). Seeds accumulate in the soil or duff, remaining viable long after this seral species has been eliminated from mature forest communities. Seeds commonly germinate in great abundance after fire. The relatively large, sweet, succulent fruit of blackberries amply "reward" animal dispersers (Janzen 1984), and some post-fire dispersal of seed from off-site is probable.

POSTFIRE REGENERATION STRATEGY

Tall shrub, adventitious bud root crown; rhizomatous shrub, rhizome in soil; geophyte, growing points deep in soil; ground residual colonizer (on-site, initial community), initial-offsite colonizer (off-site, initial community).

IMMEDIATE FIRE EFFECT ON PLANT

Trailing blackberry is described as "rather tolerant" of fire (Halpern 1989). Although it may be top-killed (Morgan and Neuenschwander 1988), underground regenerative portions of this shrub generally survive (Conrad 1987, Morgan and Neuenschwander 1988, Stewart 1978). Fires of relatively high severity or intensity, with the potential to

harm belowground regenerative structures, appear to be the most damaging to trailing blackberry (Morgan and Neuenschwander 1988).

Most trailing blackberry seed stored on-site in the soil or duff is probably unharmed by fire (Morgan and Neuenschwander 1988).

PLANT RESPONSE TO FIRE

Vegetative response: Trailing blackberry is capable of vegetative regeneration following fire through nonrhizomatous basal sprouts or root “suckers” (Conrad 1987, Morgan and Neuenschwander 1988). Basal sprouting is believed to be of primary importance, and often results in a large number of sprouts (Mitchell 1983). This low-growing shrub can also root at the stem nodes (Ingram 1931, Willoughby and Davilla 1984) and can presumably spread quickly where portions of the aboveground stem remain undamaged. Vegetative spread is generally both vigorous and rapid.

Stewart (1978) observed an average of approximately 3762 sprouts per ha (plus some surviving original stems) within two years after fire. Prior to the fire, an average of only 40 original trailing blackberry stems had been counted within the same area.

Evidence suggests that all forms of sprouting may be favoured after fires of relatively low severity or intensity which are unlikely to damage belowground regenerative structures (Morgan and Neuenschwander 1988). Expansion of trailing blackberry may be delayed on heavily burned sites (Halpern 1989, Miller and Miller 1976). Comparative cover and density values of sprouts on a 2-year-old burn in western redcedar/pachistima and western redcedar/queencup beadlily habitat types in northern Idaho are as follows (Miller and Miller 1976):

	low severity	high severity
% cover	32.5	16.0
density	1.7	1.8

Seedling establishment: Seedbanking is reportedly an important means of post-fire reestablishment for the trailing blackberry (Miller and Miller 1976). High-severity fires, which burn to mineral soil, frequently create a favourable seedbed for buried blackberry seed, and seedlings sometimes germinate in abundance (Morgan and Neuenschwander 1988). However in several instances, researchers have observed decreased seedling establishment after unusually hot fires with “much fuel consumption” (Morgan and Neuenschwander 1988, Van Dersal 1938). Other factors, such as site characteristics or climatic conditions, may have contributed to the variable responses. Seedling establishment of trailing blackberry can also occur through seed transported from off-site by birds and mammals.

Rate of post-fire recovery: Sprouting produces the most rapid early growth, as plants draw upon portions of previously established root systems (Morgan and Neuenschwander 1988). In many locations trailing blackberry has exhibited the most rapid post-fire expansion of any residual species (Halpern 1989). This shrub can dominate the herbaceous layer within 2 to 5 years after fire (Halpern 1989, Morris 1958, Yerkes 1960). Peak cover values have been reported from 0 to 5 years after fire (Halpern 1989, Mitchell 1983). Trailing blackberry is characterized by a relatively long (> 5 years) period of post-fire abundance and generally persists until suppressed by canopy closure (Halpern 1989). Trailing blackberry cover occasionally exhibits a temporary decline after rapid early growth as one or a few stems attain dominance over many initial sprouts (Miller and Miller 1976). In western redcedar habitat types of northern Idaho, Morgan and Neuenschwander (1988) observed highest cover values in the third and fifth years after fire, with trailing blackberry disappearing by the fifteenth year. However, trailing blackberry frequently remains abundant for 11 to 16 years or more after fire in the Cascade Mountains of western Oregon (Schoonmaker and McKee 1988, Steen 1966). Halpern (1989) reported that it remained prominent for at least 20 years after fire in seral Douglas-fir forests of western Oregon.

DISCUSSION AND QUALIFICATION OF PLANT RESPONSE

Fire severity and intensity can influence the rate of post-fire recovery in trailing blackberry regardless of whether regeneration occurs vegetatively or through seedling establishment. The following response has been observed in western redcedar/queencup beadlely habitat types of Idaho (Morgan and Neuenschwander 1988):

	mean frequency of occurrence					
	burn age in years					
severity	1	2	3	4	5	15
low	33	82	71	74	98	0
high	71	80	68	74	100	0

FIRE MANAGEMENT CONSIDERATIONS

Timber harvest and slash burns: Trailing blackberry commonly invades logged and slash burned sites in the Douglas-fir zone of the Pacific Northwest (Isaac 1940, Stewart 1978). Reestablishment is rapid and can occur by the second season after treatment (Morris 1958). In early post-fire years, cover of trailing blackberry is frequently as much as three times higher on slash burned sites than in adjacent undisturbed stands (Dyrness 1965). Trailing blackberry remains prominent until suppressed by the closure of the forest canopy (Halpern 1989).

Schoonmaker and McKee (1988) reported the following cover values after clearcutting and broadcast burning in the Cascades of western Oregon:

Years since treatment	2	5	10	15	20	30	40	old growth
cover(%)	2.28	2.18	1.11	9.7	20.52	7.35	0.66	0.18

Stewart (1978) observed similar increases after clearcutting and broadcast burns in Coastal Oregon:

years	pre-burn	1	3	4
cover (%)	.20	.90	2.40	.20

Competition: Trailing blackberry is favoured by fire and can aggressively compete with conifer seedlings in some post-fire communities.

Wildlife: Species which consume large amounts of blackberries are often benefited by fire (Kramp *et al.* 1983).

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Salix scouleriana
Scouler willow

FIRE ECOLOGY OR ADAPTATIONS

Willows are greatly favoured by fire in most habitats (Haeussler *et al.* 1990, Makela 1990, Viereck and Dyrness 1979). As a survivor and off-site colonizer (Keller 1980, Stickney 1981, Stickney 1989, Stickney 1990, Stickney 1991), Scouler willow is abundant following fire (Machida 1979) and has a moderate regeneration period (Kovalchik *et al.* 1988). It is adapted to fire by rapidly resprouting from the root crown (Lyon and Stickney 1976, Moir *et al.* 1997, Noste and Bushey 1987, Shiplett and Neuenschwander 1994), and establishes from seed on severely burned sites (Noste and Bushey 1987). Wind dispersed seeds facilitate rapid recolonization of burned areas (Shiplett and Neuenschwander 1994, Simmerman *et al.* 1991). In a north-west Montana study Scouler willow was found on 80% of burned sites with no previous Scouler willow presence (Stickney 1981). Stand replacing fires favour regeneration of Scouler willow (Moir *et al.* 1997), and good response from Scouler willow seedlings can be expected on sites where fire damage is thorough enough to expose mineral soil (Gruell *et al.* 1982). However, it is rarely present on sites where more than 50% of the pre-fire overstorey remains (Forsythe 1975).

Scouler willow layer groups are distinct shrub layers that occur in various habitat types and are created by stand replacing fires (Simpson 1990, Steele and Geier-Hayes 1989a, Steele and Geier-Hayes 1989b, Steele and Geier-Hayes 1990, Steele and Geier-Hayes 1992). Severe wildfires expose patches of bare mineral soil, encouraging the development of Scouler willow shrub layers (Steele and Geier-Hayes 1990, Steele and Geier-Hayes 1992). These layer groups may also develop in response to mechanical scarification in clearcuts and broadcast burns, especially where exposed soil was mounded to trap water behind the mounds, creating well-watered seedbeds of mineral soil (Steele and Geier-Hayes 1990, Steele and Geier-Hayes 1992, Steele and Geier-Hayes 1994).

Scouler willow is frequently a dominant or codominant in the persistent seral brushfields of northern Idaho. These brushfields are likely the result of dry weather patterns after canopy removal and repeated severe fires, which remove most large woody material, litter, and herbaceous fuels. Standing shrubs comprise most of the biomass, and these brushfields can burn in almost any season. If surface fuels are continuous and dry, spring fires spread readily. In the summer, brushfields are often hot and dry, and conditions are exacerbated where nighttime inversions occur. Hot, dry winds during drought conditions can drive severe fires through the shrub layer, with Scouler willow readily regenerating from seed and sprouting (Smith and Fischer 1997).

In interior Alaska, Foote (1983) identified six community developmental stages following fire in black spruce forests. These are: 1) newly burned, 2) moss-herb, 3) tall shrub-sapling, 4) dense tree, 5) hardwood or mixed hardwood-spruce, and 6) black spruce. Arising from sprouts, Scouler willow can average up to a few thousand

stems per hectare one year following wildfire depending on pre-burn density, and thus is an important part of the newly burned stage. It is then often dominant or co-dominant through the tall shrub-sapling stage of succession for about 30 years. It thereafter becomes less frequent, as larger trees outgrow and overtop it.

Sprouting occurs in response to overstorey thinning (Simmerman *et al.* 1991) where Scouler willow is well-distributed and in need of rejuvenation (Gruell *et al.* 1982). Generally, fast spreading fires produce more willow sprouts than slow fires that can damage root crowns (Smith and Fischer 1997). Density and canopy coverage frequently increase after fire because root crowns produce multiple sprouts (Noste and Bushey 1987). Four years post-fire in Alaska, Scouler willow presence was four times greater on burned sites than on adjacent unburned sites (Viereck and Dyrness 1979). In northern Idaho, Scouler willow cover was much higher on burned clearcuts than on unburned clearcuts, particularly where there had been repeated fires over a 30-year period (Mueggler 1965).

Post-fire immature stands (< 90 years) in Montana have significantly more ($p < 0.05$) presence and percent cover of Scouler willow than old growth stands (> 150 years) (Antos and Habeck 1981). The increased presence of Scouler willow in Douglas-fir/ponderosa pine stands following elimination of frequent fires is likely the result of stand opening associated with logging (Ayers *et al.* 1999). Sprouting Scouler willow creates a round growth form up to 16 feet in diameter and may as a result promote reestablishment of shade tolerant species like Douglas-fir, in turn posing a greater risk of stand replacing fires and favouring growth of Scouler willow (Steele and Geier-Hayes 1995).

Without fire, closing conifer canopies lead to the deterioration of Scouler willow (Gruell 1984). In dense second growth stands of sequoia in California, Scouler willow debris creates a fuel hazard; formerly abundant stands of Scouler willow grew in dense clones that were shaded out and killed, forming dense tangles of fuel for wildfire (Biswell *et al.* 1968).

Fire regimes for plant communities and ecosystems in which Scouler willow occurs are summarized below.

Community or Ecosystem	Dominant Species	Fire Return Interval Range (years)
silver fir-Douglas-fir	<i>Abies amabilis</i> - <i>Pseudotsuga menziesii</i> var. <i>menziesii</i>	> 200
grand fir	<i>A. grandis</i>	35-200 [1]
California montane chaparral	<i>Ceanothus</i> and/or <i>Arctostaphylos</i> spp.	50-100
Rocky Mountain juniper	<i>Juniperus scopulorum</i>	< 35 [2]
western larch	<i>Larix occidentalis</i>	25-100
Engelmann spruce-subalpine fir	<i>Picea engelmannii</i> - <i>Abies</i>	35 to > 200 [1]

	<i>lasiocarpa</i>	
black spruce	<i>P. mariana</i>	35-200
conifer bog*	<i>P. m.-Larix laricina</i>	35-200 [3]
blue spruce*	<i>P. pungens</i>	35-200 [1]
pinyon-juniper	<i>Pinus-Juniperus</i> spp.	< 35 [2]
whitebark pine*	<i>P. albicaulis</i>	50-200 [1]
Rocky Mountain lodgepole pine*	<i>P. contorta</i> var. <i>latifolia</i>	25-300+ [4, 1, 5]
Sierra lodgepole pine*	<i>P. c.</i> var. <i>murrayana</i>	35-200
Jeffrey pine	<i>P. jeffreyi</i>	5-30
western white pine*	<i>P. monticola</i>	50-200
Pacific ponderosa pine*	<i>P. ponderosa</i> var. <i>ponderosa</i>	1-47
Rocky Mountain ponderosa pine*	<i>P. p.</i> var. <i>scopulorum</i>	2-10
Arizona pine	<i>P. p.</i> var. <i>arizonica</i>	2-10 [1]
aspen-birch	<i>Populus tremuloides-Betula papyrifera</i>	35-200 [3, 6]
trembling aspen (west of the Great Plains)	<i>P. tremuloides</i>	7-120 [1, 7, 8]
mountain grasslands	<i>Pseudoroegneria spicata</i>	3-40 (10**) [4, 1]
Rocky Mountain Douglas-fir*	<i>Pseudotsuga menziesii</i> var. <i>glauca</i>	25-100 [1]
coastal Douglas-fir*	<i>P. m.</i> var. <i>menziesii</i>	40-240 [1, 9, 10]
California mixed evergreen	<i>P. m.</i> var. <i>m.-Lithocarpus densiflorus-Arbutus menziesii</i>	< 35 [1]
redwood	<i>Sequoia sempervirens</i>	5-200 [1, 11, 12]
western redcedar-western hemlock	<i>Thuja plicata-Tsuga heterophylla</i>	> 200
western hemlock-Sitka spruce	<i>T. h.-Picea sitchensis</i>	> 200
mountain hemlock*	<i>T. mertensiana</i>	35 to > 200 [1]

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

**mean

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POSTFIRE REGENERATION STRATEGY (Stickney 1989)

Tall shrub, adventitious bud/root crown; initial off-site colonizer (off-site, initial community).

IMMEDIATE FIRE EFFECT ON PLANT

Scouler willow is sensitive to fire (Kovalchik *et al.* 1988) and may be top-killed (Steele and Geier-Hayes 1995). Scouler willow crown mortality following fire can vary from 0 to 100% depending on fire severity (Owens 1982). Severe fires that destroy the organic layer may result in 100% aboveground mortality (Lyon and Stickney 1976). However, even when aboveground plant parts are destroyed by fire, underground plant parts usually survive.

DISCUSSION AND QUALIFICATION OF FIRE EFFECT

Scouler willow may be resistant to fire, and there is a greater than 65% chance that 50% of the population on a site will survive or immediately reestablish after passage of a fire with an average flame length of 30.5 cm (Powell 1994). Mortality of Scouler willow to burning was low in different prescribed fire treatments in Montana ponderosa pine stands (Bedunah *et al.* 1995).

PLANT RESPONSE TO FIRE

Sprouting: Willows sprout quickly after fire if depth of the burn in the soil is low to moderate (Hawkes 1982). When fire is intense enough to kill live foliage but does not kill the vascular cambium, Scouler willow experiences vigorous epicormic sprouting from the root crown (Agee 1994, Agee 1996, Ayers *et al.* 1999, Bedunah *et al.* 1995, Bradley *et al.* 1992a, Bradley *et al.* 1992b, Crane *et al.* 1983, Geier-Hayes 1989, Hungerford 1986, Leege 1969, Leege 1978, Lyon 1966, Lyon 1971, Lyon 1984, McNeil 1975, Mueggler 1961, Mueggler 1965, Peek 1985, Smith and Fischer 1997, Steele and Geier-Hayes 1987, Steele and Geier-Hayes 1991, Stickney 1985, Stucker and Peek 1984, West 1982). Root crowns of Scouler willow are often so large that some buds always survive, except when disturbance is really severe (Morgan 1989). New shoots have been observed growing within days of a fire (Foote 1983), and rapid sprouting after burning results in low overall plant mortality (Ayers *et al.* 1999). Multiple sprouts result in increased Scouler willow density following fire (Keay and Peek 1980). Scouler willow has a high post-fire response rating; the species population will regain its pre-burn frequency and cover in five years or less (Powell 1994).

Scouler willow increases dramatically following a variety of burn intensities, especially on relatively moist sites (Powell 1994). Basal area after three years may exceed that on unburned sites (Viereck and Dyrness 1979). Dramatic increases in volume occur over the first 15 years post-fire (Stickney 1985). Sprout height growth may be dramatic, reaching up to 3 m after the first growing season (Crane and Fischer 1986, Leege 1969, Weaver 1987). Within three years after burning, plant crowns can average over 3.4 m in height (Leege 1969). Following a prescribed summer burn in Idaho, nearly 80% of height growth of Scouler willow over a 7-year period was attained within two growing seasons (Lyon 1971). Following a summer wildfire in northern Idaho, Scouler willow reached peak cover values within 8 years (Stickney 1986).

Scouler willow plants that experience severe canopy mortality apparently concentrate their nutrients into vigorous new growth more than plants which experience only light canopy mortality. Analysis of aboveground plant part mortality classes from controlled burns in Idaho revealed that Scouler willow plants which experience top-kill exhibit the most vigorous regrowth. Twig growth (length and weight) of Scouler willow following fire was 3 to 4 times greater on plants with greater than 50% canopy mortality than on plants with less than 50% canopy mortality (Owens 1982).

Seeding: Scouler willow also has the potential to regenerate from off-site seed sources (Bradley *et al.* 1992a, Bradley *et al.* 1992b, Crane and Fischer 1986, Gruell *et al.* 1986, Halpern 1989, Hungerford 1986, Mueggler 1965, Smith and Fischer 1997, Steele and Geier-Hayes 1987, Stickney 1985, Stucker and Peek 1984, Weaver 1917), and can establish in moist mineral soil post-fire (Bradley *et al.* 1992a, Bradley *et al.* 1992b, Fischer and Bradley 1987, Geier-Hayes 1989, Smith and Fischer 1997, Steele and Geier-Hayes 1987). Sowing Scouler willow seeds on different severity burns in upland black spruce sites in Alaska showed that germination occurred only on moderately (organic layers partially consumed) and severely (ash layer present, organic material in soil consumed or nearly so to mineral soil) burned seedbeds. Severely burned sites had the best germination percentages and represented the only burn severity class where Scouler willow seedlings survived past three years (Zasada *et al.* 1983). Scouler willow establishes quickly, but the rate of cover development or increase is slow (Stickney 1981, Stickney 1982, Stickney 1986).

Germinating seed originating from off-site plants often raises Scouler willow frequency far above what would be expected from on-site surviving plants (Lyon 1971, Lyon and Stickney 1976, Stickney 1986). Stickney (1986) observed that after a stand-replacing wildfire in northern Idaho, Scouler willow seedlings made up the majority of the shrub component of the vegetation. The importance of seedlings in the post-fire community was similarly observed by Lyon (1971). He recorded the post-fire density of Scouler willow plants for 7 years, summarized below (density = # of plants > 46 cm tall per 93 m²):

		Post-fire year						
	Pre-fire – 1963	1 - 1964	2 - 1965	3 - 1966	4 - 1967	5 - 1968	6 - 1969	7 - 1970
Density	0.3	0.1	0.2	0.2	0.6	2.4	3.6	4.4

FIRE MANAGEMENT CONSIDERATIONS

Prescribed fire is widely used as a wildlife management tool to rejuvenate decadent willow stands and stimulate sprouting. In areas where Scouler willow is scattered through the vegetation and in low vigour due to overbrowsing, prescribed fires that kill aboveground plant parts and expose mineral soils are favourable. This allows existing shrubs to sprout and creates favourable seedbeds for establishment of Scouler willow. In Douglas-fir/pinegrass habitat types of Montana, burning during late summer or fall exposes 30 to 50% of mineral soil (Gruell *et al.* 1982). A quick, hot fire maximizes sprouting while slower, longer burns cause more extensive damage and reduce browse (Haeussler *et al.* 1990). The deep root system and multistemmed growth of Scouler willow allows for higher tolerance to disturbance (Ayers *et al.* 1999), and it establishes rapidly in clearcut and heavily burned areas (Edgerton 1987, Goodrich 1992). Scouler willow is favoured by conditions on burned

areas; it is scarce on areas not subjected to fire but very abundant on broadcast burns (Mueggler 1961). However, broadcast burns do not always burn hot enough to duplicate the effects of severe wildfire and create an adequate seedbed for Scouler willow, which is favoured by light soil scarification (Marcum 1971, Simpson 1990, Steele and Geier-Hayes 1987, Steele and Geier-Hayes 1989a, Steele and Geier-Hayes 1989b, Steele and Geier-Hayes 1990). Competition may limit Scouler willow establishment; the frequency and percent cover of Scouler willow were significantly less ($p < 0.05$ and $p < 0.01$ respectively) on a burned, artificially seeded site than on a burned, unseeded site (Schoennagel and Waller 1999).

On sites in northern Idaho, Scouler willow had substantially higher cover on a 30-year-old burn than after any intensive silvicultural treatment (ranging from thinning to clearcut), with no presence in the control (Irwin 1976). However, after 30 to 40 years of fire exclusion in ponderosa pine forests, Scouler willow presence increased (Barrett and Arno 1982). Logging and fire suppression allowed vigorous development of Scouler willow in a Douglas-fir/ponderosa pine community (Gruell *et al.* 1982).

Prescribed fire enhances vigorous regrowth and germination of Scouler willow, and it is effective in increasing biomass (Bedunah *et al.* 1995). In the cedar-hemlock zone of Idaho, it generally produces the most cover of any shrub species on broadcast burned areas. Cover and frequency of Scouler willow is substantially higher on broadcast burns than on areas not broadcast burned, as are the mean height values (Mueggler 1961). Total shrub volume of Scouler willow in Douglas-fir stands increased from 15 to 80% two years post-fire (Lyon 1966). Mean canopy coverage of Scouler willow increased significantly ($p < 0.01$) following selective logging and low intensity broadcast burning (intensity ~ 127 kcal/m/s) (Armour *et al.* [n.d.]). In the first year following burning, Scouler willow may produce up to 28 times as many sprouts as the previous year (Leege and Hickey 1971, Miller 1976). Scouler willow may grow significantly wider and taller ($p < 0.05$) after fall burning than after spring burning (Leege and Hickey 1971). However, fall burning removes the following winter's food supply for animals, while spring burns have substantial regrowth by summer (Leege 1968).

After stand mechanical treatment and understory burning, Scouler willow was reduced by 9% from mechanical damage and an additional 16% from fire. The surviving plants were substantially more vigorous post-treatment (Arno *et al.* 1995, Arno *et al.* 1998). While modest Scouler willow mortality may result after overstorey removal and prescribed fire, the percentage of high vigour plants increases; in one study the amount of vigorous plants increased from 15% at pretreatment to 70% post treatment (Bedunah *et al.* 1999). In northern Idaho, burning at 5-year intervals did not result in decreased vigour (Leege 1979). Scouler willow was not markedly affected by burning until it suffered deep charring of the root crown. The following table presents the change in Scouler willow population characteristics in response to different treatments (% change is relative to pretreatment conditions) (Bedunah *et al.* 1999):

Treatment	Cover reduction	Mortality	High vigour plants
Control	1	3	15
Harvest only (shelterwood cut)	33	14	60
Low consumption (shelterwood cut and low consumption burn)	62	22	71
High consumption (shelterwood cut and high consumption burn)	58	26	69

Shelterwood cuts combined with prescribed burning in a ponderosa pine forest resulted in modest Scouler willow mortality; plants remaining in the harvest only and burned treatments had higher vigour than those in the control (Ayers *et al.* 1999, Bedunah *et al.* 1999).

A summary of Scouler willow's response to different types of disturbance is presented below (Steele and Geier-Hayes 1989b):

Type of disturbance:	Clearcut, no site prep	Shelterwood cut, mechanical scarification	Clearcut, mechanical scarification	Clearcut, broadcast burn	Stand destroying wildfire
Response:	major vegetative response	minor vegetative response	major vegetative response, minor response from seed	major vegetative response, minor response from seed	major vegetative response, minor response from seed

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Sambucus caerulea = *S. nigra* ssp. *caerulea*
Blue elderberry

FIRE ECOLOGY OR ADAPTATIONS

Blue elderberry is fire tolerant (Aro 1971). Although blue elderberry is favoured by fire, its frequency and cover remain relatively low in most areas where it has been studied (Isaac 1940, Mueggler 1965, Steen 1966). In Utah blue elderberry is often prominent in burned areas where it establishes from dormant seed (Plummer *et al.* 1968).

Blue elderberry is able to resprout (Preston 1948, Stanton 1974, Van Dersal 1938), and seed buried in seed banks germinates following fire (Heit 1967). Since it is short-lived and shade intolerant, blue elderberry is usually absent from the understorey of closed-canopy forests before fire occurs and must rely on seed banks for regeneration. There may also be occasional sprouts where plants had been growing in openings in the pre-fire forest (Morgan and Neuenschwander 1988). Idaho studies found elderberry seeds consistently throughout seed bank samples despite the lack of elderberry plants in the forest understorey (Kramer 1984, Kramer and Johnson 1987, Morgan and Neuenschwander 1988). Viable seed was found to a depth of 10 cm (Kramer and Johnson 1987). In the Blue Mountains of Oregon elderberry seed was not consistently present in samples from different stands in mixed forests, and it was found in the top 2.0 cm (Strickler and Edgerton 1976).

POSTFIRE REGENERATION STRATEGY

Tree with adventitious-bud root crown/soboliferous species root sucker; initial-offsite colonizer (off-site, initial community).

IMMEDIATE FIRE EFFECT ON PLANT

Fire effects vary with season, severity and intensity, site characteristics, and the age and vigour of the plants; however, fire generally kills aboveground parts of blue elderberry which then sprout vigorously from the root crown (Little 1979, Preston 1948, Steen 1965). A severe fire might expose and kill the root and stem buds from which sprouting occurs. Fire also scarifies buried seed, and germination usually occurs the first growing season following the fire (Heit 1967, Morgan and Neuenschwander 1988).

PLANT RESPONSE TO FIRE

Blue elderberry can respond to fire by resprouting, although only one fire study in California chaparral shows blue elderberry relying on resprouting (Sauer 1977). In that study no seedlings of elderberry were found.

Fire also scarifies the hard seed coat and stimulates germination of buried seeds (Heit 1967, Stanton 1974, Wright and Bailey 1982). Buried seeds respond to fire very quickly. In northern Idaho elderberry seedlings established the first growing season after the fire (Morgan and Neuenschwander 1988). There were no new seedlings after that year. There was some resprouting of shrubs that had been growing in stand openings as well.

In Oregon (Morris 1958, Morris 1970, Steen 1966) blue elderberry responded from buried seed more strongly on logged and burned plots than on logged but unburned plots. Blue elderberry dominated several burned plots and only one or two unburned plots during the third to fifth growing seasons. Other shrubs were dominant by the 11th to 16th seasons (Morris 1970, Steen 1965).

The severity of the fire appeared to make little or no difference to the frequency of elderberry seedlings in studies of high and low severity burns after clearcutting in northern Idaho (Morgan and Neuenschwander 1988, Mueggler 1965).

Repeated fires may reduce elderberry (Mueggler 1965). Isaac (1940) stated that blue elderberry spread slowly by seed and so was eliminated by a second fire.

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Sambucus racemosa
Red elderberry

FIRE ECOLOGY OR ADAPTATIONS

Red elderberry can sprout from rhizomes or root crowns following fire (Conrad and McDonough 1972, Hungerford 1986, Lyon 1966, Stickney 1980, Van Dersal 1938). It also has the ability to store seed in seed banks so that viable seeds can germinate following fire or other disturbance even if plants are absent from the pre-fire stand (Ahlgren 1979, Hungerford 1986, Kellman 1970, Kramer 1984, Kramer and Johnson 1987, Stickney 1980).

POSTFIRE REGENERATION STRATEGY

Tree with adventitious-bud root crown/soboliferous species root sucker; rhizomatous shrub, rhizome in soil; initial-offsite colonizer (off-site, initial community).

IMMEDIATE FIRE EFFECT ON PLANT

Fire effects vary with the season, severity and intensity of fire, site characteristics, and the age and vigour of the plants. However, fire generally kills aboveground parts of red elderberry which sprouts (Conrad and McDonough 1972, Van Dersal 1938). Sprouting can occur from dormant buds on the stems following a very light fire. If stem buds are killed in a higher severity fire, sprouting can occur from rhizome or root crown buds, depending on the variety (Conrad and McDonough 1972). A very severe fire might expose and kill the rhizome or root crown and thus the plant. Fire also scarifies buried seed, and germination usually occurs the first growing season following the fire (Heit 1967).

PLANT RESPONSE TO FIRE

All red elderberry can survive either by sprouting from root crowns or rhizomes, or by colonizing a site from seed stored in seed banks (Hungerford 1986). Repeated fires may reduce elderberries (Mueggler 1965). Because of its quick sprouting, red elderberry had the third largest volume of all shrubs by the second year following prescribed fire in central Idaho (Lyon 1966, Lyon 1971). Most studies simply record the presence of red elderberry after fire, so few details of its precise fire response are available (Collins 1980, Forsythe 1975, Lyon 1984, Schoonmaker and McKee 1988, Stickney 1980, Wittinger *et al.* 1977, Zager 1980, Zamora 1975).

FIRE MANAGEMENT CONSIDERATIONS

Browsing: red elderberry appears to be well adapted to both fire and browsing. Post-fire data from a 1936 fire in an aspen-paper birch (*Populus tremuloides*-*Betula papyrifera*) community are presented below (Risenhoover and Maass 1987).

Year	Exclosure (no browsing)		Control (browsing by Moose)	
	Stems/acre	Stems/ha	Stems/acre	Stems/ha
1949	13.4	33	10.1	25
1966	23.5	58	16.6	41
1982	6.5	16	23.5	58

Apparently fire stimulated regeneration of red elderberry. Without browsing, it increased until it was overtopped by aspen and paper birch, then began to decline. With browsing it increased more slowly, but browsing kept overall stem density and height reduced, and red elderberry was still increasing in density 46 years after the fire.

Competition: Ruth (1956) suggests first piling slash in brush areas to increase fuels, then burning and planting tree seedlings to give them a head start. He also suggests that chemicals may offer some control and that slashing competing vegetation may release tree seedlings. Combinations of chemicals, mechanical treatment, and fire apparently increased the number of red elderberry seedlings (Kelpsas 1978). The same problem occurs on moist sites in British Columbia, and immediate planting on those sites is recommended since few seral shrubs are present before burning (Eis 1981). Burning before harvest to induce shrub sprouting and seed germination in Oregon brushfields, then burning again to kill those seedlings and sprouts, has also been suggested (Martin 1982).

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Shepherdia canadensis
Soopolallie

FIRE ECOLOGY OR ADAPTATIONS

Sprouting from surviving root crowns and establishment from seed transported from off-site allow soopolallie to survive fire (Noste and Bushey 1987). As fire suppression culminates in closed-canopy, old-growth forests, fire generally increases soopolallie density and vigour, although full benefits may not be realized for at least 25 years (Noble 1985).

POSTFIRE REGENERATION STRATEGY

Tall shrub, adventitious-bud root crown.

IMMEDIATE FIRE EFFECT ON PLANT

Severe fires will consume all aboveground leaves and stems of soopolallie, while light to moderate fires will leave some stems standing (Noble 1985).

PLANT RESPONSE TO FIRE

Soopolallie is normally fire resistant but can be eliminated by fire (McLean 1968). As a result it is classified as moderately resistant to burning (McLean 1968, Noste and Bushey 1987). Following a Montana wildfire, regrowth of soopolallie was slow; 4 to 5 years were required for 25 percent of the eventual crown size to be obtained (Lyon 1984). Recurrent, low-intensity surface fires are closely linked to maintaining soopolallie density and vigour in stands with lodgepole pine and trembling aspen overstories, and dry upland meadows where it dominates the shrub layer (Noble 1985).

DISCUSSION AND QUALIFICATION OF PLANT RESPONSE

Following an intense wildfire in Colorado, essentially all aboveground vegetation in the perimeter died. The fire was the most intense where dominated by lodgepole pine, with lower intensities in areas dominated by subalpine fir and Engelmann spruce. Soopolallie increased rapidly following this fire due to sprouting from surviving roots. A combination of delayed sprouting and seeds originating from outside the burn was hypothesized to be responsible for an increase in frequency over the study period. Three years after the fire, soopolallie was mainly found on sites with a somewhat lower slope, a higher pre-fire tree basal area, and a higher number of pre-fire tree stems per acre. These factors appear to be conducive to soopolallie establishment and growth (Barth 1970).

FIRE MANAGEMENT CONSIDERATIONS

Low- to moderate-intensity fires may increase vigour and density of soopolallie in old-growth stands. Berry production may also be increased for several years after fire (Barth 1970, Noble 1985).

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Sorbus sitchensis
Sitka mountain-ash

FIRE ECOLOGY OR ADAPTATIONS

A closely related species, American mountain-ash, sprouts from the bole when top-killed by fire (Cooper 1928). Sitka mountain-ash may have this ability as well, but Sitka mountain-ash sprouting has not been documented in the literature.

Some areas in which Sitka mountain-ash occurs have long intervals between fires (Atzet and McCrimmon 1990, Patten and Oliver 1986). Cool, wet maritime forests may have fire return intervals of several hundred years or more (Barbour and Billings 1988).

POSTFIRE REGENERATION STRATEGY

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

IMMEDIATE FIRE EFFECT ON PLANT

Specific information on the immediate effect of fire on Sitka mountain-ash is not available in the literature. Since it is a small tree with thin bark, it may survive light-severity fire but is probably killed by severe fire. Mature mountain-ashes (*Sorbus* spp.) have been eliminated by fires at various locations throughout the United States and Canada (Cattelino 1980).

PLANT RESPONSE TO FIRE

Sitka mountain-ash was absent from burned sites but present on adjacent unburned sites 29 years following fire in alpine heath and krummholz communities in Washington (Douglas and Ballard 1971).

Mountain-ash (*Sorbus* spp.) sprouts and seedlings appeared in the first and second post-fire years after spring and summer wildfires and spring prescribed fires in Minnesota (Ahlgren 1959).

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Spirea douglasii
Douglas' spirea, Hardhack

FIRE ECOLOGY OR ADAPTATIONS

Douglas' spirea is moderately resistant to fire (Kovalchik *et al.* 1988) and sprouts readily from the stem base and rhizomes after fire (Boggs *et al.* 1990, Hansen *et al.* 1988, Kovalchik 1987, Kovalchik *et al.* 1988). In presettlement times, wildfires were "probably common" in Douglas' spirea communities of riparian areas in Montana and Oregon; soils were usually dry by mid-summer, allowing fires from adjacent uplands to encroach upon the stand (Hansen *et al.* 1988, Kovalchik 1987). Fires were probably infrequent in the thinleaf alder-Douglas' spirea association in Oregon (Kovalchik 1987).

POSTFIRE REGENERATION STRATEGY

Small shrub, adventitious-bud root crown; rhizomatous shrub, rhizome in soil.

IMMEDIATE FIRE EFFECT ON PLANT

Douglas' spirea is probably topkilled by most fires and killed by severe fires.

PLANT RESPONSE TO FIRE

The literature contains few reports describing Douglas' spirea after fire. Because of its sprouting ability, it probably survives most fires. In central British Columbia Douglas' spirea was present with shinyleaf spirea (*Spiraea betulifolia* var. *lucida*) in seral Engelmann spruce (*Picea engelmannii*)-fir (*Abies* spp.) stands. Relative density of the *Spiraea* spp. in plots 4 to 22 years after fire was 4 percent; in plots 37 to 75 years after fire, it was 9 percent (Garman 1929). In British Columbia, McMinn (1951) studied a site in a western hemlock/Sitka spruce forest burned in 1861 and again in 1931. By 1948 Douglas' spirea had formed dense thickets with salmonberry and thimbleberry.

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Symphoricarpos albus
Common snowberry

FIRE ECOLOGY OR ADAPTATIONS

Common snowberry is classified as a “survivor” (Lyon and Stickney 1976, Stickney 1989) and has high resistance to fire (Crane 1982, McLean 1968, Noste and Bushey 1987). It is a rhizomatous species with rhizomes buried 5-12.5 cm deep in mineral soil (Haeussler *et al.* 1990, Lackschewitz 1991, Stubbendieck *et al.* 1992). After fire has killed the top of the plant, new growth sprouts from these rhizomes (Miller 1977, Neuenschwander [n.d.], Williams *et al.* 1995). This rhizomatous growth response is highly variable and depends on conditions at specific sites (Coates and Haeussler 1986, Miller 1977, Noste and Bushey 1987). Regeneration from buried seed is favoured by fires of low severity and short duration that remove little of the soil organic level (Coates and Haeussler 1986, Hawkes *et al.* 1990).

Common snowberry occurs in a wide variety of community/habitat types and plant associations. There are many fire regimes included within these plant communities (Bradley *et al.* 1992a, Davis *et al.* 1980, Fischer and Bradley 1987, Fischer and Clayton 1983).

Community or Ecosystem	Scientific name of dominant species	Fire return interval in years
Pacific ponderosa pine*	<i>Pinus ponderosa</i> var. <i>ponderosa</i>	1-47 [1]
Rocky Mountain ponderosa pine*	<i>P. ponderosa</i> var. <i>scopulorum</i>	2-10 [1]
Rocky Mountain lodgepole pine*	<i>P. contorta</i> var. <i>latifolia</i>	25-300+ [2, 3]
Rocky Mountain Douglas-fir*	<i>Pseudotsuga menziesii</i> var. <i>glauca</i>	40-140 [4, 5]
coastal Douglas-fir*	<i>P. menziesii</i> var. <i>menziesii</i>	95-242 [6, 7]

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

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POSTFIRE REGENERATION STRATEGY

Rhizomatous shrub, rhizome in soil.

IMMEDIATE FIRE EFFECT ON PLANT

Common snowberry is top-killed by fire, but belowground parts are very resistant to fire (Lyon and Stickney 1976, Miller 1977, Neuenschwander [n.d.], Stickney 1989, Williams *et al.* 1995). Variable response to fire has been reported (Coates and Haeussler 1986, Miller 1977, Noste and Bushey 1987) but in general, light- to moderate-severity fires increase stem density (Bradley *et al.* 1992a, Coates and Haeussler 1986, Donnelly 1993), and common snowberry survives even severe fires (Bradley *et al.* 1992a, Crane 1982, Noste and Bushey 1987). To eliminate rhizomatous sprouting, fire intensity must be severe enough to kill the roots and rhizome system (Agee 1994).

PLANT RESPONSE TO FIRE

Common snowberry, as a rhizomatous sprouter, is among the first to recolonize a site after fire (Miller 1977). Growth in the first post-fire year varies, but is generally considered to be good. With light to moderate soil disturbance, sprouting will return common snowberry coverage in a year (Donnelly 1993) and common snowberry may produce fruit the first year (Bradley *et al.* 1992b). Sprout height can reach one-half to three-fourths of pre-burn stem height in the first year and equal pre-burn height in four years (Noste and Bushey 1987). Another source (Donnelly 1993) states common snowberry will grow 0.3 m the first year. Cover and volume measurements consistently exceed pre-burn values the second year (Noste and Bushey 1987) and canopy cover of common snowberry increases rapidly to a maximum in 3 to 5 years after a fire and may maintain this increased coverage (Coates and Haeussler 1986, Morgan and Neuenschwander 1988). Fire severity and soil moisture content at time of burning may determine damage to the rhizome and root system of common snowberry and be responsible for variation in recovery response (Hansen *et al.* 1988).

FIRE MANAGEMENT CONSIDERATIONS

Common snowberry is one of the first species to recolonize a post-burn site. New growth provides forage and often bears increased fruit crops. Cover is provided for small wildlife species and lush vegetation can protect soil surfaces from splash erosion, but can also offer severe competition to new tree seedlings. The living rhizome systems can be important in retaining nutrients released by fire (Miller 1977). One study (Anderson and Brooks 1975) found that planting grass seed to control erosion reduced coverage of common snowberry and other native shrubs on several burned sites in Oregon.

In Saskatchewan, to burn common snowberry it is recommended waiting four days after heavy rains. In addition, if spring burning, a minimum temperature of 13° C, wind speed of 3-19 km/h, and a maximum relative humidity of 50% is suggested. After burning, a 2-year wait is needed to build up enough fuel to burn again (Bailey 1978). Common snowberry may be susceptible to frequent burning (Smith 1957). If planting common snowberry, prompt, early spring planting is required or it may experience moisture stress in the short term (Donnelly 1993).

Common snowberry has a low surface to volume ratio and will have a high flammability if there are many dead stems (Brown 1976). It is capable of producing firebrand material. When located near fire control lanes, it should be red-flagged as spot fire potential (Neuenschwander [n.d.]).

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Symphoricarpos mollis
Creeping snowberry

FIRE ECOLOGY OR ADAPTATIONS

Creeping snowberry reproduces by rhizomes following fire (Meeuwig and Bassett 1983).

POSTFIRE REGENERATION STRATEGY

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

IMMEDIATE FIRE EFFECT ON PLANT

Creeping snowberry is usually top-killed by fire (Meeuwig and Bassett 1983). Some consider it to be a weak sprouter after fire because rhizomes in the humus layer can be destroyed (Neuenschwander [n.d.]).

PLANT RESPONSE TO FIRE

Creeping snowberry sprouts from rhizomes following fire (Meeuwig and Bassett 1983).

DISCUSSION AND QUALIFICATION OF PLANT RESPONSE

In northern Idaho creeping snowberry increased to a maximum 100 percent canopy cover five years following a clearcut and low severity broadcast burn; it increased to a 95 percent cover after a high severity broadcast burn (Fetcher *et al.* 1984).

FIRE MANAGEMENT CONSIDERATIONS

Creeping snowberry is a potential spot fire hazard when near firelines (Neuenschwander [n.d.]).

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Taxus brevifolia
Pacific yew

FIRE ECOLOGY OR ADAPTATIONS

Pacific yew is susceptible to heat damage and is most often associated with forests characterized by long fire-free intervals. Fire is rare in many old-growth forests of the Pacific Northwest (Scher and Jimerson 1989). Fire intervals in forests containing Pacific yew have been estimated as follows:

Location	Fire interval	Reference
Bitterroot Mtns. ID, MT	60 years	McCune and Allen 1985
c Western Cascades, OR	100 years	Means 1982
Siskiyou, OR	20 years	Atzet and Wheeler 1982
nw CA - low elev.	500-600 years	Scher and Jimerson 1989
nw CA - mid elev.	150-200 years	Scher and Jimerson 1989

Mature moist-site stands in which Pacific yew grows as scattered individuals are often considered relics from past fires (Johnson and Simon 1987). In parts of the Northwest, stand age ranges from 80 to 250 years where fire intervals average 70 to 120 years (Crawford and Johnson 1985). Similarly, in parts of western Montana, the age of Pacific yew averages approximately 210 years where fire replacement cycles are estimated at 150 years (Crawford and Johnson 1985). This suggests that the association of Pacific yew with moist microsites conveys some protection from fire.

After fire, Pacific yew slowly reestablishes by means of bird-dispersed seed as the overstorey canopy develops.

POSTFIRE REGENERATION STRATEGY

Secondary colonizer - offsite seed.

IMMEDIATE FIRE EFFECT ON PLANT

Pacific yew has thin bark and is sensitive to heat damage (Crawford and Johnson 1985, Scher and Jimerson 1989). Plants are generally killed by even light ground fires (McCune 1983), and this species is almost always eliminated from burned stands (Crawford and Johnson 1985, McCune and Allen 1985). In western Montana, Stickney (1981) observed that all plants were eliminated from burned stands. An abundance of Pacific yew can be equated with an absence of fire (Crawford and Johnson 1985).

Plants which occasionally survive fire do so because they occur in the wettest concavities which are relatively unaffected by fire (Atzet and Wheeler 1982).

PLANT RESPONSE TO FIRE

Pacific yew reoccupies burned areas through bird-dispersed off-site seed. Although vegetative regeneration is possible after mechanical disturbance, Pacific yew's susceptibility to heat damage makes post-fire sprouting unlikely or impossible.

This plant may require shelter provided by other species for reestablishment (McCune and Allen 1985) and typically recovers slowly. Hofmann (1917) observed that seedling germination was delayed for at least 6 years after a hot slash burn in northern Idaho. Pacific yew is rare on recently burned sites, even where it was a common component of pre-burn communities (Dyrness 1965, Edgerton 1987, Steen 1966). In a northern Idaho study, Pacific yew was present on 80 percent of the pre-burn plots but was absent from all plots during the first years after fire (Stickney 1980, Stickney 1981). In parts of the northern Rocky Mountains, it is described as the "only principal residual species eliminated by fire" (Stickney 1981).

FIRE MANAGEMENT CONSIDERATIONS

Fire history: Because of its sensitivity to fire, the age of Pacific yew can be used to estimate minimum stand age (McCune 1983). However, because it establishes after initial colonization, the oldest stem is often significantly younger than the age of the stand itself (McCune 1983).

Prescribed fire: Johnson and Simon (1987) recommend against prescribed fire in Pacific yew types. Although a light underburn will not damage the duff layer, yew may be adversely affected. Scher and Jimerson (1989) note that "although prescribed burning reduces the probability of catastrophic wildfires, precautions must be exercised to maintain biodiversity by protecting temperature-sensitive species" such as Pacific yew. In some areas, prescribed and/or wildfires can contribute to the depletion of yew populations (Scher and Jimerson 1989). Broadcast burning after clearcutting has virtually eliminated yew in some areas (Pierce and Peek 1984, Scher and Jimerson 1989).

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Toxicodendron diversilobum
Poison-oak

FIRE ECOLOGY OR ADAPTATIONS

Poison-oak's primary post-fire regeneration strategy is vigorous sprouting from the root crown and/or rhizomes (Cook 1959, Pacific Northwest Extension Service 1983, Wirtz 1982).

Fire is not required for poison-oak seed germination. Keeley (1987), however, reported a significant ($p < 0.001$) increase in germination when seeds were exposed to charate. Post-fire seedlings probably originate from both soil-stored seed and fresh seed dispersed by birds.

POSTFIRE REGENERATION STRATEGY

Tall shrub, adventitious-bud root crown; rhizomatous shrub, rhizome in soil; ground residual colonizer (on-site, initial community); and secondary colonizer - off-site seed.

IMMEDIATE FIRE EFFECT ON PLANT

Fire top-kills poison-oak (Christensen and Muller 1975, Cook 1959). Wirtz (1982) reported that an October, 1953, wildfire in a coastal sage scrub/grassland community near Berkeley, California, top-killed all poison-oak present, leaving only large branches and stumps.

Rhizomes on the soil surface are probably killed by all but light-severity fire, and shallowly buried rhizomes are probably killed by moderate to severe fire. More deeply buried rhizomes are probably not killed.

PLANT RESPONSE TO FIRE

Poison-oak sprouts vigorously from the root crown and/or rhizomes after fire (Christensen and Muller 1975, Conrad 1987, McKee 1990, Pacific Northwest Extension Service 1983, Sampson 1944). It sprouts in the first post-fire growing season, and for several years thereafter (Christensen and Muller 1975, Cook 1959, Sampson 1944). Poison-oak sprouts were noted the September following the July, 1985, Wheeler Fire on the Los Padres National Forest, California. The wildfire had spread into a riparian zone containing poison-oak; pre-fire poison-oak density was unknown. By post-fire year three, poison-oak sprouts dominated most burn plots in the riparian zone (Davis *et al.* 1989).

Westman and others (1981) estimated that poison-oak fails to sprout when fire reaction intensity exceeds 200 kcal/sec/m^2 . Their estimate was derived by modelling fire behavior of a backfire set in coastal sage scrub in the Santa Monica Mountains of California, and observing sprouting the following year. The coastal sage scrub had not burned for 20 to 22 years.

Poison-oak also establishes from seed after fire, although this response is not well documented in the literature. Poison-oak seedlings were observed following site preparation and prescribed burning of an interior live oak-blue oak woodland in Madera County, California. Pre-fire poison-oak seedling density was 0 percent; seedling density at post-fire year 1 was 42 per 8,712 square feet (Frost 1989).

DISCUSSION AND QUALIFICATION OF PLANT RESPONSE

Fire response is probably related to Poison oak's successional role in the plant community. Dense poison-oak thickets may develop in chaparral that is control burned several times (Burcham 1974). Poison-oak may become locally extinct in Douglas-fir forest, however, that is burned every four years for 20 years or more (Sampson and Jespersen 1963).

FIRE MANAGEMENT CONSIDERATIONS

Urushiol volatilizes when burned, and human exposure to poison-oak smoke is extremely hazardous (Kouakou *et al.* 1992). The smoke often poisons people who think they are immune to the plant (Pacific Northwest Extension Service 1983).

Poison-oak vines are a ladder fuel (Tsiouvaras *et al.* 1989).

Goats can be used as an alternative to prescribed fire for fire hazard reduction at urban-wildland interfaces. Near Oakland, California, goats were put on a Monterey pine-redgum (*Eucalyptus camaldensis*) forest with a heavy shrub understory and on an adjacent site where the forest was managed as a fuelbreak and had less shrub cover in the understory. Goat utilization of poison-oak in the fuelbreak was 67 percent, somewhat lower than utilization of toyon, California blackberry, and coyotebrush. Annual production of poison-oak biomass before goat browsing in the fuelbreak was 99 kg/ha; it was 33 kg/ha afterwards. Total biomass of forage species was significantly ($p < 0.05$) reduced (Tsiouvaras *et al.* 1989).

A stocking rate of 600 goats per hectare on the Oakland site broke the vertical live fuel continuity in the dense shrub stand. Initial goat browsing to reduce biomass and vertical fuel continuity could be followed up by prescribed fire (Tsiouvaras *et al.* 1989).

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Vaccinium ovatum
Evergreen huckleberry

FIRE ECOLOGY OR ADAPTATIONS

The role of fire in moist coastal forests, of which evergreen huckleberry is an integral understorey component, is poorly known (Huff 1984). Many sites currently occupied by this shrub are believed to have burned at relatively infrequent intervals during presettlement times (Stuart 1987).

Consequently, specific adaptations to fire may be poorly developed in this species. Evergreen huckleberry often sprouts after disturbances such as fire, but sprouting may primarily represent an adaptation to herbivory or mechanical damage.

Evergreen huckleberry can sprout from structures such as roots or root crowns after aboveground vegetation is destroyed by fire (Hooven 1969, Westman and Whittaker 1975). Limited seedling establishment may occasionally occur as birds and mammals disperse seed from offsite. However, it is important to note that seedling establishment is rare in most western huckleberries (*Vaccinium* spp.) (Martin 1979).

POSTFIRE REGENERATION STRATEGY

Tall shrub, adventitious-bud root crown; rhizomatous shrub, rhizome in soil; initial-offsite colonizer (off-site, initial community).

IMMEDIATE FIRE EFFECT ON PLANT

Although aboveground foliage is commonly killed by fire, underground portions of evergreen huckleberry often survive (Kienholz 1929, Kruckeberg 1982). Most western huckleberries (*Vaccinium* spp.) appear to be particularly vulnerable to hot, duff-consuming fires (Martin 1979). However, older, decadent individuals can sometimes be rejuvenated by light fires which do not damage underground regenerative structures (Martin 1979, Minore 1972). Seeds of most huckleberries are susceptible to heat and are presumably killed by fire (Martin 1979).

PLANT RESPONSE TO FIRE

Vegetative response: Evergreen huckleberry commonly sprouts after aboveground foliage is damaged or destroyed by fire (Hooven 1969, Kienholz 1929, Kruckeberg 1982). Most species of huckleberry (*Vaccinium* spp.) regenerate from basal sprouts or from underground structures such as roots or rhizomes (Rowe and Scotter 1973). Westman and Whittaker (1975) report that evergreen huckleberry has a “massive root crown” rather than rhizomes. However, Hooven (1969) notes that post-fire sprouting from roots can also occur.

Recovery of evergreen huckleberry can be relatively rapid wherever sprouting occurs. Evergreen huckleberry, Pacific dogwood (*Cornus nuttallii*), vine maple, trailing blackberry (*Rubus ursinus*), Oregon grape (*Berberis* spp.), and bracken fern (*Pteridium aquilinum*) dominated within one year after a large August fire in a Douglas-fir forest of western Oregon (Hooven 1969).

Seed: Seedbanking does not appear to represent an important regenerative strategy in western huckleberries. Some seed may be brought onto the site by bird and mammal dispersers. Reestablishment by seed, if it occurs at all, is likely to be slow.

FIRE MANAGEMENT CONSIDERATIONS

Wildlife: Evidence suggests that fire suppression may be having an adverse impact on bear habitat in some areas (Unsworth *et al.* 1989, Zager 1980). Once productive seral berry fields are now being invaded by conifers. Since plants beneath a forest canopy generally produce few berries, fruit production has been steadily declining (Minore 1972). Logging treatments which include severe soil scarification or slash burns may also reduce berry yields. Even where timber harvest favours berry production, lack of cover in early years can limit bear use. However, wildfires often create diverse habitat mosaics (Zager 1980) which incorporate elements of hiding cover and favour bear use.

Prescribed fire: Flower buds tend to be more numerous on new shoots, and periodic removal of old shoots can increase flower and fruit production in many species of huckleberries (*Vaccinium* spp.) (Martin 1979). Prescribed fire has long been used to rejuvenate commercial low sweet blueberry (*V. angustifolium*) fields and to increase fruit yield (Martin 1979, Minore 1972). Spring burns, conducted when the soil is moist, are generally most effective in promoting huckleberry fruit production (Martin 1979, Kautz 1987).

Berry production: Berry production in most western huckleberries is generally delayed for at least five years after fire (Martin 1979). On some sites, production may be reduced for 20 to 30 years or longer (Martin 1979).

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Tree Layer

Abies grandis Grand fir

Adaptations to fire: Grand fir is moderately resistant to frequent surface fire. It has thin bark and is easily killed when young, but the bark is thick enough at maturity (about 5 cm) to provide resistance to low- and moderate-severity fires (Agee 1996a, Starker 1934, Crane and Fischer 1986, Davis *et al.* 1980, Fisher 1935, Hall 1976, Hall 1998). Compared to other Pacific Coast conifers, it is less fire resistant than coastal Douglas-fir but more so than western hemlock and Pacific silver fir (Flanagan 1996). Inland, it is less fire resistant than western larch, Pacific ponderosa pine, and Rocky Mountain Douglas-fir; about the same as white fir; and more fire resistant than western white pine, subalpine fir, Engelmann spruce (*Picea engelmannii*), and Rocky Mountain lodgepole pine (Rickard 1970, Flanagan 1996). Fire-scarred grand fir are susceptible to heart rot (Aho 1977, Antos and Habeck 1981, Arno and Hammerly 1977, Johnson and Simon 1987, Filip *et al.* 1983).

Grand fir does not survive crowning or severe fire. Its low, dense branching habit, flammable foliage, and tendency to develop dense stands with heavy lichen growth increase the likelihood of torching and mortality from crown fire (Crane and Fischer 1986, Davis *et al.* 1980, Fisher 1935, Starker 1934).

Fire strongly influences grand fir's ecological niche and successional role (Hall 1983, Johnson and Simon 1987). In coastal British Columbia grand fir occurs in areas of relatively low summer rainfall and high summer temperatures, suggesting that its range may be restricted to sites with higher fire frequencies compared to moister surrounding forests with longer fire return intervals (Schmidt 1957). On many Pacific Northwest sites, however, grand fir only dominates sites where fire is excluded. Fire history studies show that Garry oak, Port-Orford-cedar (*Chamaecyparis lawsoniana*), Pacific ponderosa pine, western larch, and/or coastal Douglas-fir were maintained as site dominants by frequent surface fires that eliminated young grand fir (Hall 1977, Hall 1983, Green and Jensen 1991). After cessation of Native American burning in the Willamette Valley of Oregon (around 1850), grand fir has successionaly replaced Garry oak and coastal Douglas-fir on most sites. Coastal Douglas-fir retains dominance only on the driest sites in the valley (Cole 1977). Although grand fir is not usually seral on sites with frequent fires, it may be either climax or seral on sites that experience infrequent crown fires (Hall 1983).

Fire regimes: Fires in grand fir types were historically of mixed severity, with fire behaviours ranging from frequent low-severity, nonlethal surface fire to infrequent, stand-replacing crown fire (Antos and Habeck 1981, Agee 1996a, Agee *et al.* 1990, Barrett 1982, Mutch *et al.* 1993, Arno 1980, Smith and Fischer 1997, Barrett and Arno 1991). The grand fir series can roughly be divided into warm/dry types and warm/moist types. In warm/dry types, the historical fire regime was frequent (5-50 years), low-severity fire that favoured Pacific ponderosa pine and western larch (Arno 1976, Camp 1995, Steele *et al.* 1986, Weaver 1959). For example, a mean fire return interval of 47 years is reported for the Blue Mountains (Weaver 1959), with a range of 33 to 100 years (Wischnofsky and Anderson 1983). Historically, fire severity in grand fir types of the Blue Mountains was often moderate, with a wider range of fire severities than Douglas-fir types (Agee 1996a). Dry grand fir/graminoid types with understories of elk sedge or pinegrass typically experienced frequent surface fires (10- to 25-year intervals) (Agee and Maruoka 1994, Maruoka 1994, Weaver 1961).

Fire regimes in northern Idaho and western Montana were historically similar to those in the Blue Mountains, but fire return intervals showed a wider range (3-200 years) (Antos and Habeck 1981, Arno 1976). On a dry site in the Bitterroot National Forest of western Montana, Arno (Shearer 1982, Steele and Geier-Hayes 1987) reported a mean fire return interval of 17 years between 1735 and 1900, with a range of 3 to 32 years. He attributed the short fire return interval to the relative scarcity of the grand fir series there, so that grand fir had “fire frequencies much like those in surrounding major series” such as Rocky Mountain Douglas-fir (Arno and Petersen 1983, Arno 1980).

Mixed-severity fires with longer return intervals (25-100 years) were more common on cooler, moderately moist grand fir types with Rocky Mountain maple, Pacific yew, oak fern (*Gymnocarpium dryoptera*), or sword fern (*Polystichum munitum*) understories. Fire regimes shifted to moderate severity on these wetter sites, and stand-replacement fires were more common (Agee 1996a, Agee and Maruoka 1994, Camp 1995, Arno 1980, Smith and Fischer 1997, Barrett and Arno 1991). Fire-scarred, mature grand fir trees in northern Idaho have withstood moderate-severity surface fires once or twice a century (Barrett 1982). Camp (1995) reported that fire history was complex on warm/moist forests of the eastern slope of the Cascade Range in Washington, with evidence of both frequent, low-severity fires and infrequent severe, stand-replacing fires. Sites experiencing severe fire often escaped fire through two to three surface fire cycles that occurred in surrounding forest. These long-unburned sites developed into multi-layered grand fir forest that functioned as old-growth fire refugia until the next severe fire cycle. Presettlement sites of fire refugia occurred most often on north-facing aspects, benches, valley bottoms, and stream confluences and headwalls (Camp 1995, Camp *et al.* 1997).

Long-interval (> 100 years), severe fires were most common on wet grand fir habitat types (Arno 1980, Smith and Fischer 1997, Barrett and Arno 1991). Moist types are highly productive and have large fuel loads (Hanley 1976). Barrett (1982) found that fires in grand fir of the Clearwater National Forest in northern Idaho were usually large and exhibited behavior of (1) moderate to severe surface fires that killed the grand fir but left a few fire-resistant seral conifers, and (2) running crown fires (with individual runs of several hundred acres) that killed entire stands. Even given this extreme fire behavior, there was also evidence of low-severity surface fires, particularly on north slopes, that scarred but did not kill grand fir (Barrett 1982). Barrett and Arno (1991) found that patchy, stand-replacement fires with a mean return interval of 119 years typified fire regimes in Rocky Mountain Douglas-fir/grand fir habitats of the Selway-Bitterroot Wilderness in northern Idaho. A minority of stands experienced mixed-severity fire of nonuniform spread. Long-interval, stand-replacing fire also occurred historically in the relatively moist Swan Valley of western Montana. The Swan Valley also shows evidence of a mixed fire regime, with a mosaic of stands of varying age and composition (Antos and Habeck 1981). Fire return intervals in the Swan Valley ranged from 20 to 300+ years, with a mean of 150 years (Antos and Habeck 1981, Antos and Shearer 1980).

Fire regimes for plant communities and ecosystems where grand fir is a common associate are summarized below.

Community or Ecosystem	Dominant Species	Fire Return Interval Range in Years
Pacific ponderosa pine*	<i>Pinus ponderosa</i> var. <i>ponderosa</i>	1-47 [1]
Rocky Mountain lodgepole pine*	<i>P. contorta</i> var. <i>latifolia</i>	25-300+ [2, 3]
Rocky Mountain Douglas-fir*	<i>Pseudotsuga menziesii</i> var. <i>glauca</i>	25-100 [1]
coastal Douglas-fir*	<i>Pseudotsuga menziesii</i> var. <i>menziesii</i>	95-242 [4, 5]
trembling aspen (west of the Great Plains)	<i>Populus tremuloides</i>	7-100 [6, 7]
redwood	<i>Sequoia sempervirens</i>	7-25 [8, 9]

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

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POSTFIRE REGENERATION STRATEGY (Stickney 1989):

Tree without adventitious bud/root crown; initial off-site colonizer (off-site, initial community); crown residual colonizer (on-site, initial community); secondary colonizer - off-site seed.

Young grand fir have thin bark and are easily killed by fire (Johnson 1998, Keane *et al.* 1990). Trees under 10.2 cm diameter at ground level are most susceptible to direct fire mortality (Hall 1976, Hall 1998). The bark thickens as trees age, and mature trees are moderately resistant to fire (Arno 1980, Camp 1995). Ground fires burning into the duff injure shallow roots and may kill even mature trees (Crane and Fischer 1986, Fischer 1981, Foiles *et al.* 1990, Johnson and Simon 1987, Shiplett and Neuenschwander 1994).

Baker (1929) found that grand fir seedlings in the laboratory were killed by exposing the stems to temperatures of 49° C for 10 minutes.

DISCUSSION AND QUALIFICATION OF FIRE EFFECT

Because grand fir wood does not contain decay-inhibiting properties nor exude pitch over wounds, trees that survive fire are susceptible to the entry of decay fungi through fire scars and stimulation of dormant decay by fire injury. The problem is more serious east of the Cascade Range crest because of the ubiquitousness of Indian paint fungus in the eastern portion of grand fir's range. (Aho 1977, Aho 1982, Antos and Habeck 1981, Arno and Hammerly 1977, Arno and Petersen 1983, Johnson and Simon 1987, Filip *et al.* 1983).

PLANT RESPONSE TO FIRE

Grand fir regeneration is common after fire (Antos and Habeck 1981, Schmidt 1957). Seedlings establish on burns mostly from off-site seed sources (Aho 1982, Johnson 1998, Topik *et al.* 1986). Mature grand fir that survive a fire provide an on-site seed source (Aho 1982). Fire provides a favourable seedbed. When different substrates were compared, grand fir germination was best on ash or mineral soil (Fisher 1935, Shearer 1982); however, seedling mortality may be higher on burned soils due to

higher surface temperatures on blackened compared to unburned soils (Seidel and Cochran 1981). Seedlings often establish in the first few post-fire years. For example, following a severe wildfire in a mature grand fir/queencup beadlily association in the Blue Mountains of Oregon, grand fir seedlings were first noted in study plots at post-fire year 5 (Johnson 1998). Following the Sundance Fire in northern Idaho, grand fir seedlings were first noted in post-fire years 4 to 9, with time of first emergence varying among study plots (Stickney 1985). Because grand fir seedlings are not as drought tolerant as many conifer associates, grand fir establishment is sometimes slow or delayed by drought, but grand fir is usually established as component of seral vegetation by 20 to 30 years after fire (Miller and Miller 1976, Zamora 1975). Grand fir regeneration is also common after fire thins a dense overstorey (Antos and Habeck 1981). As a shade-tolerant tree, grand fir continues to establish until canopy closure in late succession (Keane *et al.* 1990).

Low-severity fire may have little effect on grand fir. A "light" fire in an early-seral grand fir/twinflower association the Oregon Blue Mountains killed the pole-sized overstorey conifers (grand fir, Rocky Mountain Douglas-fir, and Rocky Mountain lodgepole pine), but their relative coverage remained similar to pre-fire levels during early post-fire seedling establishment. Pre-fire coverage of grand fir was 5% compared to 3% coverage at post-fire year 5 (Johnson 1998).

DISCUSSION AND QUALIFICATION OF PLANT RESPONSE

In northeastern Oregon, three wildfire sites were selected to study fire's effects on late-seral grand fir/big huckleberry associations. Two sites were severely burned, and one site was lightly underburned. The severe fires killed all overstorey and understorey grand fir. The low-severity fire was continuous with fire scorching only the basal portion of the large-diameter (76-102 cm) trees. The low-severity fire reduced overstorey grand fir coverage from 55 to 40%, and the understorey was reduced from 10 to 5%. A thicket of grand fir saplings was reduced by 30% (Johnson 1998).

Fire may aid grand fir regeneration on most sites, but grand fir may regenerate poorly after fire on south-facing slopes or on dry sites (Feller 1983, Hawkes *et al.* 1990). In a grand fir/*pachistima* habitat in the Coeur d'Alene River drainage of northern Idaho, grand fir established readily on unburned sites following clearcutting, but required shade for regeneration on clearcut and burned sites (Wittinger *et al.* 1977).

FIRE MANAGEMENT CONSIDERATIONS

Historically, low-severity surface fires and patchy, mixed-severity fires killed young grand fir and Douglas-fir in the understorey while favouring early-seral, fire-tolerant tree species including Garry oak, ponderosa pine, Douglas-fir, and western larch (Green and Jensen 1991, Kauffman 1990, Weaver 1959, Weaver 1961). Once open, park-like stands are being invaded by grand fir, other firs, and Douglas-fir, resulting in

poor regeneration of early-successional trees (Green and Jensen 1991, Arno *et al.* 1995, Camp 1995, McCune 1983, Weaver 1959, Weaver 1961). Fuel loads have increased and produced very fire-prone communities with high probabilities of crown fires (Hall 1983, Hall 1998). Fire exclusion has altered forest structure and affected understorey vegetation (Kauffman 1990). Stands have developed understories or multiple canopy layers of grand fir and other shade-tolerant species (Camp 1995). These understories may be extremely dense, often thousands of stems per acre. Without fire, understorey grand fir usually develop into thickets of stressed trees (Arno *et al.* 1995).

Underburning in grand fir stands reduces fuels and permits regeneration of pines and other fire-tolerant trees (Mohr and Kuchenbecker 1989). Several factors are considered in predicting and modelling mortality of grand fir and other conifers (Ryan and Reinhardt 1988). Underburning grand fir on steep ground generally results in high mortality (Bickford 1983). Mortality is also dependent on bark thickness, stand structure, and duration of fire. Peterson and Ryan (1986) present a model for predicting mortality of grand fir, subalpine fir, and Douglas-fir in the northern Rocky Mountains based on tree morphology, stand structure, and fire characteristics.

Fuels and fire behavior: Grand fir forests are usually highly productive, which leads to rapid fuel accumulation (Habeck and Mutch 1973). Mid-slope forests such as those occupied by grand fir are more prone to severe, stand-replacing fire than forests at lower or higher elevations (Arno 1980, Camp 1995). Habeck (1976) found that fuel loads in old-growth grand fir (> 250 years of age) in the Selway-Bitterroot Wilderness often exceeded 100 t/ha, with litter and duff layers averaging 12 cm. Such highly productive sites are subject to reburn. Barrett (1982) defines a reburn as a fire that burns in heavy downed woody fuel resulting from tree mortality in a previous fire, occurring when tree regeneration is in the seedling or sapling stage. Barrett (1982) found that on the Clearwater National Forest, the driest aspects were most likely to reburn, but potential for reburn was also present on productive north slopes.

Average heat release of live grand fir has been summarized as follows (Kelsey *et al.* 1979):

Wood	Bark	Twigs	Foliage
BTU/lb (Mj/kg)	BTU/lb (Mj/kg)	BTU/lb (Mj/kg)	BTU/lb (Mj/kg)
8,300 (19.31)	9,641 (22.43)	8,894 (20.69)	9,497 (22.09)

In a literature summary, Minore (1979) reports that fire spread in fresh grand fir slash is intermediate compared to slash of seven associated conifers. Fire spread in 1-year-old grand fir slash is slower than fire spread in 1-year-old slash of all associated conifers except western larch, in which fire spread is similar. Photo guides have been prepared for appraising slash fuels in grand fir forests of northern Idaho, and for downed woody fuels in grand fir, western larch, and Douglas-fir forests of Montana (Fischer 1981, Koski and Fischer 1979).

Fuel models: Brown (1978) and Moeur (1981) present equations for predicting crown width and foliage biomass of grand fir and associated conifers.

Keane and others (1990) predict that decomposition rates of litter in grand fir-dominated forests are an order of magnitude less than in ponderosa pine or Douglas-fir forests.

Fire behavior models: Agee (1996b) provides models for predicting stand conditions that initiate crown fire in grand fir and other western forest types based on the critical surface fire intensity needed to initiate crowning (equation 1), and for identifying conditions that allow crown fire to spread (equation 2):

$$I_o = (Czh)^{3/2} \quad (\text{equation 1})$$

where

I_o = critical surface intensity

C = 0.010 (constant)

z = crown base height

h = heat of ignition (largely a function of crown moisture content), and

$$E = Rdh \quad (\text{equation 2})$$

where

E = net horizontal heat flux, kW/m²

R = rate of spread, m/sec

d = bulk density of crown, kg/m³

h = heat of ignition, kJ/kg

Restoration: The general objective of restorative management is to develop open stands of seral conifers resembling stands maintained by historic fire regimes. Restoring presettlement stand conditions and fire regimes to grand fir habitats also reduces stand susceptibility to outbreaks of insect and fungi (Carlson and Wulf 1989). Because of dense understories of grand fir and other shade tolerant conifers, it is usually necessary to begin restoration with a "low thinning" treatment that removes excess understorey and weak understorey trees. Low-severity prescribed fire is then conducted to reduce fuel loadings, kill understorey conifers, and promote herbaceous and shrub species in the understorey. Once thinning and burning are accomplished, the stand can be maintained by periodic underburning alone, at 15- to 30-year intervals (Arno *et al.* 1995). A selection cutting that retains many of the dominant overstorey trees also helps maintain open-stand conditions when tree harvesting is an objective (Arno *et al.* 1995, Cissell *et al.* 1998).

Range productivity: Some sites are less useful for livestock grazing as a result of fire exclusion (Kauffman 1990). Hall (1977) reported that in the Blue Mountains of Oregon, forests that have been maintained as ponderosa pine/pinegrass by periodic underburning (< 50% crown cover) produce 562 to 675 kg/ha of pinegrass. In forests where fire has been excluded and grand fir and Douglas-fir have established a subcanopy (> 80% cover), pinegrass production drops to 56 to 112 kg/ha (Hall 1977).

Wildlife habitat: Historically, fire refugia sites in the grand fir and Douglas-fir-grand fir series were important habitat for late-successional animals such as northern spotted owl and American martin (Camp 1995, Camp *et al.* 1997, Carey *et al.* 1992). Camp and others (1997) provide a model for predicting occurrence of fire refugia based on topographic and physiographic variables.

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Acer glabrum
Rocky Mountain maple

FIRE ECOLOGY OR ADAPTATIONS

Rocky Mountain maple has been characterized as fire dependent, and may decline with fire exclusion (Arno and Ottmar 1994). Prolific sprouting and wind dispersal of seed of Rocky Mountain maple facilitate rapid revegetation of burned areas (Cholewa 1977, Hungerford 1986, Larsen 1929, Powell 1988, Shiplett and Neuenschwander 1994). In trembling aspen/Rocky Mountain maple communities in Colorado, this rapid regeneration results in post-fire vegetation that quickly resembles the pre-fire community (Powell 1988). Following fire, enhanced growth of Rocky Mountain maple may result in moderate growth loss of conifers and mortality of shade-intolerant conifers (Donnelly 1993). Due to aggressive competition, it may also interfere with conifer seedling establishment (Shearer and Halvorson 1967).

Rocky Mountain maple occurs as a major component or dominant in seral shrubfields in the northern Rockies. These shrubfields result from canopy removal by repeated severe fires (Chadwick 1977, Christianson *et al.* 1984, Coffman 1975, Gaffney 1941, Gratkowski 1961, Hickey 1971, Hooker and Tisdale 1974, Leege 1968, Lyon 1979, Moir *et al.* 1997, Smith and Fischer 1997, Whitford 1905, Yurich 1974). Seral shrubfields have also been maintained with prescribed fire (Leege 1979). A lack of seed combined with increased soil temperatures and moisture stress inhibit tree regeneration and maintain the shrubfields. Fuels in persistent shrubfields consist primarily of the shrubs themselves with little large downed woody material and low litter amounts; in one study conducted in northern Idaho, fuel loading averaged 19.7 tons/acre. Persistent shrubfields may burn in any season; if fuels are continuous and dry, spring fires spread readily, and in summer, hot and dry conditions are exacerbated by nighttime inversions (Smith and Fischer 1997).

Fire regimes for plant communities and ecosystems in which Rocky Mountain maple occurs are summarized below.

Community or Ecosystem	Dominant Species	Fire Return Interval Range (years)
silver fir-Douglas-fir	<i>Abies amabilis</i> - <i>Pseudotsuga menziesii</i> var. <i>menziesii</i>	> 200
grand fir	<i>Abies grandis</i>	35-200 [1]
California chaparral	<i>Adenostoma</i> and/or <i>Arctostaphylos</i> spp.	< 35 to < 100
sagebrush steppe	<i>Artemisia tridentata</i> / <i>Pseudoroegneria spicata</i>	20-70 [2]
mountain big sagebrush	<i>Artemisia tridentata</i> var. <i>vaseyana</i>	20-60 [3, 4]
coastal sagebrush	<i>Artemisia californica</i>	< 35 to < 100
California montane chaparral	<i>Ceanothus</i> and/or <i>Arctostaphylos</i> spp.	50-100 [2]
curlleaf mountain-mahogany*	<i>Cercocarpus ledifolius</i>	13-1000 [5, 6]
mountain-mahogany-Gambel oak scrub	<i>Cercocarpus ledifolius</i> - <i>Quercus gambelii</i>	< 35 to < 100
western juniper	<i>Juniperus occidentalis</i>	20-70
Rocky Mountain juniper	<i>Juniperus scopulorum</i>	< 35 [2]
western larch	<i>Larix occidentalis</i>	25-100
Engelmann spruce-subalpine fir	<i>Picea engelmannii</i> - <i>Abies lasiocarpa</i>	35 to > 200 [1]
black spruce	<i>Picea mariana</i>	35-200 [7]
blue spruce*	<i>Picea pungens</i>	35-200 [1]
pinyon-juniper	<i>Pinus</i> - <i>Juniperus</i> spp.	< 35 [2]
Rocky Mountain lodgepole pine*	<i>Pinus contorta</i> var. <i>latifolia</i>	25-300+ [8, 1, 9]
Sierra lodgepole pine*	<i>Pinus contorta</i> var. <i>murrayana</i>	35-200 [1]

Colorado pinyon	<i>Pinus edulis</i>	10-49 [2]
western white pine*	<i>Pinus monticola</i>	50-200
Pacific ponderosa pine*	<i>Pinus ponderosa</i> var. <i>ponderosa</i>	1-47
interior ponderosa pine*	<i>Pinus ponderosa</i> var. <i>scopulorum</i>	2-10
Table Mountain pine	<i>Pinus pungens</i>	< 35 to 200 [10]
trembling aspen (west of the Great Plains)	<i>Populus tremuloides</i>	7-120 [1, 11, 12]
Rocky Mountain Douglas-fir*	<i>Pseudotsuga menziesii</i> var. <i>glauca</i>	25-100 [1]
coastal Douglas-fir*	<i>Pseudotsuga menziesii</i> var. <i>menziesii</i>	40-240 [1, 13, 14]
California mixed evergreen	<i>Pseudotsuga menziesii</i> var. <i>m.</i> - <i>Lithocarpus densiflorus</i> - <i>Arbutus m.</i>	< 35
California oakwoods	<i>Quercus</i> spp.	< 35 [1]
oak-juniper woodland (Southwest)	<i>Quercus-Juniperus</i> spp.	< 35 to < 200 [2]
canyon live oak	<i>Quercus chrysolepis</i>	<35 to 200
blue oak-foothills pine	<i>Quercus douglasii</i> - <i>Pinus sabiana</i>	<35
Garry oak	<i>Quercus garryana</i>	< 35 [1]
redwood	<i>Sequoia sempervirens</i>	5-200 [1, 15, 19]
western redcedar-western hemlock	<i>Thuja plicata</i> - <i>Tsuga heterophylla</i>	> 200
western hemlock-Sitka spruce	<i>Tsuga heterophylla</i> - <i>Picea sitchensis</i>	> 200
mountain hemlock*	<i>Tsuga mertensiana</i>	35 to > 200 [1]

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

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POSTFIRE REGENERATION STRATEGY (Stickney 1989)

Tall shrub, adventitious bud/root crown; initial off-site colonizer (off-site, initial community); secondary colonizer (on-site or off-site seed sources).

IMMEDIATE FIRE EFFECT ON PLANT

Though top-killed by fire, Rocky Mountain maple generally has low susceptibility to fire due to its ability to survive via resprouting from the root crown (Monsen and McArthur 1985, Stickney 1985, Stickney 1991, Wasser 1982). Rocky Mountain maple may remain on sites where most of the understory vegetation is removed by fire (Aulenbach and O'Shea-Stone 1983). However, after moderate to severe fire, survival of Rocky Mountain maple may be substantially reduced (Bradley *et al.* 1992a, Bradley *et al.* 1992b, Coates and Haeussler 1986, Hawkes *et al.* 1990). Generally, it is only temporarily reduced by fire because often the root crowns are so large that some buds always survive (Morgan 1989).

PLANT RESPONSE TO FIRE

Rocky Mountain maple is generally favoured by fire, recovering and increasing by resprouting vigorously from the root crown (Bradley *et al.* 1992b, Crane and Habeck 1982, Hawkes *et al.* 1990, Laursen 1984, Leege 1972b, Leege *et al.* 1966, Lyon 1966, Mitchell 1984, Monsen and McArthur 1985, Reinhardt *et al.* 1994, Smith and Fischer 1997, Stickney 1986, Wasser 1982). Resprouting of Rocky Mountain maple was observed one year after a severe fire in northern Montana (Kapler 1976). After a severe crown fire in Idaho, Rocky Mountain maple resprouted within the first growing season (Steele and Geier-Hayes 1991). The greatest resprouting response, however, is observed following light severity fires, with the least response occurring after a high severity burn (Donnelly 1993). Following a broadcast burn in Idaho, Rocky Mountain maple regenerated its entire pre-burn crown volume in the first post-burn year (Zamora 1975). Following fire, Rocky Mountain maple may grow 1.2-1.5 m in three years (Donnelly 1993). Rocky Mountain maple usually increases following fire, though survival and response may be reduced by severe fire (Bradley *et al.* 1992a, Bradley *et al.* 1992b, Smith and Fischer 1997, Stickney 1982).

Rocky Mountain maple also colonizes sites after fire via wind-dispersed seed (Smith and Fischer 1997).

DISCUSSION AND QUALIFICATION OF PLANT RESPONSE

Stickney (1981) found Rocky Mountain maple to be sensitive to fire, with low survival, slow redevelopment, and slow recovery of percent cover on sites in Montana burned by wildfire and on broadcast burned sites.

FIRE MANAGEMENT CONSIDERATIONS

Rocky Mountain maple was a principal component of slowly recovering open sites in western Montana after stand-replacing fire and broadcast burning (Stickney 1981). It is highly competitive with tree species after burning (Donnelly 1993, Shearer and Schmidt 1982). Fire results in a substantial increase in Rocky Mountain maple sprouts (Steele and Geier-Hayes 1995).

Prescribed fire may initially reduce the percent cover of Rocky Mountain maple substantially, though in several years it is expected to approach or surpass pretreatment coverage (Simmerman *et al.* 1991). Percent cover of Rocky Mountain maple may increase dramatically over pre-burn levels within five years of burning, however, variable results may be expected (Coates and Haeussler 1986).

Rocky Mountain maple may experience an increase in crown volume following disturbance, resprouting more after burning than after logging treatments (Cholewa 1977). Within three years, crown diameters of Rocky Mountain maple may be equal to or greater than diameter prior to burning (Leege 1969). However, other studies have found that two years after burning, Rocky Mountain maple only recovered 8% of its pre-fire crown volume, and proportion of total shrub volume had decreased from

80% pre-fire to 14% post-fire (Lyon 1966). Three out of four harvest and burning treatments in Montana were found to effectively reduce Rocky Mountain maple shrub volume, though it began increasing again after two years post-treatment. The fourth treatment (no burning) left more individuals intact and they subsequently demonstrated less sprouting and Rocky mountain maple experienced no dramatic decline or increase in shrub volume (Schmidt 1980). After prescribed burning in Idaho, Rocky Mountain maple averaged 120 sprouts per plant; fall burning produced fewer but longer sprouts than spring burning (Leege 1972b). Mean heights for Rocky Mountain maple were found to be greater on logged/broadcast burned sites than on logged or logged/piled/burned sites in Idaho (Mueggler 1961).

Prescribed burning is used to maintain seral shrubfields and provide winter range for Elk by retarding conifer dominance, decreasing the height of browse plants, increasing browse quality and palatability, and stimulating forage plants to regenerate from seed (Leege 1968, Leege 1969, Miller *et al.* 1988, Nelson 1976, Smith and Fischer 1997). However, Rocky Mountain maple quickly grows out of reach of browsing animals due to its rapid regrowth after fire (Hooker and Tisdale 1974, Leege 1969, Weaver 1987). Miller and others (1988) found that low-temperature fires might encourage excessive growth with leaders exceeding 2 m in a single growing season. Severe burns that actually damage Rocky Mountain maple and stunt leader growth may be better utilized to reduce leader heights to a usable level of 0.5 m or less. Prescribed fire effectively improves forage availability in the short-term (Asherin 1975, Cholewa 1977, Lyon 1966, Lyon 1971, Weaver 1987, Wikeem and Strang 1983), and has been found to reduce height of Rocky Mountain maple from 2.1-4.9 m before fire to less than 2.1 m after fire (Lyon and Stickney 1966). Asherin (1975) found that by the end of the 3rd post-fire growing season, Rocky Mountain maple height exceeded 2.4 m. Four years after burning in Idaho shrubfields, 80% of twig production remained within reach of Elk (Leege 1972a, Leege 1972b). Repeated burns at 10 to 15 year intervals are expected to maintain Rocky Mountain maple availability (Leege 1972a, Leege 1972b, Leege 1979, Smith and Fischer 1997); however, it is often difficult to re-establish trees on these sites in the future (Smith and Fischer 1997). Broadcast burning of clearcuts in the northern Rockies may produce these seral shrubfields (Kingery and Graham 1994, Shearer and Schmidt 1982, Zamora 1982); in one study, shrub-dominant vegetation occurred by the 8th post-fire year (Zamora 1982). Rocky Mountain maple was present within four years on another lightly burned clearcut (Halvorson 1982).

After burning, browse production may be higher than pre-fire production initially (Cholewa 1977, Leege 1978), but annual production was observed to taper off by the 10th year, falling below pre-fire browse production (Leege 1978). Both spring and fall burning appear to increase Rocky Mountain maple palatability, based on heavy use in burned areas and browsing of larger twigs (Leege 1969). Crude protein levels may be higher in Rocky Mountain maple for up to four years following fire (Nelson 1976).

Injury due to fire may increase Rocky Mountain maple susceptibility to infection by parasitic *Cytospora* species (Dearness and Hansbrough 1934).

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Acer macrophyllum
Bigleaf maple

FIRE ECOLOGY OR ADAPTATIONS

Bigleaf maple is well adapted to fire. It sprouts prolifically from its root crown following crown destruction by fire (Haeussler and Coates 1986, Roy 1955). On moist upland sites in the Cascades, sprouting allows bigleaf maple to become a part of the immediate post-fire community when the conifer overstorey is removed or killed. Its abundance on upland sites following fire seems to change little, and it remains scattered (Halpern 1989). Seedling establishment on recently burned areas has not been reported, although it could potentially invade burned sites via seed transported from off-site by wind or birds and small mammals. Fowells (1965) reported that "In the Oregon coast range and on the western slopes of the Cascades, (bigleaf maple) frequently invades logged and burned areas, particularly in moist locations."

Bigleaf maple often grows along streams and rivers where soils are moist (Fowells 1965). Trees in these habitats may escape fire or be subjected to fires of lower intensity than those in adjacent uplands (Agee 1988). Stands of bigleaf maple and red alder bordering streams of the Tillamook Burn of Oregon were untouched by a severe fire in 1933 and survived light fires in 1939 and 1945 (Bailey and Poulton 1968).

POSTFIRE REGENERATION STRATEGY

Survivor species; on-site surviving root crown or caudex.

IMMEDIATE FIRE EFFECT ON PLANT

Most fires top-kill bigleaf maple (Haeussler and Coates 1986, Roy 1955). Severe fires which transfer heat below the mineral soil surface damage maple root crowns and thus prevent some plants from sprouting (Stickney 1981). Following prescribed burning in the western Cascades of Oregon, bigleaf maple was nearly eliminated on severely burned plots but remained abundant in lightly burned areas (Dyrness 1973).

PLANT RESPONSE TO FIRE

Bigleaf maple survives fire by producing numerous root crown sprouts (Haeussler and Coates 1986, Roy 1955). Reported annual sprout growth varies from 1-2 m (Roy 1955) and 3-4 m (Haeussler and Coates 1986). Sprout development following top-kill of bigleaf maple trees by fire in northern California is summarized below (Roy 1955):

time since fire	height of tallest sprout in clump (feet)		crown diameter of sprout clump (feet)		sprouts/clump	
	Avg	Range	Avg	Range	Avg	Range
second year	9.8	6.8-13.1	11.5	6.8-15.5	78	14-143
third year	12.8	7.5-17.1	14.7	10.4-21.5	37	8-67

In this study, the number of living sprouts was reduced drastically between the second and third growing season as weak sprouts died and growth was concentrated on fewer stems. Parent tree diameter was found to be related to the number of sprouts per tree, with large diameter trees producing the greatest number of sprouts per tree.

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Arbutus menziesii
Arbutus

FIRE ECOLOGY OR ADAPTATIONS

Arbutus depends on periodic fire to eliminate or greatly reduce the conifer overstorey (Atzet and Wheeler 1982, Veirs 1980). Post-burn regenerative modes include both the production of prolific sprouts and abundant seed (Atzet and Wheeler 1982). Following fires which kill aerial stems, arbutus sprouts vigorously via dormant buds located on an underground regenerative organ known as a burl (Fowells 1965, Harrington *et al.* 1984, Van Dersal 1938). Burls serve as a source of stored carbohydrates and aggregations of adventitious buds, enabling arbutus to rapidly occupy the initial post-burn environment (James 1984, McDonald 1981). Burl development also enhances survival after repeated burning (Atzet and Wheeler 1982, Kay *et al.* 1961). On favourable growth sites within redwood and mixed-evergreen forests, trees attain diameters and bark thicknesses capable of surviving light ground fires (Atzet and Wheeler 1982).

Exposed mineral soil seedbeds and light canopy densities associated with recent burns are conducive to arbutus seedling establishment (Atzet and Wheeler 1982, Pelton 1962, Tappeiner *et al.* 1986).

POSTFIRE REGENERATION STRATEGY

Survivor species; on-site surviving root crown or caudex; off-site colonizer; seed carried by animals or water; post-fire year 1 and 2; crown stored residual colonizer; short-viability seed in on-site cones.

IMMEDIATE FIRE EFFECT ON PLANT

Arbutus is a fire-sensitive species; aboveground portions are very susceptible to fire mortality (Atzet and Wheeler 1982, Dale *et al.* 1986, Volland and Dell 1981). Low-intensity ground fires usually top-kill arbutus seedlings and sapling-sized stems (Atzet and Wheeler 1982). Thin bark provides little insulation from radiant heat, which usually kills the cambium around the base of the stem (McDonald *et al.* 1983). Larger trees with thicker bark frequently survive light underburning on favourable growth sites within redwood and mixed-evergreen forests. Although young, vigorous trees usually exhibit bole injuries following burning, fire scars tend to heal over

rapidly. Older arbutus may initially survive light ground fires; however, bole wounds facilitate the entry of insects and disease and many fire-damaged trees eventually die (Atzet and Wheeler 1982).

PLANT RESPONSE TO FIRE

Vegetative regeneration: Following fires which kill aerial stems, arbutus initiates a rapid post-burn recovery by sprouting from adventitious buds located on an underground, globe-like structure known as a burl (James 1984, Tappeiner *et al.* 1984). Burls may reach diameters of 20 cm or more (Harrington *et al.* 1984). Since burls contain aggregations of buds, newly sprouted arbutus occur as "sprout clumps" (McDonald *et al.* 1983). Sprouts are quite tolerant of direct sunlight and develop well in the initial post-burn environment (Fowells 1965). After 10 years of growth on a good site in the northern Sierra Nevada, arbutus sprout clumps averaged 6.7 m in height and 3.1 m in crown width with an average of 15 sprouts per clump (McDonald 1981).

The initial post-burn recovery is aggressive. Arbutus sprouts sometimes grow more than 1.5 m during the first post-burn growing season (McDonald 1981). Although prolific spouting usually occurs during the first 2 years following burning, sprout numbers are drastically reduced between post-burn years 2 and 3 as growth is concentrated into multiple dominant stems. Initial trends in post-burn recovery of arbutus in northwestern California are presented below (Bolsinger 1988):

	Years post-burn		
	1	2	3
Average ht of tallest sprout in clump (feet)	4.7	7.7	10.1
Average crown diameter of sprout clump (feet)	4.5	6.8	7.6
Average sprouts per clump	17	16	13

Seedling regeneration: Arbutus seedling establishment is favoured following fire. Mineral soil seedbeds and light canopy densities of the initial post-burn environment are extremely conducive to the successful establishment and growth of arbutus seedlings (Pelton 1962, Tappeiner *et al.* 1986).

On sites where fires are not too severe, abundant seed is available from residual trees. Off-site seed is also dispersed by mammals and birds. Initial post-burn seed production of sprouted arbutus has not been reported. Minimum seed-bearing age of seedling-derived plants is 3 to 5 years (Fowells 1965).

DISCUSSION AND QUALIFICATION OF PLANT RESPONSE

Since sprout production is initially fuelled by the residual root system, post-burn sprouting potential of arbutus is strongly related to the size and vigour of the parent tree. Sprout clumps produced by larger diameter parent trees are generally taller and wider in diameter than those of smaller diameter parent trees; they also produce greater numbers of resprouts (Harrington *et al.* 1984, Tappeiner *et al.* 1984, Bolsinger 1988). Site quality apparently has almost no influence on sprout clump development for at least 6 years after fire (Tappeiner *et al.* 1984).

Limited observations on sites in southwestern Oregon indicate that previous fires and the subsequent entry of pathogens can substantially reduce the sprouting potential of parent trees greater than 43 cm DBH (Tappeiner *et al.* 1984).

FIRE MANAGEMENT CONSIDERATIONS

Broadcast burning: Where conifer regeneration is a primary management concern, broadcast burning is generally an ineffective site preparation tool following clearcutting in stands where arbutus is widespread in the understory. Not only is arbutus difficult to control with repeated burning, but sprouts often grow faster than many associated hardwoods (Kay *et al.* 1961, McDonald *et al.* 1983). Even though burning delays the recovery of arbutus for approximately one growing season, removal of logging debris promotes sprouting by exposing basal buds to solar heating and permits sprouts to grow unimpeded (McDonald *et al.* 1983). Sites where the pre-burn vegetation consists of low conifer stocking combined with high densities of arbutus are particularly prone to the rapid development of a dense arbutus understory.

Underburning: Arbutus seedlings establish readily following logging and burning of conifer-hardwood stands (Tappeiner *et al.* 1986). Light underburning at sometime during the rotation may minimize seedling establishment, thereby reducing the density of arbutus capable of sprouting after future disturbances. Control of arbutus seed trees should accompany such prescribed fire treatments.

Hardwood management: Burning should not be used as a method of slash disposal in partially cut hardwood stands where arbutus is managed for timber production. Instead, logging debris should be lopped and scattered, or piled (McDonald *et al.* 1983).

Wildlife management: Burning initially increases the palatability of arbutus browse; sprouts are utilized for up to 2 growing seasons (Dayton 1931, Van Dersal 1938).

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Betula papyrifera
Paper birch

FIRE ECOLOGY OR ADAPTATIONS

Adaptation to fire: Paper birch is well adapted to fire, recovering quickly by means of seedling establishment and vegetative regeneration (A.D. Revill Associates 1978, Lutz 1956, Viereck and Schandelmeier 1980). Seedling establishment is the most significant method of post-fire recovery. Paper birch is a prolific producer of lightweight seeds that are easily dispersed by wind and readily germinate on fire-prepared seedbeds. Young trees sprout from the root collar following top-kill, but sprouting ability decreases after about 40 to 60 years of age (Perala and Alm 1990).

Fire regime: Throughout most of Alaska and Canada, paper birch is found in boreal spruce and mixedwood forest types that burn at 50 to 150 year intervals (Day and Harvey 1981, Heinselman 1981).

Fire behavior: As a forest type, paper birch stands are one of the least flammable. The canopy often has a high moisture content and the understory is lush (Foster and King 1986). Crown fires in coniferous stands often stop at the boundary of large paper birch stands or become slow-moving ground fires (Foster and King 1986, Viereck 1975). As a result of this fire behavior, some large paper birch trees often survive fire in pure stands, and thus become seed trees for post-fire establishment (Perala and Alm 1990). During dry periods, paper birch stands will burn readily.

POSTFIRE REGENERATION STRATEGY

Survivor species; on-site surviving root crown or caudex; off-site colonizer; seed carried by wind; post-fire years 1 and 2.

IMMEDIATE FIRE EFFECT ON PLANT

Fire generally kills or top-kills most paper birch trees; the thin, flammable bark makes the bole highly susceptible to girdling even by light surface fires (Day and Harvey 1981, Foster and King 1986, Hosie 1969). Although the bark of older trees is thicker, it is also more flammable once it begins to exfoliate (Lutz 1956).

Paper birch seeds on the ground are destroyed by fire. Summer fires do not necessarily consume the catkins, but immature seeds will not ripen on killed or top-killed trees (Viereck 1973).

DISCUSSION AND QUALIFICATION OF FIRE EFFECT

In the Chippewa National Forest in Minnesota, prescribed burning in mid-May in aspen slash top-killed all 10-38 cm paper birch trees. These sprouted within a few weeks of the fire, but sprout mortality over the next few years resulted in 11 percent of the original trees dead by post-fire year 5 (Peek 1974).

Low-intensity prescribed surface fires (mean flame length > 0.3 m, mean rate of spread of 3.3 m per minute) in a 30-year-old mixed hardwood stand in central Wisconsin did not kill or top-kill any paper birch trees greater than 10 cm in trunk diameter. Most of the saplings less than 4 inches in trunk diameter, however, were top-killed (Reich *et al.* 1990).

On the Kenai Peninsula, Alaska, 22 percent of paper birch trees were unaffected, while 78 percent had dead or partially dead aerial crowns 2 years after a light surface fire (Walker 1979). Forty-two percent of top-killed trees produced sprouts.

Prescribed burning in a northern Wisconsin bracken fern (*Pteridium aquilinum*)-grassland killed 31 percent of paper birch trees present. The rest were top-killed but later sprouted. Basal area was reduced by 90 percent (Vogl 1964).

PLANT RESPONSE TO FIRE

Paper birch rapidly revegetates burned areas. Sprouts, and seedlings if seed trees are nearby, appear within the first post-fire year.

Sprout production: Young paper birch trees up to about 50 years old sprout prolifically and vigorously after fire. Sprouts appear a few weeks to 2 months after spring or summer fires but not until the following spring after late fall fires (Leege 1979, Perala 1974, Gilley 1982). They grow rapidly and are often 50-100 cm tall after one growing season (A.D. Revill Associates 1978, Leege 1979). In Minnesota, 5-year-old post-fire paper birch sprouts averaged 3 m in height (Perala 1974). Following prescribed spring fires at 5-year intervals in Idaho, paper birch averaged 31 to 58 basal sprouts per plant after each fire (Leege 1979). Sprout mortality is high in the first 5 post-fire years, leaving a few to several sprouts per clump (Perala 1974).

Seedling establishment: Mineral soils exposed by fire provide excellent paper birch seedbeds, but charred or partially removed organic layers prevent establishment. In Alaska, germination and subsequent seedling survival of artificially sown paper birch seed was abundant, practically nil, and nil on severely, moderately, and lightly burned test plots, respectively (Zasada *et al.* 1983).

Undamaged trees within a burn or trees in nearby unburned stands are necessary for post-fire seedling establishment. Where there are abundant seed trees, paper birch can easily establish tens of thousands of seedlings per acre after fire (Foote 1983, Lutz 1956, Walker 1979). In northern Saskatchewan, undamaged paper birch trees released 2.48 & 1 million seeds/ha in the first and second fall, respectively, following an April wildfire (Archibold 1980).

Because seed dispersal occurs in the fall, seedling establishment does not begin until the second post-fire year (Ohmann and Grigal 1981). Seedling establishment is generally greatest from post-fire years 2 to 5 (Ahlgren 1959, Ohmann and Grigal 1981). In Labrador, paper birch established by seed dated to within 15 years of fire, with subsequent seedling establishment lacking (Foster and King 1986).

DISCUSSION AND QUALIFICATION OF PLANT RESPONSE

In white spruce forest types in interior Alaska, paper birch establishes thousands of seedlings and sprouts shortly after fire. Sampling fire-origin paper birch stands in Alaska, Lutz (1956) observed an average of 19 760 seedlings and saplings per ha 1 year after fire.

Four years after fire, Foote (1983) observed an average of 30 000 stems/ha of both seed and sprout origin that were 1-2 m tall. Between 26 and 45 years after the fire, these thinned to a few thousand per acre.

Three years after a prescribed light surface fire in south-central Alaska, on a site stocked primarily with white spruce and paper birch, there were about 19 760 and 2470 paper birch seedlings and sprouts, respectively, per ha (Walker 1979). Seedlings averaged 15 cm and sprouts 35 cm in height.

FIRE MANAGEMENT CONSIDERATIONS

Prescribed fire can be used to prepare cut-over sites for paper birch seed regeneration. In Maine, prescribed burning following winter logging favoured paper birch establishment more than other treatments did. Burning or disking following logging exposed mineral soils on more than 70 percent of the logged area, while logging alone during the summer or winter, resulted in only 5 percent mineral soil exposure. Thirty-five seed trees per ha were left on each treatment site. Paper birch seedling establishment was as follows (Bjorkbom 1972):

Treatment	Post-treatment year 1 #/ha	Post-treatment year 10 #/ha
winter logging/disked	605 200	8200
winter logging/burned	123 700	11 900
summer logging only	162 300	4200
winter logging only	83 200	4700

Prescribed fire can be used to enhance deer and Moose winter habitat by killing late successional conifers and promoting early successional browse species such as paper birch (Walker 1979). It generally takes 3 to 5 years after fire for paper birch sprout and seedling growth to provide adequate browse for deer and Moose (Safford *et al.* 1990). Peak browse production is generally between 10 and 16 years after fire (Safford *et al.* 1990).

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Cornus nuttallii
Pacific dogwood

FIRE ECOLOGY OR ADAPTATIONS

Fire adaptations: Pacific dogwood sprouts following fire (Roper 1970). Root crown sprouting and/or epicormic branching can occur following fire (Agee 1994, Biswell *et al.* 1955).

Fire regimes: The fire regime for Pacific dogwood is dependent on the overstorey community, site conditions, and historical disturbances. In the central and south Sierra Nevada, indigenous people historically burned areas to encourage new growth in *Cornus* spp. Fires were set in the fall and burning occurred at 1- to 2-year intervals (Anderson 1996). Others also suggest that fires were common in the Sierra Nevada. Parsons and DeBenedetti (1979) suggest that fires frequently burned in sequoia and mixed conifer forest types. The poor recruitment of giant sequoia is thought to be related to fire suppression efforts in these areas (Parsons and DeBenedetti 1979, Vankat and Major 1978). Fires were also frequent in the Siskiyou region of California and Oregon; fire severities however ranged widely (Whittaker 1960).

In the Klamath Mountains of California, researchers investigated 75 plots in 1570 ha to reconstruct the fire history. They estimated the average area burned was 350 ha with 16 fires between 1627 and 1992 that were greater than 500 ha. Most fires were of low and moderate severities, although stand-replacing fires also occurred. Estimated average fire return intervals are shown below (Taylor and Skinner 1998).

Presettlement (1626-1849)	14.5 years
Settlement (1850-1904)	12.5 years
Suppression (1905-1992)	21.8 years

In the Lochsa-Selway area of Idaho, Roper (1970) reports that large fires burned in 1910, 1919, 1924, 1930, 1934 and smaller fires occurred in 1949-50 and 1967. Heavy moisture in the winter and spring months allows fuels to accumulate in this area, while hot, dry summers foster burning conditions. Climate regime is likely the primary driving factor of frequent fires in this area (Roper 1970).

Not all areas where Pacific dogwood is common burned often. Riparian areas often burn less frequently and/or burn at lower severity than the surrounding slopes (Agee 1994). In coastal redwood forests of northern California, lightning-caused fires were infrequent. In this high humidity region, the author estimates from age class and fire scar distributions that low-severity fires occurred at 250- to 500-year intervals on mesic sites, at 50-year intervals on xeric sites, and at 100- and 200-year intervals on intermediate sites (Veirs 1982).

The following table provides fire return intervals for plant communities and ecosystems where Pacific dogwood is important.

Community or Ecosystem	Dominant Species	Fire Return Interval Range (years)
silver fir-Douglas-fir	<i>Abies amabilis</i> - <i>Pseudotsuga menziesii</i> var. <i>menziesii</i>	> 200
grand fir	<i>Abies grandis</i>	35-200
Engelmann spruce-subalpine fir	<i>Picea engelmannii</i> - <i>Abies lasiocarpa</i>	35 to > 200
Jeffrey pine	<i>Pinus jeffreyi</i>	5-30
western white pine*	<i>Pinus monticola</i>	50-200
Pacific ponderosa pine*	<i>Pinus ponderosa</i> var. <i>ponderosa</i>	1-47 [1]
interior ponderosa pine*	<i>Pinus ponderosa</i> var. <i>scopulorum</i>	2-30 [1, 2, 3]
Rocky Mountain Douglas-fir*	<i>Pseudotsuga menziesii</i> var. <i>glauca</i>	25-100 [1, 4, 5]
coastal Douglas-fir*	<i>Pseudotsuga menziesii</i> var. <i>menziesii</i>	40-240 [1, 6, 7]
California mixed evergreen	<i>Pseudotsuga menziesii</i> var. <i>menziesii</i> - <i>Lithocarpus densiflorus</i> - <i>Arbutus menziesii</i>	< 35
California oakwoods	<i>Quercus</i> spp.	< 35 [1]
coast live oak	<i>Quercus agrifolia</i>	2-75 [8]
canyon live oak	<i>Quercus chrysolepis</i>	<35 to 200
Garry oak	<i>Quercus garryana</i>	< 35 [1]
California black oak	<i>Quercus kelloggii</i>	5-30 [9]
redwood	<i>Sequoia sempervirens</i>	5-200 [1, 10, 11]
western redcedar-western hemlock	<i>Thuja plicata</i> - <i>Tsuga heterophylla</i>	> 200
western hemlock-Sitka spruce	<i>Tsuga heterophylla</i> - <i>Picea sitchensis</i>	> 200
mountain hemlock*	<i>Tsuga mertensiana</i>	35 to > 200 [1]

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

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POSTFIRE REGENERATION STRATEGY (Stickney 1989)

Tree with adventitious bud/root crown; crown residual colonizer (on-site, initial community).

IMMEDIATE FIRE EFFECT ON PLANT

The immediate effect of fire on Pacific dogwood varies with fire severity. Low-consumption, early-spring fires, in mature mixed conifer ecosystems of the northern Sierra Nevada, produced temperatures hot enough to kill Pacific dogwood foliage but not hot enough to kill buds protected by bark (Kauffman and Martin 1990).

DISCUSSION AND QUALIFICATION OF FIRE EFFECT

Buds that survive a low-severity fire are released and able to regenerate (Agee 1994, Kauffman and Martin 1990).

PLANT RESPONSE TO FIRE

After being burned, Pacific dogwood typically sprouts from the root crown (Roper 1970). Parsons in a personal communication (Parsons 1989) predicts Pacific dogwood will sprout following fire "regardless of burn prescription." When 300- to 500-year-old Douglas-fir stands were logged and burned, Pacific dogwood was classified as a residual colonizer by the author (Halpern 1989).

DISCUSSION AND QUALIFICATION OF PLANT RESPONSE

Fire and logging: Much of the research regarding the post-fire recovery of Pacific dogwood involves the study of areas that were logged before being burned. Likely, fires in logged areas are different from those that burn standing forests. Following the burning of slash piles of a clearcut Douglas-fir forest type in the central Cascade Range, Pacific dogwood initially decreased in constancy and cover (Halpern 1989).

In the western Cascade Mountains of Oregon 100- to 500-year-old Douglas-fir stands were clearcut then burned the following year to determine early successional development of these forests (Dyrness 1965, Dyrness 1973). Prior to any treatments, Pacific dogwood coverage was 6.4% and 3.6% in vine maple-salal- and western sword fern-dominated communities, respectively. Five years after burning slash in these areas, Pacific dogwood coverage was 0.4% in vine maple-salal forest communities. In western sword fern communities that occupied stream banks and northern slopes, the coverage of Pacific dogwood was 1.4% in 5-year-old burned clearcuts (Dyrness 1973).

Ingram (1931) compared single- and multiple-slash burns in the Douglas-fir forests of the Columbia National Forest. Pacific dogwood density and percent composition were greater on sites that were burned the same year they were logged when compared to sites burned 1 year following logging and those sites burned twice after logging.

In the H.J. Andrews Experimental Forest of west-central Oregon, old growth stands of Douglas-fir (> 400 years) were clearcut in June and slash burned in October. Pacific dogwood had 1% cover on the sites before being logged and burned; at the end of the next growing season Pacific dogwood coverage was either 0% or lower than the 1% cutoff used to warrant reporting. There were no control comparisons available for Pacific dogwood in this study (Gashwiler 1959).

In the western Cascade Mountains of Oregon, researchers studied secondary succession following clearcutting and broadcast burning of multiple sites within Douglas-fir forests. Pacific dogwood cover increased as secondary succession progressed. Results are presented below (Schoonmaker and McKee 1988):

Number of years since clearcut and broadcast burned	2	5	10	15	20	30	40	Undisturbed (450-year-old stand)
Mean Pacific dogwood cover values (n = 3), in year 10 (n = 2)	0.04	.013	0	0.04	0.52	0.17	1.69	0.85

Fire alone: Considerably fewer studies relate to the post-fire response of Pacific dogwood to fire without logging as an additional major disturbance. After the large Oxbow fire that burned portions of western Oregon, Hooven (1969) lists Pacific dogwood as one of many species that made up the post-burn vegetation the first year following the fire. There was no information provided about fire severity or season of the burn.

In mixed conifer forests of the northern Sierra Nevada, California, four prescription fires burned on two sites where Pacific dogwood occurred. Fires were described as early spring-moderate consumption, late spring-high consumption, early fall-moderate consumption, and late fall-high consumption. Early spring prescribed fires coincided with bud break of most species, late spring burns occurred when plants were actively growing, early fall fires coincided with the end of the shrub growing season, and late fall burns corresponded with the leaf drop of most deciduous shrub species. Following late spring, early fall, and late fall fires, Pacific dogwood density increased. The early spring fire, however, had the opposite effect. The following table reports the effects of fire seasonality on Pacific dogwood; reported are the densities (number of plants/ha) (Kauffman 1986):

	Pre-burn	1st post-burn growing season	2nd post-burn growing season
Early spring	33	0	0
Late spring	0	0	34
Early fall	33	--	67
Late fall	0	42	125

FIRE MANAGEMENT CONSIDERATIONS

Fire management considerations often involve more than just the post-fire community or species response. Following an early spring fire in giant sequoia forests of Tulare County, California, Lawrence and Biswell (1972) found that the utilization of Pacific dogwood by Black-tailed Deer was significantly greater ($p < 0.01$) on logged and burned sites. The authors note that the “resulting crown sprouts were browsed so heavily that the survival of such trees seemed doubtful.”

When setting prescription fires in giant sequoia groves, the California Parks and Recreation Department first raked around Pacific dogwood as they considered this species prone to cambium damage and wanted to reduce the “visual and environmental” impacts of the fire (Harrison 1986).

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Picea sitchensis
Sitka spruce

FIRE ECOLOGY OR ADAPTATIONS

Fire is not an important factor in the ecology of Sitka spruce (Alaback 1982). Its thin bark and a shallow root system make it very susceptible to fire damage (Arno and Hammerly 1977, Brown and Davis 1973). Sitka spruce forests have a fire regime of long-interval (150 to 350+ years) severe crown or surface fires which result in total stand replacement (Parminter 1991).

POSTFIRE REGENERATION STRATEGY

Secondary colonizer - offsite seed.

IMMEDIATE FIRE EFFECT ON PLANT

The immediate effect of a cool to hot fire is damage to the cambium layer, usually resulting in death of the tree (Arno and Hammerly 1977, Brown and Davis 1973).

PLANT RESPONSE TO FIRE

Sitka spruce will invade a burned site via wind-dispersed seed from adjacent unburned forests (Ruth and Harris 1975). Wind-dispersed seed travels 30-790 m from the parent tree (Harris 1990).

FIRE MANAGEMENT CONSIDERATIONS

Arguments for and against slash burning in spruce forests recur throughout the literature. The strategy chosen will yield different results, depending on latitude.

In the northern portion of Sitka spruce's range broadcast burning will favour Sitka spruce over western hemlock, but unless Sitka spruce is planted, seedling establishment will be delayed until the next seed crop (Feller 1982, Hawkes *et al.* 1990, Ruth 1974, Ruth and Harris 1975). Ruth and Harris (1975) list the advantages of slash burning as follows:

- (1) Reduces fire hazard
- (2) Destroys advance regeneration*
- (3) Changes timber type

*This can have both positive and negative ramifications. It reduces competition with western hemlock, but growth of Sitka spruce seedlings in one study was reduced (Feller 1982).

In the southern portion of its range broadcast burning will favour the establishment of Douglas-fir (*Pseudotsuga menziesii*) mixed forest, while long-term fire exclusion will result in loss of Douglas-fir from the overstorey. This is advantageous due to the increased stumpage value of Douglas-fir and the negative impacts of the spruce weevil (Parminter 1991, Ruth and Harris 1975).

In the coastal area of Alaska, broadcast burning has been recommended to reduce the negative aesthetic value of large quantities of slash from clearcut old-growth Sitka spruce forests (Stednick *et al.* 1982).

However, removal of the slash by burning in Sitka spruce forests is not required because of the rapid decay in that moist environment (Ruth 1974). Burning is not recommended on steep slopes and where water quality may be degraded (Ruth 1974, Stednick *et al.* 1982).

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Pinus contorta var. *contorta*
Shore pine

FIRE ECOLOGY OR ADAPTATIONS

Fire is not an important factor in forest succession where shore pine occurs in southeastern Alaska (Arno and Hoff 1989). Fire is infrequent in maritime forest types and usually is of little ecological significance (Lotan *et al.* 1981). The fire interval is 150 to 350 years but may not be cyclic (Parminter 1991). The coastal cedar-pine-hemlock biogeoclimatic zone of British Columbia has little or no fire history. The presence of shade-tolerant firs and hemlocks and fire-sensitive species, such as shore pine, indicate that fire is rare in this area (Parminter 1983).

The coastal dunes where shore pine occurs in California are considered fire-free (Vogl *et al.* 1977).

The foliage of shore pine is moderately flammable. Shore pine has a moderate to low degree of fire resistance (Lotan *et al.* 1981).

POSTFIRE REGENERATION STRATEGY

Tree without adventitious-bud root crown; secondary colonizer - off-site seed.

IMMEDIATE FIRE EFFECT ON PLANT

Shore pine is sensitive to fire and probably killed by most fires.

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Pinus ponderosa
Ponderosa pine

FIRE ECOLOGY OR ADAPTATIONS

Fire is an integral part of the ecology of ponderosa pine. Studies have shown that prior to 1900, most stands experienced low-severity surface fires at intervals ranging from 1 to 30 years. Fire scars and pollen analysis trace this phenomena back to at least AD 1500. Fire has allowed ponderosa pine to dominate sites where it is the potential climax as well as sites where it is seral to more shade-tolerant tree species (Arno 1988, Habeck 1990, Henderson 1967, Lampi 1960).

Ponderosa pine has evolved with a thick bark and open crown structure that allows it to survive most fires. Mature trees will self-prune, leaving a smooth bole which reduces aerial fire spread. Other fire adaptations include deep roots, high foliar moisture content, insulated bud scales, and medium to light lichen growth (Fischer and Clayton 1983, Flint 1925, Saveland 1982). Seedlings prefer the mineral soil seedbeds created by fire (Lotan *et al.* 1981).

Fire also shapes the composition of ponderosa pine stands. In the late 1800s stands exhibited open parklike appearances with well-stocked overstories and relatively few understory trees. Fire suppression, however, has allowed the unnatural buildup of forest fuels which has increased the occurrence of stand-replacing fires. Over the last 100 years of fire suppression, seral ponderosa pine stands have been replaced by shade-tolerant climax stands (Arno 1988, Habeck 1990, Henderson 1967).

POSTFIRE REGENERATION STRATEGY

Crown-stored residual colonizer; short-viability seed in on-site cones; off-site colonizer; seed carried by wind; post-fire years 1 and 2; off-site colonizer; seed carried by animals or water; post-fire years 1 and 2.

IMMEDIATE FIRE EFFECT ON PLANT

Fire has a wide variety of potential effects on ponderosa pine. These effects vary according to size, configuration, and density of the stand, in addition to fire severity. Generally, well-spaced seedlings and saplings are able to withstand low-severity fires, as are pole-sized and mature trees. Moderate- to high-severity fires, however, will kill trees pole-sized and smaller. Mature ponderosa pines have a higher survival rate than younger trees due to their enhanced adaptations to fire (Fischer and Clayton 1983, Lampi 1960, Wright 1978).

The principal cause of mortality following fire is crown scorch rather than damage to the cambium or roots. The size of tree determines its ability to withstand scorch. A model has been developed to predict mortality using tree DBH and scorch heights as

independent variables. Fire effects are also dependent upon other factors such as season, site condition, tree age and vigour, available moisture, and occurrences of insect and disease attack (Lampi 1960, Saveland and Neuenschwander 1989).

DISCUSSION AND QUALIFICATION OF FIRE EFFECT

If fire consumes any part of the canopy, the total leaf area is reduced, thus decreasing photosynthesis. If burning results in damage to the bole or roots, nutrient and water transport is impaired. Heat from fire may kill living tissue and result in a certain amount of stress (Henderson 1967, Lampi 1960, Saveland and Bunting 1988).

Crown damage: Crown scorch appears to be the leading factor in the majority of damage to ponderosa pine. Estimation of percent crown volume scorch has been proven to be the best predictor of tree mortality following fire. Crown damage is most severe in spring and early summer due to low foliar moisture content and the succulent nature of the buds and twigs (Lampi 1960, Saveland and Bunting 1988). Survival of buds is also crucial to post-burn survival of ponderosa pine. Buds can tolerate temperatures 20° C higher than the needles can due to their protective outer scales. Therefore, large trees can sometimes survive a 100 percent crown scorch provided not all the buds are heat killed (Henderson 1967, Lampi 1960, Methven 1971, Saveland and Bunting 1988).

Root damage: Following prescribed burning of old-growth ponderosa stands in Crater Lake National Park, Oregon, mortality was higher in burned areas (19.5 percent) than in unburned areas (6.6 percent). A major factor contributing to post-fire mortality was the reduction of fine roots. Burning reduced fine-root dry weight 50 to 75 percent from 1 to 5 months after burning (Swezy and Agee 1991).

Bole damage: This pine is fire tolerant because it has a fire-resistant bark containing a 0.3-0.6 cm thick layer at 5 cm diameter (Wright 1978). It also has a very moist core of high density wood that dissipates the heat energy it receives, thus protecting the bole from lethal heat levels (Saveland and Bunting 1988). Ryan and Frandsen (1991), however, found that mature ponderosa pine trees suffered more basal injuries from smouldering fires than did immature trees because of the greater quantities of accumulated duff surrounding their boles. Cambium damage most often occurs after the passing of high-severity fires.

Young trees are most susceptible to cambium damage as a result of thinner bark and a higher occurrence of girdling (Henderson 1967, Wyant *et al.* 1986). Partially girdled trees may survive up to 25 percent basal loss if root and crown damage is minimal (Wagener 1961).

Season of burn: Ponderosa pine can withstand low-severity fires which generally occur during the wet months of early spring or late fall. A dry spring fire may occur when trees are in stress during leaf and bud burst, resulting in higher mortality rates.

Trees become dormant toward fall and thus are more fire resistant. In fall, ponderosa pine can withstand up to 50 percent crown scorch, while in spring only 30 percent (Henderson 1967, Mohr 1984).

PLANT RESPONSE TO FIRE

Ponderosa pine's response to fire will vary according to fire severity, tree age, and season. High-severity fires that occur during periods of high stress will generally result in death. Low- to medium-severity fires will generally restrict the growth and regeneration of the tree, but recovery is usually evident the following year (Lampi 1960). Immediately following fire, ponderosa pine may experience a large needle drop as a reaction to hot convectional air movement through the canopy (Saveland and Bunting 1988).

Post-fire seedling establishment: Fire creates favourable seedbeds for seedling establishment. The soil is often rich in available inorganic nitrogen that benefits tree growth (Ryan and Covington 1986). Post-fire stocking rates depend upon site characteristics, fire severity, and weather. The potential for regeneration after fire is generally considered good (Lampi 1960). On the Eldorado National Forest, California, a low-severity burn resulted in 49 400 seedlings per ha on burned sites and no seedling establishment on unburned sites (Lampi 1960). In a western Montana study, ponderosa pine produced 12 percent of the total number of sound seeds found on a burned clear-cut site over a 5-year period (Swezy and Agee 1991). A post-burn study in the Plumas National Forest, California, found that ponderosa pine had the highest post-burn percent increase of all other associated species (Lampi 1960).

Post-fire growth and recovery: Information concerning ponderosa pine's response after fire is variable. This may be attributable to the beneficial effects of reduced competition and increased nutrient availability, along with the detrimental effects of damage to the crown, cambium, and roots. Some studies found reductions in diameter and height growth (Morris and Mowat 1958, Wooldridge and Weaver 1965), while others reported increases in post-fire growth (Cooper 1960, Morris and Mowat 1958, Van Sickle and Hickman 1959, Weaver 1968).

FIRE MANAGEMENT CONSIDERATIONS

The last 100 years has produced unplanned, radical changes in stand structure, fuel loadings, and role of fire in ponderosa pine ecosystems (Arno 1988). Post-settlement fire suppression has resulted in dense stockings of shade-tolerant species and the increase of insects and disease. These results have led to other concerns such as loss of timber productivity, loss of natural diversity and aesthetic values, and the increased risk of severe fire damage to homes and harvestable timber (Henderson 1967).

Prescribed fire: Reduction of fuel loads beneath existing stands of ponderosa pine by the use of prescribed fire has proven useful in reducing the potential threat of wildfires, while also favouring natural regeneration of seral species through site preparation. In western Montana, prescribed burning on an interval of 20 to 25 years is suggested to maintain seral species and open stocking. This would also prune lower branches thus increasing timber values, while also lowering the risk of wildfire (Arno 1988, Habeck 1990, Henderson 1967, Lampi 1960).

Nutrient depletion: Prescribed fire often leads to the loss of volatile nutrients from the site, especially nitrogen (N). Following a prescribed fire on a ponderosa pine site in Oregon, all periodic annual growth increments were reduced in surviving trees four growing seasons later. Foliar N concentration was not affected by the fire; however, total foliar N content was reduced immediately after burning. Foliar N content was significantly correlated with the observed reductions in periodic annual increments (Lampi 1960, Landsberg *et al.* 1984).

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Populus balsamifera ssp. *trichocarpa*
Black cottonwood

FIRE ECOLOGY OR ADAPTATIONS

Fire Adaptations: Black cottonwood is frequently damaged by fire with low-severity burns even often causing “considerable injury” (Agee 1991, Haeussler *et al.* 1990). Young black cottonwood trees and seedlings are usually killed by fire regardless of severity (Brown 1996). Severe fire kills or top-kills even older trees. In low- and moderate-severity fires older trees with thick bark may not be top-killed (Hall and Hansen 1997, Miller 2000). In members of the *Populus* genus stem bark remains thin for longer than in other trees (Eckenwalder 1996). Though old trees have increased fire resistance due to thicker, furrowed bark, they have higher fuel loading and more heartrot, which can increase fire severity (Gom and Rood 1999). Trunks

that are not top-killed may be more susceptible to *Cytospora* spp., and other fungal pathogens; this was observed near D'Arcy in southern British Columbia (Dearness and Hansbrough 1934).

Black cottonwood sprouts from stumps, charred boles, root crowns, or lateral roots following fire (Brown 1996, Coates and Haeussler 1986, Hall and Hansen 1997, Miller 2000, Peterson and Peterson 1992) and because of this has been referred to as a fire "endurer" rather than "resister" (Agee 1991). Fire-induced sprouting is more common in the Tacamahaca section than in the Aigeiros section. Rates of sprouting are highest if fire occurs when cottonwoods are dormant (Gom and Rood 1999); this time also has the highest probability of fire, with late summer and fall, or late winter (in low snow years) most common (Hall and Hansen 1997). In general older trees sprout less than young trees (Gom and Rood 1999), and sprout survival is highest when the water table is close to the surface (Hall and Hansen 1997). In 1986, Coates and Haeussler (1986) stated that there was little information regarding black cottonwood sprout vigour after fire, or impact of fire severity on sprouting potential. Though this is still at least partially true, a study of clonal reproduction after fire in Alberta by Gom and Rood (1999) provides much useful data that is summarized in the "Fire Effects" section of this species' summary.

Black cottonwood is not only a fire "endurer" but also a fire "invader." Fire can improve seedling establishment by increasing light penetration and exposing mineral soil to allow seedling establishment if moisture is available (Agee 1991, Brown 1996). Increases in light penetration following fire also aid establishment (Brown 1996). Black cottonwood seedlings have been observed 1 year after stand-replacing fire in upland ponderosa pine/ Rocky Mountain Douglas-fir habitat on the Bitterroot National Forest, Montana. The seedlings grew in cavities in mineral soil after the root systems of large trees had burned (Smith 2001). A study of seedling establishment (artificially seeded) of balsam poplar after experimental fires in a black spruce habitat demonstrated the dependency of the species on mineral soil exposure for germination and survival. Fire was prescribed on 1 m² plots. On moderately burned plots 17 black cottonwood germinated, 1 survived 1 year, and 0 survived 3 years. On heavily burned plots 71 germinated, 39 survived 1 year, and 39 survived 3 years (Zasada *et al.* 1983).

Fire is infrequent on recently formed gravel bars, but when it does occur, damage to cottonwoods is greatest because their root systems have not developed (Hall and Hansen 1997).

Fire Regimes: Historic fire regimes have not been explicitly studied in black cottonwood communities (Arno 2001). It has been speculated that because of high moisture content and rapid decomposition of litter in riparian forests, frequency of fire is less than it is in adjacent areas (Smith and Fischer 1997). These factors could protect forested riparian sites when severe fires occur on adjacent uplands. Conversely, wind-driven fires beginning in adjacent upland communities may spread to riparian forests, particularly when fuel accumulation on upland sites has increased

as a result of fire suppression (Smith and Fischer 1997). Arno (2001) states that black cottonwood forests along major rivers of the Pacific Northwest likely burned frequently, as they were historically surrounded by communities characterized by high fire frequency, such as ponderosa pine savannas or sagebrush steppes. Fires could have easily spread from adjacent communities; prior to widespread livestock grazing and irrigation, dry grassy fuels were more continuous than they are currently (Arno 2001). When fires do occur, they are most severe in older cottonwood stands where fuel accumulation is highest (Gom and Rood 1999, Smith and Fischer 1997).

Fire regimes for plant communities and ecosystems in which black cottonwood is dominant are summarized below.

Community or Ecosystem	Dominant Species	Fire Return Interval Range (years)
grand fir	<i>Abies grandis</i>	35-200 [1]
sagebrush steppe	<i>Artemisia tridentata/Pseudoroegneria spicata</i>	20-70
Rocky Mountain juniper	<i>Juniperus scopulorum</i>	< 35 [2]
western larch	<i>Larix occidentalis</i>	25-100
Engelmann spruce-subalpine fir	<i>Picea engelmannii-Abies lasiocarpa</i>	35 to > 200
blue spruce*	<i>P. pungens</i>	35-200 [1]
Rocky Mountain lodgepole pine*	<i>Pinus contorta</i> var. <i>latifolia</i>	25-300+ [3, 1, 4]
western white pine*	<i>P. monticola</i>	50-200
interior ponderosa pine*	<i>P. ponderosa</i> var. <i>scopulorum</i>	2-10 [1]
eastern cottonwood	<i>Populus deltoides</i>	< 35 to 200 [2]
quaking aspen (west of the Great Plains)	<i>P. tremuloides</i>	7-120 [1, 5, 6]
Rocky Mountain Douglas-fir*	<i>Pseudotsuga menziesii</i> var. <i>glauca</i>	25-100 [1]
coastal Douglas-fir*	<i>P. menziesii</i> var. <i>menziesii</i>	40-240 [1, 7, 8]
western redcedar-western hemlock	<i>Thuja plicata-Tsuga heterophylla</i>	> 200 [1]
western hemlock-Sitka spruce	<i>Tsuga heterophylla-Picea sitchensis</i>	> 200 [1, 9]
elm-ash-cottonwood	<i>Ulmus-Fraxinus-Populus</i> spp.	< 35 to 200 [10, 11]

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

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POSTFIRE REGENERATION STRATEGY (Stickney 1989)

Tree with adventitious bud/root crown/soboliferous species root sucker; geophyte, growing points deep in soil; initial off-site colonizer (off-site, initial community); secondary colonizer (on-site or off-site seed sources).

IMMEDIATE FIRE EFFECT ON PLANT

Black cottonwood sprouts from the lateral roots, root crown, and bole after top-kill by fire (Gom and Rood 1999, Hall and Hansen 1997). It also establishes from seed (Agee 1991, Brown 1996, Smith 2001, Zasada *et al.* 1983). Gom and Rood (1999) studied the effects of severe fires along the Oldman River near Lethbridge, Alberta, which occurred in early and mid-April before cottonwoods' spring bud burst. The fires burned during the afternoon; strong Chinook winds increased fire severity. Black cottonwood, narrowleaf cottonwood, plains cottonwood, and hybrids among them dominated these riparian areas. All cottonwoods observed were girdled by the fires. Fire damage to trunks varied from "slight bark charring to complete trunk incineration." This caused 100% top-kill in trees even though not all canopies were

severely damaged, demonstrating the susceptibility of these cottonwood species to severe fires. The exact amount of mortality was not determined (because it is difficult to tell from which tree's lateral root system suckers originate), but mortality was definitely less than 25%, as 75% of trunks (including all species) produced coppice sprouts. The most severe fires were in older stands, likely the result of higher fuel accumulation, and there were many pits in the ground on these sites where root systems were burned. Young stands close to the river (without as much fuel loading) experienced less canopy damage but were top-killed via girdling.

Hardwood saplings (cottonwood, alder, willow) less than 5 cm in diameter experienced 100% mortality in 168 and 42 BTU/second/foot intensity fires in a white spruce, quaking aspen, and balsam poplar (likely including some black cottonwood and hybrids) stand northeast of Edmonton, Alberta (Kiil 1970).

Hall and Hansen (1997) stated that in low and moderate severity fires, older black cottonwood trees with thick bark may not be top-killed, but little quantitative data examining the relations between fire severity, tree age, and top-kill or mortality is available.

PLANT RESPONSE TO FIRE

One year after the fires along the Oldman River in Alberta (described above), approximately 75% of the top-killed cottonwoods (including all narrowleaf and black cottonwoods, and balsam poplar) produced shoots from remnants of trunks. Sprouting was not observed on control transects (on adjacent unburned areas). Of the trunks producing sprouts, 90% were from either narrowleaf or black cottonwood (the 2 species' sprouts were not easily differentiated). The average number of coppice sprouts was 17 per trunk with a range of 1 to 60. Height of the tallest sprout on each tree averaged 41 cm and ranged from 9-77 cm. The authors speculated that the high rate of sprouting was in part due to the occurrence of the fires while the cottonwoods were dormant.

The degree of canopy or trunk damage did not have much impact on coppice sprouting frequency or vigour. Sprouts were even observed from trunks that had burned to 25 cm below ground. Of trees that experienced "heavy" canopy damage, 60% produced shoots from stumps; 80% of those with "light" damage produced coppice sprouts. The average height of coppice sprouts was not significantly ($p > 0.05$) affected by the degree of trunk damage, and trunk diameter had no significant ($p > 0.05$) effect on either the number of coppice sprouts or the height of sprouts. Over 1000 root suckers were observed on the study transects, 80% of which were either narrowleaf or black cottonwood. The density of root suckers (including all species) was about 1 per 3 m². Suckers were more common near trees that had many coppice sprouts. After one growing season the average height of root suckers was approximately 1 m (Gom and Rood 1999).

Transects on the Oldman River were surveyed again 5 years after fire. At this time there was an average of 4 coppice sprouts per trunk on the 30% of the cottonwoods that still had living sprouts. Average sprout height was approximately 3 m with a range of 1.5 to 5 m. Of the 34 trunks with surviving coppice sprouts 33 were either narrowleaf or black cottonwood and 1 was plains cottonwood. Survival of root suckers over the 5 years was about 50%. A density of 1 sprout per 7 m² was observed as 550 sprouts still survived on the transects. The average height of root suckers increased from about 1 to 2.4 m. Seven root suckers were present on unburned control sites; the authors stated that these might have been induced by flooding damage 2 years prior to the survey (Gom and Rood 1999).

Gom and Rood (1999) summarized the response of several cottonwood species to fire, showing the high sprouting/suckering ability of balsam poplar and black and narrowleaf cottonwoods relative to Fremont and eastern cottonwoods:

Year of fire	Location	Age when studied (years)	Cottonwoods affected by fire	Success of asexual regeneration
about 1988	Peace River near Fort St. John, BC	~10	Balsam poplar	Very good (vigorous sucker sprouts)
about 1992	Red Deer River at Dinosaur Provincial Park, Brooks, AB	~5	Eastern cottonwood	Very poor
1990	Belly River at Lavern, AB	7	Narrowleaf cottonwood, black cottonwood	Very good
spring 1992	Oldman River at Lethbridge, AB	5	Narrowleaf cottonwood, balsam poplar, black cottonwood, eastern cottonwood	Good (sucker and coppice sprouts)
about 1986	South Saskatchewan River, Police Point Park, Medicine Hat, AB	~10	Eastern cottonwood	Very poor
summer 1973	Milk River near Manyberries, AB	24	Eastern cottonwood	Very poor
about 1995	Lower Truckee River, near Nixon, NV	~2	Fremont cottonwood	Poor
about 1994	Kane Springs Creek, south of Moab, UT	~3	Fremont cottonwood	Poor (sparse coppice sprouts)

FIRE MANAGEMENT CONSIDERATIONS

To increase the longevity of cottonwood stands managers may choose to prescribe fire when the stand is in the “pole” stage, as sprouting is high at this point (Hall and Hansen 1997). Dormant season, particularly fall, fires induce the most sprouting

(Gom and Rood 1999). Gom and Rood (1999) suggest that fire-induced sprouting, particularly in old stands, may be used to regenerate stands much the way flooding disturbance allows development of new stands. They also state that regeneration may be more rapid after fire than after flooding because the growth rate of sprouts is about 3 times greater than that of seedlings.

If the goal of prescribed fire is to maintain a cottonwood stand, 5 years of grazing exclusion after fire are recommended (Hall and Hansen 1997). Herbivory and fire interactions have been well documented for quaking aspen stands. In the Rocky Mountains (study sites ranged from Jasper National Park, Canada to Rocky Mountain National Park, Colorado), White and others (1998) found that prolific quaking aspen sprouting after fire could not overcome the effects of Elk grazing, and, with a short fire return interval, quaking aspen stands would decline.

Fire can be used to maintain black cottonwood stands, but it may cause negative impacts such as erosion and sedimentation. These impacts both on the site of fire and on sites downstream can be increased by heavy post-fire rainfall (Smith and Fischer 1997).

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Populus tremuloides
Trembling aspen

FIRE ECOLOGY OR ADAPTATIONS

Fire adaptations: Trembling aspen is highly competitive on burned sites (DeByle *et al.* 1987). Even where trembling aspen was a barely detectable component of the pre-fire vegetation, it often dominates a site after fire. Trembling aspen has adapted to fire in the following ways (DeByle 1985):

1. The thin bark has little heat resistance, and trembling aspen is easily top-killed by fire.
2. Root systems of top-killed stems send up a profusion of sprouts for several years after fire.
3. Sprouts grow rapidly by extracting water, nutrients, and photosynthate from an extant root system, and may outcompete other woody vegetation.
4. Following a fire, a new, even-aged trembling aspen stand can develop within a decade.
5. In contrast to most trees, trembling aspen is self-thinning. Without intervention, a mature forest of healthy trees can develop from dense sprouts.

Fire releases sprout primordia on roots from hormonally controlled growth inhibition; removes canopy shade; and blackens the soil surface, increasing heat absorption. Increased soil temperatures aid sprout production (Bradley *et al.* 1992, Hungerford 1988). On cold sites, trembling aspen may be unable to sprout until soil temperatures rise after fire (Hungerford 1988).

Trembling aspen is able to naturally regenerate without fire or cutting on some sites (Perala 1990), but fire may be required for regeneration on others. There are areas in Jackson Hole, Wyoming, where ungulate browsing has been light, both historically and recently, yet stems have not attained tree size since extensive fires in the 1800s (Gruell and Loope 1974).

Fuels and fire behavior: Fuels are usually more moist in trembling aspen stands than in surrounding forest. Crown fires in coniferous forests often drop to the surface in trembling aspen, or may extinguish after burning into trembling aspen only a few meters (Bevins 1984, Fechner and Barrows 1976, Rothwell *et al.* 1991). Trembling aspen stands often act as natural fuelbreaks during wildfires (Fechner and Barrows 1976), and fires sometimes bypass trembling aspen stands surrounded by conifers (Rothwell *et al.* 1991). In an analysis of fires in trembling aspen in National Forests of the Intermountain West (USFS Regions 2, 3, and 4) from 1970 through 1982, Bevins (1984) reported that wildfires that burned thousands of acres during extreme weather conditions usually penetrated less than 20 m into trembling aspen. Managers he interviewed used the terms “asbestos type” and “firebreak” to describe trembling aspen stands. Bevins reported that mixed trembling aspen-conifer types such as those on the northern Kaibab and Dixie National Forests did sustain fires, however, and burned substantial amounts of trembling aspen. Throughout all three regions, a relatively few, large fires (> 100 acres burned) accounted for 93.2 percent (or 1.12 million acres) of all trembling aspen burned.

Fire history: Before and during the mid-nineteenth century, fires were apparently more frequent, and larger acreages of trembling aspen and trembling aspen-conifer mixes burned, than any time since. A large majority of the trembling aspen stands in Jackson Hole, Wyoming, date from fires between 1850 and 1890 (Gruell and Loope 1974). In central Utah, Baker (1925) and Meinecke (1929) found few trembling aspen fire-scarred later than 1885. Earlier fire scars were common and showed a 7- to 10-year fire frequency. Since trembling aspen is fire-sensitive, the fires were probably of low severity. Extensive sampling of trembling aspen in Colorado found few fire scars dating later than about 1880 (Davidson *et al.* 1959).

These data indicate that there has been a great reduction of fire rejuvenation of trembling aspen in the West since about 1900. Extensive young stands of trembling aspen are uncommon in the West (Jourdonnais and Bedunah 1990, Shepperd 1981, DeByle *et al.* 1987). Conifers now dominate many seral trembling aspen stands. Probable contributing factors are:

1. highly effective direct control of wildfires in the last 50 years, especially in the trembling aspen type (DeByle *et al.* 1987),
2. reduction of fine fuels in trembling aspen/grass and trembling aspen/forb types due to grazing (Brown and Simmerman 1986, DeByle *et al.* 1987), and

3. cessation of deliberate burning by Native Americans (Barrett and Arno 1982, Gruell 1985, Houston 1973).

Ungulates, fire, and trembling aspen: In most areas, ungulate browsing is probably not a major factor restricting post-fire trembling aspen regeneration. Trembling aspen has increased in importance in the East despite browsing pressure from large White-tailed Deer populations. In many areas of the United States, Elk populations impact trembling aspen very little. Browsing Elk had no significant impact on trembling aspen sprout density after wildfire in New Mexico (Patton and Avant 1970). In some areas, however, fire suppression coupled with heavy ungulate browsing has reduced trembling aspen regeneration. Failure of some stands in the Great Lakes States to regenerate has been attributed to overbrowsing of sprouts by White-tailed Deer (Schier *et al.* 1985a).

Overbrowsing has particularly been noted in northwestern Wyoming, in Yellowstone and Grand Teton National Parks and the Bridger-Teton National Forest. Elk are the primary browsers of trembling aspen in this area, although where Moose populations are high, Moose have also removed considerable trembling aspen regeneration. Historic narratives and photographic evidence suggest that ungulates were a major biotic influence on trembling aspen in this region during the exploration and settlement periods. However, fires were extensive during this period, so post-fire sprouting of trembling aspen and growth of palatable grasses, shrubs, and herbs, probably produced a forage supply that dispersed browsing ungulates sufficiently for trembling aspen to regenerate (Gruell and Loope 1974).

Coring of old trembling aspen stems in Yellowstone National Park showed that most live, large trembling aspen established in a brief period between the 1870s and 1880s: a period of severe fires followed by above-normal precipitation. Elk, Moose, and Beaver populations were at a historic low, and some wolves were present. Neither this combination of conditions nor significant trembling aspen regeneration has occurred since then. Elk populations were low in the 1950s and 1960s, but fires were suppressed and the climate was dry. In the 1910s, there were numerous Elk and Beaver and few fires. After the 1988 fires, Elk numbers were high and climatic conditions were dry. In this region, even large-scale burning does not seem sufficient for trembling aspen regeneration (Gruell and Loope 1974, Kay 1993, Romme *et al.* 1995).

Prairie: Frequent fires on prairies and plains grasslands historically helped control trembling aspen invasion (Buell and Buell 1959). Fire may have been only one of several factors controlling trembling aspen, however. Drought (Hildebrand and Scott [n.d.]) and ungulate browsing may have worked in conjunction with fire to curtail woody plant invasion. Fire alone may not control trembling aspen spread (Campbell *et al.* 1994). Anderson and Bailey (1979) reported that 24 years of annual spring burning checked trembling aspen invasion onto tallgrass prairie, but actually increased the number and cover of trembling aspen sprouts in the area. Elk Island National Park, Alberta, was described by early settlers as a grassland with scattered

trembling aspen groves. By 1895, extirpation of Bison and severe reduction of other ungulates was followed by expansion of trembling aspen. Bison were reintroduced with park establishment, but fire was not. Ungulate populations rose rapidly and were culled in the 1930s and 1950s. Grassland expanded with the ungulates, while trembling aspen expanded when culling occurred (Blinn and Buckner 1989).

POSTFIRE REGENERATION STRATEGY

Tree with adventitious-bud root crown/soboliferous species root sucker; initial-offsite colonizer (off-site, initial community).

IMMEDIATE FIRE EFFECT ON PLANT

Small-diameter trembling aspen is usually top-killed by low-severity surface fire (Jones and DeByle 1985). Brown and DeByle (1987) found that as DBH increases beyond 15 cm, trembling aspen becomes increasingly resistant to fire mortality. Large trembling aspen may survive low-severity surface fire, but usually shows fire damage (Brown and DeByle 1987, Kovalchik 1987). Moderate-severity surface fire top-kills most trembling aspen, although large-stemmed trees may survive. Some charred stems that survived low- or moderate-severity fire initially have been observed to die within 3 or 4 post-fire years. Severe fire top-kills trembling aspen of all size classes.

Moderate-severity fire does not damage trembling aspen roots insulated by soil. Severe fire may kill roots near the soil surface or damage meristematic tissue on shallow roots so that they cannot sprout. Deeper roots are not damaged by severe fire and retain the ability to sucker (Gruell and Loope 1974, Tucker and Jarvis 1967, Schier and Campbell 1978, Schier *et al.* 1985b).

Mortality does not always occur immediately after fire. Sometimes buds in the crown will survive and leaf out prior to the death of the tree (Brown and DeByle 1987). Brown and DeByle (1987) reported that trembling aspen trees died over a 4-year period following fires in Wyoming and Idaho, although most individuals succumbed by the second post-fire year. Even when trembling aspen is not killed outright by fire, the bole may be sufficiently damaged to permit the entrance of wood-rotting fungi (Kovalchik 1987). According to Jones and DeByle (1985), basal scars which lead to destructive heart rot can be made on even good-sized aspen by "the lightest of fires." Basal fire scars may also permit entry of borers and other insects which can further weaken the tree (Brinkman and Roe 1975).

DISCUSSION AND QUALIFICATION OF FIRE EFFECT

Fire may kill (as opposed to top-kill) a deteriorating stand of trembling aspen. A deteriorating stand on the Sweetwater drainage of the Wind River Mountains, Wyoming, failed to sprout following a 1963 wildfire. However, another 1963 wildfire in the Wind River Mountains, near Pinedale, had the opposite effect on a

deteriorating stand of trembling aspen. Although the site was considered poor for trembling aspen due to dry, sandy soil, fire only top-killed the stand. Browsing pressure on sprouts was light, and post-fire stocking was "more than adequate" for regeneration (Gruell and Loope 1974).

The position of an individual tree on a slope, or within a stand, can influence the degree of damage caused by fire. Even when damaged, trees located near the boundaries of a fire can often maintain a live crown. These peripheral trees may receive food supplies from the roots of unburned neighbours. Trembling aspen on slopes generally show greater damage than do trees on flatter areas. Flames moving uphill often curl up the lee side of trees when fanned by upslope wind, charring the stem further up its bole. The effect of slope is particularly pronounced (up to 31-44% higher char heights) after fires of higher severity. This relationship is presented in the following table (Brown and DeByle 1987):

DBH (cm)	Probability of mortality	
	0.90	0.95
	Average char height (cm)	
10	5	12
15	14	21
20	23	30
25	32	39
	Uphill char height (cm)	
10	6	16
15	19	29
20	31	42
25	44	55

PLANT RESPONSE TO FIRE

Trembling aspen sprouts from the roots and establishes from off-site, wind-blown seed after fire (Brown and DeByle 1989, Perala 1990, Stickney 1989). It is the classic soboliferous species described by Stickney (1989): a plant that sprouts from carbohydrate-storing lateral roots (sobols).

Sprouting: Trembling aspen generally sprouts vigorously after fire. Long-term growth and survival of trembling aspen sprouts depend on a variety of factors including pre-fire carbohydrate levels in roots, sprouting ability of the clone(s), fire severity, and season of fire. Moderate-severity fire generally results in dense sprouting. Fewer sprouts may be produced after severe fire. Since trembling aspen is self-thinning, however, sprouting densities are generally similar several years after moderate and severe fire. A low-severity surface fire may leave standing live trees that locally suppress sprouting, resulting in an uneven-aged stand (Bartos and Mueggler 1979, Bartos and Mueggler 1981, Brown and Simmerman 1986, Perala 1990).

Trembling aspen burned in spring generally sprouts later in the growing season and again the following year. Fires in mid-growing season generally result in late-season sprouting. Trembling aspen burned in late summer or fall usually sprouts the next spring (Brown and Simmerman 1986).

Predicting post-fire sprouting: Applying prescribed fire in exclosures in Yellowstone National Park, Renkin and Despain (1994) found that root biomass can be estimated from basal area, and both can be used to predict local response of trembling aspen to burning. Sprout biomass produced in post-fire year 1 was positively correlated ($r^2 = 0.90$, $p = 0.013$) with both pre-fire basal area and root biomass. On average, 11.5 tonnes/ha of root mass were required to produce 0.1 tonne/ha of sprouts. Average sprout height was positively correlated with basal area and root biomass ($r^2 = 0.85$, $p = 0.004$). On average, 25 m²/ha of basal area and/or 19 tonnes/ha of root biomass were required to produce 0.5 m of sprout growth.

Examples of sprouting: After the 1988 fires in Yellowstone National Park, percentage of sprouts produced in spring, 1989, was significantly higher ($p = 0.030$) in burned stands (mean 82%) than on unburned stands (mean 60%). The percentage of sprouts in fall, 1989, was also higher ($p = 0.103$) on burned stands (mean 82%) than in unburned stands (mean 65%). In spring 1990, sprout density averaged 80 000 stems/ha in burned stands and 27 000 stems/ha in unburned stands. By fall 1991, density was 38 000 stems/ha in burned and 25 000 stems/ha in unburned stands, respectively. Mean heights were 24 cm in spring 1990 and 27 cm in spring 1991. Browsing intensity was much higher in winter and spring (45-55% of sprouts browsed) than summer and fall (5-10%). There were no significant differences in browsing among burned stands, unburned stands adjacent to burned stands, and remote unburned stands: Sprouts were heavily browsed in all stand types (Romme *et al.* 1995).

Birch-aspen: Following a 1944 summer wildfire in Maine, trembling aspen and paper birch sprouted vigorously, forming a dense stand. In 1951, there were 40 000 to 45 000 stems (both species) per acre. Trembling aspen dominated the stand; it averaged 6 m in height while paper birch averaged only 1.8 m (LaBonte and Leso 1990).

Seedling Establishment: Fire exposes mineral soil, which is an excellent seedbed for trembling aspen (Godman and Mattson 1976). Trembling aspen seedlings have been noted following severe fire in Canada. Six years after fire in northeastern Wisconsin, trembling aspen seedlings composed 20 to 35 percent of seedlings of all species present on the burn (Horton and Hopkins 1966). Kay (1993) reported good seedling establishment following 1986 fires in Grand Teton National Park and 1988 fires in Yellowstone National Park. Height growth was negligible, however, due to ungulate browsing. Density, height, and ungulate use of trembling aspen seedlings on the Yancy's Hole Burn, Yellowstone National Park, were (Kay 1993):

Transect #	Year	Number/ha	% browsed	Mean height (cm)
1	1989	177 202	--	62
	1991	32 154	100	50
2	1989	141 362	--	60
	1991	46 148	100	57
3	1989	109 522	--	53
	1991	16 660	100	75
Mean	1989	142 695	--	58
	1991	31 654	100	47

Renkin and others (1994) are conducting a similar seedling study on forested and nonforested sites in Yellowstone National Park; only preliminary data are available at this time. They found that trembling aspen seedlings were concentrated on wet microsites but widely scattered on other site types. In 1989, trembling aspen seedling density on 14 plots ranged from 0.6 to 1014 per m²; average height ranged from 5.7-27.8 cm, mean = 12.8 cm.

Trembling aspen seedlings were two to four times taller than lodgepole pine seedlings on forested plots. In 1990, all plots had persistent trembling aspen seedlings; in some cases the stem had died back but the 1-year-old roots had produced suckers. Density of surviving seedlings ranged from 0.05 to 332 per m². Average heights had increased, ranging from 9-39 cm, mean = 19.4 cm. Trembling aspen seedlings on fenced plots averaged 30 cm in height; seedlings on unfenced plots averaged 13.4 cm. Seedling survival was significantly greater ($p = 0.004$) on forested than nonforested plots. Survival was also influenced by presence of ungulates, spring flooding, disease, and intraspecific competition. Ungulate presence negatively influenced seedling survival on unfenced plots ($r = 0.97$, $p = 0.004$). Plots submerged in spring showed high seedling mortality. A fungus (*Venturia tremulae*) also contributed to seedling death or dieback (Renkin *et al.* 1994).

FIRE MANAGEMENT CONSIDERATIONS

Prescribed fire is recommended for trembling aspen (Anderson and Bailey 1979, Brown *et al.* 1989, Perala 1990, Schier and Campbell 1978). Currently, an estimated 240 ha of trembling aspen burns per year in the Intermountain Region. At that rate, it will require 12 000 years to burn the entire trembling aspen type in that region. It is likely that seral trembling aspen will be replaced by conifers; stable trembling aspen stands may become less productive (DeByle *et al.* 1987). In many areas of the West, trembling aspen stands have lived longer than they did prior to fire exclusion, and many stands are in a state of decline due to advanced age (Gordon 1976). Gruell and Loope (1974) found that in Jackson Hole, Wyoming, trembling aspen

stands begin to deteriorate after about 80 years. Houston (1973) stated that trembling aspen in Yellowstone National Park were primarily large trees ranging from 75 to 120 years of age.

Applying fire: Prescribed fire is often difficult to apply in trembling aspen stands because of the prominence of live fuels and often sparse distribution of fine dead fuels (Brown *et al.* 1989). Even if fuels are plentiful, they are usually too moist to burn easily. Prescribed fire may be possible, however, when live vegetation cures enough to contribute to fire spread rather than hinder it. The combination of dry weather and cured fuels occurs most often in early spring, late summer, and fall (Quintilio *et al.* 1991, Rothwell *et al.* 1991). The forest floor of a trembling aspen stand immediately after snowmelt is covered by matted, cured surface vegetation and deciduous leaf litter. Before leaf-out this mat is directly exposed to drying by wind and sun, which increase fuel temperature and decrease fuel moisture. Without rain, the withered leaves in the litter begin to curl, resulting in a more favourable fuelbed for combustion and heat transfer. In Alberta, these moderately severe, early season burning conditions can persist from snowmelt until the first week in June (Quintilio *et al.* 1991).

In most years, leaf fall and autumn precipitation coincide, making fall burning difficult. If September and October are dry, however, burning may be possible. Surface fuels are dead and sometimes frozen, with a continuous layer of loosely packed leaves, making trembling aspen more flammable than at any other time of year (Rothwell *et al.* 1991).

Live fuel moisture varies greatly between understory species throughout the growing season, but can be estimated well enough to determine when to light prescribed fires. Brown and others (1989) estimated that when herbaceous vegetation is the primary fine fuel, at least 50 percent curing is needed to sustain fire spread. Less than 50 percent curing may be sufficient in stands with substantial conifers. Brown and Simmerman (1986) provide a method for appraising fuels and flammability in trembling aspen to assist managers in choosing when to apply prescribed fire and help determine proper conditions for burning. Five fuel types in 19 community types common in the Intermountain West are presented, accompanied by color photographs.

Prescriptions: Aspen parkland and northern forest - Bailey (1978a, 1978b) found that in Alberta, prescribed burning in trembling aspen forests and parklands in spring was usually not successful above relative humidity of 35 to 40 percent. He recommended that prescribed burning be conducted 8 to 10 drying days after snowmelt, when air temperature is at least 18° C), relative humidity is less than 30 percent, and 10 m height open winds are 9-35 km/h.

Bailey and Anderson (1980) reported that in central Alberta, trembling aspen forest in a grassland-shrub-trembling aspen forest mosaic was the most difficult of the three vegetation types to prescribe burn. With spring burning, backfires consistently gave poor results, frequently going out within a few feet of ignition and yielding a maximum

temperature of only 288° C. Headfires were hotter but gave variable results. Most headfire temperatures ranged from 371-482° C, but 14 percent were in excess of 600° C. Fire and fuel data from the trembling aspen sites follow.

fire temperature	393 +/- 28* ° C
total fuel	13 436 +/- 354 kg/ha
ground fuel	11 704 +/- 337 kg/ha
standing woody fuel	1732 +/- 181 kg/ha

*standard error of the mean (SEM)

Perala (1974) recommended this prescription for burning trembling aspen slash in the Great Lake States:

Months for burning	dormant season (all but June, July, & August)
Fuel model*	D
Air temperature	> 18° C
Relative humidity	< 35%
Ignition component*	40 - 50
Energy release component*	14 - 17
Spread component*	4 - 7
Burning index*	13 - 21
Wind**	2.5 - 5 m/s
Number of days with less than 2.5 mm rain	> 5

*from the National Fire-danger Rating System (Deeming *et al.* 1974)

**measured 20 ft. above ground, or at average height of vegetation cover, averaged over at least a 10-minute period

Canadian Forest Fire Behavior Prediction (FBP) System: Alexander and Maffey (1993) provide examples for predicting fire spread rate, fuel consumption, and frontal intensity in trembling aspen types using the FBP System.

Forage quality and fire: Three burned trembling aspen/shrub/tall forb communities on the Caribou National Forest, Wyoming, showed increased forage quality (better Ca:P ratios, higher Elk digestibility, and higher crude protein and P levels) than adjacent unburned sites during the first post-fire year. By the second post-fire year, there were no significant differences between forage quality on burned and unburned sites. Shrubs on the unburned sites were above browse level throughout the study period, however, while shrubs on the burned site were still accessible to Elk in the second post-fire year (DeByle *et al.* 1989).

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Prunus emarginata
Bitter cherry

FIRE ECOLOGY OR ADAPTATIONS

Bitter cherry sprouts vigorously following fire (Conrad 1987, Laursen 1984, Martin and Dell 1978, Neuenschwander [n.d.], Wright 1978). Approximately 15 to 50 sprouts per plant were produced after a prescribed fire in northern Idaho (Leege 1968, Leege 1969). Post-fire regeneration also includes germination from on-site seed (Kramer 1984, Morgan 1984, Stickney 1986), and probably also from off-site seed dispersed by birds and mammals. Pre-fire canopy coverage is attained about 30 to 40 years following fire (Neuenschwander [n.d.]).

In ponderosa pine (*Pinus ponderosa*) forests of eastern Washington and the northern Rocky Mountains, where bitter cherry occurs, fire return intervals of 6 to 22 years (Weaver 1967) and 6 to 11 years (range of 2-20 years) (Arno 1976) have been described. In western Montana at the Burdette Creek winter range, Losensky (1987) describes mean fire-free intervals of 37 years.

POSTFIRE REGENERATION STRATEGY

Tall shrub, adventitious-bud root crown; small shrub, adventitious-bud root crown; ground residual colonizer (on-site, initial community); secondary colonizer - off-site seed.

IMMEDIATE FIRE EFFECT ON PLANT

Bitter cherry is top-killed or killed by severe fires (Leege 1979, Martin and Dell 1978, Morgan and Neuenschwander 1988b). High-severity fires favour bitter cherry (Morgan and Neuenschwander 1988b). Young (1983) stated that bitter cherry is "unharmful to enhanced" by fire. However, bitter cherry mortality can be high when burning occurs while plants are actively growing. Approximately 14 percent of 36

mature bitter cherry died after a single spring (late March/early April) prescribed fire on a seral brushfield in northern Idaho (Leege and Hickey 1971). Repeated spring burning of similar sites at 5-year intervals resulted in heavy bitter cherry mortality (Leege 1979). Seven plants sprouted following a fire in late March 1965. Of these, four sprouted after a second fire in May 1970, and two remained alive after a third fire in May 1975. Total mortality by 1976 was 86 percent. Leege (1979) suggested that the tendency toward fire-induced mortality in bitter cherry was probably accentuated by the advanced phenologies of plants during the second and third fires.

PLANT RESPONSE TO FIRE

Bitter cherry sprouts from the root crown following fire (Husari 1980, Richardson 1980). Since root sprouting has been documented in bitter cherry (Ferguson 1983) it probably also sprouts from roots after fire. It establishes from buried seed or seed dispersed onto burned sites (Stewart 1978). Several studies have reported rapid recovery and substantial post-fire increases in bitter cherry densities (Cattelino 1980, Gashwiler 1970, Steele and Geier-Hayes 1991). In an Oregon coastal brushfield, bitter cherry sprouted from roots 4 months after fire (Kelpsas 1978). In an Oregon red alder (*Alnus rubra*)-dominated brushfield burned on August 9, 1974, bitter cherry started sprouting within 2 to 3 weeks. By November, bitter cherry was sprouting vigorously and stems were 1 m tall (Roberts 1975).

In Montana a prescribed fire occurred on April 13, 1988. Plots were observed from mid-July to mid-September. Bitter cherry sprout twig weights on burned plots exceeded twig weights on unburned plots by a factor of 4 on southwest-facing forested types and by a factor of 9 on a southeast-facing forested type (Makela 1990).

In northern Idaho more bitter cherry sprouts are produced per surviving plant after spring fires than fall fires; however, sprout height is usually greater after fall fires. Bitter cherry recovery trends 1 year after a prescribed fire in seral brushfields in northern Idaho are presented below (Leege and Hickey 1971):

	time of fire	
	spring	fall
average # of basal sprouts per plant		
pre-fire	0.4	0.5
post-fire	18.3	15.2
average height basal sprouts (ft)		
post-fire	2.0	2.6
average crown diameter (ft)		
pre-fire	5.0	4.7
post-fire	1.9	2.3
average crown height (ft)		
pre-fire	15.1	13.9
post-fire	3.4	4.1

A seral brushfield in northern Idaho was prescribed burned in 1965, 1970, and 1975. Bitter cherry sprouted from the root crown and a few seedlings were observed. The average pre-fire crown height and diameter were 8.4 feet and 1.6 feet, respectively. Bitter cherry recovery trends 2 years after the prescribed fires are presented below (Leege 1979):

		year	
	1966	1971	1976
maximum crown height (ft)	5.2	5.5	4.1
maximum crown diameter (ft)	2.2	2.5	2.0

Idaho studies investigating post-fire successional patterns in a western redcedar (*Thuja plicata*)/queencup beadlily (*Clintonia uniflora*) habitat type found that bitter cherry regenerated predominantly from seedbanks. Seedling establishment was affected by fire severity. Bitter cherry seedlings had greatest percent cover on low-severity burns in post-fire years 1 to 3; in post-fire years 4 and 5, bitter cherry seedling percent cover was greatest on high-severity burns (Morgan and Neuenschwander 1988a, Morgan and Neuenschwander 1988b).

A seral brushfield in northern Idaho was prescribed burned on May 2, 1966; all aboveground vegetation was "totally consumed." In May 1967, 257 bitter cherry seedlings were observed. By May 1968, 14 bitter cherry seedlings remained (Leege 1978). In northeastern Idaho, on a May 14, 1975 prescribed burn, seven bitter cherry seedlings were observed in post-fire year 2. By post-fire year 4, four seedlings remained (Leege and Godbolt 1985). In Oregon a Douglas-fir (*Pseudotsuga menziesii*) clearcut was prescribed burned in late fall. Bitter cherry seedlings were present post-fire year 1 on the burned site but were not present in adjacent virgin forest (Gashwiler 1970).

In Oregon, 96 hectares of a Douglas-fir watershed were harvested over a 4-year period from the fall of 1962 to the summer of 1966. They were broadcast burned in October 1966. Bitter cherry reached peak abundance about 10 years after fire (Halpern 1989). In north-central Idaho in a grand fir (*Abies grandis*)/*pachistima* habitat type, 36 stands representing 1, 3, 8, 12, and 23 year age classes and 7 near-climax stands were clearcut and broadcast burned. Canopy cover and height of bitter cherry for each age class were as follows (Zamora 1975):

age class	canopy cover	height	
	(%)	inches	centimetres
1	0.1	9	22
3	0.1	17	44
8	0.4	25	63
12	0.9	49	122
23	0.3	38	95
near-climax	0.0	--	--

In the western Cascades bitter cherry was absent from undisturbed old-growth Douglas-fir stands, but was abundant on Douglas-fir plantations that were clearcut and broadcast burned 2 to 40 years ago. Shrub dominance decreases at 20 to 30 years with canopy closure (Schoonmaker and McKee 1988).

FIRE MANAGEMENT CONSIDERATIONS

Bitter cherry is a member of the seral brush community in northern Idaho. These brushfields provide excellent habitat for Elk. Bitter cherry is one of the dominant shrubs after fire, but grows out of reach of browsing animals within several years. If brushfields are not maintained by fire, coniferous forest may eventually establish and large ungulate habitat will be lost (Hickey 1971, Hooker and Tisdale 1974, Mueggler 1961, Yurich 1974). Bitter cherry was found to have greater frequency and crown cover on single broadcast burned sites than on unburned and piled-and-burned sites (Mueggler 1961).

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Pseudotsuga menziesii var. *menziesii*
Coast Douglas-fir

FIRE ECOLOGY OR ADAPTATIONS

Plant adaptations to fire: Coast Douglas-fir is more fire resistant than many of its associates and can survive moderately intense fires. Thick, corky bark on the lower bole and roots protects the cambium from heat damage. In addition, the tall trees have their foliage concentrated on the upper bole, which makes it difficult for fire to reach the crown (Morrison and Swanson 1990); however, it should be noted that trees are typically not free of lower branches up to a height of 10 m until they are more than 100 years old (Hermann and Lavender 1990). Moderately severe understorey burns in 50- to 60-year-old mixed and pure stands near Mount Rainier caused little cambial injury to Douglas-fir but killed most of the thin-barked western redcedar (Swanson 1976). Following the Hoh Fire in Olympic National Park, Douglas-fir's survival rate was considerably higher than Sitka spruce (*Picea sitchensis*), western redcedar, western hemlock, and bigleaf maple (*Acer macrophyllum*) (Agee and Huff 1980).

When trees are killed, Douglas-fir relies on wind-dispersed seed off-site trees to colonize the burned area. If catastrophic fires are extensive, a seed source may be limited due to the lack of seed trees. Under these circumstances, seeds come from mature trees which survive fire, survivors in small unburned pockets, or from trees adjacent to the burned area. Where seed trees are scarce, it may take 100 years or more for Douglas-fir to restock the burned area (Spies and Franklin 1988). On the other extreme, when fires do not kill all the trees in a stand, seedling establishment may begin within a year or two after burning (Huff 1984). Mineral soils exposed by fire are generally considered favourable seedbeds (Feller 1982).

Fire regime: Widely distributed as a canopy dominant in lower and middle elevation forests throughout the Pacific Northwest, Douglas-fir occupies forests with varied fire regimes. In general, the size and severity of natural fires tend to decrease, while fire frequency increases southward from western Washington to northern California (Morrison and Swanson 1990). In western Washington, Douglas-fir is a primary component of moist forests experiencing infrequent, widespread, stand-replacing fires that occur at perhaps 400- to 500-year intervals (Hemstrom and Franklin 1982, Lotan *et al.* 1981). Dry areas of the western hemlock zone in the central Oregon Cascades experience both frequent, low- to moderate-severity fires and stand-replacing fires (Morrison and Swanson 1990). The mean fire interval in these forests is between 50 and 150 years (Lotan *et al.* 1981, Means 1982). Frequent, low- to moderate-severity fires occasionally crown and create patches of even-aged stands. Underburning is more common and allows Douglas-fir to survive repeated fires. Thus uneven-aged old-growth Douglas-fir stands are more common in the central Oregon Cascades (Morrison and Swanson 1990, Stewart 1989). In mixed evergreen forests of southern Oregon and northern California, fires occurred at frequencies of 5 to 25 years (Lotan *et al.* 1981). Where Douglas-fir is seral, its great longevity allows it to maintain itself as a canopy dominant until the next catastrophic fire (Huff 1984).

POSTFIRE REGENERATION STRATEGY

Off-site colonizer; seed carried by wind; post-fire years 1 and 2; secondary colonizer; off-site seed carried to site after year 2.

IMMEDIATE FIRE EFFECT ON PLANT

Trees: Crown fires commonly kill all trees over extensive areas. Hot ground fires that scorch tree crowns and char tree boles kill variable proportions of coast Douglas-fir (Agee and Huff 1980). Rapidly spreading ground fires tend to inflict more damage to Douglas-fir crowns, while slow spreading ground fires are damaging to the bole and can kill trees through cambial heating (Peterson and Arbaugh 1989). Crown scorching from summer fires is more damaging than late summer or fall fires because more buds are killed. During late summer the buds are set and subsequent-year needles are well protected (Wagener 1961). Seedlings and saplings are susceptible to and may be killed by even low-intensity ground fires (Volland and Dell 1981).

Seed: Temperatures in excess of 60° C are lethal to Douglas-fir seeds. Thus most seeds on the forest floor will be destroyed by fire (Isaac 1943). Crown fires will kill seeds in green cones; however, green cones are relatively good insulators and are not highly flammable, and fires that are not excessively hot often only scorch the cones. Seeds can mature in scorched cones on fire-killed trees, and later disperse onto the burned area (Isaac 1943).

DISCUSSION AND QUALIFICATION OF FIRE EFFECT

The Hoh Fire in June 1978 burned 500 hectares of montane and subalpine coniferous forests. The montane forest was composed primarily of 400- to 500-year-old western hemlock and Douglas-fir. About 10 percent of this forest type was consumed by a crown fire, while most of the remaining forest experienced a hot ground fire with considerable crown scorching. All trees less than 3 m tall were killed. Mortality of overstorey trees was high, but Douglas-fir suffered the least. The fire reduced the basal area of all species as follows:

bigleaf maple	100%
western hemlock	80%
western redcedar	50%
Sitka spruce	50%
coast Douglas-fir	33%

Ninety-one percent of western hemlock overstorey trees were killed, compared with 62 percent of Douglas-fir (Agee and Huff 1980).

PLANT RESPONSE TO FIRE

Seedling establishment following fire is dependent on the spacing and number of surviving seed trees. Seedling establishment following large stand-destroying fires is slow because seed trees are killed over extensive areas. After the Tillamook Fire in northwestern Oregon, seedlings were restricted to areas around scattered survivors and near the burn edge (Wright and Bailey 1982). Conversely, Douglas-fir can quickly establish a new cohort of seedlings if there are numerous, well-spaced surviving seed trees within the burned area. Conifer seedlings were abundant after the Hoh Fire in Olympic National Park, where 38 percent of mature Douglas-fir survived. Three years after this fire, there were about 8400 conifer seedlings per ha, about half of which were Douglas-fir (Huff 1984). In the northern Cascades, Douglas-fir seedling frequency was between 80 and 100 percent on three different 4-year-old burns ranging in size from 22-166 ha (Miller and Miller 1976).

FIRE MANAGEMENT CONSIDERATIONS

Natural regeneration following slash burning: The effects of slash burning on coast Douglas-fir seedling establishment are contradictory. Some researchers report greater stocking on burned areas, while others report greater stocking on unburned areas (Morris 1970). It appears that Douglas-fir is favoured by slash burning on mesic and wetter sites in the western hemlock zone. On dry sites, natural regeneration following broadcast burning is unpredictable, and often poor, due to high soil temperatures and moisture stress (Feller 1982).

Following broadcast slash burning in clearcuts in the Coast and Cascade Ranges in Oregon and Washington, 90 percent of the area is typically moderately or lightly burned, which provides good seedbeds (Miller *et al.* 1974, Morris 1970). Severe burning, which retards Douglas-fir regeneration because of altered physical and chemical soil properties, occurs on less than 10 percent of the area. This is typically where burning stumps produce intense heat (Feller 1982). Minore (1986) found that 5-year-old coast Douglas-fir seedling height was lower on sites where slash was piled and burned than on broadcast burned sites.

On erodible granitic soils of the Klamath National Forest, hand planted Douglas-fir seedlings were 7 times more numerous on unburned than on burned plots, 6 years after planting (Strothmann and Roy 1984).

Models: Peterson and Arbaugh (1989) present a model for predicting post-fire survival of coast Douglas-fir in the Cascades. The model uses crown and bole damage variables to predict survival.

Salvage logging: Typically less than 3 percent of all merchantable timber in coast Douglas-fir stands is consumed by forest fires. Deterioration rates vary with tree size and wood type. Very little sapwood can be salvaged 3 years after fire, but heartwood

deteriorates more slowly. Salvage has been carried out for 1 to 2 years in young-growth stands, 4 to 7 years in intermediate stands, and for 5 to 10 years in old-growth stands (Kimmey and Furniss 1943).

Duff reduction: Duff consumption by prescribed burning can be predicted using weather and fuel variables on cut-over Douglas-fir sites. Generally, most duff is burned when the moisture content of the upper duff is below 30 percent. When the upper duff layer exceeds 120 percent moisture content no combustion takes place. Between these values, the percentage of duff consumed depends on the amount and moisture content of fine woody fuels (Little *et al.* 1986, Sandberg 1980).

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Quercus garryana

Garry oak

FIRE ECOLOGY OR ADAPTATIONS

Fire ecology: Historically, Garry oak was subjected to a fire regime of low-severity surface fires occurring every few years. A study in the Garry oak woodlands of Humboldt Redwoods State Park, California, revealed a history of fire every 7.5 to 13.3 years during the presettlement era (Stuart 1987). Frequent fire resulted in the open savannas typical of presettlement times in the Willamette Valley, Oregon, and the bald hills of California (Sugihara *et al.* 1987). Dead woody fuels were scant, but flashy fuels (grasses) were abundant and dry early in summer. The fire spread rate was moderated by the gentle topography typical of this cover type. Fire seldom spread into adjacent coniferous forests (Atzet and McCrimmon 1990).

Plant adaptations: Garry oak has adapted to low- to moderate-severity fire by sprouting from the bole, root crown, and roots (Griffin 1980, McDonald *et al.* 1983, Sugihara and Reed 1987). Sprouts of this species grow far more rapidly than do seedlings. Young trees not subjected to periodic top-kill by fire followed by sprouting often do not attain sexual maturity before they succumb to herbivory (Sugihara and

Reed 1987). Initial establishment of seedlings is somewhat dependent on fire also. Although this species does not require a bare mineral seedbed, seedling recruitment is greatly enhanced when the litter layer has been removed by fire (Arno and Hammerly 1977).

POSTFIRE REGENERATION STRATEGY

Tree with adventitious-bud rootcrown/soboliferous species root sucker; ground residual colonizer (onsite, initial community); secondary colonizer - offsite seed.

IMMEDIATE FIRE EFFECT ON PLANT

Crown fire generally kills this species. Moderate-severity surface fire rarely kills large trees, but smaller oaks may be killed or suffer severe cambium damage (Burns and Honkala 1990). Low-severity surface fire rarely harms mature trees, but seedlings and saplings are commonly top-killed. Animal-buried acorns are usually not affected by fire (Reed and Sugihara 1987, Sugihara *et al.* 1987).

PLANT RESPONSE TO FIRE

Most researchers report vigorous sprouting of top-killed Garry oak (Dale *et al.* 1986, McDonald *et al.* 1983, Reed and Sugihara 1987, Roy 1955, Veihneyer and Johnston 1944), although at least one researcher (Griffin 1980) has classified this species as a weak sprouter. Sugihara and Reed (1987) report more vigorous sprouting in 40-year-old than in 70-year-old oaks. Studies conducted on young, even-aged stands show good post-fire recovery of these trees. Garry oak top-killed by fall prescribed burning in Shasta and Tehama Counties, California, exhibited vigorous sprouting during the first post-fire growing season (Veihneyer and Johnston 1944). In Humboldt and Trinity Counties, California, three-year-old sprouts had grown above the browse line (McDonald *et al.* 1983).

In the absence of further fire, these sprout clumps form dense, even-aged stands. Most Garry oak woodlands of today are of this type due to fire suppression. When subjected to further fire, however, weaker meristematic tissue is killed, and individual root crowns produce fewer sprouts per clump with each fire. Continued periodic fire ultimately results in an open savanna with widely scattered, large oaks (Habeck 1962).

Fire research on Brewer oak is extremely limited. One study followed the post-fire recovery of this variety for 3 years after the Three Creeks Burn in Humboldt County, California. This "intense" wildfire top-killed most oak shrubs. At the end of post-fire year 1, sprouts varied in height from 1.3-3.4 m, with an average of 18 sprouts per clump. At post-fire year 3, many of the weaker sprouts had died, and sprouts were reduced to an average of 10 sprouts per clump. Sprout height at post-fire year 3 was not recorded (Roy 1955).

FIRE MANAGEMENT CONSIDERATIONS

Fire appears to be the dominant controlling factor involved in converting invading coniferous forests back to Garry oak woodland. If a conifer forest is the objective, managers can simply allow young invading conifers to grow. In order to halt conifer establishment and facilitate oak regeneration, a minimum frequency of prescribed burning every 5 years is recommended. Ideally, prescribed fire should be set annually. When existing conifers are 3 m or more in height, oak woodlands can be restored by removing conifers by cutting or girdling. A program of prescribed burning is then necessary for long-term maintenance (Sugihara and Reed 1987).

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Rhamnus purshiana
Cascara

FIRE ECOLOGY OR ADAPTATIONS

Cascara will sprout from the root crown following low-intensity fires (Hart 1976, Van Dersal 1938).

POSTFIRE REGENERATION STRATEGY

Survivor species; on-site surviving root crown or caudex; secondary colonizer; off-site seed carried to site after year 2.

IMMEDIATE FIRE EFFECT ON PLANT

Cascara is usually top-killed by fire (Van Dersal 1938).

Pre-burn and post-burn measurements of cascara in central Idaho were as follows (Leege and Hickey 1966):

	Avg. Live Crown Diameter (ft)	Avg. Live Crown Height (ft)	Avg. Crown Below 7ft. (%)	Avg. Dead Crown (%)	No. Basal Sprouts	Avg. Sprout Height (ft)
Pre-burn	4.0	8.0	95.0	95.0	1.0	1.5
Post-burn	3.0	4.5	100.0	100.0	18.0	3.0

1. Pre-burn measurements were taken in March, 1965. Post-burn measurements were taken in August, 1965.
2. Post-burn measurements were taken on the part of the plant which existed before treatment.
3. All cascara's aboveground parts were completely killed by the fire.

PLANT RESPONSE TO FIRE

Change in cascara measurements after prescribed burns in northern Idaho were as follows (Leege 1979):

	1965	1970	1975	1966	1971	1976
Avg. Pre-burn Crown Height (cm):	244	---	---	---	---	---
Avg. Pre-burn Crown Diameters (cm):	91	---	---	---	---	---
Avg. No. Basal Sprouts per Plant:	18	12	7	---	---	---
Avg. Sprout Height (cm):	91	76	76	---	---	---
Max. Crown Height 2 Years After Burn (cm):	---	---	---	183	137	122
Max. Crown Diameter 2 Years After Burn (cm):	---	---	---	137	107	107

FIRE MANAGEMENT CONSIDERATIONS

Fire intervals on southern Oregon sites where cascara occurs range from 30 to 60 years, to longer intervals of 100 to 320 years. These understories are generally free from heavy fuels, giving rise to low-intensity fires. In some plant associations that include cascara, however, high-intensity, stand-replacing fires occur approximately every 60 to 150 years (Atzet and McCrimmon 1990).

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Thuja plicata
Western redcedar

FIRE ECOLOGY OR ADAPTATIONS

Western redcedar fire resistance is low to moderate (Fischer and Bradley 1987). Its thin bark, shallow root system, low dense branching habit, and highly flammable foliage make it susceptible to fire damage (Fischer and Bradley 1987, Parminter 1983). However, it often survives fire because of its large size (Fischer and Bradley 1987). Old western redcedar trees are commonly fire scarred in northern Idaho (Minore 1983). Western redcedar is more severely damaged by fire than any of its associates along the coast region but is less susceptible than Engelmann spruce (*Picea engelmannii*), western hemlock, and subalpine fir in interior regions (Minore 1990).

The frequency of fire in western redcedar stands tends to be low (Boggs *et al.* 1990, Turner 1985). In most of the western redcedar forests from southern British Columbia to northern California moderate to severe wildfires occur at long intervals between 50 to 350 years (Arno 1985). In streamside and seepage areas dominated by western redcedar the mean fire interval is greater than 200 years. In western redcedar habitats on lower and middle slopes the mean fire interval is 50 to 150 years (Arno and Davis 1980). In the Selway-Bitterroot Wilderness those stands dominated by western redcedar had the longest fire regime (Habeck 1976).

POSTFIRE REGENERATION STRATEGY

Tree without adventitious-bud root crown; secondary colonizer - off-site seed.

IMMEDIATE FIRE EFFECT ON PLANT

Western redcedar is commonly killed by fire. Because of their large size, however, old western redcedar trees can often survive if they are not completely girdled by fire (Fischer and Bradley 1987). Shallow roots under the duff layer are often scorched when the duff layer burns and even surface fires may kill western redcedar (Minore 1983). Fire injury to roots can lead to fungal infection, chronic stress, and growth losses (Ryan 1990). The most common causes of fire mortality are root charring and crown scorching (Spalt and Reifsnyder 1962).

PLANT RESPONSE TO FIRE

After fire, western redcedar will readily establish on bare mineral soil seedbeds via off-site wind dispersed seeds (Feller 1982, Graham *et al.* 1988). Although unburned soil benefits western redcedar regeneration more than soil that has been scorched, slash burning favours western redcedar by creating more mineral soil surfaces in cutover areas (Minore 1990).

FIRE MANAGEMENT CONSIDERATIONS

Riparian stringers supporting western redcedar may act as firebreaks because the moist duff does not readily burn (Fischer and Bradley 1987). Old-age western redcedar stands have heavy fuel loads, but a large proportion of this material is in the form of deep duff layers and downed, rotting log material. These stands could support slow moving fires at best; once ignited, however, such heavy fuel materials could support long-lasting fires (Habeck 1976).

Fire-killed western redcedar often shows little deterioration even after 5 years. The bark usually remains intact on dead trees for 5 years. Fire mortality produces no immediate reduction in strength of western redcedar poles, and some large trees remain salvageable for almost 100 years after being killed by fire (Minore 1983).

When slash from decadent western redcedar-western hemlock stands was burned, a greater proportion of western redcedar than of western hemlock slash was consumed. This was a result of greater longitudinal and horizontal fracturing of the western redcedar. When fracturing does not occur, western hemlock slash is at least as flammable as western redcedar slash. Fire spreads faster in western redcedar when the slash from both species is 1 year old. Western redcedar slash does not drop its foliage. The slash of western redcedar is less flammable when chipped. One study showed that the fire hazard normally associated with cutting of western redcedar poles was reduced by skidding entire pole-size trees to the landing, where the slash was chipped and blown over the edge (Minore 1983).

Slash from western hemlock-western redcedar-Alaska-cedar forests produce greater nutrient losses to the atmosphere when the slash composition has a greater proportion of Alaska-cedar and western redcedar. One can expect smaller nutrient losses when western hemlock makes up the majority of the slash (Feller 1988).

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Amphibians

Ambystoma macrodactylum Long-toed Salamander

DIRECT FIRE EFFECTS ON ANIMAL

There are no data available on fire effects on salamanders. It is likely that fire has little direct effect on Long-toed Salamanders. Adults are rarely active above ground; when they are active, it is usually under cool, moist weather conditions (Howard *et al.* 1983) that occur outside the fire season. During the dry season, adult Long-toed Salamanders are generally either in burrows, where they are relatively safe from fire, or under moist rotten logs or moist vegetation mats that are not likely to burn except under extreme fire weather conditions.

If caught in the open during a fire, Long-toed Salamanders would probably be killed. They are very slow-moving (Anderson 1967), and probably cannot escape even slow-moving fire. Even if missed by fire, they probably could not survive the heat.

High temperatures are lethal to Long-toed Salamanders. In the laboratory, adults from northeastern Oregon and western Idaho were killed by water temperatures that ranged from above 33-36° C (Howard *et al.* 1983).

HABITAT RELATED FIRE EFFECTS

Adult and sub-adult Long-toed Salamanders use logs and large branches for cover, and larvae use floating and submerged downed woody debris of all size classes for cover (Anderson 1967). Fire that increases downed woody debris while retaining some overhead shade probably improves habitat structure of Long-toed Salamanders.

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Bufo boreas Western Toad

DIRECT FIRE EFFECTS ON ANIMALS

There is no published information on mortality of Western Toads from fire. The fact that there are no reports of high mortality for any herpetile species may indicate that amphibians and reptiles are not highly vulnerable to fire (Means and Campbell 1981). Kahn (1960) reported that western fence lizards (*Sceloporus occidentalis*) survived a serious chaparral fire by remaining in the soil beneath rocks. Western Toads similarly could survive fire by remaining in the soil beneath rocks, entering animal burrows, or by escaping to water; survival in retreats under flammable materials (logs, stumps, and boards) would depend on fire severity and moisture conditions. Komarek (1969) states that animals appear to respond to fire with adaptive behaviours which minimize mortality; he reports that experiments with different types of prescribed fires resulted in no discernible amphibian mortality. Frogs escaped a backing fire by travelling ahead of the fire, then burying themselves under wet leaves and soil in a small depression (Komarek 1969).

HABITAT RELATED FIRE EFFECTS

Western Toads occupy diverse habitats, some of which experience fire relatively frequently (Douglas-fir forests in drier areas [Agee 1993]), and some of which rarely experience fire (riparian zones tend to act as fire breaks but will burn during extended dry conditions [Crane 1982]).

No specific information describing the response of Western Toads and their habitat to fire was available in the literature. Western Toads are vulnerable to changes in both terrestrial and aquatic habitat. They are, however, found in slightly higher abundance in early seres of Douglas-fir forests (Raphael 1988). Severe fires that burn surface objects such as logs and stumps would immediately decrease available hiding cover for Western Toads, but post-fire sprouting of shrubby species would result in a longer-term, overall increase in low hiding cover (Crane 1982). Most willow species sprout after topkill by fire (Kovalchik *et al.* 1988), so the amount and thickness of willow clumps increases after fire. Crown fires would reduce shade and surface humidity and decrease the amount of daylight time toads could spend active after a fire. Fires during early spring could affect egg masses by reducing shade and increasing water temperatures. Any substantial change in runoff rates, erosion, or water tables caused by fire could degrade breeding sites.

It is likely that there is a change in the relative amounts of different types of prey organisms in the post-fire diet of Western Toads. Immediately after fire, many insects are present but those requiring shade do not adapt well to the more open conditions. In the longer term there are differential responses to fire among prey organisms; ant populations were one-third higher in burned areas than in unburned areas, but beetles tend to decrease on burned areas (Chandler *et al.* 1983).

FIRE USE

Specific information concerning prescribed fire as it affects Western Toads was not available in the literature. Conservation of surface objects used for hiding cover is important in any management decision (Olson 1992).

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Birds

Accipiter striatus
Sharp-shinned Hawk

DIRECT FIRE EFFECTS ON ANIMALS

Direct, fire-caused mortality of Sharp-shinned Hawks has not been reported in the literature. Nestlings and eggs are probably vulnerable to fire; most fledglings and adults could easily escape fire (Lehman and Allendorf 1989).

Some authors (Komarek 1969, Tewes 1984) have described the attraction of hawks to readily available prey at fires and on fresh burns.

HABITAT RELATED FIRE EFFECTS

The effects of fire on Sharp-shinned Hawk habitat are related to habitat structure and to prey abundance and availability. The Sharp-shinned Hawk is most benefited by a mixture of habitats. Fire in dense conifers tends to thin understories and open canopies, making them less suitable for Sharp-shinned Hawk nesting habitat; severe fire can destroy nest trees, roost sites, and perching sites (Lehman and Allendorf 1989, Snyder and Snyder 1975). However, open canopies are more suitable for hunting. Thus, the Sharp-shinned Hawk is vulnerable to either extreme: loss of nesting habitat with fire, or the lack of open foraging areas without fire (Snyder and Snyder 1975). Lehman and Allendorf (1989) stated that lack of fire, with concomitant increases in the density of vegetation, can result in an increase in Sharp-shinned Hawk numbers. However, Sharp-shinned Hawks occur in the following fire-dependent (*sensu* Wright and Bailey [1982]) ecosystems: ponderosa pine, Douglas-fir, redwood, sequoia, and chaparral (Lehman and Allendorf 1989).

Lawrence (1966) reported that predatory birds increased in burned chaparral for the first 2 post-fire years, but declined the third year. Sharp-shinned Hawks were more abundant in the burned area in the first post-fire years, probably due to the increased vulnerability of prey. Declines in later post-fire years were attributed to increased vegetative cover.

In the Southwest, Sharp-shinned Hawk prey populations and diversity decreased during long fire-free intervals; the loss was attributed to a reduction in grassy understorey and in structural diversity caused by lack of fire (Dodd 1988).

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Anas platyrhynchos
Mallard

DIRECT FIRE EFFECTS ON ANIMALS

Fire can and often does destroy mallard nests. However, some females seem devoted to hatching their clutch enough to return to nests to hatch undamaged eggs (Hodson 1965, Leedy 1950).

HABITAT RELATED FIRE EFFECTS

Burning in late May in Manitoba's pothole region showed a drastic decline in mallard nests initiated immediately following burning. Nest initiations rose again in late June (Fritzell 1975). Mallards are early nesters and are adversely affected by spring burns. Also they prefer nesting in dense cover, which is susceptible to heavy burning (Fritzell 1975). Fires before May 10 in Manitoba negatively affect nesting success, and fires after May 10 affect nesting success of later-nesting species (Ward 1968). Also, large scale autumn burns may remove vegetation that is important for capturing snow, which in turn recharges marshes during spring.

Spring burning to remove grass cover showed a slight decrease in mallard nesting on a North Dakota wildlife refuge. On average there were 13 percent fewer of all nesting ducks, including mallard, on plots that were mowed and burned compared to undisturbed plots (Martz 1967). Fires on another North Dakota refuge conducted over a 4-year period showed a greater number of nest successes on plots burned in August and September compared to June fires (Higgins 1986). By the fourth growing season nest success was still greater on the burned plots later, although there was no significant difference between the number of nests on the plots burned in August and September, and the plots burned in June.

FIRE USE

Fires can be used to reduce predator activity through elimination of hiding cover (Fritzell 1975). Rotating spring fires have proved effective for enhancing waterfowl habitat in Manitoba. To ensure the maximum area is available for nesting, burning should be done in small parcels (Ward 1968).

Fire can be used to establish red goosefoot (*Chenopodium rubrum*), an important duck food, by reducing impenetrable reed (*Phragmites* spp.) thickets and breaking solid stands of meadow grass. To avoid harmful effects on ducks burning should be done at times other than during the primary nesting season or shortly before (Higgins 1986). Any burning can reduce nesting cover, however. Autumn fires could potentially destroy rank grasses needed for cover the following nesting season, so some cover should be left at all times. In northern prairies burning should not be conducted any more frequently than every two to three years (Higgins 1986).

Duebbert and others (1981) recommend fire for rejuvenating prairie pothole regions of cool- and warm-season grasses. Cool-season native grasses should be burned from late March through mid-May or mid-August through mid-September. Warm-season native grasses should be burned between mid-May and mid-June (Duebbert *et al.* 1981).

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Buteo jamaicensis
Red-tailed Hawk

DIRECT FIRE EFFECTS ON ANIMALS

Fire directly reduces Red-tailed Hawk reproductive success if the fire crowns in occupied nest trees (Landers 1987). Fires that kill or otherwise alter unoccupied nest trees may disrupt reproduction if acceptable nest trees are scarce. Red-tailed Hawks are reported to be attracted to fire and smoke (Dodd 1988). They have been reported feeding on grasshoppers fleeing from fires (Landers 1987). Low-severity fires probably have little direct effect on Red-tailed Hawks. Landers (1987) commented that light winter burning probably does no substantial harm to raptors.

HABITAT RELATED FIRE EFFECTS

Red-tailed Hawks occur in the following 10 major fire-dependent plant associations in the western United States: grasslands, semidesert shrub-grasslands, sagebrush (*Artemisia* spp.)-grasslands, chaparral, pinyon-juniper (*Pinus* spp.-*Juniperus* spp.) woodland, ponderosa pine, Douglas-fir, spruce-fir (*Picea* spp.-*Abies* spp.), redwood (*Sequoia sempervirens*), and giant sequoia (*Sequoiadendron giganteum*) forests (Lehman and Allendorf 1989).

Suppression of fires in large expanses of treeless areas may benefit Red-tailed Hawks. In southern Saskatchewan, the control of fires on the once open prairies and the planting of trees and shrubs has resulted in a semiopen, tree-grassland mosaic and consequent territory expansion and population increase of Red-tailed Hawks (Palmer 1988).

Although fire may reduce potential nest trees, it may also create snags for perch sites and enhance the foraging habitat of Red-tailed Hawks. Red-tailed Hawks often perch on snags created by lightning strikes (Baker 1974). They often use fresh burns when foraging due to increased prey visibility (Dodd 1988, Landers 1987, Mason 1981, Nichols and Menke 1984). Regular prescribed burning helps to maintain habitat for many prey species of Red-tailed Hawks (Collins 1980, Dodd 1988, Landers 1987, Lehman and Allendorf 1989, Mason 1981).

Several studies indicate that many prey populations increase rapidly subsequent to burning in response to increased food availability (Dodd 1988, Landers 1987). Fire suppression in grasslands was detrimental to small bird and mammal populations due to organic matter accumulation and reduced plant vigour (Wagle 1981).

The suppression of natural fire in chaparral has resulted in reduced seral stage diversity and less edge (Dodd 1988) which has probably affected Red-tailed Hawks in these communities. Red-tailed Hawks are more abundant in recently burned chaparral areas than in unburned areas due to greater visibility and less cover for prey (Nichols and Menke 1984). Additionally, Red-tailed Hawks are favoured by fires that open up or clear pinyon-juniper woodlands (Mason 1981). Raptors associated with pinyon-juniper woodlands depend upon edges of openings created by fire and scattered islands of unburned woodlands (Dodd 1988).

In the first year following a severe fire in grassland, ponderosa pine, Douglas-fir, and mountain big sagebrush (*Artemisia tridentata* ssp. *vaseyana*) habitat types on the Salmon National Forest, several Red-tailed Hawks were observed within the burn. They were not observed in the area before the fire (Collins 1980). Following a fire in a mountain big sagebrush community on the Bridger-Teton National Forest, Red-tailed Hawks were more commonly observed using an area that experienced a severe fall fire than in a nearby area burned by a low-severity spring fire (McGee 1976). Red-tailed Hawks have also been observed hunting on recently burned areas in Colorado County, Texas (Baker 1940).

Although fire is often beneficial to Red-tailed Hawk prey species, Yensen and others (1992) reported that in the Snake River Birds of Prey Area, southwestern Idaho, fire may reduce populations of Townsend's ground squirrels (*Spermophilus townsendii*).

FIRE USE

Prescribed fire can be beneficial to Red-tailed Hawk populations by enhancing habitat and increasing the prey base (Dodd 1988, Landers 1987). Prescribed burning plans should strive for creation of maximum interspersion of openings and edge, with high vegetative diversity. Habitats should be maintained in a random mosaic. In most cases, burning plans must be integrated with proper range management. Reseeding of perennial grasses as well as rest from livestock grazing may be necessary to achieve desired goals. Burning should be deferred until nesting is completed in areas where impact to breeding red-tailed hawks may occur (Dodd 1988). After logging, Benson (1979) suggested broadcast burning rather than piling slash to reduce high temperature fires which may be destructive to soil organisms and small mammals. For more information regarding the use of prescribed fire in specific habitats for the benefit of raptors, see Dodd (1988).

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Falco sparverius
American Kestrel

DIRECT FIRE EFFECTS ON ANIMALS

Direct mortality in raptors due to fire is rare (Lehman and Allendorf 1989). Adults can probably easily escape fire. However, fire could directly reduce American Kestrel populations if the fire destroys occupied nest trees. American Kestrels have been reported to be attracted to fire and smoke in search of prey (Komarek 1969, Stoddard 1963). They have been observed dashing close to flames, sometimes landing on stumps or fallen branches in thick smoke (Sedgwick and Knopf 1986, Smallwood *et al.* 1982). Low-severity fires probably have little effect on American Kestrels. Landers (1987) commented that light winter burning probably does no substantial harm to raptors.

HABITAT RELATED FIRE EFFECTS

American Kestrels occur in the following 10 major fire-dependent plant associations in the western United States: grasslands, semidesert shrub-grasslands, sagebrush (*Artemisia* spp.)-grasslands, chaparral, pinyon-juniper (*Pinus* spp.-*Juniperus* spp.) woodland, ponderosa pine, Douglas-fir, spruce-fir (*Picea* spp.-*Abies* spp.), redwood (*Sequoia sempervirens*), and giant sequoia (*Sequoiadendron giganteum*) forests (Lehman and Allendorf 1989). American Kestrels occur in fire-dependent longleaf pine communities in the eastern United States (Hoffman and Collopy 1988, Hoffman and Collopy 1987).

Although fire may reduce potential nest trees, it may also create snags for nest and perch sites and enhance the foraging habitat of American Kestrels. In the Sierra Nevada, nesting American Kestrels were two to three times more numerous in a burned-over forest than in an unburned forest nearby. This difference was attributed to the greater availability of nest cavities in the burned forest (Balgooyen 1976). At Sagehen Creek, California, American Kestrels breed (but do not winter) in burned forests and along edges between sagebrush and forest habitats. American Kestrels do not use areas of thick cover because they require an open understorey in which to manoeuvre and visually locate prey. American Kestrels often use fresh burns when foraging due to increased prey visibility (Dodd 1988, Landers 1987, Smallwood *et al.* 1982). A decrease in the frequency of ground fires leads to an increase in vegetative cover and, therefore, has a negative impact on habitat quality for American Kestrels (Balgooyen 1976, Hoffman and Collopy 1988). In the Sierra Nevada, Balgooyen

(1976) found that open areas created by a severe fire in ponderosa pine and lodgepole pine (*Pinus contorta*)-red fir (*Abies magnifica*) forests provided only temporary habitat for American Kestrels. Eleven to twelve years after the fire, brush vegetation including deerbrush (*Ceanothus integerrimus*) and snowbrush ceanothus (*C. velutinus*) formed dense cover in the burned areas (Balgooyen 1976).

American Kestrels are favoured by fires that open up or clear pinyon-juniper woodlands (Mason 1981). Raptors associated with pinyon-juniper woodlands depend upon edges of openings created by fire and scattered islands of unburned woodlands (Dodd 1988). In pinyon-juniper woodlands on the Humboldt National Forest, California, American Kestrels were observed only on burned areas and only during the second season. Surveys were conducted in only two seasons (Mason 1981).

American Kestrels congregate at both controlled and naturally occurring fires to hunt along the edge (usually the windward side) for insects, small mammals, and reptiles (Palmer 1988, Smallwood *et al.* 1982, Sprunt 1954). Howell (1932) reported seeing 13 southeastern American Kestrels feeding over a "raging" marsh fire. During a January fire in scrublands near Immokalee, Florida, 15 American Kestrels were observed hunting along the approximate 150 m windward edge of the fire. The linear concentration (1 bird/10 m) was a hundredfold greater than that on utility lines in the area that same winter. American Kestrels preyed exclusively on insects which flew away from the fire into the wind (Smallwood *et al.* 1982).

FIRE USE

Prescribed fire can be beneficial to American Kestrel populations by enhancing habitat and increasing the prey base (Dodd 1988, Landers 1987, Lehman and Allendorf 1989). In the sandhills communities of Florida, fire suppression has caused some sites to have dense understories, particularly of fire-intolerant rosemary (*Ceratiola ericoides*). Such sites may be unsuitable for southeastern American Kestrels and a program of prescribed burning in these habitats is recommended (Hoffman and Collopy 1987). Several studies indicate that many prey populations increase rapidly subsequent to burning in response to increased food availability (Dodd 1988, Landers 1987). Fire suppression in grasslands was detrimental to small bird and mammal populations due to organic matter accumulation and reduced plant vigour (Wagle 1981).

Prescribed burning plans should strive for creation of maximum interspersed openings and edge, with high vegetative diversity. Habitats should be maintained in a random mosaic of open areas and standing trees and snags should be conserved. In most cases, burning plans must be integrated with proper range management. Reseeding of perennial grasses as well as a period of rest from livestock grazing may be necessary to achieve desired goals. Burning should be deferred until nesting is completed in areas where impact to breeding American Kestrels may occur. After logging, Baker (1974) suggested broadcast burning rather than piling slash to reduce

high temperature fires which may be destructive to soil organisms and small mammals. For more information regarding the use of prescribed fire in specific habitats for the benefit of raptors, see Dodd (1988).

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Haliaeetus leucocephalus
Bald Eagle

DIRECT FIRE EFFECTS ON ANIMALS

Bald Eagles have continued nesting during wildfire and returned to the nest the following year (Murphy 1991).

HABITAT RELATED FIRE EFFECTS

Because forest structure (density and height class) determines avian community composition, changes in forest structure lead to changes in avian communities (Diem and Zeveloff 1980, Smith 1980). A stand-replacing fire will, therefore, likely change Bald Eagle use of a forest. Fires that destroy old-growth forest can reduce eagle populations (Yellowstone National Park 1991). If low-intensity, litter-reducing fires are not allowed to burn in old-growth forests, stand-replacing, high-intensity crown fires can result (Covington and Moore 1992).

Fires create snags, which are important perching and nesting sites for Bald Eagles. Snags can possibly increase potential for lightning-caused fire when standing, and when fallen, they increase fuel loading (Lyon 1977). These increased potentials may be hazardous in areas where fire control for maintaining Bald Eagle populations is

necessary. There have been no studies to determine if the hazards of snags outweigh their benefits to eagles. Snag attrition rates have been listed for lodgepole pine forests following fire (Lyon 1977). Old-growth eastern white pine (*Pinus strobus*) forests in Ontario continually recruit snags in the absence of fire because of their uneven-aged structure (Quinby 1991). Catastrophic fires in mature and old-growth forests can create even-aged conditions which may stop continuous snag recruitment (Quinby 1991).

FIRE USE

Fire can be used to reduce litter build-up, control disease, remove less vigorous species, and allow more vigorous trees to reach maturity, thus providing old-growth habitat for Bald Eagles (Harrington and Sackett 1992).

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Molothrus ater
Brown-headed Cowbird

DIRECT FIRE EFFECTS ON ANIMALS

There were no references in the literature concerning direct or indirect mortality of Brown-headed Cowbirds due to fire. Adults could easily escape fire. Nests and young are vulnerable to fire but unlikely to be exposed to fire because in most areas fire seasons are not coincident with early nesting seasons. Literature references are mostly on the topic of bird use of recently burned versus unburned plots. Brown-headed Cowbird use of burns is likely to be related to food availability and availability of perches. It is also likely to be related to the presence of host species nests.

In Connecticut a 1982 census revealed that Brown-headed Cowbirds were more abundant on old fields that had been burned in spring of 1981 than on unburned fields. Vegetation in these plots consisted of shrubs and saplings (mostly red maple [*Acer rubrum*]) (Euler and Thompson 1978).

In Kansas a survey of birds was made on successional (i.e., undisturbed) plots dominated by shrubs and prairie plots dominated by grasses that had been mowed and burned. Brown-headed Cowbirds occurred only on the shrub plots; none were observed on the mowed and burned plots. There was an extreme drop in the numbers of other bird species on the mowed and burned plots as well (Cink and Lowther 1989). On the Curtis Prairie, Wisconsin, there was little difference between Brown-headed Cowbird recapture rates on unburned plots and on recently burned plots (Loiselle and Blake 1984).

In South Dakota a prescribed fire was conducted in mixed-grass prairie in May 1983. A bird census was conducted on the burned area and on an adjacent unburned area in June and July of the same year. Brown-headed Cowbirds were observed on both burned and unburned areas with no significant differences in numbers of observations (Huber and Steuter 1984).

In Arizona an October 1973 census of nonbreeding birds in ponderosa pine (*Pinus ponderosa*) stands showed Brown-headed Cowbirds with a slightly higher prominence value (product of number of individuals and frequency of occurrence) on unburned plots than on plots that had been burned by wildfire in May 1972 (Blake 1982).

HABITAT RELATED FIRE EFFECTS

Effects of Fire Exclusion: Fire exclusion in mixed-grass prairie has allowed shrubs to invade grasslands. In North Dakota Brown-headed Cowbird density was higher on shrubby plots than on shrubless plots (Arnold and Higgins 1986).

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Pandion haliaetus
Osprey

DIRECT FIRE EFFECTS ON ANIMALS

Ospreys are probably able to escape fire, however, nests may be destroyed.

HABITAT RELATED FIRE EFFECTS

Information was not found in the literature on habitat-related fire effects of the Osprey; however, fires will presumably create and destroy snags used by Ospreys. Additionally, the short-term effects of a riparian fire may affect the Osprey's food supply. Removal of streamside vegetation increases the risk of streambank erosion, reduces available habitat and raises stream temperatures, all of which could potentially reduce fish populations in the stream. However, the long-term effect of fire on fish populations could be beneficial. The thinning and removal of conifers along streams by fire and stimulation of deciduous vegetation promotes cover, provides shading, and allows development of terrestrial insects important in the diet of fish (Wright and Bailey 1982).

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Phasianus colchicus
Ring-necked Pheasant

DIRECT FIRE EFFECTS ON ANIMALS

Fires in Ring-necked Pheasant habitat during the nesting season can destroy Ring-necked Pheasant nests, eggs, and broods (Erwin and Stasiak 1979). During an April 17th wildfire in mixed growth of herbaceous and woody plants in Castalia, Ohio, several Ring-necked Pheasant nests and eggs were destroyed (Leedy 1950). In Nebraska, Erwin and Stasiak (1979) reported that 38 Ring-necked Pheasant nests containing a total of 336 eggs were destroyed in a native grassland by an early spring prescribed fire. The authors did not mention if any nests survived the fire. Adult Ring-necked Pheasants reacted to the fire by flying from the grass only after the fire had approached to within a few meters. Several times Ring-necked Pheasants were observed flying through flames that at times reached 5-7 m in length. An adult Ring-necked Pheasant hen flew through the flames and landed in the smoking ashes just behind the fire line. She did not appear to be injured (Erwin and Stasiak 1979).

HABITAT RELATED FIRE EFFECTS

Ring-necked Pheasants may be both positively and negatively affected by fire occurring in their habitat. The effect of fire on Ring-necked Pheasants during the first post-fire year is generally negative if valuable cover is destroyed and alternate cover is not available (George *et al.* 1978, Higgins *et al.* 1989, McAttee 1945). However, fire is an important factor in creating and maintaining Ring-necked Pheasant habitat and often has a positive effect over the long term (George *et al.* 1978, Higgins *et al.* 1989, Hughes 1985).

Short-term effects - Scattered patches of tall, dense (undisturbed) cover are important for Ring-necked Pheasant survival (Westemieier 1984). Spring burning of grain stubble, irrigation ditchbanks, barrow pits, railroad rights-of-way, and agriculturally idle areas removes important nesting cover for that year. Removal of herbaceous cover in abandoned fields or along fencerows destroys crowing territories, nesting sites, and general escape cover for Ring-necked Pheasants both in summer and winter (McAttee 1945). Fall burning of grain stubble destroys protective foraging cover (Weigand and Janson 1976).

Marsh vegetation may provide the only early nesting cover in some areas. Burning of marshes just prior to the nesting season is especially harmful to Ring-necked Pheasants (McAttee 1945).

Remnant tracts of sagebrush in southern Idaho provide important winter loafing and escape cover for Ring-necked Pheasants. Fire may cause sagebrush to revert to grasslands that do not provide Ring-necked Pheasant winter habitat, and it may take 15 to 20 years for a useful shrub component to reestablish in a dryland setting (Leptich 1992).

Long-term effects - Although Ring-necked Pheasants may be negatively affected by fire the first post-fire year, periodic disturbance of stands is desirable and often essential to maintain Ring-necked Pheasant habitat. To provide residual grass-forb cover, suitable areas of cover must be established and then the stand must be "rejuvenated" every few years by fire or other disturbance (Frank 1984).

On the Rathbun Wildlife Area in south-central Iowa, there was a marked decline in available nesting cover for Ring-necked Pheasants immediately following a spring prescribed fire. The quality of nesting cover then gradually improved in subsequent years until it reached or exceeded pre-fire and control levels. In addition to altering species composition and improving plant vigour, prescribed burning removed accumulated litter resulting in an overall improvement of nesting cover. Following a July prescribed fire, brood and renesting cover were reduced. Brood cover seemed to show rapid improvement due to the rapid growth of canopy-forming forbs such as common ragweed (*Ambrosia artemisiifolia*) and goldenrod (*Solidago* spp.). Nesting cover improved gradually in ensuing years (George *et al.* 1978).

FIRE USE

Prescribed burning can be used to maintain Ring-necked Pheasant habitat (Baxter 1984, Westmeier 1984). In some areas, prescribed burning to improve sharp-tailed grouse (*Tympanuchus phasianellus*) habitat has also improved habitat for Ring-necked Pheasants (Bendell 1974, Vogl 1967). Cover must always be available for nesting, brood-rearing, loafing, and roosting. If Ring-necked Pheasant habitat is burned, alternate cover must be available. Westmeier (1984) suggests the following 3-year management rotation for Ring-necked Pheasant habitat in Illinois:

- first year - no disturbance
- second year - March fire (possibly followed by July haying or light grazing)
- third year - light grazing, rotary mowing, or seed harvesting (by combine) in late summer or fall.

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Sialia curruoides
Mountain Bluebird

DIRECT FIRE EFFECTS ON ANIMALS

There are no reports of Mountain Bluebird mortality due to fire. Nests and nestlings are probably vulnerable to severe fire. Nesting cavities may be lost in severe fires including crown fires.

HABITAT RELATED FIRE EFFECTS

Mountain Bluebirds are strongly attracted to early post-fire (1-2 years) communities, particularly those with many standing dead trees. Woodpecker activity in these communities provides many nest cavities (Hutto 1995). Fire initiates a tree-hole nesting cycle in Grand Teton and Yellowstone National Parks. Woodpeckers concentrate in newly burned areas to feed on insects (particularly bark beetles) and to nest in standing dead trees (Taylor 1979), especially those with decay prior to fire

(Saab and Dudley 1995). Mountain Bluebirds and other secondary cavity nesters use the holes after the woodpeckers abandon them. In northern Wyoming, northern three-toed woodpeckers (*Picoides tridactylus*) and black-backed three-toed woodpeckers (*P. arcticus*) begin leaving burned areas about 4 to 5 years after the fire. Hairy woodpeckers (*P. villosus*) continue to excavate holes for years, until snags finally decay to the point of falling down. At around 50 post-fire years, tree-hole nesting species usually leave the area even if a few snags are still present (Taylor 1979).

A 1982 study in the Little Firehole River watershed within Yellowstone National Park indicated that Mountain Bluebirds were only common in a 90 ha area that burned in 1949. The authors speculated that Mountain Bluebird numbers were high in the late 1700s and early 1800s because up to 50 percent of the area was covered by early post-fire stands; as forests matured into the twentieth century Mountain Bluebird numbers probably decreased (Romme and Knight 1982). In other areas of Yellowstone National Park, Mountain Bluebirds were observed in burned areas in the first post-fire year (Gniadek 1977). Mountain Bluebirds were associated with post-fire plots in severely burned lodgepole pine stands in Yellowstone National Park. By the second post-fire year Mountain Bluebirds were nesting in holes excavated by northern three-toed woodpeckers. Peak Mountain Bluebird breeding density occurred on 12-post-fire year plots with a small peak on 30-year-old plots. There were no Mountain Bluebirds on 57-year-old and older plots (Taylor 1979). In Grand Teton National Park, Mountain Bluebirds were present in the first post-fire year on severely burned plots, increasing to 15 pairs per 40 ha in the second year. They were not observed on moderately burned or unburned plots, or on 43-year-old burned plots (Taylor 1979).

In Alberta fire-originated forests dominated by quaking aspen, balsam poplar (*Populus balsamea*), and lodgepole pine, Mountain Bluebirds were present in 14-year-old stands but not present in stands that had been clearcut, nor in 30-, 60-, or 80-year-old stands (Westworth and Telfer 1993). In Montana Mountain Bluebirds increased from post-fire year 2 to post-fire year 4 in both burned, and burned and logged areas (Harris 1982).

In California Mountain Bluebirds were observed in large numbers (15.2 pairs per 40 ha) in post-fire years 5, 6, and 7 in mixed coniferous forest. Woodpecker activity was decreasing by the time of the study (Bergeron and Jodoin 1989). In the Sierra Nevada a severe fire occurred in a Jeffrey pine (*Pinus jeffreyi*)-white fir community in 1960. Birds were censused in 1968 and 1975. Mountain Bluebird densities were highest on burned plots with standing dead trees and openings in 1968. They were still present on burned plots in 1975; these plots had developed dense brushfields (Bock *et al.* 1978).

Later post-fire communities may not be as attractive as early successional stages; Raphael and others (1987) noted a decreasing trend for Mountain Bluebirds in burned habitat in the Sierra Nevada between 1966 and 1985. In Yellowstone

National Park Mountain Bluebirds densities 5 to 25 years after fire depend on the number of standing snags. At 25 to 30 post-fire years Mountain Bluebird numbers may begin to decline depending on the rate of snag loss. From 30 to 50 post-fire years Mountain Bluebirds depart as canopy closure develops (Taylor and Barmore 1980).

Based on data from Brawn and Balda (1988), Hejl (1994) hypothesized a Mountain Bluebird population decrease in southwestern ponderosa pine due to fire exclusion and consequent closure of forests.

FIRE USE

Savannas and open stands of mature pine are natural Mountain Bluebird habitats that require recurrent fire for maintenance. Prescribed fire is usually beneficial to Mountain Bluebirds, especially if it controls shrubs and understorey trees (Pinkowski 1979). Mountain Bluebird densities 5 to 25 post-fire years depend on the number of standing snags with nest cavities (Taylor and Barmore 1980). In Wyoming, mountain big sagebrush (*Artemisia tridentata* var. *vaseyana*) communities, bordered by Douglas-fir and quaking aspen stands, were burned with both spring and fall prescribed fires (not on the same plots). Mountain Bluebirds were present as nonbreeders at higher densities on the burned plots than on unburned plots (both treatments) (McGee 1976). In Nevada pinyon-juniper woodlands burned with prescribed fire for 3 consecutive years, Mountain Bluebird populations were the same on burned and unburned plots in the first post-fire year, but were higher on burned plots in the second year. Nest cavities were lost to the fire the first year, but by the second season cavity numbers were increasing (Mason 1981).

Some bird species are eliminated from burned areas, but other species are attracted to them, creating a similar amount of species diversity (Despain 1978). Prescribed fire in Wind Cave National Park, South Dakota, temporarily improved foraging habitat for Mountain Bluebirds by reducing surface cover. There was little change in overall tree and snag density on these sites (Bone and Klukas 1990). Burned areas less than 5 years old, particularly those that had severely burned, are highly valuable for many cavity nesting species including Mountain Bluebirds (Taylor 1979). For enhancement of wildlife habitat it is strongly recommended that severely burned areas be allowed to develop naturally rather than be subject to salvage logging. When salvage is considered essential (for sanitary reasons or as a firebreak), it is recommended that a substantial area be left uncut rather than thinning the entire burn (Hutto 1995).

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Sialia mexicana
Western Bluebird

DIRECT FIRE EFFECTS ON ANIMALS

There are no reports of direct Western Bluebird mortality due to fire; nestlings and nests are probably vulnerable, however. Fast-moving, intense chaparral fires have the potential to kill birds and/or create extreme stresses (panic, dehydration) (Nichols and Menke 1984).

HABITAT RELATED FIRE EFFECTS

Post-fire successional communities are usually attractive to Western Bluebirds, especially in the first few years following the fire (Saab and Dudley 1995). In Kern County, California, Western Bluebirds were recorded prior to a July 1954 prescribed fire in scrub woodlands dominated by blue oak (*Quercus douglasii*), gray pine (*Pinus sabiniana*), interior live oak (*Q. wislizenii*), and wedgeleaf ceanothus (*Ceanothus*

cuneatus). By 3 years after the fire, Western Bluebird densities had nearly doubled (Lawrence 1966). In southwestern Idaho a ponderosa pine-Douglas-fir forest that burned in 1992 was censused in 1994. Most of the trees on the plot were dead. Cavity nesters in the area included Western Bluebirds (17 nests on 3163 ha) (Saab and Dudley 1995). Based on data from Brawn and Balda (1988), Hejl (1994) hypothesized a Western Bluebird population decrease for southwestern ponderosa pine forests since fire exclusion has led to increased amounts of closed forest.

Snag availability in post-fire communities is a substantial positive influence on Western Bluebird activity (Schreiber and DeCalesta 1992). On the Olympic Peninsula Western Bluebirds were detected on 30- to 50-year-old Douglas-fir plantations in dense patches of fire-killed snags; they were not detected on plots without snags nor in younger and older plots with snags (Zarnowitz 1982). Hutto (1995) emphasizes the importance of snags and strongly recommends against salvage cutting of burned forests; the abundance of newly killed trees and concomitant woodpecker activity is particularly valuable to Western Bluebirds for nest cavities. If salvage logging is unavoidable (for sanitary or firebreak reasons) then some areas should be left untouched rather than thinning the entire unit (Hutto 1995).

Loss of vegetative cover due to fire is detrimental to most small birds, leaving them vulnerable to raptors, especially in large, severe fires which leave few unburned refuges. In San Dimas, California, Western Bluebirds initially declined in immediate post-fire chaparral communities, probably due to lack of fruit (Wirtz 1982).

FIRE USE

Savannas and open stands of mature pine are natural Western Bluebird habitat that require recurrent fire for maintenance. Prescribed fire is usually beneficial to Western Bluebird especially if it reduces shrubs and understory trees (Pinkowski 1979).

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Zenaida macroura
Mourning Dove

DIRECT FIRE EFFECTS ON ANIMALS

Literature addressing the direct effects of fire on Mourning Doves is lacking; however, fire can destroy Mourning Dove nests. Adult Mourning Doves are probably able to escape fire.

HABITAT RELATED FIRE EFFECTS

Fires may affect Mourning Dove nesting habitat by destroying nest trees and therefore increasing the occurrence of ground nesting. In Texas, a 2-year study of Mourning Dove nesting on a grassland infested with woody vegetation showed that a low-severity fire had little effect on either mesquite trees or their use as nesting sites by Mourning Doves.

However, on a similar area earlier treated with herbicides and burned in late March, the loss of the larger mesquite trees as nest sites was followed by the occurrence of more ground nesting (Soutiere and Bolen 1973).

Soutiere and Bolen (1973) found that current year burns provided better ground-nesting habitat than did older burns except under drought conditions. The highest densities of ground nesting pairs were found in the current year's burn and decreased each successive year thereafter. The degree of ground cover became less attractive to ground-nesting doves as the proportion of cover approached the unburned condition. Also, burning reduced the amount of available litter but added to the suitability of the area by increasing the amount of open space where doves might collect nest materials.

The effects of fire in a drought year could be disastrous to Mourning Dove nesting. Spring fires in a drought year may delay the development of suitable ground-nesting habitat (Soutiere and Bolen 1973).

FIRE USE

Mourning Doves generally will not scratch in litter for seeds and will avoid areas with dense vegetation when feeding (Landers 1987). For these reasons Mourning Doves commonly forage on newly burned areas. Mason (1981) found that Mourning Doves often foraged in 2-year-old burns on a (*Pinus monophylla-Juniperus osteosperma*) woodland site burned in winter or fall. The burns provided weedy areas for foraging, snags for perching, and open areas for loafing.

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Mammals

Canis lupus
Gray Wolf

DIRECT FIRE EFFECTS ON ANIMALS

No direct fire effects on Gray Wolves have been noted.

HABITAT RELATED FIRE EFFECTS

The effect of fire on Gray Wolf habitat is best defined by how fire affects Gray Wolves' prey. Beaver, Elk, Moose, and deer are fire-dependent species, requiring the plant communities that persist following frequent fires (Hansen *et al.* 1973, Kramp *et al.* 1983). Edwards (1985) reported that after fire, Moose populated the area around Wells Gray Park, British Columbia, where they were previously unknown.

This was followed by a marked increase in Gray Wolves. Other studies in Alaska, Michigan, Minnesota, and Canada show an increase in Moose populations following fire (Hansen *et al.* 1973, Heinselman 1973, Viereck and Schandelmeier 1980).

Now absent from the old-growth forests of Minnesota, caribou once were an important prey for Gray Wolves here. These forests do not provide enough food to sustain other ungulates for Gray Wolves to prey on. Due to fire exclusion, these old-growth forested areas have increased, checking ungulate populations and consequently limiting Gray Wolf populations (Heinselman 1973).

FIRE USE

Fire can be used to create browse for ungulates which, in turn, provides prey for Gray Wolves. In Minnesota Heinselman (1973) concluded that enough early post-fire plant communities must exist within a Gray Wolf pack's territory to support a surplus of deer, Moose, and Beaver for prey. Adequate hiding cover should be maintained for the ungulates. If they are abundant then Gray Wolf populations have a better chance of thriving.

Gray Wolves prosper best when they have a large area, relatively free from human disturbance, in which to roam, and when there is a surplus of ungulates (Herman and Willard 1978). Frequent fires that promote ungulate browse in and around areas that are at least moderately remote offer ideal Gray Wolf habitat.

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Castor canadensis
Beaver

DIRECT FIRE EFFECTS ON ANIMALS

Information on the direct fire effects on Beavers was not found in the literature; however, they can probably easily escape fire. Since lodges are typically built over water, they are probably at little risk of being destroyed by fire.

HABITAT RELATED FIRE EFFECTS

Fire occurring in riparian areas often benefits Beaver populations (Kelleyhouse 1979). Beavers are adapted to the early stages of forest succession. Quaking aspen, willows, alders, and red-osier dogwood, prime Beaver food trees, all sprout vigorously after fire. As succession progresses, these trees become too large for Beavers to use or are replaced by climax trees (Wright and Bailey 1982). Recurring fires within parts of boreal forests have allowed aspen and willow to replace

coniferous forests. This change favours Beaver populations, since both species are important food sources. Fire may also help create more open bodies of water (Kelleyhouse 1979).

FIRE USE

Fire can be used to maintain Beaver habitat in a subclimax state, thus ensuring adequate food supply for Beavers (Kelleyhouse 1979, McCune 1982, Wright and Bailey 1982). High Beaver populations in many areas are the direct result of the extensive clearcutting and forest fires which were characteristic of the northern forests until recent years (Patric and Webb 1953, Wright and Bailey 1982).

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Martes americana Marten

DIRECT FIRE EFFECTS ON ANIMALS

Martens could be trapped in a rapidly spreading, intense fire. However, direct fire mortality probably has little effect on the population as a whole.

HABITAT RELATED FIRE EFFECTS

Fires that completely consume the understorey and/or reduce the canopy closure to less than 30 percent are detrimental to Marten populations in the short-term. However, fires that create a mosaic of diverse habitats provide the best cover for Marten and their prey in the long-term.

Marten populations tend to increase several decades after fire as adequate food and cover are replaced (Koehler and Hornocker 1977, Koehler *et al.* 1975, Viereck and

Schandelmeier 1980). Koehler and others (1975) suggested that low-intensity fires on mesic sites where canopy cover is maintained at greater than 30 percent may not adversely affect Marten habitat.

FIRE USE

Prescribed fire can be used to create a diversity of forest communities, which, over time, may support more Marten. Fire can create and maintain openings where abundant fruits, insects, Ground Squirrels, and voles provide summer food. Smaller burns may be more beneficial because they are less restrictive to Marten movement in winter, and cover is reestablished more quickly. Islands of climax forest containing logs, stumps, and slash for denning should always be maintained. Islands of forest should be no farther apart than 50 m (Koehler and Hornocker 1977, Koehler *et al.* 1975).

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Mustela vison Mink

DIRECT FIRE EFFECTS ON ANIMALS

There are no reports of direct mortality of Mink due to fire. Because Mink are highly mobile, semi-aquatic animals and often den underground, it seems unlikely that fire-caused mortality is ever substantial.

HABITAT RELATED FIRE EFFECTS

There are no reports in the literature linking fire-caused habitat changes to Mink. Fire along streambanks that reduces cover and downed logs would have a negative impact on mink activity. Reduction of fish and crayfish due to changes in stream conditions would adversely affect Mink. Conversely, fire that resulted in increased

snag numbers and stream channel downfalls, shrub density, and herbaceous vegetation cover would probably encourage Mink activity.

Odocoileus hemionus

Black-tailed Deer

DIRECT FIRE EFFECTS ON ANIMALS

Although uncommon, Black-tailed Deer can be trapped and killed by fast-moving fires (McTaggart-Cowan 1956, Hines 1973).

HABITAT RELATED FIRE EFFECTS

The effects of fire on Black-tailed Deer habitat are widely varied and well documented in the literature. In general, fires that create mosaics of forage and cover are beneficial. Deer seem to prefer foraging in burned compared to unburned areas, although preference may vary seasonally (Biswell 1989, Davis 1976, Davis 1977, Johnson 1989, Keay 1977, Keay and Peek 1980, Klinger *et al.* 1989, Willms *et al.* 1980). This preference may indicate an increase in plant nutrients which usually occurs following fire (Asherin 1973, Hobbs and Spowart 1984, Severson 1987). Hobbs and Spowart (1984) warned about making conclusions regarding the benefits of fire based on forage studies alone. Their study of fire on nutrition in Colorado revealed increases in the quality of deer diets due to changes in forage selection--not increases in nutrients of previously selected forage.

Burning in grassland communities reduces litter that otherwise inhibits new growth of grasses. Fire rejuvenates and improves these grasslands, which are important winter range in some areas (Johnson 1989, Willms *et al.* 1980). Burning sagebrush communities can result in significant increases of herbaceous plants by reducing decadent sagebrush that outcompetes more nutritious and palatable species (Smith 1985, USDA Forest Service 1973). However, in areas where sagebrush is the only cover, its complete removal can be detrimental to Black-tailed Deer populations (USDA Forest Service 1973).

Antelope bitterbrush is a highly preferred browse species on some Black-tailed Deer winter ranges and is sensitive to burning (Gruell 1986, Wagstaff 1980). Burned bitterbrush takes longer to recover than bitterbrush disturbed by other means (Wagstaff 1980). Burned bitterbrush grows slower, is less dense, and plants are smaller than unburned specimens. Bitterbrush responds variably to fire intensity, temperature, and season (Gruell 1986). Late summer fires in Idaho killed two-thirds of the bitterbrush, while a moderate-intensity spring fire in Montana killed one-third. A summer fire of moderate intensity in Oregon destroyed the entire stand of bitterbrush (Gruell 1986).

Shrubs and forbs in pinyon (*Pinus* spp.)-juniper (*Juniperus* spp.) communities tend to increase the first few years following fire, providing valuable browse (Blackburn *et al.* 1975, McCulloch 1969). Black-tailed Deer seem to use these areas more after 15 years (McCulloch 1969, Stager and Klebenow 1987). Stager and Klebenow (1987) reported that the beneficial effects of fire for Black-tailed Deer in pinyon-juniper stands can last as long as 115 years. However, Bunting (1987) concluded that burning of these stands becomes increasingly difficult as stands grow older because fine fuels in the understory are reduced. He stated that burning should take place at early successional stages and at intervals based on the fire tolerance of desirable forage species. Everett (1987) warned that pre-burn conditions in pinyon-juniper stands will most likely determine the post-fire plant composition. If perennial shrubs are present before a burn, they will come back following fire. If no shrubs are present, perennial grasses will develop (Blackburn *et al.* 1975).

FIRE USE

Fire can be used to stimulate browse, create openings in dense, inaccessible plant communities, and reduce slash, as well as increase nutrient content and palatability of forage (Dasmann and Dasmann 1963, Gruell 1986, Nelson 1976). Gruell (1986) listed several factors that influence post-fire plant composition, including the severity, size, and season of the burn, fuel type, post-burn foraging intensity, and the pre-burn plant community composition. He stated that surface fires of moderate intensity following thinning or selection cuts can improve Douglas-fir or ponderosa pine (*Pinus ponderosa*) forests for Black-tailed Deer by promoting regeneration of crown-sprouting shrubs and preparing the seedbed for herbs and shrubs. A mosaic of seral stages is best for Black-tailed Deer (Gruell 1986).

In areas where chaparral adjoins oak woodlands, prescribed burns can create access through the chaparral to the understory forage of the oak woodlands (Klinger *et al.* 1989). Biswell (1989) recommended burning chaparral every 30 years to create a mosaic of young stands. Late summer or early fall burning promotes the highest seed crop for most species in these plant communities. Wallmo and others (1981) listed several recommendations for burning chaparral communities to improve Black-tailed Deer habitat.

Fire can control pinyon-juniper woodlands by maintaining them in a subclimax state (Blackburn *et al.* 1975). Small burns are more beneficial than large burns to Black-tailed Deer because they tend to use burned areas close to cover. The optimum width for burns in these communities may be less than 0.4 km (Blackburn *et al.* 1975). To maintain forage in bunchgrass communities, burning at 4- to 6-year intervals in winter or early spring is recommended (Johnson 1989).

Burning can control sagebrush in areas where it has dominated grasslands and reduced deer forage (USDA Forest Service 1973). Where Gambel oak grows thick and impenetrable, fire can open stands and provide valuable winter range for Black-tailed Deer (Kunzler and Harper 1980). Kufeld (1983) recommended burning Gambel oak in autumn during or immediately following leaf fall and building fire

breaks 8 m wide around the areas to be burned. Because Gambel oak recovers quickly following fire, particularly at low elevations where Black-tailed Deer winter, its growth must be monitored and retarded to improve Black-tailed Deer habitat (Kunzler and Harper 1980).

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Peromyscus maniculatus
Deer Mouse

DIRECT FIRE EFFECTS ON ANIMALS

Causes of direct mortality due to fire include burns, heat stress, asphyxiation, physiological stress, trampling by other animals, and predation. Indirect causes include loss of food supply, loss of nest sites, predation, increased parasitism and disease, increased competition, and changes in social interaction. Small mammals such as the Deer Mouse often survive fire by moving into underground burrows or by moving to unburned areas (Forde 1983). Mortality within burrows is difficult to assess but hypothesized to be low (Kaufman *et al.* 1990, Kaufman *et al.* 1988). Wirtz

(1982) reported that Deer Mice survived chaparral fires in burrows. No dead animals were found after prescribed fire in mixed-grass prairie, a community inhabited by Deer Mice (Springer 1988a). There are a few reports of direct mortality of Deer Mice from fire. Chew and others (1959) found two carcasses of *Peromyscus* species in 0.7 ha transect after a chaparral wildfire in an area supporting both Deer Mice and California Mice (*P. californicus*).

Attempts to radiotrack Deer Mice during a prescribed fire were largely unsuccessful; one female burrowed under a 20-cm diameter log that was scorched by the fire but did not burn. The mouse survived the fire (Springer 1988b). In west-central Oregon Douglas-fir stands, Gashwiler (1959) observed Deer Mice on clearcut and slash-burned (October) areas while fires were still active; some were captured within 0.6 m of a smoldering fire. He reported that 12 of 16 (75% of) mice marked prior to the fire were recaptured on the burned area within 15 days of fire initiation. In November, 13 of the 16 original marked animals were recaptured on the burned area. The total number of Deer Mice captured on the burn (Chew *et al.* 1959) was three times the number of Deer Mice captured on the adjacent unburned control plot (Gashwiler 1959).

In some instances, Deer Mice leave the burn area immediately after a fire, possibly due to the presence of loose ash or to a lack of food. Tevis (1956) reported that one-third of the Deer Mice marked before a broadcast (slash) fire were recaptured in the post-fire period; all but four were captured on the edge of the burn but none were recaptured on the burned area. Colonizers did not enter the burned area until rainfall packed down the deep ash layer. By 2.5 weeks after the fire, Deer Mouse numbers were twice the pre-fire level (Tevis 1956). Four Deer Mice marked prior to a prescribed fire in oak savanna were not caught again after the fire; the cause for their absence was unknown (possibilities include fire mortality, predation, death by other causes, and emigration) (Tester 1965).

HABITAT RELATED FIRE EFFECTS

In many communities Deer Mouse abundance was higher on burned areas than on adjacent unburned areas by the first growing season after fire. In other communities there was no clear response, and in some communities Deer Mice decreased after fire. Deer Mice are often the first animals to invade an area that has been burned (Baker 1968, Forde 1983, Mason 1977). Burned areas often support increased numbers of insects and seeds of annual plants which are beneficial to Deer Mice (Hooven 1973). In many reports Deer Mouse abundance was negatively correlated with amount of litter (Halvorsen and Anderson 1983). Fire in grassland immediately reduces litter and aboveground vegetation; total biomass usually is higher than pre-fire levels by the summer following a spring prescribed fire (Schramm and Willcutts 1983). Deer Mice in grasslands tend to use burned plots more than adjacent unburned plots (Peterson *et al.* 1985, Schramm and Willcutts 1983). In Minnesota tallgrass prairie, prairie Deer Mouse populations were negatively associated with litter depth; large beetles (a favoured food of Deer Mice) were associated with sparse litter

(Tester and Marshall 1961). Fire in ecotones may increase available habitat for prairie Deer Mice. In Wisconsin Deer Mice were only found on frequently burned areas where woodland had been successfully converted to brush-prairie (Beck and Vogl 1972).

The success of the Deer Mouse on burned areas is attributed to its nocturnal habits, erratic movements, tolerance of open space/bare ground, and lack of competition (Ream 1981). In California the ratio of Deer Mice to California Mice decreases with succession from grassland created by prescribed fire to mature chaparral (Bell and Studinski 1972). In Yellowstone National Park, Deer Mice were able to find adequate food the first growing season after wildfire, even though plant cover was less than 10 percent (Despain 1978). In Kansas tallgrass prairie Deer Mice selected recently burned areas over areas that had burned in previous years. These areas were characterized by a large proportion of exposed soil, lush vegetation, and little or no plant litter (Kaufman *et al.* 1988). In Arizona ponderosa pine forests, the increased number of Deer Mice after fire was attributed to increased food and cover in the form of stumps and fallen logs; the highest Deer Mouse populations occurred in the areas with significantly more cover and forbs (Lowe *et al.* 1978).

In northern Idaho, Deer Mice were the most commonly trapped small mammal on the Trapper Peak Burn (in subalpine fir [*Abies lasiocarpa*]) 3 years after fire (Ferrell and Olson 1952). In Kansas tallgrass prairie Deer Mice increased after fire largely due to immigration from unburned areas. The positive response to fire was evident by July following an April fire, and continued through the following spring (Kaufman *et al.* 1990, Kaufman *et al.* 1988). In eastern Oregon grass and forb-dominated flood meadows, Deer Mouse numbers were higher on control plots than on burned plots the first year following a fall prescribed fire. Deer Mouse numbers were, however, four times greater on burned areas than on control areas the third winter after the fire (Cornely *et al.* 1983). In northern California brushfields Deer Mouse numbers remained relatively constant in burned areas even though the Deer Mouse population crashed due to drought in control areas (Cook 1959). In California chaparral Deer Mice disappeared immediately after a wildfire, were present within 1 year after the fire, and reached a maximum population the third year after the fire (Quinn 1979).

The frequency of fire affects Deer Mouse abundance. In Kansas tallgrass prairie, Deer Mouse abundance was higher the first year after fire on plots burned every 4 years than on annually burned plots. The average relative density of Deer Mice in all 4 years of a 4-year fire cycle was also higher than the average relative density with annual fire (Kaufman *et al.* 1990). A similar result was obtained in New Brunswick mixed-grass prairie; annual fires resulted in lower Deer Mouse abundance than fires at longer intervals (Springer 1988a).

Although Deer Mouse populations generally increase within a year after fire, effects are variable, especially in nonforested habitats. Lists of reports describing positive, negative, and neutral responses to fire follow.

In the following studies, Deer Mice were more abundant on burned areas than on adjacent unburned areas, or were more abundant on burned areas than on the same area prior to fire. Numbers in parentheses indicate post-fire year(s) of peak Deer Mouse abundance.

Grassland and Prairie

California: annual grassland (Lawrence 1966); central Wisconsin: spring prescribed fire in marshland (1) (Halvorsen and Anderson 1983); South Dakota: spring prescribed fire in mixed-grass prairie (1) (Forde 1983) 2 years after the fire Deer Mouse numbers had dropped to below pre-fire levels (Bock and Bock 1983, Bone and Klukas 1990, Forde *et al.* 1984); Kansas: spring and fall prescribed fire in tallgrass prairie (1); numbers declined to pre-fire levels by the second year (Kaufman *et al.* 1990) southern Illinois: plots in annually burned tallgrass prairie had higher Deer Mouse densities than unburned plots (Schramm and Clover 1994) New Brunswick: mixed-grass prairie (1) (Springer 1988a).

Deciduous woodlands

Minnesota: prescribed fire in bur oak (*Quercus macrocarpa*) savanna and tallgrass prairie (Tester 1965).

Chaparral and Scrub

California: chaparral (3) (Bell and Studinski 1972, Quinn 1990); Deer Mice were not present in pre-fire samples, nor on control plots, but were common in burned plots (2) (Wirtz *et al.* 1988).

Pinyon-Juniper

Nevada: severe prescribed fire reducing pinyon-juniper to grassland (1) (Mason 1977); Utah: chained and burned pinyon-juniper (2) (Baker and Frischknecht 1973); Colorado: pinyon-juniper (Douglass 1989).

Sagebrush

Nevada: fall prescribed fire in sagebrush/grass (McGee 1976); Wyoming: fall prescribed fire in mountain big sagebrush (*Artemisia tridentata* ssp. *vaseyana*)/grassland (2) (McGee 1982).

Forest

Oregon: clearcut and slash-burned Douglas-fir (Hooven 1973); California: clearcut and slash-burned Douglas-fir (1) (Tevis 1956); Arizona: ponderosa pine (1) (Lowe *et al.* 1978), severe spring wildfire in ponderosa pine (Campbell *et al.* 1977); South Dakota: annual prescribed fire in ponderosa pine and adjacent grasslands (Shown 1982); Colorado: wildfire in lodgepole pine (Roppe and Hein 1978); Wyoming: wildfire

in lodgepole pine (Stanton *et al.* 1990); southeastern Manitoba: clearcut and slash-burned jack pine (1) (Sims and Buckner 1973); northeastern Minnesota: cut and burned jack pine stands (1,3) (Ahlgren 1966); north-central Ontario: logged and slash-burned upland black spruce (*Picea mariana*) and northern hardwoods (Martell 1984).

In the following studies Deer Mice were less abundant on burned plots than on adjacent unburned plots or were less abundant on burned plots than on the same plots prior to fire:

Grassland

Illinois: prescribed fire in restored tallgrass prairie; there was no resident population of Deer Mice on adjacent unburned areas to supply immigrants (Springer and Schramm 1972).

Chaparral

California: chaparral (Lawrence 1966).

Sagebrush

Washington: wildfire in antelope bitterbrush-big sagebrush (Gano and Rickard 1982); eastern Idaho: severe wildfire in big sagebrush/grassland; Deer Mice used both burned and unburned areas (Halford 1981); southwestern Idaho: prescribed fire in shrub-steppe; Deer Mouse abundance 1 year after fire was lower on burned and seeded grasslands than on partially burned or control plots (Groves and Steenhof 1988).

Forest

Wyoming: Deer Mice were abundant on both burned and unburned coniferous forest plots; peak abundance occurred in August on unburned plots (Spildie *et al.* 1991).

In the following studies, Deer Mice showed no preference for either burned or unburned plots:

Grassland

Southeastern Arizona: big sacaton (*Sporobolus wrightii*) (Bock and Bock 1978); Minnesota: fall prescribed fire in tallgrass and shortgrass prairie sampled 10 months after the fire (Chance 1986).

Chaparral

Southern California: coastal sage scrub (Price and Waser 1984).

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Procyon lotor
Raccoon

DIRECT FIRE EFFECTS ON ANIMALS

Raccoons are very mobile and probably escape most fires. There are no reports of direct Raccoon mortality due to fire (Landers 1987, Nichols and Menke 1984). Dead insects and small mammals on fresh burns may be attractive to Raccoons (Benseler 1968). Sunquist (1967) reported on the reactions of Raccoons to a controlled fire on the Cedar Creek Natural History Area in east-central Minnesota, which burned 10 ha of savanna habitat. The burn area was not heavily utilized by Raccoons before the fire and even less utilization occurred after. During the 4 days prior to the fire, three of four Raccoons visited or traveled through the area to be burned seven times and spent approximately 2 hours and 15 minutes (total time) in the area. The Raccoons did not enter the burn area on the day of the fire although they rested at different locations within 0.4-0.8 km of it. During the 4 days after the fire all four Raccoons visited or traveled through the burned area six times and spent approximately 2 hours and 30 minutes (total time) in it.

HABITAT RELATED FIRE EFFECTS

Fire that creates a mosaic of burned and unburned areas is probably the most beneficial to Raccoons. Lynch (1941) reported that in Gulf Coast marshes, Raccoons were favoured by "spotty cover burns" (burning the area when there is from 8-13 cm of standing water present). The unburned marsh vegetation provided cover for Raccoons. Longhurst's (1978) observations at the Hopland Field Station in California showed that populations of Raccoons increased in young to intermediate chaparral and grassland-chaparral interspersions. Populations showed a downward trend in both mature chaparral and extensive grasslands.

Periodic fire may also help to maintain Raccoon food. Insects and the fruit of various plants are important in the diet of Raccoons. Populations of insects may increase or decrease as a result of fire depending on fire severity, habitat, and number of years after fire. Effects of late winter controlled burning in broom sedge (*Carex scoparia*) habitat on arthropod density and biomass were studied by Hurst (Hurst 1971). Results of summer sampling revealed that burning increased both density and biomass of most insect orders. The apparent cause of the increases was an increased insect food supply in the form of succulent plant growth following burning in 4- to 5 -year-old broom sedge habitat.

Oaks, persimmons, plums, cherries, and grapes can be severely reduced by fire in the short term. However, except for grapes, these woody species require openings for establishment. Edges of burns along forested areas may be common regeneration sites for many of these plants. Many fruiting shrubs such as blackberries (*Rubus* spp.), blueberries (*Vaccinium* spp.), and huckleberries (*Vaccinium* ssp., *Gaylussacia* spp.) do not fruit the year of burning but produce the most fruit 2 to 4 years after fire pruning (Hon 1981, Landers 1987).

FIRE USE

Areas supporting fire-sensitive mast and fruit producing hardwood species (e.g., large oaks and persimmon) should be protected from burning until they have established (Hon 1981, Landers 1987).

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Sylvilagus floridanus
Eastern Cottontail

DIRECT FIRE EFFECTS ON ANIMALS

Taylor (1981) reviewed twelve studies of fire effects on small mammals and found little evidence of direct mortality. Small mammals are often able to escape fire by retreating to underground burrows (Taylor 1981). Komarek (1969) reported that there was no evidence of direct mortality or even singed fur of either juvenile or adult Eastern Cottontails in 225 prescribed fires (from late fall to early spring) in southern pine forests. Small mammals including Eastern Cottontails have been collected from unburned areas of cover within burned plots (Komarek 1969, Simpson 1972). In Alabama no radio-collared Eastern Cottontails were killed by either severe or low-severity prescribed fire, and no Eastern Cottontail deaths were recorded in the immediate post-fire period (12 days) in pine-hardwood woodlands and adjacent pastures (King *et al.* 1991). In southeastern Illinois tallgrass prairie, Eastern Cottontails were observed escaping winter prescribed fire (Westemeier and Buhnerkempe 1983). In Oklahoma a hot fire in prairie woodlands (August) killed a number of box turtles (*Terrapene* spp.) but living Eastern Cottontails were observed during and after the fire (Bigham *et al.* 1964).

The rate of fire spread is a major factor in direct mortality of small mammals. After a fire that spread 0.6 to 0.8 m/sec with flame heights up to 6 m and flame widths up to 11 m, carcasses of marsh rabbits (*Sylvilagus palustris*) and one cotton rat (*Sigmodon* spp.) were found. However, no dead mammals were found after a fire that spread at a rate of 0.1 m/sec (Taylor 1981). Komarek (1969) found dead marsh rabbits but living Eastern Cottontails after a fire; the implication was that Eastern Cottontails were better able to escape the fire than their congeners. Kelsall and others (1977) asserted that forest fires were less destructive to small mammals than grass fires because forest fires usually move slower than grass fires.

In eastern Nebraska a search conducted immediately after an April prescribed fire in tallgrass prairie turned up a litter of nestling Eastern Cottontails that had been injured by the fire but were able to run off. During the fire one adult Eastern Cottontail was observed behind the fireline, appearing to be disoriented and possibly singed. The researchers were unable to capture the animal for closer examination. No injured or killed Eastern Cottontails were found after two previous similar fires (Erwin and Stasiak 1979).

Komarek (1969) did not list Eastern Cottontails as attracted to fire and smoke, but they were listed as present on both black burns and on burned areas that had greened up.

Most of the effect of fire on vertebrates is the abrupt habitat change following the fire (Erwin and Stasiak 1979). Concentration of Eastern Cottontails into unburned patches increases vulnerability to predators; predators hunted extensively in unburned cover areas in Georgia slash pine-longleaf pine woods after spring prescribed fires (Simpson 1972).

HABITAT RELATED FIRE EFFECTS

Fire's effects on habitat depend on fire characteristics. Soils lose fewer nutrients in low-severity fire than in severe fire. Severe fire volatilizes nutrients and occasionally decreases wettability of the soil surface. Low-severity fire increases herb diversity and stimulates growth, particularly among native legumes. Improved nutritional levels in forage species have been reported after fire. Soil fertilization may increase Eastern Cottontail ovulation rates (Hill 1981).

In southeastern Illinois tallgrass prairie Eastern Cottontails preferred 3-year post-fire communities that had not been mowed over unburned plots and 3-year post-fire plots that had been mowed (Westemeier and Buhnerkempe 1983). In south-central Iowa prescribed fires resulted in declines in Eastern Cottontail habitat quality during the first few post-fire months, but habitat quality improved thereafter until it met or exceeded pre-fire levels (George *et al.* 1978).

In Oklahoma Cross Timbers habitats, pastures (some in post oak-blackjack oak stands) were treated with herbicides (two types) in 1983 to control shrubs, then burned in 1985 to maintain shrub control. There was a gradual decline in Eastern Cottontail populations on all treatments which was attributed to population cyclicity. However, Eastern Cottontail density was higher on herbicide-only treated pastures and on herbicide-burned pastures than on plots that were not treated with herbicide and not burned. The herbicide treatments reduced shrub height but increased stem density. Fire encouraged the growth of herbaceous plants. The authors concluded that herbicide with or without fire has no adverse impacts on resident Eastern Cottontail populations, and that treatment areas had more preferred habitat than control areas (Lochmiller *et al.* 1991).

In Alabama shortleaf pine-hardwood woodlands Eastern Cottontail populations were similar on annually and biennially burned plots. Annually burned plots usually had little fuel and thus experienced low-severity fire that burned less than 50 percent of aboveground vegetation. On biennially burned plots fuels were plentiful and supported severe fire that removed all herbaceous vegetation. Eastern Cottontails chose artificial brushpiles more frequently on biennially burned plots than on annually burned plots in immediate post-fire periods. Eastern Cottontails moved off of severe fire plots during the immediate post-fire period (King *et al.* 1991, Stribling and Speake 1991].

In Florida cattle ranges in slash pine-palmetto flatwoods are maintained in open condition by frequent prescribed fire. Eastern Cottontails use saw-palmetto patches for cover and saw-palmetto is encouraged by frequent fire (Komarek 1963).

FIRE USE

Prescribed fire is the most useful tool for enhancing Eastern Cottontail habitat since it can be used to control the amount of brushy cover and available forage (Chapman *et al.* 1982). In New York prescribed fire every third year in shrub stands within pitch pine-oak woodlands maintains shrub cover adequate for Eastern Cottontails (Reiners 1965).

In Pennsylvania manipulation of hawthorn (*Crataegus* spp.) and alder (*Alnus* spp.) can be achieved with prescribed fire. Hawthorns are an important food for Eastern Cottontails in the area, and can be encouraged by periodic application of fire, since hawthorns sprout after top-kill by fire (Burgason 1976).

In the Southeast pine woodlands are managed with frequent fire; Eastern Cottontail habitat is usually at least adequate in managed pine stands. Additional benefits of fire include reduction of Eastern Cottontail parasites. Pine plantations are good Eastern Cottontail habitat for the first five growing seasons after site preparation. They deteriorate with increased canopy closure and do not improve until prescribed fire (usually initiated in the ninth season) and/or thinning (usually initiated in the fifteenth season) are implemented. To benefit Eastern Cottontails, fire should be used at a frequency sufficient to maintain open conditions and discourage broomsedge, but at long enough intervals to retain some shrub cover and winter browse. There is a need to balance annual fire, which increases summer forage, and longer-interval fires to maintain shrubby areas. Prescribed fire for Eastern Cottontail management therefore needs to be planned so as to leave patches of areas in different post-fire stages, with sufficient annual burn plots to provide summer forage (Hill 1981).

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Tamiasciurus hudsonicus
Red Squirrel

DIRECT FIRE EFFECTS ON ANIMALS

Most Red Squirrels probably escape most fires. In Minnesota oak savanna one Red Squirrel was found dead after a prescribed April fire that consumed all litter, killed most shrubs and young trees, and killed a few large trees (Tester 1965).

HABITAT RELATED FIRE EFFECTS

Kirkpatrick and Mosby (1981) suggested that prescribed fire in southern pine-hardwood sapling and older stages is unlikely to affect resident Tree Squirrels. This may apply to Red Squirrels, since important habitat includes mature trees unlikely to be adversely affected by low-severity fire. Fire severe enough to cause basal fire

wounds may increase cavities available for food caches. Basal fire wounds are unlikely to increase cavities useful as nests for Red Squirrels (Kirkpatrick and Mosby 1981).

In Yellowstone National Park lodgepole pine stands monitored for presence of birds and mammals during post-fire succession, Red Squirrels were only present in stands with closed canopies (Taylor 1974). In north-central Colorado Red Squirrels were not present on 8-year-old burned areas but were present in adjacent unburned lodgepole pine stands (Roppe and Hein 1978).

The maintenance of many mature coniferous forest types is often dependent on fire. Ponderosa pine, Douglas-fir, lodgepole pine, whitebark pine, and spruces are either dependent on stand-replacing fires for regeneration or on low-severity fires for maintenance. Even though severe fire is immediately destructive of Red Squirrel habitat, the long-term maintenance of most coniferous forests is dependent on fire (Kozlowski and Ahlgren 1974).

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Ursus americanus
Black Bear

DIRECT FIRE EFFECTS ON ANIMALS

Direct fire-caused mortality probably has little effect on populations as a whole (Landers 1987).

HABITAT RELATED FIRE EFFECTS

Fires that favour early and mid-seral fruit-producing shrubs and plentiful grasses and forbs are beneficial to bears. Many bear foods are enhanced by fire (Asherin 1973, Edwards 1954, Hall and Shay 1981, Hanson 1979, Heinselman 1973). Fire can also provide a medium for insect invasion, which could provide food. Huckleberries and blueberries are more productive on recently burned sites compared to unburned sites (Books 1972, Hall and Shay 1981, Hanson 1979, Heinselman 1973). However, hot, duff-consuming fires can destroy shallow rhizomes (Hall and Shay 1981). Fire can also reduce important food species in the short-term (Landers 1987). A study in western Montana found that berry production was low for globe huckleberry (*Vaccinium globulare*) in mature stands and stands burned 60 to 100 years previously (Martin 1979). Berry production was best on sites burned between 25 and 60 years previously or on clearcuts that were broadcast burned 8 to 15 years previously.

FIRE USE

Fire can be used to create and maintain seral plant communities important to bears for food. Young and Beecham (1986) recommended the adoption of a "let burn" policy for wildfires and the use of prescribed fire for enhancing bear food. Unsworth and others (1989) suggested broadcast burning logging slash (or leaving it untreated) rather than piling and burning, which can destroy some important shrubs. Landers (1987) stated that burning at 3-year intervals in open slash pine (*Pinus elliotii*) forests would optimize fruit production in blackberries, blueberries, and common gallberries (*Ilex glabra*). However, certain fire-sensitive fruit producers should be protected for longer time periods. Landers listed recommendations for burning in pocosins and hardwood swamps.

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