

**CONSERVATION STRATEGY**

**for**

**SPALDING'S CATCHFLY**  
*(Silene spaldingii Wats.)*

**by**

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## CONSERVATION STRATEGY FOR *SILENE SPALDINGII*

Taxon: *Silene spaldingii* Wats.

Common Name: Spalding's catchfly  
Spalding's silene  
Spalding's campion

Family: Caryophyllaceae  
Pink or Carnation family

States/Province where taxon occurs: United States: Idaho, Montana, Oregon,  
Washington. Canada: British Columbia

Current Federal Status: Threatened

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## **INTRODUCTION**

### **PURPOSE AND GOALS**

The purpose of this document is to provide guidelines to ensure the future survival of *S. spaldingii*. It presents current, available biological information and status of the species, provides the reasons for listing and ongoing threats to the species, summarizes the conservation measures that are in effect, and identifies future management procedures to conserve the species. This information is critical to understanding the needs of the species and will be used in the recovery planning process to ensure the long-term viability of this species across its range.

### **METHODS**

The information presented in this document represents a compilation from previous Status Reports (Heidel 1980; Schassberger 1988; Kagan 1989; Lorain 1991; Gamon 1991; Heidel 1995; Lichthardt 1997) and summary of information not otherwise presented in a Status Report. Element Occurrence Records (EORs) from the Conservation Data Centers (CDC) and Natural Heritage Programs (NHP) in each of the four states and British Columbia provided the best available information on current and historical occurrences. All pertinent literature including Status Reports, Management Plans, research and monitoring studies, and published and unpublished reports were utilized to prepare this document. Questionnaires were sent digitally to all persons working with *S. spaldingii* to provide the opportunity for them to relate pertinent information to be considered in the recovery planning process. A meeting was conducted to provide a forum for interested parties to express, share, and discuss their viewpoints, experience, conservation ideas, and results of work with this species, and to evaluate the Conservation Strategy document. Rangelwide and state distribution maps are included based on current Geographic Information Systems (GIS) files for this species from each of the NHP and CDCs. Other maps show which EORs were combined into populations in each state and locations of populations with 500 or more individuals as potential conservation priority sites. Pertinent literature cited, documentation of letters and telephone calls, completed questionnaires, and all data and reports used to prepare this Conservation Strategy are provided as Administrative Record to the U.S. Fish and Wildlife Service (USFWS).

## SPECIES STATUS

A Petition to List as Threatened or Endangered (Biodiversity Legal Foundation et al. 1995) was filed in February of 1995. Following a 90-Day Finding in November 1998 (63 FR 63661, USFWS 1998), a Proposed Threatened Status in December 1999 (64 FR 67814, USFWS 1999), and Notice of Proposed Critical Habitat Determination in April 2000 (65 FR 21711, USFWS 2000), *S. spaldingii* was listed as a Threatened species on 10 October 2001 (66 FR 51598, USFWS 2001). The legal status of *S. spaldingii* has not changed since the time of final listing. The current legal status of *S. spaldingii* according to federal, state, province, and private agencies is as follows:

### Canada

**Federal:** No status designated under the Species at Risk Act or the Canada Wildlife Act. (Douglas et al. 2002; BC Species and Ecosystems Explorer 2003)

### **British Columbia:**

Global Rank: **G2** (imperiled globally because of rarity or because other factors demonstrably make it very vulnerable to extinction) [British Columbia CDC (BCCDC) 1999; BC Species and Ecosystems Explorer 2003].

Provincial Rank: **S1** (critically imperiled in the province of British Columbia because of rarity or some factor(s) making it very susceptible to extirpation or extinction) (BCCDC 1999; BC Species and Ecosystems Explorer 2003).

BC Status: **Red List.** Includes any indigenous species or subspecies that have, or are candidates for Extirpated, Endangered, or Threatened status in British Columbia (BC Species and Ecosystems Explorer 2003).

### United States

**Federal:** **Threatened** (66 FR 51598, USFWS 2001) under the Endangered Species Act (ESA) of 1973. Taxon likely to be classified as Endangered within the foreseeable future throughout all or a significant portion of its range. Also determined that Critical Habitat is prudent, however, it will not be designated until available resources and priorities allow (66 FR 51598, USFWS 2001).

### **Idaho:**

Global Rank: **G2** (see description above) [Idaho CDC (IDCDC) 2003a].

State Rank: **S1** (critically imperiled in the state of Idaho because of extreme rarity or because some factor of its biology makes it especially vulnerable to extinction) (IDCDC 2003a).

Idaho Native Plant Society: **GP2** (Global Priority 2) - includes taxa with a Global Rank of G2 (IDCDC 2003a).

Bureau of Land Management (BLM): **Type 1 Sensitive Species** - includes Threatened, Endangered, Proposed and Candidate species. These species are listed by the USFWS as Threatened or Endangered, or they are proposed or Candidates for listing under the ESA (IDCDC 2003a).

Idaho Department of Parks and Recreation: State law (Idaho Code 18-3911) to protect wildflowers and shrubs along highways (Moseley 1992).

**Montana:**

Global Rank: **G2** (see description above) [Montana NHP (MNHP) 2003a].

State Rank: **S1** (critically imperiled in the state of Montana because of extreme rarity or because some factor of its biology makes it especially vulnerable to extinction) (MNHP 2003a).

**Oregon:**

Global Rank: **G2** (see description above) [Oregon NHP (ONHP) 2003a].

State Rank: **S1** (critically imperiled in the state of Oregon because of extreme rarity or because some factor of its biology makes it especially vulnerable to extinction) (ONHP 2003a).

State Status: **Endangered.** In danger of becoming extinct or extirpated from Oregon (ONHP 2003a).

ONHP: **List 1.** Threatened or Endangered throughout range (ONHP 2003a).

**Washington:**

Global Rank: **G2** (see description above) [Washington NHP (WNHP) 2003a].

State Rank: **S2** (imperiled in the state of Washington because of rarity or because other factors demonstrably make it very vulnerable to extinction) (WNHP 2003a).

State Status: **Threatened.** Likely to become Endangered in Washington (WNHP 2003a).

## TAXONOMY AND DESCRIPTION OF SPECIES

### TAXONOMY AND EARLY COLLECTIONS

*Silene spaldingii* Wats. (Spalding's catchfly, Spalding's silene, Spalding's campion) is a member of the pink or carnation family (Caryophyllaceae). The majority of plants in this family, including *S. spaldingii*, have distinctly swollen nodes and opposite, simple leaves. The genus *Silene* is characterized by a calyx that forms a distinct tube, petals with short appendages, and normally 3 styles. The approximately 400 species in this genus occur primarily in the North Temperate Zone and are especially abundant in Eurasia (Hitchcock et al. 1964; Hitchcock and Cronquist 1973).

The missionary Henry Spalding collected the first specimen around 1846 (Oliphant 1934) probably from near the Clearwater River in central Idaho. The type specimen label reads "*Silene new?* Clear Water, Oregon, Rev. Mr. Spalding" (*Spalding* 37926 GH) (Hitchcock and Maguire 1947; Hitchcock et al. 1964). At the time, the Clearwater River in Idaho was in Oregon Territory. In a letter to Susan Tucker, C. V. Piper wrote "the original specimens collected by Spalding were supposed to come from the vicinity of Lapwai, Idaho, but no one has since found it there" (Piper 1903). *Silene spaldingii* was formally described and named by Sereno Watson in 1875 (Watson 1875). He states the type specimen came from "on Clearwater in central Idaho, Spalding". He was not aware of any collections other than Spalding's when he published his description.

Other historical collections of *S. spaldingii* include:

1881: W.C. Cusick collected specimens (*Cusick* 970 NY; *Cusick* 970 GH) on the Imnaha River in eastern Oregon (P. Holmgren, *in litt.* 2003; S. Yelton, *in litt.* 2001; Piper 1903).

1882: W.C. Cusick collected a specimen (*Cusick* OSU) at "Wallowa and Imnaha, dry soil" in eastern Oregon (A. Liston, *in litt.* 2003).

1894: R.S. Williams collected specimens (*Williams* NY; *Williams* GH) at Columbia Falls in Flathead County, Montana (P. Holmgren, *in litt.* 2003; S. Yelton, *in litt.* 2001).

1898: W.C. Cusick collected specimens (*Cusick* 2084 GH; *Cusick* 2084 WS; *Cusick* RM; *Cusick* OSC) on the "Dry hills of the Wallowa region, blooming from June to Sept." and "Dry hills near Wallowa? Lake" in eastern Oregon (EOR OR 001) (S. Yelton, *in litt.* 2001; L. Kinter, *in litt.* 2003a; R. Hartman, *in litt.* 2003; A. Liston, *in litt.* 2003).

1903: Mrs. Susan Tucker collected specimens (*Tucker* 129 GH; *Tucker* 129 WS) on Tucker Prairie, 12 miles west of Cheney in Spokane County, Washington (EOR WA 004) (S. Yelton, *in litt.* 2001; L. Kinter, *in litt.* 2003a).

1925: A.H. Eastwood and H. St. John collected a specimen (*Eastwood & St. John* 13222 WS) south of Winona, Whitman County, Washington (EOR WA 003) (L. Kinter, *in litt.* 2003b).

1949: R. Daubenmire collected a specimen (*Daubenmire* 4930 WS) 5 miles west of Colton in Whitman County, Washington (EOR WA 006) (L. Kinter, *in litt.* 2003b).

1951: R. Daubenmire collected a specimen (*Daubenmire* 5118 WS) at the west edge of Pullman in Whitman County, Washington (EOR WA 002) (L. Kinter, *in litt.* 2003a).

1964: R. Daubenmire collected a specimen (*Daubenmire* 6429 WS) two miles west of Craigmont in Lewis County, Idaho (EOR ID 007) (L. Kinter, *in litt.* 2003a).

## **SPECIES DESCRIPTION**

### **Non-technical Description**

*Silene spaldingii* is a herbaceous perennial plant usually growing 8 to 24 inches in height, and occasionally to 30 inches (Hill and Gray 2000). One to several erect stems arise from a persistent underground stem just beneath the soil surface which sits atop a long, narrow taproot, up to 85 cm in length (Menke 2003). Commonly four to seven pairs, occasionally up to 10 pairs (J. Hill, *in litt.* 2003a), of broadly lance-shaped leaves are arranged opposite each other on the stem and measure two to three inches in length and up to 1.5 inches in width (J. Hill, *in litt.* 2003a). The flower cluster is sparsely branched. The flowers consist of fused floral bracts that form a flaring tube about 0.5 inches long and enclose the majority of the five flower petals. Ten distinctive veins run along the length of this tube. The white to cream to greenish petals, and occasionally pinkish petals (J. Hill, *in litt.* 2003a), consist of a lower section ("claw") concealed inside the floral tube and a short flared portion ("blade") that expands a short distance beyond the floral tube, making for a rather inconspicuous flower. The petal blade is shallowly two-lobed, with generally four (up to six), short, lance-shaped appendages at the junction of the claw and blade. The mature fruit has one chamber and can hold up to 150 seeds (Lesica and Heidel 1996). The leaves, stems and floral tube bear gland-tipped hairs that render them extremely sticky. Insects often become stuck on these hairs giving the plant its common name "catchfly." The preceding description was adapted from Hitchcock et al. (1964). Line drawings and photographs of *S. spaldingii* are included in the Appendices (Figures 1A and 2A – Appendix A; Photos 1 through 16 – Appendix E). Slides and photographs of *S. spaldingii* are also on file at each state's NHP or CDC.

### **Technical Description**

The following technical description of *S. spaldingii* is found in Hitchcock et al. (1964):

Villous-tomentose and more or less viscid-pubescent perennial from a simple or branched caudex, 2 to 6 dm tall; cauline leaves in 4 to 7 pairs, oblanceolate below to lanceolate above, 6 to 7 cm long, and 0.5 to 1.8 cm broad, opposite, sessile, and slightly connate; flowers several to many in a leafy and usually compact cyme; calyx tubular-campanulate, about 15 mm long at anthesis, becoming more nearly clavate-campanulate in fruit, 10-nerved; corolla white

(?), the claw of the petals about 15 mm long, not auriculate above; the blade very short, ovate, about 2 mm long, entire to shallowly emarginate; appendages 4 (5 or 6), ovate-lanceolate, about 0.5 mm long; carpophore about 2 mm long, glabrous; styles 3; capsule 1-celled; seeds light brown, about 2 mm long, corrugate-wrinkled and inflated.

### Local Field Characters

Although *S. spaldingii* is similar in appearance to a number of other species of the genus that occur in similar habitats, *S. scouleri*, *S. douglasii*, *S. cseri*, *S. oregana*, and *S. scaposa*, some distinctive features enable accurate field identification. In Montana, Washington, and Idaho, *S. scouleri* and *S. douglasii* are common associates (Schassberger 1988; Gamon 1991; Lichthardt 1997). *Silene scouleri* is very similar vegetatively; however, there are some differences. The lower leaves of *S. scouleri* are much longer and taper to a narrow apex, and the remaining leaves are strongly reduced upward. Lower to middle leaves of *S. spaldingii* are shorter and broadly lanceolate and only weakly reduced upwards. When the plants are flowering, the very short (2 mm) petal blades (flared portion of petal) of *S. spaldingii* are diagnostic: they are entire or shallowly notched with four appendages. In contrast, the petal blades of *S. scouleri* are 6-7 mm long, deeply bi-lobed with only two appendages. *Silene scouleri* flowers earlier than *S. spaldingii* (Lichthardt 1997; J. Hill, *in litt.* 2003a). The dry, curled petal blades of post-bloom *S. scouleri* can appear much shorter than those of full-bloom flowers. This feature can result in misidentification as *S. spaldingii* if close inspection to determine the true petal length is not done (Lichthardt 1997). *Silene douglasii* also shares habitat with *S. spaldingii*, but generally has multiple, more slender stems, and narrower leaves. It is rarely sticky-pubescent and possesses stems that tend to be decumbent (Hitchcock et al. 1964), while those of *S. spaldingii* are generally erect. In Montana, *S. cseri* differs from *S. spaldingii* by its annual habit and longer petal blades, and *S. oregana* has petal blades that are deeply four-lobed and much longer and narrower (Schassberger 1988). In Oregon, a similar-appearing subspecies of *S. scaposa* is differentiated by its slightly earlier bloom time and taller growth form (Youtie 1990).

The late bloom time for *S. spaldingii*, mid-July through September (Kagan 1989; Gamon 1991) and occasionally into October (J. Hill, *in litt.* 2003a), helps distinguish it from associated *Silene* species that flower from May or June through August (Hitchcock et al. 1964). The majority of the surrounding native bunchgrass community dries up during the summer drought period, July through October (Daubenmire 1970), and the pale green foliage of *S. spaldingii* contrasts with the surrounding dried, straw-colored grasses. This feature aids greatly in field surveys for this species (Kagan 1989; B. Benner, *in litt.* 1993); however, the sticky, glandular foliage of *S. spaldingii* is often covered with wind blown debris, including dust, spider webs, various plant parts, insects, etc. (Gamon 1991) which can effectively camouflage the plants and make surveying for them difficult (F. Caplow, *in litt.* 2003a).

## DISTRIBUTION AND ABUNDANCE

### SUMMARY

*Silene spaldingii* is distributed within two distinct areas of occurrence: a larger tri-state area in northeastern Oregon, eastern Washington, and adjoining north-central Idaho, and a smaller disjunct area in northwestern Montana which extends slightly into British Columbia, Canada. These areas include five distinct physiographic areas: 1) Palouse Grasslands of southeastern Washington and adjacent Idaho, 2) Canyon Grasslands along major river systems in the tri-state area of Washington, Idaho and Oregon, 3) Channeled Scablands of east-central Washington, 4) Wallowa Plateau in northeastern Oregon, and 5) Intermontane Valleys of northwestern Montana and British Columbia (Figure 3A – Appendix A) (See descriptions of these physiographic regions in the Environment and Habitat section of this report).

The number of occurrences and plants of *S. spaldingii* reported since petitioning and final listing has increased as a result of increased survey efforts. Currently *S. spaldingii* is known from 130 EORs and 66 populations with 12 in Idaho, 8 in Montana (includes the B.C. EOR), 8 in Oregon, and 38 in Washington (Figures 4A through 7A – Appendix A). The majority of *S. spaldingii* populations are small in size, and located on small, privately-owned parcels, i.e., 65% of populations have <100 plants each, 77% of EORs occur on 10 acres or less in size, and 52% of populations are privately-owned. The majority of *S. spaldingii* plants, 81%, are located in nine larger populations from 500 to >10,000 plants each (Figure 8A – Appendix A). Five extirpations are known rangewide. Potential unsurveyed habitat exists in all physiographic regions in which *S. spaldingii* occurs, particularly the Canyon Grasslands.

### DISTRIBUTION AND ABUNDANCE AT PETITIONING AND LISTING

Previously, at the time of the Petition to List *S. spaldingii* as threatened or endangered (Biodiversity Legal Foundation et al. 1995) in February 1995, *S. spaldingii* was known from 72 EORs and a total number of plants estimated at 13,894 in Idaho, Montana, Oregon, Washington, and British Columbia (Heidel 1995). In the final listing rule (USFWS 2001) in October 2001, *S. spaldingii* was known from 94 EORs, which were grouped together into 52 populations, with a total plant count estimated at 16,500 in Idaho, Montana, Oregon, Washington, and British Columbia (USFWS 2001; E. Rey-Vizgirdas, *in litt.* 1999).

### CURRENT DISTRIBUTION AND ABUNDANCE

Currently, *S. spaldingii* is known from 130 EORs in Idaho, Montana, Oregon, Washington, and British Columbia (BCCDC 1999; IDCDC 2003b; MNHP 2003b; ONHP 2003b; WNHP 2003b; K. Gray, *in litt.* 2003a; B. Benner, *in litt.* 2003a; L. Garner, personal communication, 2003). One hundred seventeen of them are extant records, eight are historical or have location problems, and five are assumed extirpated (Table 1 below; Tables 1A through 4A – Appendix B).

**Table 1 - Current Classification of *Silene spaldingii* Element Occurrence Records.**

State/Province	Extant	Historical	Location Problems	Extirpated	Total EORs
British Columbia	1	-	-	-	1
Idaho	17	-	-	2	19
Montana	10	1	1	-	12
Oregon	24	-	-	1	25
Washington	65	2	4	2	73
<b>TOTAL</b>	<b>117</b>	<b>3</b>	<b>5</b>	<b>5</b>	<b>130</b>

Applying the approximately 1-mile distance criterion used to combine EORs into populations for final listing (E. Rey-Vizgirdas, *in litt.* 1999), the 117 extant EORs represent 66 populations including 12 in Idaho, 8 in Montana (includes the BC EOR), 8 in Oregon, and 38 in Washington for a total of 24,365 plants. Table 2 below shows the distribution of EORs, populations, and plants within the counties of occurrence in each state and British Columbia. Tables 5A through 8A - Appendix B show plant counts for each EOR and population, and which EORs were combined into populations. The plant count represents the most recent count reported for each EOR (Tables 9A through 12A – Appendix B).

**Table 2 – Current Distribution of Element Occurrence Records, Populations, and Plant Counts of *Silene spaldingii*.**

State/Province	County	EORs	Populations	Plant Counts
<b>British Columbia</b>		1	(combined w/ MT)	100
<b>TOTAL</b>		<b>1</b>		<b>100</b>
<b>Idaho</b>	Idaho	5	4	171
	Lewis	1	1	9
	Nez Perce	11	7	4,516
<b>TOTAL</b>		<b>17</b>	<b>12</b>	<b>4,696</b>
<b>Montana</b>	Flathead	3	3	372
	Lake	1	1	200
	Lincoln	3	2	10,146
	Sanders	3	2	85
<b>TOTAL</b>		<b>10</b>	<b>8</b>	<b>10,803</b>
<b>Oregon</b>	Wallowa	24	8	3502
<b>TOTAL</b>		<b>24</b>	<b>8</b>	<b>3,502</b>
<b>Washington</b>	Asotin	6	2	997
	Lincoln	25	11	2,623
	Spokane	15	9	350
	Lincoln/Spokane	-	1	540
	Whitman	18	14	609
	Adams	1	1	145
<b>TOTAL</b>		<b>65</b>	<b>38</b>	<b>5,264</b>
<b>TOTAL</b>		<b>117</b>	<b>66</b>	<b>24,365</b>

The current reported total plant count, 24,365, represents a 75% increase from the 13,894 plants reported in 1995 at the time of the petitioning (Heidel 1995) and a 48% increase from the 16,500 plants reported in 2001 at final listing (E. Rey-Vizgirdas, *in litt.* 1999; USFWS 2001). The increases in numbers of plants reported are a result of discovering new populations or finding larger population extents rather than recruitment of new individuals into populations or increases in abundance of the species. Primarily, the increases between listing and currently have resulted from continuing survey efforts in the Canyon Grasslands of Idaho (Hill et al. 1999; Hill and Gray 2000; Hill et al. 2001; Lichthardt and Gray 2002; Menke 2003; Gray and Lichthardt 2003a, 2003b; IDCDC 2003b) and Channeled Scablands of Washington (B. Benner, *in litt.* 1999, 2003c; Caplow 2001, 2002a, 2002b; Weddell 2002; WNHP 2003b) following the discovery of *S. spaldingii* in habitat types characteristic of these regions in 1993 (B. Benner, *in litt.* 1993; Mancuso and Moseley 1994; J. Hill, *in litt.* 2003a; IDCDC 2003b). A large population was also located at Zumwalt Prairie Preserve in 2002 on the Wallowa Plateau, Oregon, following purchase of this former private parcel by The Nature Conservancy (Elseroad and Taylor 2002a).

Although there have been large increases in the total number of plants reported since petitioning and final listing, caution should be used in interpreting total plant counts. These numbers likely do not represent the actual numbers of plants present at *S. spaldingii* locations. Plant numbers can vary among years due to prolonged dormancy where some plants can remain undetected belowground for up to three consecutive years (Lesica 1997a). The proportion of the population exhibiting prolonged dormancy each year varied from 11-74% during five years of demographic monitoring at Dancing Prairie, Montana (Lesica 1997a). Thus, it is likely that the actual population sizes are greater than the counts of visible plants. Furthermore, methods of counting plants varied between locations, from actual counts of all visible plants at the majority of *S. spaldingii* locations to estimates made by walking stratified random belt transects and counting visible plants at Dancing Prairie (P. Lesica, *in litt.* 2003a, 2003b). The 10,000-plant count for Dancing Prairie is a good minimum estimate, and 50,000 plants would probably be a reasonable upper limit (P. Lesica, *in litt.* 2003b).

## EXTANT OCCURRENCES

### States and Province of Occurrence

#### British Columbia, Canada

*Silene spaldingii* is rare in Canada, known only from Tobacco Plains in southeastern British Columbia, where it was first observed by L. Roe in 1992 and collected by M. Miller in 1995 (Douglas et al. 2002). Currently, the Conservation Data Centre, Victoria, British Columbia, Canada, lists one EOR for *S. spaldingii* (BCCDC 1999; BC Species and Ecosystems Explorer 2003). This EOR has an estimated minimum size of 100 plants. It is located 0.4 miles north of the U.S. border, approximately 0.8 miles northeast of the nearest documented occurrence (EOR MT 008 -Tobacco Plains North) in Lincoln County, Montana, and 4.3 miles north of the Dancing Prairie Preserve, Lincoln County, Montana (EOR MT 001). This is the first record for Canada and the most northerly record for *S. spaldingii* (Miller and

Allen 1997). This EOR was combined with the Montana EOR MT 008 – Tobacco Plains North, into a British Columbia-Montana population (Table 6A – Appendix B; Figure 5A – Appendix A).

## Idaho

In Idaho, *S. spaldingii* is currently known from three counties: Nez Perce, Idaho, and Lewis. Fifty-eight percent of populations and 96% of the plants in Idaho occur in Nez Perce County. The discovery of this species in the Canyon Grasslands of the Snake River (Nez Perce County) in 1993 (Mancuso and Moseley 1994) increased potential habitat considerably, and ongoing surveys in this area have resulted in increases in the number of EORs and total number of individuals known in the state within the last 10 years. The largest population in Idaho and the second largest population rangewide, 3995 plants (J. Hill, *in litt.* 2003b; J. Lichthardt, *in litt.* 2003a; C. Menke, *in litt.* 2003a, 2003b, 2003c; IDCDC 2003b), is located at Garden Creek Ranch in Canyon Grassland habitat on Craig Mountain. Four of the most recent (2001 and 2002) EORs occur in Canyon Grasslands on other areas of Craig Mountain. Two recent occurrences (2002 and 2003) are located in Canyon Grasslands in the Salmon River drainage. Fifty-nine percent of EORs and 98% of plants in Idaho occur within Canyon Grasslands. The remaining *S. spaldingii* occurrences in Idaho are small and isolated, occurring on Palouse Grassland remnants (Figure 3A – Appendix A).

Currently in Idaho, *S. spaldingii* is known from 17 extant EORs, 12 populations, and a total of 4,696 plants; 9 of the 12 populations contain <100 plants each (Table 2; Table 5A - Appendix B). A summary of each *S. spaldingii* EOR in Idaho, including site name, county of occurrence, number of plants recorded for each survey date, area size, elevation, slope, aspect, ownership, habitat type, other rare plants, threats and comments, is provided in Table 9A - Appendix B. Figure 4A - Appendix A shows locations of EORs and populations of *S. spaldingii* in Idaho.

## Montana

In Montana, *S. spaldingii* is currently known from four counties: Lincoln, Flathead, Sanders and Lake. Thirty percent of populations and 94% of plants occur in Lincoln County. The Dancing Prairie Preserve in Tobacco Plains, Lincoln County, supports the largest known population of *S. spaldingii* rangewide with over 10,000 plants, which represents 41% of the known plants rangewide. Prior to 2002, *Silene spaldingii* was known from two regions in northwest Montana about 80 miles apart, one in the Tobacco Plains/Eureka area and the other in the Niarada/Flathead Lake area. In 2002 an occurrence at Lost Trail National Wildlife Refuge (NWR) was located between the two main sites (M. Mantas, *in litt.* 2003; L. Garner, personal communication, 2003). All Montana EORs are located in intermontane valleys (Figure 3A – Appendix A).

Currently in Montana, *S. spaldingii* is known from 10 extant EORs, 8 populations, and a total of 10,803 plants; 4 of the 8 populations contain <100 plants each (Table 2; Table 6A - Appendix B). A summary of each *S. spaldingii* EOR in Montana, including site name, county of occurrence, number of plants recorded for each survey date, area size, elevation,

slope, aspect, ownership, habitat type, other rare plants, threats and comments, is provided in Table 10A - Appendix B. Figure 5A - Appendix A shows locations of each EORs and populations of *S. spaldingii* in Montana.

## Oregon

In Oregon, *S. spaldingii* is currently known only from Wallowa County. All extant EORs are located in the high plateau area in northeastern Oregon (Figure 3A – Appendix A). Zumwalt Prairie has the largest known population of *S. spaldingii* in Oregon with 1,721 plants. Wallowa Lake, Clear Lake Ridge and Crow Creek also support large populations (over 400 plants each). The remaining populations are small, isolated occurrences, scattered within the population and agricultural areas of the Wallowa Valley. Although extensive Canyon Grassland habitat is present within the Snake, Imnaha, and Grande Ronde river drainages of Wallowa County, currently no occurrences have been located in the Canyon Grasslands of Oregon (J. Kagan, *in litt.* 2003a).

Currently in Oregon, *S. spaldingii* is known from 24 extant EORs, 8 populations, and a total of 3,502 plants; 4 of the 8 populations contain <100 plants each (Table 2; Table 7A – Appendix B). A summary of each *S. spaldingii* EOR in Oregon, including site name, county of occurrence, number of plants recorded for each survey date, areal size, elevation, slope, aspect, ownership, habitat type, other rare plants, threats and comments, is provided in Table 11A - Appendix B. Figure 6A - Appendix A shows locations of EORs and populations of *S. spaldingii* in Oregon.

## Washington

The majority, 57%, of *S. spaldingii* populations rangewide occur in Washington, where *S. spaldingii* is known from five counties: Lincoln, Whitman, Spokane, Asotin, and Adams. Lincoln County supports 50% of the known plants and 29% of the populations in Washington. In the 1991 status review for Washington (Gamon 1991), *S. spaldingii* was known from 17 extant EORs in three counties: Whitman, Asotin, and Spokane. The majority of EORs were small, only one had more than 100 individuals, and 76% were privately owned. It was not known from Lincoln or Adams counties. Gamon (1991) identified the western and northern edges of the known range in Washington as needing additional inventory work. Surveys in the last 10 years, most of which were conducted on BLM lands, have focused on this area and many new occurrences have been identified. Since that status review, an additional 48 EORs have been identified, with the majority of these in Lincoln and Spokane counties. The only known extant occurrence in Adams County was identified in 2002 (B. Benner, *in litt.* 2003a). Eighty percent of extant occurrences and 83 % of plants in Washington are located in the extensive sagebrush steppe areas of the Channeled Scablands that include all EORs in Lincoln, Spokane, and Adams counties and those EORs in the northwest corner of Whitman County. The remaining EORs are located in Palouse Grasslands that includes the majority of Whitman and Asotin counties (Figure 3A – Appendix A). Extensive areas of Canyon Grasslands also occur in Washington in the Snake and Grande Ronde river drainages (Figure 3A – Appendix A); EOR WA 009 likely occurs within Canyon Grasslands.

Currently in Washington, *S. spaldingii* is known from 65 extant EORs, 38 populations, and a total of 5,264 plants; 26 of the 38 populations contain <100 plants each (Table 2; Table 8A - Appendix B). A summary of each *S. spaldingii* EOR in Washington, including site name, county of occurrence, number of plants recorded for each survey date, area size, elevation, slope, aspect, ownership, habitat type, other rare plants, threats and comments, is provided in Table 12A - Appendix B. Figure 7A - Appendix A shows locations of EORs and populations of *S. spaldingii* in Washington.

### Areal Size of Extant Occurrences

Areal size classes of extant *S. spaldingii* occurrences are provided in Table 3 below. Information on areal extent of each EOR was only recorded for 65 of the 117 extant occurrences of *S. spaldingii*.

**Table 3 – Areal Size Classes of Extant *Silene spaldingii* Element Occurrence Records.**

Acres	Idaho	Montana	Oregon	Washington	TOTAL
<1	11		6	2	19
1-10	4	7	10	10	31
11-20			1	4	5
21-30				2	2
31-40				1	1
41-50				1	1
>50	1	2	2	1	6
<b>Total EORs</b>	<b>16</b>	<b>9</b>	<b>19</b>	<b>21</b>	<b>65</b>

The majority of EORs, 77%, were located in areas 10 acres or less in size. Most of Idaho EORs, 11 of 16 (69%), were located on less than one acre. The largest areas of occurrence recorded for EORs were 640 acres with 571 plants (EOR OR 020 - Crow Creek) and 651 acres with 67 plants (EOR MT 005 - Crosson Valley). The largest population, Dancing Prairie, Montana, (EOR MT 001), has 10,000 plants on 251 acres. The second largest population, Garden Creek Ranch, Idaho (EOR ID 006), has 3995 plants on 86 acres (J. Hill, *in litt.* 2003b).

### Size (Number of Plants) of Extant Populations

The number of plants in each *S. spaldingii* population was summarized within size classes and is provided in Table 4 below (also see Tables 5A through 8A – Appendix B). Forty-three of the 66 populations, 65%, are small in size, consisting of <100 plants each. These 43 populations support a total of 1,051 plants that represent 4% of the total plants rangewide.

**Table 4 - Size Classes of *Silene spaldingii* Populations.**

Size Class	States				Total
	Idaho	Oregon	Montana	Washington	
0-9	3	3		9	15
10-49	4	1	3	11	19
50-99	2		1	6	9
100-499	2	1	3	8	14
500-999		2		4	6
1000-2000		1			1
2000-3000					
3000-4000	1				1
5000+			1		1
<b>Total Populations</b>	<b>12</b>	<b>8</b>	<b>8</b>	<b>38</b>	<b>66</b>

Fourteen of the 66 populations, 21%, consist of 100-499 plants each. Nine of the 66 populations, 14%, consist of 500 or more plants each, which in total support 81% of plants rangewide. At least one of these larger populations is located in each state of occurrence (Figure 8A – Appendix A) and include:

1. Montana: Dancing Prairie (MT Population 1), >10,023 plants
2. Idaho: Garden Creek Ranch (ID Population 5), 3,995 plants
3. Oregon: Zumwalt Prairie (OR Population 8), 1,721 plants; Wallowa Lake (OR Population 1), 513 plants; Crow Creek (OR Population 3), 844 plants
4. Washington: Miller Ranch (WA Population 20), 540 plants; Twin Lakes (WA Population 27), 627 plants; Blue Mountain (WA Population 25), 997 plants; Coal Creek (WA Population 24), 500 plants

Two other areas have high potential to support populations of 500 or more plants in the future. Although these populations do not currently meet the 1-mile criterion for combining EORs into populations, they are in close proximity (two-three miles) of each other (Figures 4A and 7A – Appendix A), and suitable, unsurveyed habitat exists between them (Caplow 2002b; B. Weddell, *in litt.* 2003a; F. Caplow, *in litt.* 2003b; J. Lichthardt, *in litt.* 2003b; K. Gray, *in litt.* 2003b). These sites include: 1) Populations 22, 36, and 37 in the Swanson Lake Wildlife Area, Washington, which if combined would total 759 plants, and 2) Populations 1, 7, and 11 on northern Craig Mountain, Idaho, which if combined would total 506 plants. Both of these sites occur in close proximity to other large populations as well. The Swanson Lake populations are within two-three miles of the Twin Lakes population (Population 27), and the Craig Mountain populations are within three miles of the Garden Creek Ranch population (Population 5). Lichthardt and Gray (2002) recommend that the three northern Craig Mountain populations be included in the Garden Creek population.

## Ownership of Extant Populations

*Silene spaldingii* populations occur on lands with a variety of ownership types including: 1) **Private ownership** [includes private, private with volunteer registry, and private conservation group (TNC)], 2) **Joint ownership** (includes TNC and federal, private and federal, private and state, private with volunteer registry and state), 3) **Federal ownership**, and 4) **State ownership** (BCCDC 1999; IDCDC 2003b; MNHP 2003b; ONHP 2003b; WNHP 2003b) (Table 5 below; Tables 6A through 9A – Appendix B).

**Table 5 - Ownership of *Silene spaldingii* Plants and Populations.**

Ownership	Total Plants (Populations)					
	BC/ Montana	Idaho	Montana	Oregon	Washington	Totals
Private	223(1)	109 (7)	107 (4)	23 (4)	163 (8)	<b>625 (24)</b>
Private, volunteer registry					143 (7)	<b>143 (7)</b>
Private, TNC			10,023 (1)	2,122 (2)		<b>12,145 (3)</b>
TNC and Federal		3,995 (1)				<b>3,995 (1)</b>
Private and Federal				1357 (2)	800 (2)	<b>2,157 (4)</b>
Private and State					90 (1)	<b>90 (1)</b>
Private/registry and State					203 (2)	<b>203 (2)</b>
Federal		320 (3)	350 (1)		3,266 (12)	<b>3936 (16)</b>
State		272 (1)	200 (1)		599 (6)	<b>1,071 (8)</b>
<b>Totals</b>	<b>223 (1)</b>	<b>4,696 (12)</b>	<b>10,680 (7)</b>	<b>3,502 (8)</b>	<b>5,264 (38)</b>	<b>24,365 (66)</b>

Thirty-four populations and 12,913 plants of *S. spaldingii* occur on privately-owned lands; this represents 52% of the populations and 53% of plants rangewide. Thirty of these populations, which support in total 545 plants, are small populations consisting of < 100 plants each. The remaining four populations are larger populations with >100 plants each. Three of these four larger populations, which support 12,145 plants, are owned by TNC; two of these populations consist of >500 plants each. The remaining larger population is the BC/Montana population with 223 plants. See Tables 5A through 8A – Appendix B.

Eight populations and 6,445 plants of *S. spaldingii* occur on lands with joint ownership, this represents 12% of the populations and 26% of the plants rangewide. Two of these populations, WA Populations 3 and 12, are small populations consisting of < 100 plants each. The remaining six populations are larger populations with >100 plants each. Four of these six larger populations consist of >500 plants each. See Tables 5A through 8A – Appendix B.

Twenty-four populations and 5,007 plants of *S. spaldingii* occur on federal- or state-owned lands; this represents 36% of the populations and 21% of the plants rangewide. Eleven of these 24 populations, which support in total 413 plants, are small populations consisting of <100 plants each. The remaining 13 populations, which support in total 4,594 plants, are

larger populations with >100 plants each. Three of these 13 populations consist of populations with >500 plants each. See Tables 5A through 8A – Appendix B.

Ownership of the nine populations with >500 plants includes:

1. Montana: Dancing Prairie, >10,023 plants (Private/TNC)
2. Idaho: Garden Creek Ranch, 3,995 plants (Private/TNC and BLM)
3. Oregon: Zumwalt Prairie, 1,721 plants (Private/TNC); Wallowa Lake, 513 plants (Private and USFS); Crow Creek, 844 plants (Private and USFS)
4. Washington: Miller Ranch, 540 plants (BLM), Twin Lakes, 627 plants (Private and BLM); Blue Mountains, 997 plants (USFS); Coal Creek, 500 plants (BLM)

### EXTIRPATED OCCURRENCES

Sixteen *S. spaldingii* EORs were stated as extirpated in the final listing rule (USFWS 2001). This information was based on documentation in a status review table (E. Rey-Vizgirdas, *in litt.* 1999). According to 1998 and 2000 EORs (IDCDC 1998, 2000; MNHP 1998, 2000; ONHP 1998, 2000; WNHP 1998, 2000) and status reports (Schassberger 1988; Gamon 1991; Lorain 1991; Lichthardt 1997), only five EORs rangewide (EORs ID 004, ID 002, OR 002, WA 005, WA 007) were officially classified as "Assumed Extirpated" by CDCs and NHPs at the time of final listing (Tables 1A through 4A – Appendix B). Table 6 below lists the EORs determined to be extirpated in the status review table (E. Rey-Vizgirdas, *in litt.* 1999) and the official rank assigned by the CDC or NHP at the time of final listing. The status review table lists 17 EORs as extirpated although only 16 EORs were quoted in the final rule.

**Table 6 - Rank of *Silene spaldingii* Element Occurrence Records Determined to be Extirpated at Final Listing.**

EORs determined to be Extirpated at Final Listing (E. Rey-Vizgirdas, <i>in litt.</i> 1999)	CDC/NHP Rank at Final Listing (1998 and 2000 EORs)
ID 004 (Lawyer Creek)	Extirpated
MT 006 (Columbia Falls)	Historical
MT 007 (Niarada)	Exact Location Unknown
WA 002 (Pullman West)	Exact Location Unknown
WA 003 (Hill South of Winona)	Exact Location Unknown
WA 004 (Tucker Prairie)	Exact Location Unknown
WA 005 (Liberty Lake)	Extirpated
WA 006 (Kramer Prairie)	Extant (Surveyed with 0 plants in 1995 & 200 in 2000)
WA 007 (SW of Lamont)	Extirpated
WA 009 (Upper Wawawai)	Extant (Surveyed with 0 plants in 1995 & 3 in 2002)
WA 010 (Spaulding Road)	Extant (Surveyed with 0 plants in 1995)
WA 016 (Gooseneck Steppe I)	Extant (Surveyed with 0 plants in 1995)
WA 017 (Gooseneck Steppe II)	Extant (Surveyed with 0 plants in 1995)
WA 018 (Smoot Hill)	Historical
WA 019 (Johnson-Pullman Road)	Historical
WA 022 (Strangland Road)	Extant (Surveyed with 0 plants in 1995)
WA 024 (Kramer Prairie NNE)	Extant (Surveyed with 0 plants in 1995)

The seven extant EORs were considered extirpated because the most recent surveys at these locations found no plants (Wentworth 1996). However, an EOR is not generally classified by the CDC or NHP as “Assumed Extirpated” until repeated surveys show no plants or the habitat is destroyed (Heidel 1995). Further complicating this issue is the fact that large proportions of *S. spaldingii* populations can remain dormant underground for up to three years (Lesica 1997a) (see Prolonged Dormancy in the Life History and Biology section of this report). Of the five EORs officially classified as “Assumed Extirpated” by CDCs and NHPs at the time of final listing, only three were included in the status review table (E. Rey-Vizgirdas, *in litt.* 1999) utilized for the final rule: 1) EOR ID 004 (Lawyer Creek), 2) EOR WA 005 (Liberty Lake), and 3) EOR WA 007 (SW of Lamont). The two remaining extirpated EORs, i.e., Daubenmire Stand 161 (previously numbered EOR ID 002) (Lichthardt 1997) and EOR OR 002 (Darr Flat) (Schassberger 1988), were not included on the list of extirpated EORs in the status review table (E. Rey-Vizgirdas, *in litt.* 1999).

It is unclear why historic records and records where the exact location is unknown were listed as extirpated in the final listing. Inconsistencies regarding the definition of a ‘historic record’ may have contributed to this interpretation. The MNHP defines a “Historical” occurrence as one “known only from records over 50 years ago; may be rediscovered” (MNHP 2003a). The WNHP defines it as “Historical occurrences only are known, perhaps not verified in the past 20 years, but the taxon is suspected to still exist in the state” (WNHP 2003a). The IDCDC defines it as “Historical occurrence (i.e., formerly part of the native biota; implied expectation that it might be rediscovered or possibly extinct)” (IDCDC 2003a). The BCCDC defines it as “Historical occurrence; usually not verified in the last 40 years, but with the expectation that it someday may be rediscovered” (BCCDC 1999). The NatureServe website (2003) defines a historical record as “possibly extirpated/possibly extinct.” Heidel (1995) defines historic records as “those which were documented over 20 years ago and which preliminary fieldwork indicates that they may be extirpated”. There is disagreement even within the same state over the status of an EOR. In the last status report for Montana (Schassberger 1988), the Columbia Falls occurrence (EOR MT 006) is listed as “historic”, yet Heidel (1995) considered this EOR to be extirpated. Recently, P. Lesica (*in litt.* 2003a) mentions this EOR is from a historic collection with only general locality information provided on the label, and since the exact location is not know, it is not possible to say whether it is extirpated or not.

According to the 2003 EORs (IDCDC 2003b; MNHP 2003b; ONHP 2003b; WNHP 2003b) and last status reports (Schassberger 1988; Gamon 1991; Lorain 1991; Lichthardt 1997), only five EORs rangewide are officially classified as "Assumed Extirpated." These extirpated EORs include: 1) Daubenmire Stand 161 (previously numbered EOR ID 002), 2) EOR ID 004 - Lawyer Creek, 3) EOR OR 002 – Darr Flat, 4) EOR WA 005 – Liberty Lake, and 5) EOR WA 007 – SW of Lamont (Table 1; Tables 1A through 4A – Appendix B). At EOR ID 004 - Lawyer Creek, four plants were observed in 1980. Since then the area has undergone extensive highway construction and infestation of large areas by the non-native invasive species, yellow starthistle (*Centaurea solstitialis*). A survey in 2001 revealed no plants and because of the small population and disturbance in the area, this population was assumed extirpated (Lorain 1991; Lichthardt 1997; Lichthardt and Gray 2002). The other

extirpated Idaho occurrence, Daubenmire Stand 161 (previously identified as EOR ID 002) where *S. spaldingii* was noted and collected in a grassland stand sampled by R. Daubenmire in the 1950s (Daubenmire 1970), appears to have been plowed. This occurrence was listed as assumed extirpated and the remaining portion of this site has since been combined with the extant occurrence EOR ID 007 – Cold Spring Creek (Lichthardt 1997). The EOR WA 005 - Liberty Lake occurrence, where 10 plants were observed in 1982, has been subdivided and the plants likely extirpated (Gamon 1991). The other extirpated Washington occurrence, EOR WA 007 - SW of Lamont, has been significantly degraded and the plants are assumed extirpated (Gamon 1991). The extirpated site in Oregon, EOR OR 002 - Darr Flat, was discovered in 1983, but had no plants in 1986. The population was presumed to have been lost due to the heavy livestock grazing that occurs in the area (Schassberger 1988).

### **ERRONEOUS REPORTS**

An EOR originally assumed to occur in the McCall area of Adams County, Idaho, actually was an EOR from Adams County near Macall, Washington (Lorain 1991; Gamon 1991). It has not been relocated. Six Idaho EORs, [Moscow South/Cameron Prairie (EOR ID 003), Reubens Cemetery (EOR ID 005), Thorn Springs Creek (EOR ID 006), Cold Spring Creek South (EOR ID 007), Lapwai Lake Southeast (EOR ID 008), Talmaks Campground East (EOR ID 009)], were determined to be based on misidentifications (Lichthardt 1997). These records were dropped from the IDCDC database and the EOR numbers were reassigned to new populations. Post-bloom *Silene scouleri*, whose petals had dried up making them appear much shorter, was misidentified as *S. spaldingii* at these sites (Lichthardt 1997).

### **POTENTIAL UNSURVEYED HABITAT**

Potential unsurveyed habitat exists in all physiographic regions known to support *S. spaldingii* rangewide, i.e., the Palouse Grasslands in eastern Washington and adjacent Idaho, the Channeled Scablands in east-central Washington, the Canyon Grasslands along major rivers in the tri-state area of Idaho, Oregon and Washington, the Wallowa Plateau in northeastern Oregon, and the Intermontane Valleys in northwestern Montana and British Columbia.

One of the largest areas of unsurveyed potential habitat for *S. spaldingii* is Canyon Grasslands. Extensive areas of Canyon Grasslands occur along the Snake, Salmon, Clearwater, Grande Ronde, and Imnaha rivers in Idaho, Oregon and Washington (Tisdale 1986a) (Figures 3A and 9A – Appendix A); however, only a small portion of this region in Idaho, primarily Craig Mountain, has been surveyed for *S. spaldingii*. Due to the steep, rugged terrain of Canyon Grasslands, conversion to cropland, residential and urban development, and other human-induced fragmentation has been minimal, and large areas of suitable, contiguous habitat remain. The presence of the second largest population of *S. spaldingii* rangewide within Idaho Canyon Grasslands strongly suggests the species occurs in other portions of the Canyon Grasslands. The Idaho fescue-prairie junegrass (*Festuca idahoensis-Koeleria cristata*) [FEID-KOCR] habitat type/plant association (Tisdale 1986a; Johnson and Simon 1987) that supports many of the *S. spaldingii* occurrences in Idaho

Canyon Grasslands, comprises significant portions of Canyon Grasslands in Oregon and Washington as well (Tisdale 1986a; Johnson and Simon 1987).

Based on known locations of *S. spaldingii* in Idaho Canyon Grasslands, potential habitat in Canyon Grasslands is predictable. All known sites occur on northerly aspects (NW to N to NE) occupied by mesic Idaho fescue grasslands between 1,380 and 4,000 feet (J. Lichthardt, *in litt.* 2003b, J. Hill, *in litt.* 2003a, 2003b; K. Gray, *in litt.* 2003a; Lichthardt and Gray 2002; C. Menke *in litt.* 2003c; IDCDC 2003b). Above this elevational range, northerly aspects generally support shrub and/or forest vegetation; below this elevational range, northerly aspects generally support grasslands too dry for Idaho fescue or *S. spaldingii*. The elevational range of mesic Idaho fescue grasslands communities on northerly aspects likely varies between different regions of the Canyon Grasslands depending on the elevational extent of trees on northerly aspects. On Craig Mountain, forests extend to lower elevations on northerly aspects in the northern part of the mountain than on the southern part of the mountain (Mancuso 1993).

Specific areas with potential unsurveyed habitat are listed below for each state/province, including those EORs that state "needs further surveying," " cursory surveys," or "more thorough survey needed." Much unsurveyed potential habitat exists on privately-owned lands and permission to inventory these areas is difficult to obtain.

### **British Columbia**

1. Intermontane Valleys: Unsurveyed rough fescue grasslands in intermontane valleys similar to those known to support *S. spaldingii* in British Columbia have potential to support *S. spaldingii*.
2. EORs indicating further survey work is needed: The BC EOR does not indicate further survey is needed at this site.

### **Idaho**

1. Palouse Grasslands: Portions of the Camas Prairie on Nez Perce Reservation land have potential habitat that has not been surveyed (A. Sondenaar, personal communication, 2003).
2. Canyon Grasslands:
  - a. Garden Creek Ranch on Craig Mountain: Approximately 300-500 acres of potential unsurveyed habitat remain to be surveyed at the Ranch. Previous surveys have covered 700-900 acres of the approximately 1,200 acres of mesic Idaho fescue habitat types (FEID-KOCR) capable of supporting *S. spaldingii* at the Ranch (Hill and Gray 1999; J. Hill, *in litt.* 2003a; C. Menke, *in litt.* 2003a, 2003b). Of suitable habitat surveyed at the Garden Creek Ranch, approximately half supported *S. spaldingii* and half did not (J. Hill, *in litt.* 2003a).

- b. Other Craig Mountain areas: Billy Creek, Middle Creek and possibly Redbird Creek drainages have potential unsurveyed habitat (J. Lichthardt, *in litt.* 2003b; K. Gray, *in litt.* 2003b). The Eagle Creek drainage also likely supports potential habitat. A possible occurrence of *S. spaldingii* was located in this drainage late in the 2003 field season (K. Gray, personal communication, 2003).
  - c. Canyon Grasslands other than Craig Mountain: Canyon Grasslands have extensive coverage in Idaho (Tisdale 1986a) (Figure 9A – Appendix A) and much of them have had little survey work. This includes the Clearwater River from its confluence with the Snake River at Lewiston to Kooskia, the Salmon River from its confluence with the Snake River to Riggins, and the Snake River north of Craig Mountain to Lewiston and south of Craig Mountain to Brownlee Dam. Large tributaries that drain into these rivers, i.e., Potlatch Creek, Rice Creek, Rock Creek, Lapwai Creek, etc., likely contain potential habitat as well. The Lapwai Creek area is where Reverend Spalding first collected the type specimen for *S. spaldingii*; however, no extant occurrences are known from this area.
3. EORs indicating further survey work is needed (IDCDC 2003b):
- a. EOR ID 002 – Genesse South
  - b. EOR ID 006 – Garden Creek Ranch
  - c. EOR ID 008 – Red Bird Triangulation Point
  - d. EOR ID 014 – Salmon River Oxbow
  - e. EOR ID 015 – Camp Creek
  - f. EOR ID 017 – Billy Creek North

### **Montana**

- 1. Intermontane Valleys: Potential unsurveyed habitat likely exists in intermontane valleys of northwestern Montana, including those in the Bob Marshall Wilderness, Flathead Indian Reservation, Lost Trail NWR, and the National Bison Range (C. Bjork, personal communication, 2003a).
- 2. EORs indicating further survey work is needed: No Montana EORs indicated further survey work was needed (MNHP 2003b).

### **Oregon**

- 1. Wallowa Plateau: Zumwalt Prairie has potential habitat that has not been surveyed, both on the Zumwalt Prairie Preserve (R. Taylor, *in litt.* 2003a;) and private lands surrounding the Preserve (J. Kagan, *in litt.* 2003a, 2003b).

2. Canyon Grasslands:

- a. Canyon Grasslands are extensive in Oregon (Tisdale 1986a) (Figure 9A – Appendix A), and have received little specific inventory for *S. spaldingii*. This includes the Imnaha River, the Grande Ronde River (including large tributaries like Joseph Canyon), and the Snake River from Brownlee Dam to the Washington border. Some rare plant survey work was done in the fescue grasslands of the Canyon Grasslands along the Snake River in 1987, but these surveys did not focus specifically on *S. spaldingii* and no occurrences were located (J. Kagan, *in litt.* 2003a, 2003c).
- b. Portions of the Nez Perce Tribe Precious Lands Preserve in the Grande Ronde River drainage occur in potential Canyon Grasslands habitat that has not been surveyed (A. Sondenaar, personal communication, 2003). During 2003 field surveys, a possible occurrence of *S. spaldingii* (discovered too late in the season for positive identification) was located on the Precious Lands Preserve in Joseph Canyon (A. Sondenaar, *in litt.* 2003b).

3. EORs indicating further survey work is needed (ONHP 2003b):

- a. EOR OR 005 – Downey Gulch North
- b. EOR OR 006 – Clear Lake Ridge
- c. EOR OR 020 – Crow Creek.

## **Washington**

1. Canyon Grasslands:

- a. Potential unsurveyed Canyon Grassland habitat in Washington includes the lower Grande Ronde River and the Snake River from the Washington/Oregon border to Little Goose Dam. Large canyons draining into these rivers, especially those containing abundant northerly slopes, like Wawawai Canyon, should also be surveyed. The EOR WA 009 – Upper Wawawai is likely a Canyon Grassland site; it occurs in Wawawai Canyon in the upper breaks of the Snake River on a steep, northerly slope at 2300 feet (WNHP 2003b). A possible occurrence of *S. spaldingii* (too early in the season for positive identification) was located within Canyon Grasslands on Lime Hill near the confluence the Snake and Grande Ronde rivers (C. Bjork, personal communication, 2003b).
- b. Portions of the Nez Perce Tribe Precious Lands Preserve on the lower Grande Ronde River support potential unsurveyed Canyon Grasslands habitat that has not been surveyed (A. Sondenaar, personal communication, 2003).

2. Channeled Scablands:

- a. The majority of known occurrences in Channeled Scablands are located in the three-tip sagebrush-Idaho fescue (*Artemisia tripartita*-*F. idahoensis* [ARTR2-FEID] habitat type (B. Benner, personal communication, 2003a). This type covers the majority of Lincoln County, northeast Adams County, and portions of Douglas and Okanagan counties (Figure 10A – Appendix A), and penetrates drier zones to the south as a topographic climax restricted to northerly slopes and loessal soils (Daubenmire 1970). All unsurveyed areas supporting this habitat type have high potential to support *S. spaldingii*.
- b. Portions of the extensive shrub-steppe area in Lincoln County in the Swanson Lake Wildlife Area and the Twin Lakes area to the north support potential unsurveyed habitat (Weddell 2002; Caplow 2002b; F. Caplow, *in litt.* 2003b; B. Weddell, *in litt.* 2003a).
- c. The area between the Crab Creek population (WA Population 24) and the Coal Creek population (WA Population 18), including the Rocky Ford population (WA Population 23) supports potential unsurveyed habitat (B. Benner, *in litt.* 2003b).
- d. State lands within the “channels” have not yet been surveyed and likely support *S. spaldingii* habitat (B. Benner, *in litt.* 2003c).
- e. Private lands in Lincoln and Spokane counties have potential unsurveyed habitat. Many private land parcels acquired by BLM through land exchanges in this area have had *S. spaldingii* occurrences (B. Benner, *in litt.* 2003d).

3. EORS indicating further survey work is needed (WNHP 2003b):

- a. EOR WA 009 – Upper Wawawai
- b. EOR WA 016 – Gooseneck Steppe I
- c. EOR WA 017 – Gooseneck Steppe II
- d. EOR WA 020 – Steptoe Butte
- e. EOR WA 050 – Sourdough Ridge

## ENVIRONMENT AND HABITAT

### SUMMARY

Major geologic events have shaped the terrain and influenced the soils and climate in the areas of *S. spaldingii* occurrence. *Silene spaldingii* occurs in five physiographic regions characterized by distinctive physical and biological features. Within the larger tri-state area it occurs in 1) Palouse Grasslands in southeastern Washington and adjacent Idaho, 2) Channeled Scablands in east-central Washington, 3) Canyon Grasslands along the Snake, Salmon, Clearwater, Grande Ronde, and Imnaha rivers, and 4) the Wallowa Plateau in northeastern Oregon. Within the smaller disjunct area of northwestern Montana and adjoining British Columbia, *S. spaldingii* occurs in 5) Intermontane Valleys (Figure 3A – Appendix A).

*Silene spaldingii* occurs within a wide range of elevations, reaching its extremes in the Hells Canyon area in the tri-state area of northeastern Oregon, eastern Washington and adjacent Idaho. Here it occurs as low as 1,380 feet along the lower Salmon River in Idaho to 5,100 feet on the Wallowa Plateau in northeastern Oregon. *Silene spaldingii* appears to be located in areas where enough soil moisture accumulates or is retained into the growing season, i.e., swales, northerly aspects, microtopographic features, etc, to provide the necessary mesic conditions it requires. *Silene spaldingii* shows a close association with northerly aspects, particularly in the drier portions of its range, i.e., Canyon Grasslands and Channeled Scablands, although all aspects have been recorded. Soils are characteristically deep with silt-sized particles, being considerably loess- and ash-influenced in the tri-state area and glacially-influenced in the disjunct northwestern Montana/British Columbia area. Climate of the area supporting *S. spaldingii* varies considerably, from moderately cold to relatively mild winters, and warm to hot, dry summers. Mean annual temperature varies from approximately 45 to 55 °F, mean annual precipitation varies from 11 to 25 inches, and seasonal distribution of precipitation varies from winter-high precipitation in the tri-state area of occurrence to summer-high precipitation in the disjunct northwestern Montana area (Climate Diagrams – Appendix C).

*Silene spaldingii* occurs in grassland, shrub and forest habitat types across its range. The grassland types are mesic fescue grasslands of the Pacific Northwest Bunchgrass Grasslands (Tisdale 1983), and are generally characterized by high cover of perennial bunchgrasses, a relatively abundant and diverse perennial forb component, often a minor shrub component, and a well-developed cryptogam layer. Shrub and forest habitat types include sagebrush – fescue and open-canopy pine-fescue types. The fescue associations in these shrub- and tree-dominated communities are very similar to the mesic fescue grassland habitat types (Daubenmire 1968b, 1970). Within the tri-state area of occurrence, *S. spaldingii* is associated with Idaho fescue (*F. idahoensis*), and in the disjunct northwestern Montana area of occurrence, *S. spaldingii* is associated with rough fescue (*F. scabrella*). *Silene spaldingii* usually occurs with low cover and frequency in relatively undisturbed, late-successional communities that are in relatively good ecological condition.

## PHYSICAL FEATURES

Interplay between the physical features of *S. spaldingii*'s environment, i.e., geologic, edaphic, topographic habitat features and climate, provide the mesic conditions necessary to support *S. spaldingii* and the fescue communities with which it is associated. Major geologic events, which have shaped the terrain and influenced soils and climate where *S. spaldingii* occurs, indicate a close association between the disjunct area of occurrence in northwestern Montana and the tri-state area of occurrence in Washington, Idaho and Oregon.

### Geologic Characteristics

Major geologic events shaped the area where *S. spaldingii* occurs. During the Cenozoic era (65 million years ago) and the Miocene Era (15 million years ago), numerous lava flows erupting from deep fractures near the Washington-Idaho-Oregon border spread across thousands of square miles, filling the Columbia Basin with deep basalt bedrock. Continental glaciers moved down large trenches in northern Washington, Idaho, and Montana, and as they recessed, melt-water formed the large glacial Lake Missoula in northwest Montana. The full volume of the lake was released when an ice dam broke, resulting in catastrophic flooding across east-central Washington. The torrents of water and ice stripped sediment layers and cut through underlying basalt. Recurring glacial floods created the braided erosion channels of the Channeled Scablands, and large deposits of suspended sediment and glacial outwash were deposited in south-central Washington. Uplifting of the Wallowa Mountains and the Seven Devils Mountains in the Hells Canyon area during the Mesozoic Era (225-65 million years ago), and down-cutting by the Snake, Salmon, Clearwater, Grande Ronde, and Imnaha rivers during the Pliocene (10 million years ago) occurred as well. During the Pleistocene (1 million years ago), strong winds carried silt-sized particles (loess) from the outwash deposits in south-central Washington and deposited layers of loess over the tri-state area of *S. spaldingii* occurrence. The major deposition occurred in the Palouse region of southeastern Washington and adjacent Idaho, creating the large earthen dunes characteristic of this area. Later, volcanic eruptions from Glacier Peak in Washington (12,000 yr ago), Mt. Mazama in Oregon (6600 yr ago), and Mt. St. Helens in Washington (23 yr ago) deposited layers of ash over the tri-state area of *S. spaldingii* occurrence. (This information was adapted from Alt and Hyndman 1978, 1984, 1986, 1989; Johnson and Simon 1987; Mueller and Mueller 1997).

### Edaphic Characteristics

Soils that support *S. spaldingii* reflect the geology of the area in which it occurs. In the tri-state area of occurrence, soils are considerably loess- and ash-influenced (Daubenmire 1970; Tisdale 1986a; Tisdale 1986b; Johnson and Simon 1987). *Silene spaldingii* in this area is closely associated with wind-blown soils (loess) that originated from outwash deposits in south-central Washington. The loess and ash layers have eroded in many areas, especially on southerly slopes in areas of high relief. However, major deposits remain on northerly slopes of Canyon Grasslands, the large loess dunes of the Palouse Grasslands, and the mounds of biscuit and swale topography in the Channeled Scablands (Daubenmire 1970; Franklin and Dyness 1973; Tisdale 1986a; Johnson and Simon 1987). Deep, ash- and loess-influenced

soils have high moisture-holding capacity and high water infiltration rates (Ross and Savage 1967; Johnson and Simon 1987). These properties increase the water holding capacity of soils that support *S. spaldingii*, a feature that is especially critical in the tri-state area where summer precipitation is limited. In the disjunct northwestern Montana area of occurrence, *S. spaldingii* is located on glacially-influenced soils, i.e., deep alluvial or eolian soils (loess) deposited by glacial Lake Missoula. These soils are moderately deep, silty loams that are sparsely gravelly (Schassberger 1988). In both the tri-state area and the disjunct Montana area of occurrence, *S. spaldingii* is associated with generally deep loamy soils that are not excessively drained (P. Lesica, *in litt.* 2003a, 2003b).

Mesic fescue grasslands that support *S. spaldingii* are highly productive, a factor related to the high ash and/or loess content of the soils. Ash soils have high amounts of nutrients in upper surface layers, and loessal deposits are normally high in base saturation and can hold large amounts of nutrients (Johnson and Simon 1987; Buckman and Brady 1960). Productivity, as measured by total cover and dry weight, is nearly three times greater in mesic fescue communities, where soils are deep and ash- and loess-influenced, than in the drier bluebunch wheatgrass communities where ash and loess have eroded and soils are generally residual in nature (Johnson and Simon 1987). As a consequence of this high productivity, much of the rolling loess dunes in the Palouse Grasslands, areas of loess that survived the glacial floods in the Channeled Scablands, the Wallowa Plateau of Oregon, and Intermontane Valleys of Montana have been converted to cropland or pastures (Habeck 1967; Daubenmire 1978; Lichthardt and Moseley 1997b; Noss et al. 1995). Due to the steep, rugged terrain of Canyon Grasslands and portions of the Channeled Grasslands, conversion to cropland is not feasible. These areas remain relatively intact although the productive grasslands they support have been utilized extensively for domestic livestock grazing (Daubenmire 1970; Tisdale 1986a; Johnson and Simon 1987).

### **Topographic Features**

*Silene spaldingii* occurs at a wide range of elevations and percent slopes. The extremes in elevation rangewide occur within the tri-state area near the Idaho-Oregon border spanning the Hells Canyon area. Here, *S. spaldingii* occurs at the lowest known elevation, 1,380 feet in Canyon Grasslands on the lower Salmon River and the highest known elevation, 5,100 feet on the Wallowa Plateau in northeastern Oregon. In Idaho, elevations range from 1,380 feet in Canyon Grasslands to 3,850 feet in Palouse Grasslands of the Camas Prairie (IDCDC 2003b; K. Gray, *in litt.* 2003a). In Washington, elevations range from 1,550 feet in the Channeled Scabland area to 3,200 feet in the foothills of the Blue Mountains (WNHP 2003b). In Montana, elevations range from 2,700 feet at Dancing Prairie to 3,800 feet in the Hog Heaven Range (MNHP 2003b). In Oregon, elevations range from 3,700 feet in the Crow Creek area to 5,100 feet on Clear Lake Ridge (ONHP 2003b). Rangewide, slopes of *S. spaldingii* sites vary from 0-70%, with the majority occurring between 20-40% (IDCDC 2003b; MNHP 2003b; ONHP 2003b; WNHP 2003b). (Note: Slope information was reported for only 66 of the 117 extant EORs)

The primary aspects of *S. spaldingii* EORs rangewide are northwest, north, and northeast (Table 7 below). Eighty-two percent of the EORs for which aspect was recorded occur

within this range of aspects, with the highest percentage, 46%, occurring on north aspects (IDCDC 2003b; MNHP 2003b; ONHP 2003b; WNHP 2003b; K. Gray, *in litt.* 2003a; B. Benner, *in litt.* 2003a). (Aspect was recorded for 105 of 117 extant EORs. More than one aspect was often recorded for an EOR resulting in a total of 151 recorded aspects).

**Table 7 - Aspects of *Silene spaldingii* Element Occurrence Records.**

State	S	SW	W	NW	N	NE	E	SE	Flat	Total
Idaho	0	1	3	9	6	6	2	0	0	-
Montana	0	0	0	1	3	1	2	0	0	-
Oregon	0	1	0	1	10	5	1	0	1	-
Washington	2	0	7	19	51	12	3	0	4	-
<b>TOTAL</b>	<b>2</b>	<b>2</b>	<b>10</b>	<b>30</b>	<b>70</b>	<b>24</b>	<b>8</b>	<b>0</b>	<b>5</b>	<b>151</b>
<b>Percentage</b>	<b>1</b>	<b>1</b>	<b>7</b>	<b>20</b>	<b>46</b>	<b>16</b>	<b>6</b>	<b>0</b>	<b>3</b>	<b>100</b>

Northerly aspects are important distributional factors for mesic fescue communities and *S. spaldingii* in the more arid portions of its range, i.e., the Canyon Grasslands and drier portions of the Channeled Scablands, where they are restricted to these more mesic aspects (B. Benner, *in litt.* 1993, 2003a; F. Caplow, personal communication, 2003; Hill and Gray 1999; J. Hill, *in litt.* 2003a, 2003b; Menke 2003; Lichthardt and Gray 2002). Soil moisture of northerly aspects is significantly greater than that of southerly slopes at the same elevation due primarily to differences in insolation, soil temperature, soil depth, and soil properties between these slopes (Blinn and Habeck 1967; Daubenmire 1970; Aller et al. 1981; Tisdale 1986a; Mancuso 1993). Northerly aspects receive less direct insolation and thus are cooler and have less evaporation than southerly aspects at the same elevation. Snow often remains longer in the spring on northerly aspects, and the deep loess- and ash-influenced soils of these sites have high moisture-holding capacity. North-facing slopes in the Palouse Grasslands of southeastern Washington and adjacent Idaho have an equivalent of 150-200 cm of annual precipitation (Rockie 1950) compared to 52-60 cm for level areas (Klages 1942). The mesic fescue communities that support *S. spaldingii* can extend into generally drier regions, i.e., Canyon Grasslands, as topographic climaxes by taking advantage of mesic conditions on north-facing slopes (Daubenmire 1970; Johnson and Simon 1987). Also, at zones of contact between major vegetation zones, more mesic zones extend into drier zones as topographic climaxes (Daubenmire 1970).

Although more pronounced on steep slopes, the mesic conditions created on northerly aspects are evident even on low slope gradients and microtopographic features as well. Large occurrences of *Silene spaldingii* in Idaho Canyon Grasslands are commonly restricted to the northerly slopes of low hills in Redemsky Flats, an area of low elevational gradient on rolling benchland terrain at the Garden Creek Ranch (Hill and Gray 1999; J. Hill, *in litt.* 2003a 2003b; Menke 2003; Lichthardt and Gray 2002). *Silene spaldingii* in Channeled Scablands is often associated with the northerly aspects of the mounds in biscuit and swale topography (F. Caplow, personal communication, 2003) or on the north slopes of swales or depressions (B. Benner, *in litt.* 1993).

In more mesic areas, i.e., Palouse Grasslands, Wallowa Plateau, Intermontane Valleys, and more mesic portions of the Channeled Scablands, where annual precipitation is sufficient to support mesic fescue communities as climatic climaxes (Daubenmire 1970; Franklin and Dyrness 1973), northerly aspects may not be as important distribution factors for *S. spaldingii*. The Idaho fescue-snowberry and sagebrush-fescue habitat types that commonly support *S. spaldingii* in these areas often occurs on southerly aspects or flat ground as well as northerly aspects (see plot data in Daubenmire 1970). Although known occurrences of *S. spaldingii* are commonly associated with northerly aspects in the Palouse Grasslands (WNHP 2003b; IDCDC 2003b), this may be due to the steepness of the slope that has prevented cultivation and conversion to cropland rather than a species distribution factor (Heidel 1980). Vegetation differences are evident with varying aspect in Palouse Grasslands also, but many northerly aspects are moist enough to support shrub thickets and small trees (Aller et al. 1980; Lichthardt and Moseley 1997b). *Silene spaldingii* appears to not tolerate dense shade of shrub thickets (B. Benner, *in litt.* 1993); competition from woody plants likely reduces reproduction of many prairie species (Menges 1995).

*Silene spaldingii* is often associated with a other topographic terrain features that tend to accumulate and/or retain soil moisture, i.e., low-lying areas, talus or basalt outcrops, particularly in the drier portions of its range. Moisture tends to accumulate in low-lying areas such as shallow swales, ravines, depressions, or slope bottoms. Talus areas often have good moisture storage beneath them where water evaporation is minimal and accessible to vegetation surrounding the talus (Johnson and Simon 1987). Deep-rooted plant species like *S. spaldingii* are particularly adapted to access these subsurface water sources. In areas of layered basalt, water accumulates within the basalt layers during the wetter time periods and seepage between the layers can slowly release moisture well into the growing season (Johnson and Simon 1987; Johnson 1994).

### **Physiographic Regions and Specific Habitat Features**

*Silene spaldingii* occurs in distinct physiographic regions across its range including: 1) Palouse Grasslands of eastern Washington and adjacent Idaho, 2) Canyon Grasslands of the Snake, Salmon, Clearwater, Imnaha, and Grande Ronde rivers in Washington, Idaho and Oregon, 3) Channeled Scablands of east-central Washington, 4) Wallow Plateau in northeastern Oregon, and 4) Intermontane Valleys of northwestern Montana and adjoining British Columbia (Figure 3A – Appendix A). Each of these regions consists of relatively homogeneous physical features that distinguish one region from the other. *Silene spaldingii* is associated with specific physical habitat features within each of these regions.

#### **Palouse Grasslands**

The term “Palouse” has been used to describe not only grasslands in the Palouse Region of southeastern Washington and adjacent Idaho, but also grasslands in Utah, Montana and British Columbia (Stoddard 1941; Caldwell 1961; Tisdale 1983; Lichthardt and Moseley 1997a). For purposes of this report, Palouse Grasslands refers to those grasslands in the Palouse Region of southeastern Washington and adjacent Idaho (Figure 3A – Appendix A).

In Washington, the Palouse Grasslands are bordered on the west and north by contact with the Channeled Scablands (Caldwell 1961), and on the south by the Blue Mountains and the Grande Ronde River. In Idaho, the Palouse Grasslands are bordered on the north and east by contact with the Clearwater Mountains and the Clearwater River, and on the west and south by the Snake and Salmon rivers (Ertter and Moseley 1992; Lichthardt and Moseley 1997a). (See Figure 3A – Appendix A).

The majority of the Palouse Grasslands occurs in southeastern Washington and is referred to as the "Palouse Hills" (Gamon 1991); this area covers the majority of Whitman and Asotin counties and portions of Spokane, Columbia and Walla Walla counties (Figure 3A – Appendix A). The Palouse Grasslands in Washington correspond to the "true steppe" (*Agropyron-Festuca* vegetation zone) and "meadow steppe" (*Festuca – Symphoricarpos* and *Festuca-Rosa* vegetation zones) described by Daubenmire (1942, 1970) (Figure 10A – Appendix A). The Palouse Grasslands in Idaho include "The Palouse" north of the Clearwater River, which includes portions of Latah and Benewah counties, and the "Camas Prairie" south of the Clearwater River (Lichthardt and Moseley 1997a), which includes the majority of Lewis and Nez Perce counties and portions of Idaho county (Figure 3A – Appendix A).

The Palouse Hills of Washington and The Palouse of Idaho consist of gently rolling hills of loess historically deposited by wind from outwash deposits in south-central Washington. The hills have a dune-like form with a southwest/northeast alignment. This has resulted from a combination of factors including prevailing southwesterly winds, differential erosion by precipitation, headward erosion by snow accumulations on northerly slopes during winter, and mass movement (slumping) during the wet season (Aller et al. 1981). Occasionally taller granitic hills, "steptoes", protrude above these dunes. Idaho received the most loess, with soil depths up to 150 feet (Ross and Savage 1967). The Camas Prairie was not in the main path of loess deposition, and consequently the soils are shallower and lack the dune-like appearance of The Palouse and the Palouse Hills (Barker et al. 1983). The Camas Prairie is higher in elevation than The Palouse and is correspondingly moister and cooler (Ertter and Moseley 1992).

Elevations of *S. spaldingii* occurrences within the Palouse Grasslands of Washington vary from 2,300-2,700 feet in the primary area of the Palouse Hills to 2,900 feet on slopes of Steptoe Butte and 3,200 feet in the foothills of the Blue Mountains (WNHP 2003b). In the Palouse Grasslands of Idaho, elevations range from 2,600-2,680 feet on The Palouse to 3,560-4,400 feet on the Camas Prairie (IDCDC 2003b).

Due to the deep, fertile soils of the Palouse Grasslands, large portions of this region has been converted to cropland (Kaiser 1961, Caldwell 1961; Tisdale 1961; Ertter and Moseley 1992), and Palouse Grasslands are considered a critically endangered ecosystem (Noss et al. 1995). Only remnants of the former native prairie remain, and they are surrounded by large expanses of cultivated fields. *Silene spaldingii* occurrences in this region are restricted to small remaining patches of native grassland that occur as: 1) "eyebrows" on northerly slopes too steep to plow, 2) small strips at corners of fields that have escaped cultivation, 3) remnants preserved in cemeteries, 4) areas interfingering into forest zones (Blue Mountain

foothills), 5) areas near exposures of underlying basalt near creek channels, and 6) open slopes of some of the steptoes (IDCDC 2003b; WNHP 2003b; R. Crawford, *in litt.* 2003).

Habitat types in which *S. spaldingii* is known to occur in Palouse Grasslands include: 1) Idaho fescue-snowberry (*Festuca idahoensis-Symphoricarpos albus*) [FEID-SYAL] (Daubenmire 1970), 2) Idaho fescue-rose (*F. idahoensis-Rosa* spp.) [FEID-Rosa] (Daubenmire 1970), 3) bluebunch wheatgrass-Idaho fescue (*Agropyron spicatum-Festuca idahoensis*) [AGSP-FEID] (Daubenmire 1970), and 4) Idaho fescue-bluebunch wheatgrass (*Festuca idahoensis-Agropyron spicatum*) [FEID-AGSP] (Tisdale 1986; Simon and Johnson 1987). (See descriptions of these habitat types in the Biological Features section that follows).

The Palouse Grasslands are included in the Palouse Dry Steppe ecoregion described by Bailey (1995), the Palouse Hills and Tri-State Uplands Sections of the Columbia Intermontane Province described by Ross and Savage (1967), the Columbia Basin and Blue Mountains Provinces described by Franklin and Dyrness (1973), and the Palouse Unit described by Ertter and Moseley (1992).

### **Canyon Grasslands**

Canyon Grasslands occur on slopes of deep river canyons associated with the major river systems in the tri-state area. This area includes the Imnaha River in Oregon, the Grande Ronde River in Washington and Oregon, the Clearwater River in Idaho to Kooskia, the Lower Salmon River in Idaho to Riggins, and the portion of the Snake River from Brownlee Dam in Idaho and Oregon to the Little Goose Dam in Washington (Tisdale 1986a) (Figures 3A and 9A – Appendix A). Currently, *S. spaldingii* is known only from Canyon Grasslands along the Snake and lower Salmon Rivers in the area of Craig Mountain, Idaho, and along the Snake River breaks in upper Wawawai Canyon in Washington, i.e., EOR WA 009 – Upper Wawawai.

Erosive down-cutting by these large rivers carved through the many layers of the Columbia River Basalt Flows and into the older rock formations below them, creating highly dissected canyon slopes with high elevational gradient and often abrupt changes in aspect (Tisdale 1986a; Johnson and Simon 1987; Mancuso 1993; Mancuso and Moseley 1994). Aspect and associated differences in moisture and soil properties appear to be the overriding factor in determining natural community patterns across the Canyon Grassland landscape (Mancuso 1993). Southerly and northerly aspects at the same elevation have very different soils and vegetation types. At the highest elevations on Craig Mountain (~5,200 feet), coniferous forests occur on northerly slopes and grasslands occur on southerly slopes. At low to mid-elevations (~1,300-3,800 feet), northerly slopes support the mesic fescue grasslands in which *S. spaldingii* is known to occur, while southerly slopes support xeric bluebunch wheatgrass grasslands. Because of the highly dissected terrain of the canyons, the xeric bluebunch wheatgrass and the mesic Idaho fescue communities occur as intermixed patches across the landscape (Hill and Fuchs 2002). The mesic Idaho fescue sites capable of supporting *S. spaldingii* occur with regularity across the landscape, but vary in size and distance from each other from as little as 20 feet to several hundred feet (Hill and Fuchs 2002).

Known *S. spaldingii* occurrences in the Canyon Grasslands are restricted to northerly aspects, i.e. northwest to north to northeast, between 1,380 and 4,000 feet, on slope gradients of 5-70%, and on all slope positions (Hill and Gray 1999; Lichthardt and Gray 2002; J. Hill, *in litt.* 2003a, 2003b; Menke 2003; IDCDC 2003b; WNHP 2003b). The lowest known elevation for *S. spaldingii*, 1,380 feet, in Canyon Grasslands (and rangewide) occurs along the lower Salmon River, and the highest elevation in Canyon Grasslands, ~4,000 feet, occurs on Garden Creek Ranch on Craig Mountain (IDCDC 2003b; K. Gray, *in litt.* 2003a). Although present on steep northerly slopes, many of the known *S. spaldingii* sites in Canyon Grasslands are located on low slope gradients [i.e., Redemsky Flats at Garden Creek Ranch (Hill and Gray 1999; Hill and Gray 2000; Hill et al. 2001)], similar to those in other portions of its range, i.e., Palouse Grasslands in Washington and Idaho, Dancing Prairie in Montana, or Zumwalt Prairie in Oregon. Due to the rugged terrain and difficult access, Canyon Grasslands are contiguous and intact with minimal human-induced fragmentation; however, much of the region has been seriously degraded by weed invasion.

Habitat types and plant associations that are known to support *S. spaldingii* in Canyon Grasslands include: 1) Idaho fescue-snowberry (Daubenmire 1970), 2) Idaho fescue-rose (Daubenmire 1970), and 3) Idaho fescue-prairie junegrass (*F. idahoensis-Koeleria cristata*) [FEID-KOCR] (Tisdale 1986; Johnson and Simon 1987). (See descriptions of these habitat types in the Biological Features section that follows).

The Canyon Grasslands are included in the Tri-State Uplands Sections of the Columbia Intermontane Province described by Ross and Savage (1967), the Middle Rocky Mountain Steppe ecoregion described by Bailey (1995), the Wallowa-Snake Province described by Johnson and Simon (1987), the Canyon Grasslands described by Tisdale (1986a), and Canyons Unit described by Ertter and Moseley (1992).

### **Channeled Scablands**

The catastrophic floods that cut through the loess and underlying basalt in east-central Washington created the Channeled Scablands, a complex of braided channels, shallow, poorly drained coulees, basalt mesas, isolated islands of loess, and gigantic gravel bars and other flood deposits (Mueller and Mueller 1997). The course of the floods split into three main channels, the easternmost Cheney-Palouse Channel, the middle Crab Creek Channel, and the westernmost Grand Coulee Channel (U.S. Department of the Interior (USDI) 1974; Mueller and Mueller 1997). *Silene spaldingii* occurrences are known only from the Crab Creek and Cheney-Palouse Channels (WNHP 2003b).

A variety of terrain features occur in the Channeled Scablands, and habitat features for *S. spaldingii* in this region are varied as well. The majority of *S. spaldingii* occurrences in this region are associated with the "channels" (B. Benner, personal communication, 2003b). The western-most extension of *S. spaldingii*'s range in the Channeled Scablands (and rangewide) occurs in the Crab Creek Channel. In this area, *S. spaldingii* occurs on long continuous expanses of north-facing slopes of the Coal Creek and Crab Creek coulees and "eyebrows" at Seven Springs. Occurrences in this area are often located just above and beside (but rarely

below) edges of talus patches (B. Benner, *in litt.* 1993). At Miller Ranch in the Cheney-Palouse Channel, *S. spaldingii* occurs on gentle north-facing slopes, north-facing slopes below rock outcrops, above depressions, or on mounds in biscuit and swale topography (B. Benner, *in litt.* 1993; WNHP 2003b).

A common feature in both flooded and unflooded scablands is biscuit and swale topography, consisting of small biscuit-like mounds of loess that are regularly dispersed over the scabland surface (Daubenmire 1970). Much of the upland portion of the Channeled Scablands consists of biscuit and swale topography (Gamon 1991). *Silene spaldingii* in biscuit and swale topography occurs on the loess mounds rather than the shallow soils of scabland swales between the mounds (B. Benner, *in litt.* 1995). At Fairchild Air Force Base (AFB), *S. spaldingii* typically occurs on the north sides of these mounds (Caplow 2002a; F. Caplow, personal communication, 2003).

At Turnbull Wildlife Management Area, *S. spaldingii* occurs in native fescue grass understories in open-canopy ponderosa pine (*Pinus ponderosa*) stands (Weddell 2002; Caplow 2002b). At the Swanson Lakes Wildlife Area, *S. spaldingii* occurs on north-facing slopes of basalt outcrops (photos), often between drier rigid sagebrush (*Artemisia rigida*) communities on top of the outcrop and a weedy grass/forb community below (Weddell 2002; Caplow 2002b).

The lowest elevation for *S. spaldingii* in Channeled Scabland, 1,550 feet, occurs at Rock Creek in Adams County. The highest elevations for *S. spaldingii* in Channeled Scabland, 2,450 feet, are in the vicinity of Fairchild AFB at Strangland and Ladd Roads, Spokane County (WNHP 2003b). Northerly aspects are common for *S. spaldingii* occurrences in many portions of the Channeled Scablands (B. Benner, *in litt.* 1993, 1995; Caplow 2002a; 2002b; Weddell 2002). Aspect and soils appear to be natural distribution factors for *S. spaldingii* in Channeled Scablands, where topographic climaxes of mesic steppe types can occur on northerly slopes and loessal soils within drier steppe zones (Daubenmire 1970). Slope gradients are variable. On BLM lands, *S. spaldingii* occurrences in Lincoln and Spokane counties range from flat to over 55% with an average around 35% (B. Benner, *in litt.* 1993, 1995).

Habitat types that are known to support *S. spaldingii* in Channeled Scablands include: 1) Sagebrush-Idaho fescue [*Artemisia tridentata*-*Festuca idahoensis* (ARTR-FEID)]; *A. tripartita*-*F. idahoensis* (ARTR2-FEID)] (Daubenmire 1970), 2) ponderosa pine types [*Pinus ponderosa*-*Symphoricarpos albus* (PIPO-SYAL)]; *P. ponderosa*-*F. idahoensis* (PIPO-FEID)] (Daubenmire 1968b; Cooper et al. 1991), 3) Idaho fescue-snowberry (Daubenmire 1970), and 4) bluebunch wheatgrass-Idaho fescue (Daubenmire 1970). (See descriptions of these habitat types in the Biological Features section that follows).

The Channeled Scablands are included in the Columbia Basin and Blue Mountains Provinces described by Franklin and Dyrness (1973) and in the Intermountain Semi-desert ecoregion described by Bailey (1995).

## **Wallowa Plateau**

On the high, dissected basalt plateau of the Wallowa Valley and Zumwalt Prairie in Wallowa County, northeastern Oregon, *S. spaldingii* is located on remaining occurrences of the native bunchgrass prairie. The area where *S. spaldingii* is known to occur is bordered on the north and east by Joseph Canyon and the Wallowa National Forest, on the south by the Wallowa Mountains and on the east by the Imnaha River. *Silene spaldingii* is located on terminal moraines created by glaciers from the Wallowa Mountains that flank Wallowa Lake. It also occurs in the Chief Joseph Monument area between the moraines at their northern end. *Silene spaldingii* occurs on north slopes of both moraines and major ridgetops in this region in deep loess mixed with glacial till soils (ONHP 2003b). At Clear Lake Ridge, *S. spaldingii* occurs on lower to upper, gentle to moderately steep slopes (Elseroad and Taylor 2002b). In the Crow Creek area at the head of Joseph Canyon, *S. spaldingii* is located on low slopes of broad ridges and ridgebrows, some of which support biscuit and swale topography. On the broad summit of one ridge in the Crow Creek area, *S. spaldingii* occurs in wetter microsites in a dry meadow (ONHP 2003b). On the elevated regions of Zumwalt Prairie, *S. spaldingii* occurs on deep soils in swales occurring in otherwise flat land and on slopes (bottom to midslope positions) with aspects ranging from 5-255° and slopes ranging from 1-10% (Elseroad and Taylor 2002a; R. Taylor, *in litt.* 2003a). In the Wallowa Valley, where most of the original bunchgrass prairie has been converted to agricultural purposes or residential/urban areas, *S. spaldingii* occurs on small isolated prairie remnants. The highest elevation on the Wallowa Plateau (and rangewide), 5,100 feet, occurs on Clear Lake Ridge, and the lowest elevation, 3,700 feet, occurs in the Crow Creek area (ONHP 2003b; Elseroad and Taylor 2002a, 2002b).

Habitat types and plant associations known to support *S. spaldingii* on the Wallowa Plateau include: 1) Sagebrush-Idaho fescue (*Artemisia ludoviciana-Festuca idahoensis*) (not described), 2) Idaho fescue-snowberry (Daubenmire 1970), and 3) Idaho fescue-prairie junegrass (ridgetop) (Johnson and Simon 1987). (See descriptions of these habitat types in the Biological Features section that follows).

The plateau grasslands of the Wallowa Valley and Zumwalt Prairie merge on the east and north with extensive areas of Canyon Grasslands associated with the Imnaha, Snake and Grande Ronde Rivers. Currently, no occurrences of *S. spaldingii* have been reported from these Canyon Grasslands in Oregon.

The Wallowa Plateau is included in the Wallowa-Snake Province described by Johnson and Simon (1987), the Blue Mountain Province described by Franklin and Dyrness (1973), and the Middle Rocky Mountain Steppe ecoregion described by Bailey (1995).

## Intermontane Valleys

In northwestern Montana and adjoining British Columbia, *S. spaldingii* is known from the deep valleys between the Purcell Mountains and Salish Mountains on the west and the Whitefish Range on the east. These intermontane valleys were associated with glacial activity of the continental ice sheet. They are located in the overthrust belt area of the Rocky Mountain Trench that was filled by the Flathead Lobe of the continental glacier (Alt and Hyndman 1986).

Previous to the 2002 field season, occurrences of *S. spaldingii* were known from two regions about 80 miles apart, a northern group in the Tobacco Plains area and a southern group in the Niarada/Flathead Lake area. The northern Tobacco Plains area is located in a valley that was glaciated and the terrain is typically rolling kettle and moraine topography, with *S. spaldingii* occurring in the bottom of shallow swales, on cool slope exposures, and ridgetops with relatively deep soils (Lesica 1991b, 1997a, 1999; B. Martin, *in litt.* 2003). Within the large Dancing Prairie population on Tobacco Plains (EORs MT 001 and 008), *S. spaldingii* occurs in patches separated by approximately 50 to 100 meters (Lesica 1993). The southern Niarada/Flathead Lake area is located in intermontane valleys that were once covered by glacial Lake Missoula. *Silene spaldingii* occurs in the Niarada area on low to moderate slopes, bottoms of draws, and in or along small drainages (without running water) often along the lower treeline, or near scattered trees, and at Wild Horse Island in Flathead Lake on northwest-facing slopes in gravelly silt-loam soils (MNHP 2003b). During the 2002 field season, a large population of *S. spaldingii* was located at Lost Trail NWR that occurs in intermontane valleys roughly midway between the northern Tobacco Plains area and the southern Niarada/Flathead Lake area (M. Mantas, *in litt.* 2003). *Silene spaldingii* occurs in rolling swales in this area (L. Garner, personal communication, 2003).

The lowest elevation of known *S. spaldingii* occurrences in the Intermontane Valleys of Montana, 2,700 feet, occurs in the Tobacco Plains area, and the highest elevation, 3,800 feet, occurs in the Hog Heaven Range in the Niarada area. Aspects vary from no aspect (flat) to northwest to north to northeast to east (Lesica 1991b; MNHP 2003b).

Habitat types known to support *S. spaldingii* in Intermontane Valleys of northwestern Montana and adjoining British Columbia include rough fescue types, *Festuca scabrella*-*F. idahoensis* (FESC-FEID) and *F. scabrella*-*Agropyron spicatum*/*Stipa comata* (FESC-AGSP/STCO) (Mueggler and Stewart 1980). (See descriptions of these habitat types in the Biological Features section that follows).

The Intermontane Valleys of Montana are included in the Canada Province within the Rocky Mountain System described by Hunt (1974) and the Middle Rocky Mountain Steppe ecoregion described by Bailey (1995).

## Climate

In general, the climate across the range of *S. spaldingii* is characterized by warm to hot, dry summers and cool to cold, moist winters. Major influences on the climate within *S. spaldingii* range are the Cascade Mountains, the Rocky Mountains, and the Pacific Ocean (Western Regional Climate Center 2003a). As westerly-moving moist air from the ocean crosses the Cascade Mountains, major releases of moisture occur along western slopes. Warming and drying of the air as it descends along the eastern slopes creates a strong rain shadow that makes the western Columbia Basin very arid. The most western known occurrence of *S. spaldingii* is located near Odessa, Washington, which has a mean annual precipitation of 10.4 inches (Western Regional Climate Center 2003b). As the floor of the Columbia Basin rises eastward toward the Rocky Mountains, orographic lifting of the air results in a gradual increase in annual precipitation, reaching approximately 24 inches at Moscow in the Idaho Palouse Grasslands (Western Regional Climate Center 2003b). The orographic effect of the northern Rocky Mountains creates the fescue grassland zones that support *S. spaldingii* in the tri-state region of its distribution in eastern Washington, northeast Oregon, and adjacent Idaho (R. Crawford, *in litt.* 2003). Air masses continue to move eastward, releasing moisture on the western slopes of the Rocky Mountains, and creating small rain shadows in intermontane valleys that transect their path, such as Tobacco Plains near Eureka and the Little Bitterroot Valley near Niarada, Montana. The low annual precipitation of these valleys, i.e., from 11 inches at Lone Pine to 15 inches at Eureka and Polson (Western Regional Climate Center 2003b), is largely responsible for the development of the fescue grasslands that support *S. spaldingii* (R. Crawford, *in litt.* 2003).

Winter climate is noticeably influenced by the moderating effect of maritime air from the Pacific Ocean that is borne eastward by prevailing westerly winds (Tisdale 1983; Johnson and Simon 1987). A high proportion (45-65%) of the precipitation occurs during the winter months (Daubenmire 1942; Mueggler and Stewart 1980). The mild, moist winters in the tri-state area permit intermittent photosynthesis and slow growth of many plants during the winter months (Weaver 1958; Daubenmire 1970; Tisdale 1983). The phenologically active state of Pacific Northwest Bunchgrass communities in response to high winter precipitation and mild winter temperatures in the tri-state area of *S. spaldingii* occurrence contrasts sharply with all North American grasslands east and south of the Northern Rockies (Daubenmire 1942; Mack 1986). Summers typically have low relative humidity, rapid evaporation, abundant sunshine, and a period of intense drought during the mid- and late summer when minimum precipitation and maximum temperatures occur (Tisdale 1983). Because abundant moisture is available during the cooler months and very little in the warmer months, many native bunchgrasses and forbs in *S. spaldingii* communities are dormant in summer but photosynthetically active from autumn through spring (Weaver 1958; Daubenmire 1970; Tisdale 1983).

Regional variations in this general climatic pattern reflect the great differences in elevation, geography, and topography across the range of *S. spaldingii*. Climate in the Canyon Grasslands is a semi-arid, Mediterranean climate with hot, dry summers and mild winters with limited and ephemeral snow in canyon bottoms (Tisdale 1986a). There are marked differences in annual precipitation and temperature with changes in elevation. In general,

precipitation increases approximately 5 inches with each 1,000 foot rise in elevation (Johnson and Simon 1987). At Craig Mountain, weather stations at 4,760 feet and 1,040 feet recorded 26.5 and 13.4 inches annual precipitation, respectively (Mancuso and Moseley 1994). Based on these figures, annual precipitation of approximately 15-23 inches might be expected at elevations from 1,400 to 4,000 feet where *S. spaldingii* is known to occur on Craig Mountain, Idaho. Mean summer temperatures below 3,000 feet range from 80-90 °F and at the lowest elevations (ca. 800 to 900 feet) daily maximums often exceed 100 °F (Johnson and Simon 1987). In the intermontane valleys of northwestern Montana and British Columbia, a Canadian climate with snowy, cold winters and moderately warm summers prevails (Visher 1954). In the Wallowa Valley, winter months are colder than the Columbia Basin due to drainage of cold air from surrounding mountain slopes (Johnson and Simon 1987).

The pattern of precipitation and temperature through the season differ across the range of *S. spaldingii* distribution. Long-term precipitation and temperature data (Western Regional Climate Center 2003b) from various locations across *S. spaldingii*'s range were utilized to create climate diagrams that are used to show relationships between soils, vegetation and climate (Walter and Lieth 1967) (Climate Diagrams – Appendix C). Monthly average temperature and precipitation were plotted together on a scale where 10 °C temperature corresponds to 20 mm of precipitation (converted to °F and inches). The area where the precipitation line dips below the temperature line indicates dry season and water shortage for plants, and the area where the precipitation line is above the temperature line indicates moist season. Table 8 below shows annual mean temperature, annual mean precipitation, percent of precipitation during the growing season for *S. spaldingii* (i.e., May, June, July, August), and the season of the year with the highest amount of precipitation for a number of locations across the range of *S. spaldingii*.

Annual temperatures range from a low of 43.5 °F at Enterprise, Oregon, on the Wallowa Plateau to a high of 54.6 °F at Wawawai, Washington, in Canyon Grasslands. Annual precipitation ranges from a low of 10.4 inches at Odessa, Washington, in Channeled Scablands to a high of 23.5 inches at Moscow, Idaho, in Palouse Grasslands. The percentage of precipitation occurring during the growing season of *S. spaldingii* ranges from a low of 19% at Wawawai, Washington, to a high of 44% at Eureka, Montana. At over half of the sites, winter is the season with the highest amount of precipitation; this feature characterizes all the sites in Washington. The annual precipitation is relatively low at locations in Montana, although the percentage of precipitation occurring during the growing season is relatively high.

The climate diagrams demonstrate some major differences in seasonality of precipitation between physiographic regions. In general, areas in the Channeled Scablands and Palouse Grasslands show winter-high and summer-low precipitation patterns while the Intermontane Valley areas of Montana show winter-low and summer-high precipitation patterns. For example, the annual precipitation is very similar for Sprague, Washington (14.6 inches), and Eureka, Montana (14.5 inches); however percentage of precipitation occurring during the growing season for *S. spaldingii* is 22% at Sprague and 44% at Eureka.

**Table 8 – Climatic Variables and Seasonality within *Silene spaldingii* Range.**

Location	Annual Mean Temp. (F)	Annual Mean Precip. (in)	% Precip. Growing Season	Winter High Precip.	Spring High Precip	Summer High Precip.	Fall High Precip.
<b>Idaho</b>							
Moscow	47.3	23.5	22	X			
Craigmont	43.9	21.6	34		X		
Lewiston	52.4	12.7	33		X		
<b>Montana</b>							
Eureka	45.2	14.5	44			X	
Lonepine	45.2	11.2	34				X
Polson	45.7	15.3	41			X	
<b>Oregon</b>							
Enterprise	43.5	13.1	40		X		
<b>Washington</b>							
Anatone	44.5	20.4	29	X			
Wawawai	54.6	17.4	19	X			
Pullman	47.0	21.4	21	X			
Rosalia	47.0	17.8	24	X			
Spokane	48.0	16.1	24	X			
Sprague	47.6	14.6	22	X			
Odessa	48.8	10.4	23	X			

Some minor differences in annual precipitation and temperature and seasonality of precipitation are associated with the three primary vegetational zones of the Washington steppe area where *S. spaldingii* occurs, i.e., meadow steppe (FEID-SYAL; FEID-*Rosa*), true steppe (AGSP-FEID), and shrub steppe (ARTR-FEID; ARTR2-FEID) (Figure 10A – Appendix A). Areas supporting meadow steppe have higher annual precipitation and receive more of the rainfall in autumn than the shrub steppe areas. These climatic differences affect the climax structure of the communities chiefly in the shrubby elements, with sagebrush and Wyeth's buckwheat (*Eriogonum heracleoides*) associated with the dry autumn climate and snowberry and rose associated with wet autumn climate. The meadow steppe is cooler and wetter than the true steppe zone, and the true steppe is both wetter and warmer than the shrub steppe. Microclimatic differences are also associated with aspect. Topographic climaxes of mesic steppe types occur on northerly aspects within drier steppe zones. (Adapted from Daubenmire 1970).

Although annual precipitation levels are similar, the low humidity and low precipitation occurring during the growing season makes the Pacific Northwest Bunchgrasses in the tri-state area ecologically more arid than the Great Plains (Stoddart 1941). The low percentage of precipitation occurring during the growing season in Pacific Northwest Bunchgrass areas is seldom sufficient to be of benefit to growing plants (Stoddart 1941). Most plants in the tri-state area of *S. spaldingii* occurrence depend on stored soil moisture during the growing season. Soil in this area is similar to a large reservoir replenished during fall, winter and spring but thoroughly emptied of its water during summer. In early summer the entire soil to

a depth of 5 or more feet gradually loses most of its available supply of water to transpiring plants (Weaver 1958). In the Great Plains, a larger percentage of the annual precipitation occurs during the growing season and is of immediate value to the vegetation (Stoddart 1941). Climate of Intermontane Valleys of northwestern Montana are more similar to that of the Great Plains with relatively high precipitation occurring during the growing season.

## BIOLOGICAL FEATURES

### Plant Communities

*Silene spaldingii* occurs in grassland, shrub, and forest habitat types across its range. Regional botanical references indicate *S. spaldingii* occurs in "sagebrush, scabland, or woodland flora" (Hitchcock and Maguire 1947; Hitchcock et al. 1964). Herbarium records indicate that it has been found in sagebrush plains, grasslands, and open-canopy ponderosa pine (Heidel 1979). The associated perennial bunchgrass, forb, shrub and tree species vary across the range of *S. spaldingii*; however, the presence of a native fescue (*Festuca*) bunchgrass species is a consistent feature among all habitat types.

In the Palouse Grasslands, Canyon Grasslands, Wallowa Plateau, and Intermontane Valleys, *S. spaldingii* is associated with mesic fescue grasslands of the Pacific Northwest Bunchgrass Grasslands (Tisdale 1983). These grasslands are composed of two major bunchgrass series, the drier bluebunch wheatgrass (*Agropyron spicatum* = *Pseudoroegneria spicata*) series and the more mesic fescue (*Festuca*) series. *Silene spaldingii* is confined to the fescue series that occupies the cooler and moister sites in the grassland zone at higher elevations, and northerly aspects at lower elevations. Within the tri-state area of occurrence, *S. spaldingii* is associated with Idaho fescue (*F. idahoensis*), and in the disjunct northwestern Montana area of occurrence, *S. spaldingii* is associated with rough fescue (*F. scabrella*). The structure of these grassland communities consists of: 1) domination by perennial bunchgrasses of which a fescue (*Festuca*) species is always a major component, 2) a varied and conspicuous forb component, 3) often, but not always, a shrub component, occurring as scattered, dwarfed individuals or taller patches/colonies, 4) a well developed cryptogam layer, and 5) occasionally scattered tree species (Daubenmire 1970; Mueggler and Stewart 1981; Tisdale 1986a; Johnson and Simon 1987).

In the Channeled Scablands, *S. spaldingii* also occurs in shrub and forest plant communities including sagebrush-fescue and open-canopy pine-fescue communities. The fescue associations in these shrub- and tree-dominated communities are very similar to the mesic fescue grassland habitat types of the Pacific Northwest Bunchgrasses. Structure of the sagebrush-fescue communities includes: 1) domination by irregularly scattered sagebrush species, creating a taller conspicuous layer in the big sagebrush type and an inconspicuous layer in the shorter three-tip sagebrush type, 2) high cover of perennial bunchgrasses of which Idaho fescue is always a major component, 3) varied and conspicuous forb component, and a 4) well-developed cryptogam layer. Structure of the forest communities in which *S. spaldingii* occurs is similar to the fescue grasslands described above with a dominant tree layer superimposed (Daubenmire 1968b).

Primary Pacific Northwest Bunchgrass grassland habitat types include: 1) Idaho fescue-snowberry, 2) Idaho fescue-rose, 3) Idaho fescue-prairie junegrass, 4) bluebunch wheatgrass-Idaho fescue; Idaho fescue-bluebunch wheatgrass, and 5) rough fescue types (Daubenmire 1970; Mueggler and Stewart 1981; Tisdale 1986; Johnson and Simon 1987). Primary shrub habitat types include: 1) big sagebrush-Idaho fescue and 2) three-tip sagebrush-Idaho fescue (Daubenmire 1970). Forest habitat types include: 1) ponderosa pine-Idaho fescue and 2) ponderosa pine-snowberry (Daubenmire 1968b; Cooper et al. 1991). The majority, 73%, of *S. spaldingii* occurrences are within grassland habitat types. Of the remaining occurrences, 20% are within shrub habitat types and 7% within forest habitat types (Table 13A – Appendix B) (IDCDC 2003b; MNHP 2003b; ONHP 2003b; WNHP 2003b).

### **Grassland Habitat Types**

Grassland habitat types are climatically determined, occurring in areas with low annual precipitation, often less than 15 inches (Habeck 1967), and intense summer drought. In the tri-state area of *S. spaldingii* occurrence, soils dry out to a depth of 50 cm during the summer drought period. This is too deep for the establishment of conifer trees, even the drought-tolerant ponderosa pine (Daubenmire 1968a). Heavy utilization of moisture by grass roots reduces water content from the surface downward as a result of direct evaporation plus transpiration. The taproots of conifer tree seedlings, which start growth in the spring, are unable to descend rapidly enough to keep ahead of the advancing soil moisture depletion. Idaho fescue and bluebunch wheatgrass start growth with fall rains, and continue to grow intermittently during intervals of above-freezing weather throughout winter. They have developed enough to endure the period of enforced aestivation during the summer drought period (Daubenmire 1968a). Scattered ponderosa pine can establish in some of the grassland habitat types in moister microsites, forest/grassland ecotones, or during exceptionally wet years; however, a period of low rainfall and high evapotranspiration during the hottest months prevents the establishment of full, dense forest (Daubenmire 1978; Conner et al. 2001).

#### Idaho fescue-snowberry

The Idaho fescue-snowberry (*Festuca idahoensis-Symphoricarpos albus*) [FEID-SYAL] habitat type (Daubenmire 1970) is a mesic bunchgrass grassland type with an inconspicuous dwarf shrub element. Idaho fescue and bluebunch wheatgrass are the dominant bunchgrasses along with lesser amounts of prairie junegrass (*Koeleria cristata* = *K. macrantha*). The bluebunch wheatgrass in this type has a rhizomatous habit. The dwarf shrub element, which includes both snowberry (*Symphoricarpos albus*) and rose species (*Rosa nutkana* and *R. woodsii*), can occur as scattered, solitary stems that do not project above the bunchgrasses or as taller thickets in a mosaic within the grassland. Occasionally, individual taller shrubs of black hawthorn (*Crataegus douglasii*), serviceberry (*Amelanchier alnifolia*), and/or chokecherry (*Prunus virginiana*) may occur. The community also includes a diverse and conspicuous perennial forb component and the ground is covered by lichens and mosses.

Idaho fescue-rose: The Idaho fescue-rose (*Festuca idahoensis-Rosa* spp.) [FEID-Rosa] habitat type (Daubenmire 1970) is similar to the Idaho fescue-snowberry habitat type except snowberry is absent.

Both the Idaho fescue-snowberry and the Idaho fescue-rose habitat type are representative of the meadow steppe described by Daubenmire (1970) (Figure 10A – Appendix A). These habitat types are located in Palouse Grasslands, Canyon Grasslands, and the Wallowa Plateau (Figure 3A – Appendix A).

#### Idaho fescue-prairie junegrass

The Idaho fescue-prairie junegrass (*Festuca idahoensis-Koeleria cristata*) [FEID-KOCR] plant association described by Johnson and Simon (1987) and habitat type described by Tisdale (1986a) is a mesic bunchgrass grassland type **without** a shrub element. Both the "low elevation" and "ridgetop" types of this plant association (Johnson and Simon 1987) support *S. spaldingii* occurrences (IDCDC 2003b; ONHP 2003b). The "low elevation" type is common in Canyon Grasslands of Idaho (Figure 3A – Appendix A), occurring below 4,000 feet on northerly facing slopes. The "ridgetop" type is common on the Wallowa Plateau of northeastern Oregon (Figure 3A – Appendix A). Both types have high cover of Idaho fescue and bluebunch wheatgrass with lesser amounts of prairie junegrass. Idaho fescue typically dominates. Both types have a rich perennial forb component.

#### Bluebunch wheatgrass-Idaho fescue; Idaho fescue-bluebunch wheatgrass

The bluebunch wheatgrass-Idaho fescue (*Agropyron spicatum-Festuca idahoensis*) [AGSP-FEID] habitat type (Daubenmire 1970) and the Idaho fescue-bluebunch wheatgrass (*Festuca idahoensis-Agropyron spicatum*) [FEID-AGSP] plant association (Tisdale 1986; Johnson and Simon 1987) are similar. The principal bunchgrasses in this drier grassland type are bluebunch wheatgrass and Idaho fescue; bluebunch wheatgrass often dominates. Lesser amounts of Sandberg's bluegrass (*Poa sandbergii* = *P. secunda*) are also present and prairie junegrass is lacking. Perennial forbs are few and inconspicuous. Generally, a shrub element is lacking in these types; however, if shrubs are present they are usually either Wyeth's buckwheat (*Eriogonum heracleoides*) or rabbitbrush (*Chrysothamnus nauseosus*). Snowberry or rose species do not occur in this type. This habitat type is representative of the "true" steppe described by Daubenmire (1970) (Figure 9A – Appendix A). This grassland habitat type supports *S. spaldingii* in Palouse Grasslands and Channeled Scablands (Figure 3A – Appendix A).

#### Rough fescue

The rough fescue habitat types that support *S. spaldingii* include a type associated with Idaho fescue and a type associated with both Idaho fescue and bluebunch wheatgrass. The rough fescue-Idaho fescue (*Festuca scabrella-Festuca idahoensis*) [FESC-FEID] habitat type (Mueggler and Stewart 1980) is a bunchgrass grassland type with rough fescue (*Festuca scabrella*) and Idaho fescue (*F. idahoensis*) as the principal bunchgrasses. Prairie junegrass is consistently present in lesser amounts. Perennial forb diversity is high and shrubs are

scarce. The rough fescue-bluebunch wheatgrass habitat type in which *S. spaldingii* occurs is usually the needle and thread (*Stipa comata*) phase (*F. scabrella-Agropyron spicatum/Stipa comata*) [FESC-AGSP/STCO] habitat type (Mueggler and Stewart 1980) (MNHP 2003b). This habitat type is dominated by rough fescue, Idaho fescue, and bluebunch wheatgrass. Needle-and-thread (*Stipa comata*), prairie junegrass and a variety of forbs are also present (Mueggler and Stewart 1980). Scattered ponderosa pines and rose shrubs (*Rosa woodsii*) occur at a number of rough fescue *S. spaldingii* occurrences (Schassberger 1988; MNHP 2003b).

Rough fescue habitat types that support *S. spaldingii* are located in the Intermontane Valleys of northwestern Montana (Figure 3A – Appendix A). Rough fescue is present in all the intermontane valleys west of the Continental Divide (Stickney 1960). Intermontane Valleys in northwestern Montana are transitional between the Pacific Northwest Bunchgrasses in the Columbia Basin and the Mixed Prairie of the Great Plains (Mitchell 1958; Habeck 1967; Koterba and Habeck 1971; Mueggler and Stewart 1980; Antos et al. 1983; Tisdale 1947, 1983). Idaho fescue and/or bluebunch wheatgrass share dominance with species characteristic of the Great Plains east of the Continental Divide. The transition is related to a change in climate characterized by winter-high precipitation of the Columbia Basin to a summer-high precipitation pattern of the Great Plains (Mitchell 1958; Antos et al. 1983).

### **Shrub Habitat Types**

The shrub habitat types in which *S. spaldingii* occurs include sagebrush-fescue habitat types representative of the sagebrush steppe described by Daubenmire (1970) (Figure 3A – Appendix A). Although there are occasional occurrences of *S. spaldingii* in the big sagebrush-Idaho fescue type in the shrub steppe area of Lincoln County, Washington, the majority of sites are located within the three-tip sagebrush-Idaho fescue habitat type (B. Benner, personal communication, 2003b). Daubenmire (1970) noted that lower autumn precipitation than either meadow or true steppe areas of eastern Washington was associated with three-tip sagebrush locations. In Oregon on the terminal moraines of Lake Wallowa, *S. spaldingii* occurs on a distinct undescribed prairie sagebrush-Idaho fescue (*Artemisia ludoviciana-Festuca idahoensis*) [ARLU-FEID] type (Kagan 1989) (Figure 3A – Appendix A).

#### Big sagebrush-Idaho fescue

In the big sagebrush-Idaho fescue habitat type (*Artemisia tridentata-Festuca Idahoensis*) [ARTR-FEID] (Daubenmire 1970), big sagebrush is the principal shrub species and forms a discontinuous taller layer above the herbaceous bunchgrass layer. Big sagebrush averages approximately 1 meter in height, but is capable of growing up to 2 meters in some areas. The shorter bunchgrass layer is dominated by bluebunch wheatgrass and Idaho fescue. Sandberg's bluegrass is also present; prairie junegrass is absent, and perennial forbs are well represented.

### Three-tip sagebrush-Idaho fescue

In the more mesic three-tip sagebrush-Idaho fescue (*Artemisia tripartita-Festuca idahoensis*) [ARTR2-FEID] habitat type (Daubenmire 1970), three-tip sagebrush is the principal shrub species. Three-tip sagebrush is approximately half the size of big sagebrush and does not conspicuously project above the bunchgrass layer, giving the community a relatively uniform appearance. Sandberg's bluegrass, and often prairie junegrass and small needlegrass (*Stipa columbiana* = *S. occidentalis*), are present. Perennial forbs that are associated with meadow steppe types (FEID-SYAL and FEID-*Rosa* spp.) are well represented in this type. Big sagebrush occurs in disturbed areas within the three-tip sagebrush-Idaho fescue type.

### **Forest Habitat Types**

The forest habitat types supporting *S. spaldingii* occur primarily in areas where grasslands and pine woodlands form a mosaic in the Channeled Scablands. Here the FEID-SYAL grassland habitat type meets and is intermingled with ponderosa pine woodlands (Gamon 1991). Ponderosa pine is the only coniferous tree species and forms an open canopy. The ability to rapidly elongate the root system in an environment of high moisture deficiency is the primary competitive advantage ponderosa pine has with respect to other tree species of this region (Daubenmire 1968a). These forest types have less severe soil drought during the summer than FEID-SYAL or FEID-*Rosa* grassland habitat types, and ponderosa pine seedlings are able to establish (Daubenmire 1968b).

### Ponderosa pine-Idaho fescue

The ponderosa pine-Idaho fescue habitat type (*Pinus ponderosa-Festuca idahoensis*) [PIPO-FEID] (Daubenmire 1970; Cooper et al. 1991) has a bunchgrass understory dominated by Idaho fescue with lesser amounts of bluebunch wheatgrass. Prairie junegrass is present, and arrowleaf balsamroot (*Balsamorhiza sagittata*) and yarrow (*Achillea millefolium*) are conspicuous perennial forbs.

### Ponderosa pine-snowberry

The moister ponderosa pine-snowberry habitat type (*Pinus ponderosa-Symphoricarpos albus*) [PIPO-SYAL] (Daubenmire 1970) has a conspicuous low shrub layer composed of snowberry, rose, spiraea (*Spiraea betulifolia*), and a rich grass and forb component. This type is essentially like the shrub phase of the FEID-SYAL plant community with a superimposed tree layer (Daubenmire 1968b).

## **Habitat Types and Physiographic Regions**

Many of the habitat types in which *S. spaldingii* occurs are specific to a physiographic region, i.e., sagebrush steppe types in the Channeled Scablands of east-central Washington, rough fescue types in Intermontane Valleys of northwestern Montana, and prairie sagebrush-Idaho fescue in the Wallowa Plateau. Other habitat types are more wide-ranging, occurring

in a number of different physiographic regions, i.e., Idaho fescue-snowberry in Palouse Grasslands, Canyon Grasslands, Channeled Scablands, and Wallowa Plateau (Table 9 below).

**Table 9 - *Silene spaldingii* Habitat Types and Physiographic Regions.**

Habitat Types	Physiographic Regions				
	Palouse Grasslands	Canyon Grasslands	Channeled Scablands	Wallowa Plateau	Intermontane Valleys
FEID-SYAL	X	X	X	X	
FEID-Rosa	X	X			
FEID-KOCR (low)		X			
FEID-KOCR (ridgetop)				X	
AGSP-FEID	X		X		
FEID-AGSP	X				
ARTR-FEID			X		
ARTR2-FEID			X		
ARLU-FEID				X	
FESC-FEID					X
FESC-AGSP/STCO					X
PIPO-FEID	X		X		
PIPO-SYAL			X		

*Silene spaldingii* is known from the Idaho fescue-bluebunch wheatgrass habitat type in the Palouse Grasslands (WNHP 2003b); however, no occurrences are known from this habitat type in the Canyon Grasslands of Idaho although the habitat type is common in this region (Hill and Gray 1999; J. Hill, *in litt.* 2003a; IDCDC 2003b).

#### Associated Species

Based on current EORs (IDCDC 2003b; MNHP 2003b; ONHP 2003b; WNHP 2003b), species that are most commonly associated with *S. spaldingii* vary between physiographic regions and also within physiographic regions (Table 14A – Appendix B). Some species are commonly associated rangewide with *S. spaldingii*. Three native bunchgrasses, bluebunch wheatgrass (*Agropyron spicatum*), Idaho fescue (*Festuca idahoensis*), and prairie junegrass (*Koeleria cristata*), are present in all physiographic regions. Snowberry (*Symphoricarpos albus*) and/or rose species (*Rosa* spp.) are present in all regions. Rangewide, Black hawthorn (*Crataegus douglasii*) and ponderosa pine (*Pinus ponderosa*) are the most common tree species, while yarrow (*Achillea millefolium*), arrowleaf balsamroot (*Balsamorhiza sagittata*), prairie smoke (*Geum triflorum*), sticky geranium (*Geranium viscosissimum*), stoneseed (*Lithospermum ruderales*), yampah (*Perideridia gardneri*), and slender cinquefoil (*Potentilla gracilis*), are commonly associated forbs. The most commonly associated invasive non-native species include Japanese brome (*Bromus japonicus*), St. Johnswort (*Hypericum perforatum*), and Kentucky bluegrass (*Poa pratensis*).

Some species are restricted to specific physiographic regions, i.e., rough fescue (*Festuca scabrella*) to the Intermontane Valleys of northwestern Montana, big sagebrush (*Artemisia tridentata*) and threetip sagebrush (*A. tripartita*) to the Channeled Scablands of east-central Washington., twin arnica (*Arnica sororia*) and field chickweed (*Cerastium arvense*) to the Canyon Grasslands, and the invasive, non-native species, spotted knapweed (*Centaurea maculosa*) to Montana.

### **Discrepancies/Canyon Grassland Habitat Types**

A common habitat type in which *S. spaldingii* occurs in Canyon Grasslands is one that is dominated by Idaho fescue and bluebunch wheatgrass with lesser amounts of prairie junegrass. Scattered throughout the community are small-statured, single-stemmed snowberry and/or rose plants that do not project above the bunchgrasses. Occasionally these shrubs form small clumps within the bunchgrass. Perennial forbs are abundant and conspicuous. This community appears to fit very closely the FEID-SYAL or FEID-*Rosa* spp. habitat types described by Daubenmire (1970). The only difference is that the Canyon Grassland community has caespitose bluebunch wheatgrass rather than rhizomatous bluebunch wheatgrass.

It should be noted that neither of the primary references for Canyon Grassland habitat types (Tisdale 1986a; Johnson and Simon 1987) list this habitat type. Using either the Johnson and Simon or the Tisdale plant community keys, these communities key to the Idaho fescue-prairie junegrass habitat type; however, this type does not have a shrub component. Johnson and Simon (1987), whose study area included the Canyon Grasslands of the Imnaha River and portions of the Snake River in Oregon, state that the FEID-SYAL (Daubenmire 1970) habitat type is not common in the Wallowa-Snake Province, existing infrequently as small stands. Tisdale (1986a), whose study area included the lower Salmon, lower Clearwater, lower Grande Ronde rivers and the Snake River from Brownlee Dam, Idaho, to Little Goose Dam, Washington, originally had included a FEID-SYAL type in earlier reports (Tisdale 1979), but later eliminated this type because of its limited extent. He concluded that this type was "virtually absent from the canyon grasslands" (Tisdale 1986a).

This raises the question to what extent the Idaho fescue-snowberry and Idaho fescue-rose communities occur within the Canyon Grasslands system. A number of botanists have recorded this type in Washington Canyon Grasslands, i.e., Bear Creek on the Snake River just north of the Oregon border (Wilderman et al. 2000), and in Idaho Canyon Grasslands on Craig Mountain (Hill and Gray 2000; Hill et al. 2001; Hill and Fuchs 2002, 2003; Hill and Weddell 2003; Lichthardt and Gray 2002; Gray and Lichthardt 2003b; Menke 2003) and along the Salmon River (K. Gray, *in litt.* 2003a; IDCDC 2003b). Of 107 plots established at 32 sites at the Garden Creek Ranch, Idaho, approximately 40% had either snowberry or rose species as associated species (Menke, *in litt.* 2003e). Other occurrences of *S. spaldingii* in Idaho Canyon Grasslands on Craig Mountain, EORs ID 001, 008, 009, 010, 015, 016, 017, and along the Salmon River south of Craig Mountain, EORs ID 014 and 018, occur within the FEID-SYAL habitat type as well (IDCDC 2003b; K. Gray, *in litt.* 2003a).

For identification of habitat types where *S. spaldingii* occurs in Canyon Grasslands, it may be more appropriate use the Daubenmire (1970) plant community key for those communities with a snowberry and/or rose shrub element and the Johnson and Simon (1987) or Tisdale (1986a) key for those communities without a shrub element.

### **Dominance and Frequency**

*S. spaldingii* has low cover and frequency in the communities in which it occurs. Plants occur in a patchy distribution usually occupying only a small portion of the apparently suitable habitat (B. Benner, *in litt.* 1993; Hill and Gray 2000; Lichthardt and Gray 2002). Of approximately 800 acres of suitable habitat surveyed at Garden Creek Ranch, Idaho, *S. spaldingii* occurs in an area of 86 acres, and due to its patchy distribution, the actual occupied area is much less than 86 acres (J. Hill, *in litt.* 2003b). In two plots established within *S. spaldingii* occurrences by R. Daubenmire in the 1950s and 1960s, *S. spaldingii* had low canopy cover, i.e., 1% and <0.5% (Daubenmire 1970). In 10 feet X 10 feet plots containing *S. spaldingii* at Garden Creek Ranch, basal cover for *S. spaldingii* was low, varying from trace amounts to < 2-3% (K. Gray, *in litt.* 2003c).

### **Successional Phenomena**

The majority of the communities in which *S. spaldingii* occurs are considered late successional or climax communities (Schassberger 1988; Kagan 1989; Gamon 1991; Lorain 1991; Lichthardt 1997). These communities represent the “potential natural community” (PNC), also referred to as “habitat type” (Daubenmire 1970; Horton 1972; Tisdale 1986a) or “plant association” (Johnson and Simon 1987), for a particular site. The PNC is the plant community that would develop at a site under present environmental conditions without disturbances (Society for Range Management 1983). The composition and pattern of these communities across the landscape are related to climate, soil and physiographic factors (Daubenmire 1970; Tisdale 1986a). Although they may be modified or temporarily altered by normal disturbances such as fire, under natural conditions and normal disturbance regimes, these communities return to the PNC with time. Alterations in the normal disturbance regimes can alter the composition and structure of these communities and physical parameters such as soil moisture regimes. Habitat fragmentation and alterations in fire regime apparently have resulted in increases in deciduous shrubs or ponderosa pine on steppe and shrub steppe fragments in Washington (R. Crawford, *in litt.* 2003; Weddell, *in litt.* 2003b).

Weed invasion can cause irreversible changes in physical conditions and plant composition that interrupt normal successional pathways and prevent a return to the PNC (Daubenmire 1970; Tisdale 1986a; Mack 1986). (see Threats section of this report).

## LIFE HISTORY AND BIOLOGY

### SUMMARY

*Silene spaldingii* is a long-lived, perennial plant species with one to several stems from an underground caudex surmounting a long taproot. Aboveground vegetation of this herbaceous geophyte dies back at the end of the growing season. Plants emerge in the spring or can remain dormant underground through the growing season for up to 3 consecutive years (Lesica 1997a). Most plants spend nearly half of their summers dormant beneath the ground. This prolonged dormancy appears to be an important life strategy for survival in arid environments (Lesica 1997a). Recruitment does not appear to occur every year (Lesica 1997a). Seeds germinate mainly in the spring and form rosettes the first year. Generally, vegetative stems are produced the second year, and plants begin flowering during or after the third season (Lesica 1997a, 1999). Growth forms can vary from year to year and are not necessarily correlated with age of the plant, i.e., flowering plants can produce vegetative plants or even rosettes in subsequent years. *Silene spaldingii* blooms in late summer and is thought to be an obligate or near-obligate outcrossing species whose primary pollinator is the bumblebee *Bombus fervidus* (Lesica 1993; Lesica and Heidel 1996; Baldwin and Brunsfeld 1995). Reproduction is solely by seed. Genetic analyses of populations across *S. spaldingii*'s range indicate a northern and a southern grouping of genetically similar populations (Baldwin and Brunsfeld 1995).

### GROWTH

*Silene spaldingii* distribution appears to be patchy, with apparently suitable unoccupied habitat between patches (B. Benner, *in litt.* 1993; Hill and Gray 1999, 2000; Hill et al. 1999, 2001; Hill and Fuchs 2002, 2003; Lichthardt and Gray 2002; Menke 2003). At Dancing Prairie, Montana, *S. spaldingii* clusters are separated by 50-100 meters (Lesica 1993). Plants occur in clusters of varying sizes and density. The density of plants within clusters is typically low, but can vary from areas with several meters between plants to denser areas. Up to 17-18 plants have been observed in one-meter square (Hill and Gray 2004). Less often, plants occur singly (Hill and Gray 2000).

At Garden Creek Ranch, emerging plants appeared aboveground in the spring as either a stemmed plant, a rosette plant (without a visible stem), or occasionally, as a stemmed/rosette plant (Figure 2A - Appendix A; Photos 5 through 8 - Appendix E). Stemmed plants either became reproductive or remained vegetative later in the growing season. Rosette plants remained as rosette plants during the growing season, they did not form stems later in the season. The aboveground portions of 75% of rosettes had senesced by August. A stemmed plant one year can be a rosette plant the following year (B. Benner, *in litt.* 2003d; Hill and Gray 2004), or a rosette plant one year can be a rosette plant the following year (Hill and Gray 2004). Rosette plants can be either seedlings or older plants and the distinction between them is difficult (Hill and Weddell 2003; Hill and Gray 2004). The life spans of individual plants have not been determined, but plants appear to be long-lived, with 72% still present after five years (Lesica 1997a).

Belowground, an adult plant can support numerous whitish buds for new growth on the caudex (Figure 2A - Appendix A; Photos 9 and 10 - Appendix E). Up to 14 buds have been counted on one plant. This is many more than the number of stems or rosettes that form the following year. Live caudex buds were also observed on rosettes that had senesced aboveground at the end of the growing season (Hill and Weddell 2003). The taproot of adult plants can grow up 85 cm in length (Menke 2003).

Plant height appears to vary with reproductive status, annual weather patterns, and perhaps surrounding vegetation. Reproductive stems are usually taller than non-reproductive stems. In 1999 and 2002, the average height of reproductive stems was 33.5 cm and 26.5 cm and the height of non-reproductive stems was 20.8 cm and 10.6 cm, respectively (Hill and Gray 2000; Hill and Weddell 2003). Plant height may also be related to the height of surrounding vegetation. Menke (2003) observed that *S. spaldingii* appears to grow to a height that minimally overtops surrounding vegetation, hypothesizing this might be a reason for the shorter plants she observed in burned grasslands where bunchgrasses were shorter than in unburned sites. Hill and Gray (2000) observed *S. spaldingii* plants in a snowberry patch growing up to 75 cm, a height that minimally over-topped the surrounding snowberry bushes.

The majority of *S. spaldingii* plants produce one stem; however, plants do produce multiple stems. Production of multiple stems appears to vary between regions and between years. Two separate studies conducted at Garden Creek Ranch in the Canyon Grasslands of Idaho, showed 26% and 28% of plants were multi-stemmed (Hill and Gray 2000; Hill and Weddell 2003). Of the 241 multi-stemmed plants recorded in the 1999 study, 59% had two stems, 25% had three stems, 7% had four stems, 7% had five stems, 1% had six stems and 1% had eight stems (Hill and Gray 2000). Of the 44 multi-stemmed plants recorded in the 2002 study, 83% had two stems and 17% had three stems (Hill and Weddell 2003). Thirteen stems have been observed on one plant (J. Hill, *in litt.* 2003a). Usually multi-stemmed plants possessed either all reproductive stems or all non-reproductive stems, but occasionally a multi-stemmed plant possessed both reproductive and non-reproductive stems (Hill and Gray 2000; Hill and Weddell 2003). At Fairchild AFB in the Channeled Scablands in Washington, Caplow (2001, 2002a) reported 12 % multi-stemmed plants in 2001 and 20% in 2002. Of the three multi-stemmed plants recorded in 2001, 67% had two stems and 33% had 3 stems. Of the 16 multi-stemmed plants recorded in 2002, 81% had two stems, 13% had four stems, and 6% had 5 stems. At Dancing Prairie in Intermontane Valleys of Montana, multiple stemmed plants are rare (P. Lesica, *in litt.* 2003b). In demography studies at Dancing Prairie, Lesica (1997a) noted 7% multiple flowering stems.

## PHENOLOGY

*Silene spaldingii* plants emerge from winter dormancy in the spring. At Dancing Prairie, Montana, *S. spaldingii* plants were not apparent aboveground on 8 May 1996, but were apparent by 22 May 1995 (Lesica 1997a). The majority of plants at Garden Creek Ranch, Idaho, begin emerging in late April. Occasional plants emerge up to 2-3 weeks later in the season (J. Hill, *in litt.* 2003a). In Whitman County, Washington, plants emerge aboveground in late May (B. Weddell, *in litt.* 2003a). Plants emerge after early to mid-June at Fairchild AFB in Spokane County, Washington (F. Caplow, *in litt.* 2003b).

*Silene spaldingii* is a late-blooming plant that begins flowering in mid-July across its range. Peak bloom occurs the last two weeks of July in Montana (Schassberger 1988) and into August in Oregon (Kagan 1989) and Canyon Grasslands of Idaho (Hill and Gray 2000; Hill and Weddell 2003). The flowering period often extends late into the growing season. Oregon collections of flowering individuals have been made as late as September (Kagan 1989; Gamon 1991) and flowering plants have been observed as late as mid-October in Canyon Grasslands (J. Hill, *in litt.* 2003a). Initiation of flowering can vary somewhat with climatic and site conditions, with flowering beginning earlier on flat sites or southerly slopes than on steeper northerly slopes. Although *S. spaldingii* blooms and fruits during the hottest and driest time of the year, the mesic northerly aspects and protection from the drying effects of the sun and prevailing winds (Gamon 1991) likely enables the late flowering of this species.

Fruit capsules and seeds mature during late July and August, with seed dispersal occurring from mid-August through October. Initiation and length of senescence depends on climatic conditions, beginning earlier and extending for a longer period of time during drought years. During the droughty 1988 field season, stems were senescing as flowers were just opening (Schassberger 1988). Senescence usually begins on the lower leaves of the plant and extends upward. Aboveground portions of the plant die back at the end of the growing season with cold temperatures and the plant becomes dormant, surviving underground through the winter season.

The phenology of *S. spaldingii* differs markedly from the phenology of the majority of plants in the associated plant community. The major bunchgrasses and up to 1/3 of the perennial forbs in mesic fescue communities in the tri-state area of occurrence are photosynthetically active from late autumn through early July entering a period of aestivation (summer dormancy) in the summer drought period (Daubenmire 1970; Tisdale 1986). *Silene spaldingii* is dormant during the winter and does not exhibit a period of aestivation during the summer drought period.

## REPRODUCTION

### Flowering and pollination

*Silene spaldingii* reproduces solely by seed. It lacks rhizomes or other means of reproducing vegetatively. Three to twenty flowers are borne at the top of the stem in a sparingly branched inflorescence. The number of flowers per reproductive plant and flowering (the proportion of plants in a population that flower) varies between years and between sites. The mean number of flowers per reproductive plant at Dancing Prairie, Montana, varied from 3.94 in 1996 to 7.84 in 1990, and flowering varied from a low of 11% in 1994 to a high of 68% in 1989 (Lesica 1997a). At Garden Creek Ranch, Idaho, the mean number of flowers per reproductive plant varied from 9.4 in 1999 to 6.6 in 2003, and flowering varied from 78% in 1999 to 70% in 2003 (Hill and Gray 2000; Hill and Weddell 2003). At Fairchild AFB, Washington, the mean number of flowers per reproductive plant was 21.4 in 2001, 16.1 in

2002 and 9.8 in 2003, and flowering was 100% in 2001 and 97% in 2002 (Caplow 2001; Caplow 2002a; F. Caplow, *in litt.* 2003c).

The flower of *S. spaldingii* is protandrous. Within a particular flower, anthers mature and dehisce pollen before the stigmas expand and become receptive. This discourages pollination within the same flower. However, because two or more flowers may be open on the plant at the same time, the possibility for pollination by another flower on the same plant is also present. This system promotes out-crossing while allowing the possibility of selfing (Lesica 1988a; 1991a, 1993; Lesica and Heidel 1996).

Pollinators are critical to the persistence of *S. spaldingii*. Loss of fitness, i.e., lower proportion of fruits matured, seeds per fruit, germination, seedling growth and survival, was observed in plants from which pollinators were excluded (Lesica 1993). The bumblebee, *Bombus fervidus* (Photo 15 – Appendix E), was shown to be a significant pollinator for *S. spaldingii* at five sites in Idaho (EOR ID 006), Montana (EOR MT 001), Oregon (EOR OR 006), and Washington (EOR WA 027 and 048) where 83% of visits to *S. spaldingii* were by this bumblebee species (Lesica and Heidel 1996). *Bombus fervidus* is common throughout most of temperate North America occurring in grasslands and meadows where it builds nests on or near the surface of the ground (Hobbs 1966). Conversion to agricultural crop production, application of pesticides, domestic livestock grazing, and fire may be detrimental to populations of *Bombus fervidus* (Sugden 1985; Lesica 1991a, 1993; Lesica and Heidel 1996).

It is possible that noctuid moths are significant pollinators for *S. spaldingii* as well. Many of these mostly night-pollinating moths are known to pollinate, as well as predate seeds, of other species of *Silene* (Brantjes 1978; Pettersson 1992; Proctor et al. 1996; Young and Gravitz 2002; Westbrook 2003; Kephart 2003; Reynolds 2003). Diurnal pollination is low and affected by other co-occurring flowering plants for common varieties of *Silene douglasii*; however, noctuid moths visit some populations of the rare *S. douglasii* var. *oraria* at high rates, and serve both as pollinators and seed predators (Kephart 2003). *Silene spaldingii* pollination studies conducted during daylight hours and including dusk and dawn periods (1-2 hours past sunset and prior to sunrise) showed four visits by noctuid moths at Clear Lake Ridge Preserve (Lesica and Heidel 1996). Nighttime observations may show higher rates of pollination visits by these moths. The significant predation of *S. spaldingii* seeds by noctuid larvae observed at Clear Lake Ridge (R. Taylor, *in litt.* 2003a) further indicates pollination by these moths may be significant.

### **Seed Maturation and Dispersal**

The percentage of flowers successfully maturing to seed-filled capsules varies between sites, with 58% observed at Dancing Prairie (Lesica 1993) and approximately 70% at Garden Creek Ranch (Menke 2003). Mature fruits appear to produce relatively large numbers of seeds, with up to 150 per capsule (Lesica and Heidel 1996).

Seeds are apparently dispersed from the top of the upright capsule by stem movement caused by wind or passing animals. Seeds often stick to the glandular foliage (Photo 14 - Appendix

E), and it is possible that passing animals pick the sticky seeds up on their coats (F. Caplow, *in litt.* 2003b; K. Gray, *in litt.* 2003b). When wet, seeds have a very thin coat that may be somewhat mucilaginous, a factor that may be important as a means of dispersal (P. Lesica, *in litt.* 2003b). Seeds and seedheads are commonly eaten by deer and elk, and possibly rodents and birds, and may be transported in the digestive tract of these animals for considerable distances.

### **Germination**

*Silene spaldingii* seeds germinate considerably better with a 30-day period of cold stratification, suggesting that *S. spaldingii* germinates primarily in early spring (Lesica 1988b). Rosettes with 6-14 leaves formed after 60 days. They ceased growth but remained green for another 60 days, then the leaves senesced. In approximately 45 days most individuals produced new leaves. It was hypothesized that *S. spaldingii* germinates and grows in the spring while the soil is moist, and aboveground portions senesce during the hot dry summer months and produce new growth with fall rains and/or shortened photoperiod (Lesica 1988b, 1999).

Another germination study did not demonstrate the senescent period observed in the Dancing Prairie seedlings. Seeds collected at the Garden Creek Ranch, Idaho, germinated following cold stratification in November 1998 and produced rosettes with 10-12 leaves by July 1999. Growth slowed, but rosettes stayed green through the winter. Stem development began in April 2000 and some plants bolted by May. Five plants were transplanted into a *S. spaldingii* location at Garden Creek Ranch. Following transplantation, one plant produced buds and a flower but senesced before seed set at the end of the 2000 field season (Hill et al. 2001). None of the transplanted plants have been observed in subsequent years (Hill and Fuchs 2002, 2003).

### **Seedling Phenology**

Identification of *S. spaldingii* seedlings is difficult. Germinating seeds first form rosettes (visible stem lacking); however, adult plants can also produce rosettes, and differentiation between seedling and adult rosettes is ambiguous (Hill and Weddell 2003). In demography and germination studies at Dancing Prairie, true seedlings are stemless rosettes with glabrous leaves. Older plants will sometimes produce what appears to be a rosette (pseudorosette), but a very short stem between leaf pairs can usually be observed, and the leaves or stem usually have some glandular hair (P. Lesica, *in litt.* 2003b). At Garden Creek Ranch, rosette leaves produced by germinating seeds in greenhouse studies, had glabrous upper and lower surfaces with a few, scattered hairs on their edges. They did not develop the sticky, gland-tipped hairs on upper and lower leaf surfaces characteristic of adult plants until they bolted (Hill et al. 2001). At Garden Creek Ranch, eight rosettes attached to large, established caudices of adult plants demonstrated various leaf surface details, ranging from glabrous to dense, gland-tipped hairs. These adult rosettes showed no visible stem aboveground. Careful excavation of soil around the caudex area showed no evidence of a stem below ground either. The petioles of the rosette leaves attached directly to the caudex an inch or so belowground (Hill and Weddell 2003; Hill and Gray 2004).

In initial demographic studies at Garden Creek Ranch (Hill and Weddell 2003), it was thought that size of rosette leaves may distinguish rosettes produced by seedlings from rosettes produced by adult plants. Two types of rosette plants were noted, those with smaller 1-2 cm-long leaves and those with larger 8-10 cm-long leaves. Rosette leaves first produced by seedlings would likely be relatively small because of the limited resources of the seed. Of the nine rosettes from which soil was removed around the caudex, the eight adult rosettes were of the large-leaved variety and the ninth, a suspected seedling, was of the small-leaved variety (Hill and Weddell 2003). The second year of demographic studies at Garden Creek Ranch (Hill and Gray 2004); however, showed that many small-leaved rosettes the first year of the study were also small-leaved rosettes the second year of the study. This suggests that young plants may produce small-leaved rosettes for a number of consecutive years before producing stemmed plants. A small-leaved rosette may be a recruit, a newly germinated seedling, or it could be at least two years old and not a true recruit. This complicates the determination of recruits in studies without previous long-term monitoring.

The observed differences in seedling phenology may be related to the response of this species to different environmental and habitat factors across its range, or it is also possible there is a genetic basis for differences (Lesica 1997; P. Lesica, *in litt.* 2003b). Researchers should be aware that rosettes are not always recruits, but can also be adult plants with established, mature caudices or young plants at least two years old. Long-term monitoring is essential for determining which rosettes are seedlings and which rosettes are produced by older plants.

Many seedlings of associated native forbs are similar in appearance to *S. spaldingii* seedlings. In Canyon Grasslands at Garden Creek Ranch, seedlings of the native perennial forbs *Senecio integerrimus*, *Solidago missouriensis*, *Frasera albicaulis*, and *Penstemon glandulosus* produce rosettes that closely resemble *S. spaldingii* seedlings. The main diagnostic feature for *S. spaldingii* seedlings was the presence of a few scattered hairs along the edges of the leaves, many of which bent at a right angle and pointed backward toward the petiole of the leaf. These hairs were also observed along the edges of the rosette seedlings germinated in greenhouse studies (Hill et al. 2001; Hill and Weddell 2003; Hill and Gray 2004).

## GENETICS

Genetic analyses of *S. spaldingii* individuals from populations in Idaho, Montana, Oregon and Washington revealed a northern grouping composed of Dancing Prairie, Montana, and Lamona, Washington, populations and a southern grouping composed of Kramer Prairie, Washington, Garden Creek Ranch, Idaho, and Clear Lake Ridge, Oregon, populations. Plants in the northern group were more genetically similar to each other than they were to any plants in the southern grouping, and plants in the southern group were more genetically similar to each other than they were to any plants in the northern grouping. The populations within the Hells Canyon area, i.e., Garden Creek and Clear Lake Ridge, were shown to be very similar. The majority of sites showed outcrossing to be the primary breeding system, although a high degree of selfing was indicated in the Dancing Prairie plants, likely

indicating low pollinator activity. Genetic differentiation among populations was expected to be high due to the fragmented nature of the *S. spaldingii* habitat and low migration of gametes among populations; however, genetic drift was negligible and there was little genetic differentiation among populations. This could indicate that geographic distribution was much more extensive than is currently observed, and that drift is negligible because of a long generation time (Adapted from Baldwin and Brunfeldt 1995). Genetic studies have not been conducted on the numerous *S. spaldingii* populations located after this study.

## PROLONGED DORMANCY

*Silene spaldingii* plants exhibit prolonged dormancy in which they can survive underground during the growing season for up to three years and reappear in subsequent years (Lesica and Steele 1994). Since the perennating buds of *S. spaldingii* are below ground, those plants that are dormant cannot be detected aboveground. Lesica (1997a) determined that *S. spaldingii* plants at Dancing Prairie spend nearly half of their summers in a dormant condition. The proportion of plants that are dormant can vary between years. At Dancing Prairie, a low of 11% and a high of 74% were observed. Prolonged dormancy is likely possible due to the deep-seated taproot acting as a storage organ, which may allow the plant to remain underground when producing aboveground growth is not advantageous. Prolonged dormancy tended to be higher in summers preceded by a wet summer and dry fall (P. Lesica, *in litt.* 2003b). In dry years many plants can remain dormant during the growing season (Lesica 1995; Heidel 1995; B. Benner, *in litt.* 1999). This may be an important life history strategy in semi-arid environments (Lesica 1997a).

Prolonged dormancy has also been observed at other *S. spaldingii* populations. At Fairchild AFB in eastern Washington, wide annual fluctuations in number of plants and plant locations were observed in four years of population monitoring, indicating plants can become dormant for one or more years and then re-emerge (Caplow 2001, 2002a). Four-year monitoring studies at ten sites on BLM parcels in Washington showed a number of plants that were present one year, absent the next, and then present again the following year (B. Benner, *in litt.* 1999). A 4-year population monitoring study at the Garden Creek Ranch, Idaho, showed the proportion of the population that did not emerge at sites where they were present the year before were 33% in 2000, 33% in 2001, and 10% in 2002. The lower dormancy rate in 2002 appears to be related to a wildfire occurring at the site the previous fall (Hill et al. 2001; Hill and Fuchs 2002, 2003).

The presence of prolonged dormancy and the variation in the proportion of dormant plants between years complicates determination of the size of *S. spaldingii* populations, and makes it difficult to determine population trends or assess management (J. Hill, *in litt.* 2003a; F. Caplow, *in litt.* 2003b). Surveys may be hampered in years when large proportions of populations are dormant during the growing season and may be erroneously reported to be extirpated (Lesica and Steele 1994). Long-term monitoring is essential for determination of population size at *S. spaldingii* sites. Approximately 96% of dormant plants can be detected in the third year of recording (Lesica 1997a).

## DEMOGRAPHIC DYNAMICS

In a long-term demography study at Dancing Prairie, Lesica (1997a) determined that *S. spaldingii* had four life history stages: 1) seedling, 2) dormant, 3) non-reproductive, and 4) reproductive. The dormant life stage (which can last for one or more summers) played an important role in the life history of *S. spaldingii*. Fifty percent of population growth was due to survival and growth of plants in the dormant stage. Growth of the population was stable even though recruitment was episodic, occurring in only 2 of the 7 years for which it could be accurately measured. Seventy-two percent of plants observed in 1989 were still present in 1994. The long life span of *S. spaldingii* likely enables populations to persist many years without recruitment. Flowering, growth, and recruitment demonstrated a strong biennial periodicity with high levels occurring every other year, suggesting performance in one year may impact performance in the subsequent year.

Preliminary demographic studies in Canyon Grasslands of Idaho (Hill and Gray 2000; Hill et al. 2001; Hill and Fuchs 2002, 2003) showed some differences in demographic patterns and the rosette life stage compared to the Dancing Prairie studies. For three consecutive years of counting and flagging stemmed plants at a four-acre *S. spaldingii* site, the population size stayed remarkably constant, approximately 200 plants each year. The population at this site did not appear to exhibit the biennial periodicity observed at the Dancing Prairie site. Even though the period of the study (1999-2001) was a particularly droughty period, dormancy rates were fairly low, approximately 33% for both the second and third years. During these three years no *S. spaldingii* rosettes were observed; however, numerous rosettes were observed the fourth year, many occurring either at the base of stemmed plants or at flagged locations where stemmed plants had occurred the year before. Careful removal of soil around the caudex area of nine of these rosettes revealed that eight of them were attached to established, mature caudices (Photo 9 and 10 – Appendix E). Some of these caudices also supported stemmed plants. These rosettes were not seedlings (recruits) but were produced by mature adult plants. Approximately 25% of rosettes recorded within permanent transects at this site were still alive in August (Hill and Weddell 2003). At Dancing Prairie, it was determined that, with rare exception, all *S. spaldingii* rosette plants were new recruits (Lesica 1997a, 1999); however, many rosettes observed at the Garden Creek Ranch were produced by adult plants. Similar observations have been made at BLM sites in eastern Washington, where data from individually marked plants suggest that some rosettes are probably plants that have been vegetative with elongated internodes, or even reproductive plants, in past years (B. Benner, *in litt.* 2003d).

# POPULATION ECOLOGY

## SUMMARY

*Silene spaldingii* populations interact with biotic and abiotic components of their environment in a variety of ways. A mutually beneficial interaction exists with *S. spaldingii*'s primary pollinator, *Bombus fervidus*; however, interactions with other fauna, i.e., predation and herbivory by native ungulates and domestic livestock, rodents, and insects, appear to be detrimental to varying degrees. Livestock grazing can also affect the plant community that supports *S. spaldingii* by altering its composition and promoting weed invasion. *Silene spaldingii* apparently tolerates competition and some shading from the dominant bunchgrasses and the shrubs and other perennial forbs in the native community in which it occurs; however, it does not appear to persist in areas where the native vegetation has been displaced by invasive, non-native plant species. Invasive species have superior adaptations for exploiting resources, especially in disturbed areas, and can cause irreversible ecological changes in the invaded community.

Interactions of *S. spaldingii* populations with the abiotic components of its environment, i.e., fire and moisture, vary across its range. *Silene spaldingii* and the mesic fescue communities with which it is associated, have evolved with fire and have adaptations to survive normal fire regimes (Daubenmire 1970; Tisdale 1986; Johnson 1998). However, historic fire intervals for these communities are difficult to determine (Weddell 2001; Gray 2001), and the presence of invasive non-native weed species, habitat fragmentation, excessive livestock grazing, fire suppression, and increasing human activity have altered fire regimes (D'Antonio and Vitousek 1992; R. Crawford, *in litt.* 2003). In the highly productive rough fescue grasslands at Dancing Prairie in Intermontane Valleys of Montana, fire has been used to prevent woody encroachment and decrease large litter accumulations that may limit *S. spaldingii* recruitment (Lesica 1999). Fire likely has less ecological benefit in the less productive Idaho fescue grasslands of the tri-state area where low summer precipitation restricts woody encroachment and litter accumulation is less. Furthermore, fire can create conditions that favor establishment and spread of highly competitive, disturbance-adapted weed species (Christensen and Burrows 1986; Hobbs and Huenneke 1992; Peters and Bunting 1994; Given 1994; Agee 1996; Asher 1998; Asher et al. 1999; Sheley et al. 1999b; USDI 2000; Keeley 2002; Parks 2003).

*Silene spaldingii* is restricted to mesic grasslands, requiring a certain amount of moisture, not too wet or too dry, through the growing season for optimum growth and reproduction. Droughty conditions have a negative effect on *S. spaldingii* populations by limiting growth and reproduction and exacerbating negative impacts from livestock grazing and invasive weeds (Hill and Gray 2000; Lesica 1988c; B. Benner, *in litt.* 2003c).

## BIOTIC ENVIRONMENT

### Interactions with Fauna

Most *S. spaldingii* interactions with fauna are negative to some degree, with one notable exception. *Silene spaldingii* has a mutually beneficial relationship with its primary pollinator, the bumblebee *Bombus fervidus*. The bee is essential to reproduction and outcrossing of *S. spaldingii*, which only reproduces by seed, and *S. spaldingii* provides a nectar nutritional source for the bee (see Pollination Section above).

Negative interactions with fauna include herbivory and predation of *S. spaldingii* by native and domestic herbivores, i.e., deer and elk, rodents, insects, and domestic livestock (Lesica 1988a; Kagan 1989; Youtie 1990; Lorain 1991; Gamon 1991; Heidel 1995; B. Benner, *in litt.* 1999; Hill and Gray 2000; Caplow 2001; Caplow 2002a; Hill and Weddell 2003). Grazing herbivory by native ungulates (deer and elk) and domestic livestock typically removes the top portion of a stem and usually all flowers or capsules present (Photos 11 and 12 – Appendix E). Trampling of these large herbivores often damages plants and the soil disturbance from trampling can promote weed invasion. Damage caused by rodents, including voles and pocket gophers, varies from herbivory of roots to entire plants. Burrows and runways cause soil disturbance, which can also promote weed invasion. Herbivory of flowers and foliage and seed predation has been observed by insect larvae (Photos 13 and 14 – Appendix E) and insects, including grasshoppers, rove beetles, and seed weevils. Some insects become trapped on the gland-tipped, sticky hairs that cover the plant.

Overgrazing by domestic livestock can result in major alterations in the structure and composition of the native plant community that supports *S. spaldingii* and promote weed invasion (Daubenmire 1970; Tisdale 1961, 1986a). (See Threats Section of this report.)

The effects of domestic livestock grazing herbivory on *S. spaldingii* appear to depend on the moisture conditions of the site and the timing and the degree of grazing pressure. *Silene spaldingii* plants in more mesic sites, which are often more vigorous and grow to greater heights, are damaged less by livestock grazing than plants in drier sites (Heidel 1979). Although *S. spaldingii* appears to have survived in some areas that have been traditionally grazed by domestic livestock (R. Taylor, *in litt.* 2003a), it is primarily found in locations that have received low grazing pressure, i.e., those sites at a distance from water, or where grazing has occurred at a time when plants were dormant (B. Benner, *in litt.* 1999). *Silene spaldingii* plants have more time and moisture resources for regrowth if grazed early in the season; however plants are more likely to be incidentally grazed because livestock are spread out. Plants grazed later in the season have less time and moisture resources for regrowth, but livestock are more likely to be concentrated around water sources and therefore less likely to graze *S. spaldingii* (B. Benner, *in litt.* 2003c). Light grazing in very early spring appears to have minimal effects; however, grazing is especially detrimental to *S. spaldingii* with late summer grazing, or heavy grazing (Kagan 1989; R. Taylor, *in litt.* 2003a; G. Glenne, *in litt.* 2003a).

The negative impacts on *S. spaldingii* from herbivory by native fauna and domestic livestock can cause considerable reductions in annual reproductive effort and seed crop, and represent

significant threats to the long-term viability of *S. spaldingii*. (See Threats Section of this report.)

### Interactions with Flora

Interactions with other plant species in *S. spaldingii*'s environment typically involve competition, which results in mutually adverse effects to organisms that utilize a common resource in short supply. Much competition occurs between bunchgrasses and forbs, not only for light but also for water and nutrients (Weaver 1958).

*Silene spaldingii* persists in productive mesic grassland communities that typically have high cover of large bunchgrasses, a diverse forb component, and often scattered shrub species. This indicates that *S. spaldingii* is successfully competing with the native flora that comprise these communities. The long taproot of *S. spaldingii* likely contributes to its persistence in these communities. Weaver (1915) studied the root systems of many forbs in southeastern Washington before all of the Palouse Grasslands were converted to cropland. The rooting zone of 12 of 15 forbs studied were deeper than roots of the surrounding bunchgrasses. The structure of their root systems had little or no provision for much absorption in the surface foot of soil, but had strong taproots that absorb to depths of 5-12 feet or more and thus below the general depth of bunchgrass roots, i.e., 1.5 to 2 feet for Idaho fescue and 4-5 feet for bluebunch wheatgrass (Weaver 1958). The scattered distribution typical of native forbs in the mesic fescue communities, including *S. spaldingii*, may be related to inter- and intra-specific competition of the deep forb taproots within the lower portions of the soil profile.

*Silene spaldingii* does not appear to persist in otherwise suitable areas where the native vegetation has been displaced by invasive, non-native plant species (Schassberger 1988; Lorain 1991; Gamon 1991; Lichthardt 1997), indicating it cannot compete successfully with these plant species. Menke and Muir (2003) observed that *S. spaldingii* populations at study sites at the Garden Creek Ranch were disjunct from weed patches and hypothesized that *S. spaldingii* may be keeping weeds out due to competitive exclusion. However, many other *S. spaldingii* populations at Garden Creek Ranch have weed infestations within them, some of them extensive and dense (Hill et al. 1999; Hill and Gray 1999; Hill and Gray 2000; Hill et al. 2001; Hill and Fuchs 2002, 2003; Hill and Weddell 2003; Hill and Gray 2004). The ability of *S. spaldingii* to persist in dense weed infestations is unknown. Mature *S. spaldingii* plants appear to be able to persist for a period of time. Numerous, apparently healthy, reproducing plants have been observed in dense infestations of Japanese brome and yellow starthistle at Garden Creek Ranch (Photo 16 - Appendix E) (Hill and Gray 2000). These infestations likely required a number of years to develop. The long taproot of *S. spaldingii* may enable access to deep water unavailable to the invasive species (Kephart and Paladino 1997). *Silene spaldingii* will likely not be able to persist at these sites because seedlings are the most vulnerable stage in competition of slow-growing perennial species with fast-growing annual invasive weeds (Harris 1967; Hironaka 1990). A demographic study of the rare plant *Arabis fecunda* (Lesica and Shelly 1996) found that competition with spotted knapweed did not reduce the reproductive output of mature plants, but restricted recruitment and population growth.

A number of non-native, invasive weed species occur at *S. spaldingii* locations across its range of distribution. Habitat degradation from weed invasion is the most serious and imminent threat to *S. spaldingii* across its range. (See Threats Section of this report.)

## ABIOTIC ENVIRONMENT

### Fire

Historically, fire probably had little long-term negative effect on *S. spaldingii* or the mesic fescue community with which it is associated. These communities evolved with natural fires, which occur in mid- to late summer when lightning strikes are most common and the vegetation is dry. The bunchgrasses and native perennial forbs are tolerant of fire and resprout readily from underground storage organs and large root masses (Daubenmire 1970). Fire does not appear to affect the survival of established *S. spaldingii* plants. Cool season prescribed fires (spring and fall) at Dancing Prairie, Montana, did not increase mortality or decrease growth of older *S. spaldingii* plants (Lesica 1999; P. Lesica, *in litt.* 2003b). Total counts of stemmed plants over a five-year period at a *S. spaldingii* site at the Garden Creek Ranch in Idaho indicate a late-September wildfire that occurred during this period had little effect on established plants (Hill and Fuchs 2003; Hill and Weddell 2003; Hill and Gray 2004).

The bunchgrasses and majority of native perennial forbs in *S. spaldingii* communities complete their life cycles by the onset of summer drought. They enter a dormant state (aestivation) during the summer drought period from mid-July through October; this period coincides with the normal fire season. Fires are least harmful to these plants when they are dormant and not actively growing (S. Bunting, personal communication, 1999). Natural fires occurring at this time minimally disrupt reproduction and subterranean carbohydrate storage (Antos et al. 1983). *Silene spaldingii* is a notable exception, however, and is one of the few late-blooming perennial forbs in these communities. It does not enter summer aestivation and is actively growing and reproducing during the normal fire season. Fires occurring during the normal fire season can severely reduce the annual reproductive output and seed crop for *S. spaldingii* (Hill and Weddell 2003). Historically, however, fire return intervals were likely long enough to allow sufficient recruitment in non-fire years to maintain this long-lived species.

Although considerable controversy exists regarding historic fire intervals in the tri-state area of *S. spaldingii* occurrence, much evidence indicates historic fire-return intervals in this region may have been quite long. Prior to the arrival of white settlers, the fire-return intervals in the sagebrush-steppe areas probably varied between 60-100 years (Whisenant 1990). The widely spaced bunchgrasses in pristine sagebrush-steppe represented a discontinuous fuel bed that did not easily carry fire (Whisenant 1990; Brooks 1999). In areas with big sagebrush (*Artemisia tridentata*) that is killed by fire, it takes up to 60 years to reestablish (Fire Effects 2003). The use of fire by Native Americans in steppe environments of the tri-state area is not well understood (Weddell 2001), but appears to have been limited. Since bison were not present in the steppe area (Tisdale 1961; Daubenmire 1970; Mack and Thompson 1982) and deer and elk tended to remain close to forests, Native Americans had

little incentive to burn the steppe areas for hunting purposes (Daubenmire 1970; Tisdale 1986a). Due to the sparse resources of both man and his prey in bunchgrass and shrub steppe areas, Native Americans may not have burned them as regularly as areas with richer resources, i.e. bottomlands, meadows, and forests. Historic journals indicate Native Americans burned these areas from mid-July to early October which was the season for gathering, preparing, and storing provisions for the winter (Shinn 1980). This was also the normal fire season and the time when bunchgrasses were summer dormant. Kaiser (1961) considered the minimal use of fire by Native Americans in pre-settlement time in the Palouse Grasslands region allowed for the accumulation of annual plant growth over a long period of time which resulted in building into the soil a high level of organic matter and plant food nutrients. The presence of mosses and lichens as major components of *S. spaldingii* communities (Hill and Weddell 2003; Hill and Gray 2004; Gamon 1991) indicates that fire return intervals may have been quite long. Mosses and lichens experience high mortality with fire (Hill et al 2003; Hill and Weddell 2003) and require many years or decades to recover (Belnap and Eldridge 2001).

Historic fire-return intervals for Intermontane Valleys in northwestern Montana appear to be considerably shorter than those in the tri-state area, possibly due to frequent Indian-set fires (Arno 1980; Barrett and Arno 1982). At Dancing Prairie Preserve the mean presettlement fire-return interval for western Montana valleys was ~9 years (Barrett and Arno 1982) and was estimated to be 6.4 years (range 2-13 years) for the north end of the Tobacco Valley in southeastern British Columbia (Dorey 1979). Frequent wildfires in the past tended to favor grassland development and served to interrupt a very slow but certain invasion of these grasslands by coniferous forest species (Koterba and Habeck 1971).

Fire plays essential roles in many grassland ecosystems by preventing woody invasion or removing heavy accumulations of litter that suppress biodiversity in highly productive grasslands. Prescribed burning had positive effects on reproductive output of the rare prairie species, *Silene regia* (Menges 1995). In the rough fescue grasslands of Dancing Prairie, Montana, periodic fire is used to prevent woody encroachment of ponderosa pine (Lesica 1999; Lesica and Martin 2003). Furthermore, in the absence of grazing and fire, large accumulations of litter from the dominant grass, rough fescue, may inhibit *S. spaldingii* seedling recruitment (Lesica 1997a). At Dancing Prairie, *S. spaldingii* density and flowering were highest at sites with reduced interference from the dominant grass rough fescue (Lesica 1997a). Prescribed fall and spring fires within *S. spaldingii* sites at Dancing Prairie (Lesica 1999) resulted in enhanced seedling recruitment and an increase in population size. Lesica concluded that fire had a positive effect on the population dynamics of *S. spaldingii* by removing litter and creating safe sites for recruitment. The beneficial effects of the fire at Dancing Prairie were short-term. Although there was a 1-year increase in recruitment following fire, this increase was largely negated by increased mortality of new recruits in subsequent years (Lesica 1995). Fire may not be necessary to maintain viable populations of *Silene spaldingii* at Dancing Prairie (B. Martin, *in litt.* 2000; P. Lesica, *in litt.* 2003b).

Fire may have fewer beneficial effects on the climatic grasslands in the tri-state area of occurrence of Oregon, Idaho and Washington. For the majority of this area, fire is not necessary to prevent woody encroachment and maintain these grasslands. Limited soil

moisture in summer prevents trees from becoming established in this zone even in the absence of fire (Daubenmire 1968a). Soil dries out to 50 cm in the Idaho fescue-snowberry and Idaho fescue-rose habitat types, which is too deep for conifer seedling survival (Daubenmire 1968a). Fire has little effect on the distribution of shrubs in the meadow steppe habitat types, since snowberry and rose resprout readily from rhizomes and return to a prefire state within three years (Daubenmire 1970; Fire Effects 2003). In some remaining meadow steppe areas in Washington, increases in woody species have been observed, especially near forests or woody riparian areas (R. Crawford, *in litt.* 2003). Fire has a relatively greater effect on the shrub steppe habitat types in which *S. spaldingii* occurs in the Channeled Scablands of Washington. Fire readily kills *Artemisia tridentata*; it will not resprout and regeneration is by seed. Fire often kills the aerial portions of *A. tripartita*, but it can resprout from the root crown following fire (Fire Effects 2003). The decline in sagebrush in these habitat types following fire, releases resources for grasses and forbs.

The Idaho fescue grasslands of the tri-state area are not as productive as the rough fescue grasslands at Dancing Prairie, and therefore have less litter to limit *S. spaldingii* recruitment. Whereas litter depths at the Dancing Prairie can reach depths of ~ 9-10 cm five years post-fire (Lesica 1999), at a Canyon Grasslands *S. spaldingii* site that had not burned in at least 15 years, litter depth averaged only 1.3 cm (Hill and Weddell 2003). Another fire study at the Garden Creek showed only a thin layer of detached ground litter that averaged 23% cover of the ground surface in an Idaho fescue community that had not burned for over 15 years (Hill et al. 2003). Litter accumulation differences between Idaho fescue grasslands and rough fescue grasslands may also be related to climatic differences between the tri-state area and northwestern Montana, which could affect litter decomposition rates. Low litter accumulations despite relatively long fire intervals may be due to more rapid litter decomposition. The mild, wet winters of the tri-state area may be more conducive to litter decomposition than the colder, drier winter climate in northwestern Montana (Climate Diagrams – Appendix C). Flushes of seed germination were not observed at *S. spaldingii* locations at the Garden Creek Ranch following two wildfires in 2000 and 2001 (Menke and Muir 2003). Although Hill and Weddell (2003) observed many *S. spaldingii* rosettes following these fires, the majority of them were not recruits but produced by older established adult plants.

Some interactions of *S. spaldingii* with fire are not easily classified as negative or positive. An increase in the number of flowers produced per plant of *S. spaldingii* was observed following cool-season prescribed fires at Dancing Prairie (Lesica 1999) and a wildfire occurring in late September at Garden Creek Ranch (Hill and Weddell 2003; Menke 2003). Lesica reported that this condition was apparent for only 2-3 years following the burn treatments. At Garden Creek Ranch, fewer of these flowers developed mature fruits compared to those in unburned areas (Menke 2003). The late-season fire at Garden Creek Ranch appeared to break prolonged dormancy of *S. spaldingii* plants. Total population count at the site the year following the fire, 270 plants, was a 35% increase over counts made in the previous 3 years, i.e., 201 in 1999, 200 in 2000, and 202 plants in 2001. Also, in the year following the fire, the number of plants returning at flags marking locations of plants the previous season, was considerably greater than previous years. In 2000 and 2001 an average of 66% plants returned at flagged locations from the previous year (Hill et al. 2001; Hill and

Fuchs 2002). In 2002, the year following the fire, 90% of the plants flagged in 2001 returned in 2002. The effect of fire breaking prolonged dormancy in *S. spaldingii* is unknown. If prolonged dormancy is a life history strategy to survive drought conditions, it may be questioned whether the enforced breaking of prolonged dormancy by fire is beneficial.

Although historically, fire probably had little long-term negative effect on the mesic fescue communities that supported *S. spaldingii*, the current pervasive presence of a number of invasive, non-native weed species makes the effect of fire problematic in these communities. Fire can create conditions that favor the establishment and spread of highly competitive, disturbance-adapted weed species (Christensen and Burrows 1986; Hobbs and Huenneke 1992; Peters and Bunting 1994; Given 1994; Agee 1996; Asher 1998, Asher et al. 1999; Sheley et al. 1999b; USDI 2000; Keeley 2002; Parks 2003). The presence of weeds can also affect the continuity of fine fuels and contribute to increases in fire frequency and intensity (Whisenant 1990; Peters and Bunting 1994). Alteration of the natural fire regime can make even highly fire-adapted plant communities vulnerable to weed invasion (Keeley 2002). Since the mesic fescue communities that support *S. spaldingii* evolved with fires that occurred primarily during the summer months, prescribed out-of-season fires conducted in early spring and late fall may have detrimental effects on the communities that support *S. spaldingii*. (See Threats Section of this report).

### Moisture

*Silene spaldingii* and the communities with which it is associated require mesic conditions, not too much or too little moisture, to persist. Little information exists regarding the relationship of *S. spaldingii* with excessive moisture conditions; however, more mesic areas and years of above average precipitation appear to favor growth and reproduction of *S. spaldingii*. *Silene spaldingii* is more vigorous and grows to greater heights in more mesic sites (Heidel 1979). Years with above average precipitation are considered to be “good years” for *S. spaldingii* and are associated with less prolonged dormancy (Lesica 1994). Surveys conducted during ‘good years’ may more closely reflect the actual population size (Heidel 1995). *Silene spaldingii* appears to tolerate light grazing in early spring when moisture reserves are plentiful for recovery and regrowth (B. Benner, *in litt.* 2003c).

Low moisture is a more prevalent condition within *S. spaldingii* habitat than excessive water, and is associated with less vigorous growth, lowered reproduction effort, and higher senescence of *S. spaldingii* (Lesica 1988c, 1988d; B. Benner, *in litt.* 1999; Hill and Fuchs 2002, 2003). Annual precipitation within *S. spaldingii* range is low, varying from 10-24 inches, and soil moisture becomes particularly limiting during the hot, dry summers that typify the climate across *S. spaldingii* range (Stoddart 1941, Weaver 1958; Tisdale 1986a; Daubenmire 1970; Habeck 1967) (Table 8; Climate Diagrams – Appendix C). *Silene spaldingii* has adaptations, a long taproot and prolonged dormancy, to survive periods of low moisture conditions; however, prolonged drought conditions can present a threat to *Silene spaldingii*. (See Threat Section of this report for further discussions regarding the effect of prolonged drought on *S. spaldingii*.)

## THREATS

### SUMMARY

*Silene spaldingii* is threatened by several factors. Substantial portions of *S. spaldingii* habitat in the Palouse Grasslands, Wallowa Plateau, Channeled Scablands, and Intermontane Valleys have been converted to croplands, urban and residential development, and non-native range grasses. The effects of habitat fragmentation threaten small patches of native vegetation that remain in these areas, and the small populations they support are subject to pollinator limitation, inbreeding depression, and genetic pressures associated with small population size. Several areas of *S. spaldingii* habitat in rugged, inaccessible terrain, including the Canyon Grasslands and portions of the Channeled Scablands, remain intact and contiguous with minimal human-induced fragmentation; however, these areas are seriously threatened with habitat degradation from weed invasion, livestock grazing, alteration of fire regimes, and herbicide drift. Invasive, non-native plant species have superior adaptations for exploiting resources, especially in disturbed areas, and can cause irreversible ecological changes in the invaded community. Predation of *S. spaldingii* by herbivores (deer, elk, domestic livestock, rodents and insects) has caused reductions in reproductive effort and seed crop. Existing regulatory mechanisms to protect this species are inadequate, especially the protection of populations on private land. Prolonged drought has negative impacts on growth and reproduction, and global warming has potential to threaten the long-term existence *S. spaldingii*. Factors that threaten *S. spaldingii* are discussed within five categories in this section, including: 1) the present or threatened destruction, modification, or curtailment of habitat or range, 2) over-utilization for commercial, recreational, scientific, or educational purposes, 3) disease or predation, 4) the inadequacy of existing regulatory mechanisms, and 5) other natural or manmade factors affecting its continued existence.

Threats of greatest concern to the continued existence of *S. spaldingii* include, in order of priority: 1) habitat degradation from weed invasion and livestock grazing, 2) habitat loss and fragmentation and associated genetic pressures of small populations, i.e., pollinator limitation, inbreeding depression, loss of populations, 3) alteration of fire regimes, including fire suppression, increasing fire frequencies, and out-of-season fires, 4) predation by herbivores, including domestic livestock, native ungulates, rodents and insects, 5) herbicide drift, and 6) prolonged drought and global warming.

#### **PRESENT OR THREATENED DESTRUCTION, MODIFICATION, OR CURTAILMENT OF *SILENE SPALDINGII* HABITAT OR RANGE**

*Silene spaldingii* has restricted habitat and geographic range. It is limited to mesic fescue grassland communities of the Pacific Northwest Bunchgrasses (Tisdale 1986a), sagebrush-fescue communities, and open-canopy pine-fescue communities in eastern Washington, adjacent Idaho, northeastern Oregon and northwestern Montana. These habitats have suffered considerable decline, including habitat loss and habitat degradation, within the past century.

## Habitat Loss

Habitat loss, a decline in habitat that involves the destruction or conversion to land uses or vegetation types that do not support *S. spaldingii* or its associated community (Noss et al. 1995), has been extensive within *S. spaldingii* range. The greatest loss of *S. spaldingii* habitat has resulted from conversion to croplands, urban and residential development, and planting of non-native grasses on rangelands. The following discussion of habitat loss is applicable to both habitat that currently supports known occurrences of *S. spaldingii* and habitat that has the potential to support, but currently does not support, *S. spaldingii*.

### Conversion to Cropland

Prairies and grasslands have dwindled throughout the United States because their soils are typically fertile and tillable and have been converted to cropland (Noss et al. 1995; Conner et al. 2001). Because of the exceptionally rich, deep, loessal soils of the native bunchgrass habitat which supports or supported *S. spaldingii*, much of the area has been converted to dryland (unirrigated) agriculture (Kaiser 1961; Daubenmire 1970, 1978; Ertter and Moseley 1992; Lichthardt and Moseley 1997a, 1997b). Regions that have suffered large areal losses of *S. spaldingii* habitat due to conversion to croplands include the Palouse Grasslands of southeastern Washington and adjacent Idaho, the Intermontane Valleys of northwestern Montana, the Wallowa Plateau in northeastern Oregon, and portions of the Channeled Scablands (Mitchell 1957; Tisdale 1961; Kaiser 1961; Caldwell 1961; Habeck 1967; Daubenmire 1970, 1978; Schassberger 1988; Kagan 1989; Gamon 1991; Ertter and Moseley 1992; Lorain 1991; Lichthardt 1997; Lichthardt and Moseley 1997a, 1997b) (See Figure 3A – Appendix A). The Palouse Grasslands region suffered the greatest loss of *S. spaldingii* habitat. Over 99% this region has been lost to agriculture and is considered a critically endangered ecosystem (Noss et al. 1995) (this reference includes the Palouse Grasslands and Wallowa Plateau regions described in this paper). The Channeled Scablands of east-central Washington has had 47% of native vegetation destroyed (Noss et al. 1995).

European settlement began in the Palouse Grasslands and Channeled Scablands regions in the 1860s and 1870s (Kaiser 1961; Meinig 1995; Lichthardt and Moseley 1997a, 1997b). These regions generally had enough annual precipitation to support dryland wheat farming with good yields of peas and wheat in the moister meadow steppe areas of the Palouse Grasslands (Daubenmire 1970, 1978). All areas that could be plowed were in cropland by the late 1800s and early 1900s (Kaiser 1961; Meinig 1995; Lichthardt and Moseley 1997a, 1997b). Livestock grazing continued on the remaining areas in rugged terrain or where soils were otherwise impractical to plow (Tisdale 1961; Daubenmire 1978; Lichthardt and Moseley 1997a, 1997b).

The loss of large portions of *S. spaldingii* habitat to croplands has undoubtedly reduced its abundance from pre-settlement times as it has for other native plant species endemic to these regions, i.e., Jessica's aster (*Aster jessicae*) and Palouse goldenweed (*Haplopappus liatrifomis*) (Ertter and Moseley 1992). However, we do not know to what degree it has been reduced because there is little information indicating how much *S. spaldingii* was present prior to conversion to cropland. The little historical evidence that exists in the ca.

65-year period from its discovery at Lapwai, Idaho, in 1846 until conversion to cropland was complete in 1910, indicates it was collected only a few times during this time period, considerable time periods occurred between collections, and the areas of collection were widely separated from each other. The only botanical reference for this area written before the majority of conversion to cropland had occurred (Watson 1875) lists only the type specimen occurrence in central Idaho made 29 years earlier. During the pre-conversion period *S. spaldingii* was collected in Canyon Grasslands and the Wallowa Plateau of Oregon (1881, 1882, 1889), in northwestern Montana (1894), and in Channeled Scablands (1903) in Washington (Piper 1903; Heidel 1980; S. Yelton, *in litt.* 2001; L. Kinter, *in litt.* 2003a; R. Hartman, *in litt.* 2003; A. Liston, *in litt.* 2003; P. Holmgren, *in litt.* 2003). No collections of *S. spaldingii* were made in the Palouse Grasslands prior to or during the conversion to croplands. A 1901 flora, "Flora of the Palouse", which examined the area in a 21-mile radius of Pullman, Washington, did not list the species (Piper and Beattie 1901). The first collections from the Palouse Grasslands were by Dr. R. Daubenmire in 1949 (L. Kinter, *in litt.* 2003b) after conversion to croplands.

This information indicates *S. spaldingii* may not have been a common plant prior to conversion to cropland, and that it had a fairly extensive range. This agrees with descriptions of "uncommon", "not common", and "collected but a few times, even though its range is fairly extended" used to describe this species in numerous botanical references that followed (Hitchcock and Maguire 1947; Davis 1952; St. John 1963; Hitchcock et al. 1964). The current distribution of *S. spaldingii* in areas that have not been converted to croplands, i.e., the Canyon Grasslands and portions of the Channeled Scablands, also indicate that it is not a common plant and occurs in a patchy distribution with abundant unoccupied, apparently suitable habitat between patches (J. Hill, *in litt.* 2003a; B. Benner, *in litt.* 1993). Irregardless, the presence of numerous extant *S. spaldingii* occurrences on remaining native prairie fragments in regions converted to croplands (IDCDC 2003b; MNHP 2003b; ONHP 2003b; WNHP 2003b) indicates that its original abundance was likely considerably reduced when large portions of these regions were lost to agriculture.

The loss of *S. spaldingii* habitat to croplands was a significant threat in the past; however, there is relatively low risk of further habitat loss due to this factor since most tillable land has been tilled (Kaiser 1961; Meinig 1995; Lichthardt and Moseley 1997a, 1997b). Although no longer an imminent threat in itself, the consequences resulting from conversion to cropland are largely responsible for current imminent threats to *S. spaldingii* and its habitat. The small, isolated fragments of native grassland habitat that remain suffer from edge effects and are threatened by habitat degradation from weed invasion, livestock grazing, alteration of fire regimes, and herbicide drift from surrounding agricultural fields (Lichthardt 1997). The small *S. spaldingii* populations these fragments support, suffer from pollination limitation, genetic consequences of small population size, and high risk of local extirpations (Lesica 1993; Lesica and Heidel 1996). (See the Other Factors portion of the Threats section of this report for further discussion of habitat fragmentation, pollination and dispersal problems, and genetic consequences of small populations.) Other threats resulting from conversion to cropland are irreversible soil changes associated with cultivation, i.e., loss of fertile topsoil from erosion (Kaiser 1961; Barker et al. 1983) and loss of up to 50% of original carbon within the first 40-50 years following cultivation (Conner et al. 2001). These factors would

likely hamper or preclude restoration of native bunchgrass grasslands in regions converted to croplands (Allen 1995).

### **Residential and Urban Development**

Metropolitan and urban centers within *S. spaldingii* habitat, including Spokane and Pullman in Washington, Moscow and Lewiston in Idaho, and Kalispell in Montana, have increased in population size in the last two decades (Populstat 2003), and continue to expand to meet the needs of increasing population growth. Habitat is lost and further fragmented as areas are converted to subdivisions and shopping centers, schools, hospitals, and the associated facilities to support them, such as highways, streets, sewer systems and treatment plants, and water storage facilities. Urban encroachment encourages surrounding landowners to sell their grasslands for high-dollar developments (Conner et al. 2001).

Rural population growth has increased in all counties that support *S. spaldingii* over the last two decades (Populstat 2003), making habitat loss to residential development in rural areas an increasing threat as well. Growth in human population and per capita income and the resulting demand for property and services, low returns to the ranching industry, increasing desirability for open-space, coupled with advances in telecommunications, have contributed pressure to convert grasslands into large-lot, rural homesites and ranchettes (Conner et al. 2001). Habitat fragmentation increases and soil and water resources are lost to development as increased density of habitation leads to increased demand for water and land (Conner et al. 2001). Human settlement is expanding into new areas and technologies allow exploitation of undisturbed habitats (Noss et al. 1995). For example, alternative sewage treatment systems, such as composting toilets, allow development at sites where soils fail to meet percolation requirements for septic tank construction.

Active housing development and facilities to support them are an on-going threat to known populations of *S. spaldingii* and potential habitat. Residential development threatens to eliminate a *S. spaldingii* population and habitat near Redbird Ridge in Idaho (Lichthardt 1997). Subdivisions have been developed at a *S. spaldingii* population near Liberty Lake in Washington, and the population there is assumed extirpated (Gamon 1991; WNHP 2003b). Residential development immediately adjoining the large occurrence of *S. spaldingii* at Dancing Prairie Preserve in Montana has destroyed potential habitat, increased the likelihood of weed invasion, and reduced management options such as controlled burning on the preserve; continued development in this area is expected (B. Martin, *in litt.* 1998, 2003). In Idaho, road development at the EOR ID 004 - Lawyer Creek site is likely responsible for extirpation of *S. spaldingii* at this location (Lichthardt 1997).

Increased use of popular recreational sites within *S. spaldingii* habitat, such as Wallowa Lake in Oregon, Hells Canyon National Recreation Area along the Snake River in Oregon and Idaho, and Flathead Lake in Montana, has contributed pressure to expand recreational facilities and accommodations. A large *S. spaldingii* site near Wallowa Lake in Oregon is close to existing recreational facilities and residences, and may be threatened by development in the future (E. Rey-Vizgirdas, personal observation, 1998).

The loss of *S. spaldingii* habitat to urban, residential and recreational development and the habitat fragmentation associated with these losses are ongoing, imminent threats to *S. spaldingii*. The risk of further losses of *S. spaldingii* habitat to these factors is high.

### **Planting of Non-native Grasses**

The planting of non-native grasses and/or cultivars of native grass species in pastures and rangelands results in further loss and fragmentation of *S. spaldingii* habitat. Large-scale rangeland revegetation is utilized for forage production for livestock, erosion control and watershed rehabilitation. Non-native grasses, such as crested wheatgrasses (*Agropyron cristatum* and *A. desertorum*), smooth brome (*Bromus inermis*), and Kentucky bluegrass (*Poa pratensis*), are typically used because they are widely-adapted, easily produced and established, grazing tolerant, and provide competitive weed control (Daubenmire 1978; Monsen and McArthur 1995; Roundy et al. 1997).

However, these plantings have been shown to decrease biodiversity and disrupt the integrity of native grassland and shrub-steppe ecosystems (Belnap 1995; Roundy et al. 1997). Many of these non-native grasses have aggressive characteristics, and they compete with native plants for nutrients, water and favorable sites. They have invaded adjacent native communities and negatively affect the ability of the natives to reclaim the site (D'Antonio and Vitousek 1992; Belnap 1995). Dominance by the introduced species creates monocultures that support little diversity and are susceptible to insect damage and pathogens in epidemic proportions since they are uniformly susceptible (Belnap 1995; Fairbanks and Andersen 1995; Roundy et al. 1997). Cultivars, non-native species, or non-local native species can dilute or change the genetic makeup of remnant native populations through gene flow (genetic pollution) resulting in the loss of local genomes (Linhart 1995; Roundy et al. 1997). Many of the revegetation projects have failed due to replacement by less desirable non-native species or lack of genetic diversity needed to maintain adaptation in dynamic environments (Belnap 1995; Munda and Smith 1995; Roundy et al. 1997).

The decrease in biodiversity, disruption of integrity and displacement of the native grassland and shrub-steppe ecosystems due to rangeland revegetation with non-native species represent significant threats to *S. spaldingii* habitat, especially in the Channeled Scablands of Washington (B. Benner, *in litt.* 1993, 2003c; Weddell, *in litt.* 2003a). The planting of non-native range grasses in this area is one of the factors responsible for elimination of *S. spaldingii* from much of its original habitat in this area (B. Benner, *in litt.* 1993; Noss et al. 1995). Two Washington EORs that indicate evidence of this threat include the Swanson Lake Population 37 (EOR WA 079) and the Miller Ranch Population 20 (EORs WA 032-037, 042, 047). The Swanson Lake occurrence includes a site where a single *S. spaldingii* plant was located in a disturbed area with 70% crested wheatgrass at a former Conservation Reserve Program (CRP) site (B. Benner, *in litt.* 2003a; WNHP 2003b). In a portion of the Miller Ranch occurrence, surveys in appropriate *S. spaldingii* habitat, i.e., steep, north-facing slopes, revealed non-native, seeded grasses but no *S. spaldingii* (B. Benner, *in litt.* 1993).

The loss of *S. spaldingii* habitat to planting of non-native grasses is a continuing, imminent threat. However, a major trend toward the use of native species to maintain genetic and

ecological integrity of rangeland ecosystems is underway (Roundy et al. 1997; Richards et al. 1997; Darris 2003). This trend should help to reduce this threat in the future; however, at present, there are a number of constraints to the use of native species for this purpose. Seed availability for native grass species, especially those adapted to a local area, is very limited (Belnap 1995; Monsen and McArthur 1995; Roundy et al. 1997; Bermant and Spackeen 1997; Boyer 2003; Gisler et al. 2003; Kendig 2003; Darris 2003). Also critically lacking is genetic information for identifying the appropriate number and distribution of collection sites, and for determining the geographic and ecological distance that seed should be transferred from original source populations (Erickson 2003). Further complicating the issue is the fact that many sites are inaccessible to normal site preparation and seeding practices (Monsen and McArthur 1995), and successful establishment and survival is dictated by adverse and unpredictable climatic conditions present in semi-arid climates (Bleak et al. 1965).

### **Habitat Degradation**

Habitat degradation has occurred extensively in remaining *S. spaldingii* habitat. The primary causes of habitat degradation are: 1) weed invasion, 2) herbicide drift, 3) livestock grazing, and 4) alteration of fire regimes. Habitat degradation is a decline in habitat that alters the structure, function, and composition of the habitat (Noss et al. 1995). “Structure” refers to the spatial arrangement of various components of the ecosystem. “Composition” refers to the types of species present, their abundance, and relationships. “Function” refers to the types and rates of ecological processes (Franklin et al. 1981; Noss 1990).

Although large portions of *S. spaldingii* habitat in moderate, accessible terrain have been lost to agricultural croplands and development, extensive areas of *S. spaldingii* habitat remain in steep and/or rugged and inaccessible terrain (or areas otherwise not conducive to cultivation) where little human-induced fragmentation has occurred. These areas are located primarily in the Canyon Grasslands (Lichthardt 1997; Hill and Gray 1999; J. Hill, *in litt.* 2003b; IDCDC 2003b) and portions of the sagebrush steppe areas in the Channeled Scablands (Weddell 2002; Caplow 2002b; B. Benner, *in litt.* 1993; B. Benner, *in litt.* 2003a, 2003b, 2003d; WNHP 2003b). Also, within the fragmented portions of the Palouse Grasslands, Channeled Scablands, Wallowa Plateau, and Intermontane Valleys of Montana, certain areas were never converted to croplands but remained as relatively extensive areas of rangelands. The largest tracts within this category include Dancing Prairie in Montana (Heidel 1995; Lesica 1997a, 1999; MNHP 2003b) and Zumwalt Prairie in Oregon (Else road and Taylor 2002a; ONHP 2003b).

The following discussion of habitat degradation is applicable to both habitat that currently supports *S. spaldingii* and habitat that has the potential to support, but currently does not support, *S. spaldingii*.

## Weed Invasion

### Invasive Weeds and Habitat Degradation

Degradation from weed invasion is considered one of the major threats to *S. spaldingii* habitat (IDCDC 2003b; MNHP 2003b; ONHP 2003b; WNHP 2003b; Kagan 1989; Schassberger 1988; Gamon 1991; Lorain 1991; Heidel 1995; Lichthardt 1997; Caplow 2001, 2002a, 2002b; Elseroad and Taylor 2002a, 2002b; Gray and Lichthardt 2003a, 2003b; Lichthardt and Gray 2002; Mancuso 1993; Mancuso and Moseley 1994; Menke 2003; Hill 1998, 2000, 2001a, 2001b; Hill and Fuchs 2002, 2003; Hill and Gray 1999, 2000, 2004; Hill and Weddell 2003; Hill et al. 1999, Hill et al. 2001, 2003; Lesica 1998; Lesica and Martin 2003; Lesica and Shelly 1996; Tisdale 1961, 1986a; Weddell and Lichthardt 1998; Weddell 2001, 2002; Wentworth 1996; USFWS 1998, 1999, 2001; B. Benner, *in litt.* 1995, 2003c; F. Caplow, *in litt.* 2003b; K. Gray, *in litt.* 2003a, 2003b, 2003c; P. Lesica, *in litt.* 2003a, 2003b; J. Lichthardt, *in litt.* 2003a, 2003b; J. Hill, *in litt.* 2003a, 2003b; B. Martin, *in litt.* 2003; C. Menke, *in litt.* 2003b, 2003e; B. Weddell, *in litt.* 2003a, 2003c).

Invasive weeds degrade *S. spaldingii* habitat by altering its structure, composition, and function. Weeds outcompete, invade and displace native plant communities, altering species composition and relationships, and reducing species diversity. Weeds form monocultures that change the physical structure of the native communities, increase soil erosion resulting in changes in soil structure and chemical composition, and alter microclimate. Weeds alter ecological processes such as community productivity, soil water and nutrient dynamics, community successional patterns and disturbance cycles (D'Antonio and Vitousek 1992; Woods 1997; Walker and Smith 1997; Cox 1999; Olson 1999a).

The success of invasive weeds in degrading habitat can be attributed to a number of factors. Many invasive weeds occur outside of their natural range, being accidentally or deliberately introduced into new environments in which they did not evolve. Consequently they have no natural enemies to limit their reproduction and spread (Sheley et al. 1999a; Olson 1999a; Wilson and McCaffrey 1999; USDI 1994; USDI 1996b). Invasive weeds are capable of displacing native species because they are superior competitors and can utilize site resources such as water, nutrients, and light, more efficiently than native species (D'Antonio and Vitousek 1992; Olson 1999a). Invasive weed species also compete with native species for pollinators (Lesica and Heidel 1996). Invasive weeds often have rapid germination and growth, relatively short maturation times, high reproductive and seed output, a large seedbank, and efficient dispersal mechanisms that can facilitate their spread (Holdgate 1986; Hobbs 1991; Olson 1999a). Some weed species, such as Russian knapweed (*Centaurea repens*) and spotted knapweed (*C. maculosa*), have alleopathic characteristics, producing chemicals that inhibit other plant species (Whitson 1999; Sheley et al. 1999b). Weed invasion success is also enhanced if an invasive weed species comes from a climate that is similar to that in the invaded habitat (Holdgate 1992; D'Antonio and Vitousek 1992). Many invasive weeds, such as cheatgrass, common crupina and yellow starthistle, have their origins in the extensive arid regions where Europe, Asia, and Africa meet, including the Mediterranean area. These species are particularly adapted to the Mediterranean type

climates in the intermountain west (Mack 1986; Tisdale 1986a; Mancuso and Moseley 1994; Thill et al. 1999; Sheley et al. 1999c; Mosley et al. 1999).

Invasive weeds are especially efficient in utilizing limited soil water resources in semi-arid grasslands. Many perennial weed species such as leafy spurge and Russian knapweed have deep root systems up to 30 feet that can access water sources not available to many native plants (Olson 1999a). Others, like St. Johnswort, have roots that spread laterally just under the ground surface and likely monopolize critical summer precipitation (Hill and Gray 2000). The winter annual life cycle of many invasive weed species, i.e., annual bromes (cheatgrass, Japanese brome, soft brome, rattlesnake brome), yellow starthistle, and common crupina, enables greater use of soil moisture early in the growing season. Winter annual weed species begin growth in late fall and winter and can monopolize soil moisture resources before some native plant seedlings, like *S. spaldingii*, begin to grow in the spring (Sheley et al. 1999a; Thill et al. 1999; Mosley et al. 1999). Although the native bunchgrasses, bluebunch wheatgrass and Idaho fescue, also grow through late fall and winter (Daubenmire 1970; Weaver 1958), root growth during winter is much greater for cheatgrass than for a bluebunch wheatgrass seedling (Harris 1967). This difference in depth allows cheatgrass to absorb water from a much greater part of the soil profile than bluebunch wheatgrass. This advantage is most evident at the seedling stage of native plants. Commonly, annual plants cannot displace established perennial plants; however, annual plants can prevent perennial seedlings from establishing by outcompeting them at the seedling stage (Harris 1967). Perennial seedlings that develop slowly are at a disadvantage in competing with very efficient, fast-growing annuals like Japanese brome and yellow starthistle (Hironaka 1990).

The success of invasive weed species is related to their ability to alter ecological processes. Plant community successional pathways can be interrupted, and the introduced species creates a new stable state (Heady et al. 1992; Cox 1999) preventing re-establishment of the climax community, i.e., the cheatgrass domination of intermountain steppes (Tausch et al. 1993, 1995). These new stable-state, weed-dominated communities are also the sites of greatest vulnerability to newly introduced non-native invasive plants (Tisdale 1986a). Yellow starthistle has invaded sizeable areas in the Clearwater and lower Snake River valleys during the past decade, replacing annual bromes, which had been dominant there for many years (Tisdale 1986a). Invasive weeds can alter hydrological patterns and conditions, change soil structure and chemistry, and affect moisture-holding capacity of soils, resulting in increased erosion. Lower basal cover and crusting of exposed soils caused reduced infiltration and increased runoff in spotted knapweed plots versus bunchgrass plots (Lacey and Marlow 1990; Olson 1999a). Increasing frequency and intensity of fires can occur when fire-tolerant species such as cheatgrass contribute to increased fuel loads and inflammability. These fires further enhance the dominance of cheatgrass that recovers rapidly following fire and can suppress the growth of native species (Whisenant 1990; D'Antonio and Vitousek 1992; USDI 2000).

## Evidence of Weed Threat

Invasive, non-native weeds cause major alterations to structure, composition and function of native grassland vegetation. The presence of invasive weeds within *S. spaldingii* occurrences indicates alteration in structure and composition of *S. spaldingii* habitat has occurred, and likely, functioning is being negatively impacted as well. This section provides information on the pervasive presence of invasive non-native weed species within known *S. spaldingii* occurrences, and indicates the degree of habitat degradation caused by them. Little actual documentation exists of invasive weeds causing actual decreases in *S. spaldingii* populations; this information is needed.

A number of invasive, non-native weed species have been reported at *S. spaldingii* occurrences across its range of distribution. Based on Element Occurrence Records and other reports (IDCDC 2003b; MNHP 2003b; ONHP 2003b; WNHP 2003b; Caplow 2001, 2002a; Elseroad and Taylor 2002a, 2002b; Lichthardt and Gray 2002; Gray and Lichthardt 2003b; Hill and Gray 1999, 2000, 2004; Hill et al. 1999, 2001; Hill and Fuchs 2002, 2003; Hill and Weddell 2003), the most commonly reported invasive weed species are listed in Table 10 below. Tables 15A through 18A - Appendix B list invasive, non-native weed species reported for each EOR in each state.

**Table 10 – Common Invasive Weed Species at *Silene spaldingii* Occurrences.**

Invasive Weed Species		WA	ID	OR	MT
Common Name	Scientific Name				
Japanese brome	<i>B. japonicus</i>	X	X	X	
Cheat	<i>B. secalinus</i>	X			
Cheatgrass	<i>B. tectorum</i>	X	X		
Yellow starthistle	<i>Centaurea solstitialis</i>	X	X		
Diffuse knapweed	<i>C. diffusa</i>	X			
Spotted knapweed	<i>C. maculosa</i>				X
Rush skeletonweed	<i>Chondrilla juncea</i>	X			
Canada thistle	<i>Cirsium arvense</i>	X	X		
Teasel	<i>Dipsacus sylvestris</i>	X	X		
Leafy spurge	<i>Euphorbia esula</i>		X		
St. Johnswort	<i>Hypericum perforatum</i>	X	X	X	X
Dalmatian toadflax	<i>Linaria dalmatica</i>	X			
Kentucky bluegrass	<i>Poa pratensis</i>	X	X	X	X
Sulfur cinquefoil	<i>Potentilla recta</i>		X	X	X
Jim Hill mustard	<i>Sisymbrium altissimum</i>	X	X		
Ventenata	<i>Ventenata dubia</i>	X	X	X	

Some weed species are known only from *S. spaldingii* locations in specific states or regions, i.e. spotted knapweed in the intermontane valleys of northwestern Montana, rush skeletonweed and diffuse knapweed in the Channeled Scablands of Washington; whereas other species have a more widespread occurrence, i.e., Kentucky bluegrass, St. Johnswort, Japanese brome, and ventenata. The genera *Bromus* (annual bromes) and *Centaurea*

(knapweeds) are well represented at *S. spaldingii* locations. Annual bromes and knapweeds are considered some of the most problematic weeds in western states and cover large portions of *S. spaldingii* range (Tisdale 1961, 1986a; Roché et al. 1986; Roché and Roché 1988; Lacey 1990; Sheley et al. 1999b, 1999c; Mosley et al. 1999; Whitson 1999; DiTomaso 2000).

The following paragraphs list specific *S. spaldingii* occurrences or potential habitat where infestations of non-native, invasive weeds have been reported, and include indications of their rate of increase, degree of habitat degradation, and, in some cases, the direct effect on *S. spaldingii*.

Spotted knapweed is one of the most abundant and rapidly spreading invasive weed species within *S. spaldingii* range; it is present in every county in Washington, Idaho and Montana (Sheley et al. 1999b). Spotted knapweed has been reported at two *S. spaldingii* occurrences in Montana, EOR MT 001 – Dancing Prairie and EOR MT 003 – Black Bear Ranch (Table 16A – Appendix B). Black Bear Ranch, which had a total of 30 *S. spaldingii* plants in 1983, had a total count of 11 plants in 1990 and a heavy knapweed invasion where plants formerly occurred (MNHP 2003b).

Kentucky bluegrass is prevalent in *S. spaldingii* habitat. This invasive weed species has been reported within *S. spaldingii* occurrences in Idaho at EORs ID 005, 006, 007, 008, 009, 010, in Montana at EORs MT 001, 003, 005, in Oregon at EORs OR 001, 006, 007, 016, 019, and in Washington at EORs 006, 071, 078 (Tables 15 A through 18A – Appendix B). At the Kramer Prairie occurrence (EOR WA 006) in Washington, Kentucky bluegrass was not present in vegetation plots established in the 1950s (Daubenmire 1970); however, it was the most abundant invasive weed species in 1998 despite protection from fire and livestock grazing (Weddell and Lichthardt 1998; B. Weddell, *in litt.* 2003c). Increases in Kentucky bluegrass have been observed associated with increases in deciduous shrubs or ponderosa pine on steppe and shrub steppe fragments in Washington apparently resulting from habitat fragmentation and alterations in fire regime (R. Crawford, *in litt.* 2003). At Garden Creek Ranch (EOR ID 006), Kentucky bluegrass is common within *S. spaldingii* locations (Hill and Gray 1999, 2000, 2004; Hill et al. 1999, 2001; Hill and Fuchs 2002, 2003; Hill and Weddell 2003; Lichthardt and Gray 2002; Gray and Lichthardt 2003b; Menke 2003). Kentucky bluegrass is present within long-term monitoring transects for *S. spaldingii* at the Garden Creek Ranch (Hill and Weddell 2003; Hill and Gray 2003).

St. Johnswort, even though controlled periodically by bio-control insects, can develop very large, dense infestations (Tisdale 1976) that can threaten *S. spaldingii* and its habitat (Hill and Gray 2000). This invasive species has been reported within *S. spaldingii* occurrences in Idaho at EORs ID 006, 009, 012, in Montana at EORs MT 001, 011, in Oregon at EOR OR 007, and in Washington at EORs WA 006, 016, 029, 044, 050 (Tables 15A – 18A – Appendix B). This invasive species blooms at the same time as *S. spaldingii*, and competes with *S. spaldingii* for its pollinator, *Bombus fervidus* (Lesica and Heidel 1996). During pollination studies at Garden Creek Ranch, one-third of the time when the bumblebee pollinator for *S. spaldingii* started to visit *S. spaldingii* flowers, it ended by switching to St. Johnswort flowers (Lesica and Heidel 1996). St. Johnswort was considered one of the worst weed

threats at six *S. spaldingii* locations at the Garden Creek Ranch in 1999 (Hill and Gray 2000). At the Kramer Prairie occurrence in Washington, St. Johnswort was not present in vegetation plots established in the 1950s (Daubenmire 1970); but was present in 1998 despite protection from fire and livestock grazing (Weddell and Lichthardt 1998; B. Weddell, *in litt.* 2003c). St. Johnswort was the dominant weed species within *S. spaldingii* populations at the Fairchild AFB in Washington (EOR WA 044), with cover varying from 2-51% in 2001 and 18-30% in 2002 (Caplow 2001, 2002a).

Large areas of the Canyon Grasslands have been degraded and/or displaced by yellow starthistle, particularly portions in Idaho and Washington. This invasive weed species has been reported within *S. spaldingii* occurrences in Idaho at EORs ID 006, 008, 009, 010, 012, 014, 018 and in Washington at EOR WA 017 (Tables 15A and 18A – Appendix B). Yellow starthistle has displaced large tracts of Canyon Grassland communities in a relatively short period of time at the Garden Creek Ranch, spreading from approximately 150 acres in 1987 to over 2000 acres in 1998 (J. Hill, *in litt.* 1999; Hill and Gray 1999). Yellow starthistle was considered one of the worst weed threats at six *S. spaldingii* locations at the Garden Creek Ranch in 1999 (Hill and Gray 2000). Yellow starthistle is found in the vicinity of all *S. spaldingii* populations in Idaho (Lichthardt 1997). Yellow starthistle was present within long-term monitoring transects for *S. spaldingii* (Hill and Weddell 2003; Hill and Gray 2004) and other research plots (Menke 2003) at the Garden Creek Ranch.

Sulfur cinquefoil occurs frequently at *S. spaldingii* locations. This invasive weed species has been reported within *S. spaldingii* occurrences in Idaho at EORs ID 006, 008, 009, 010, 018, in Montana at EOR MT 001, in Oregon at EORs OR 001, 026, and Washington at EORs WA A large infestation of sulfur cinquefoil was present within the boundaries of one of the *S. spaldingii* clusters at Zumwalt Prairie in Oregon (EORs OR 022-026) (Elseroad and Taylor 2002a). Sulfur cinquefoil is the primary weed concern at Dancing Prairie in Montana (EORs MT 001, 011) (P. Lesica, *in litt.* 2003a, 2003b). Increases in sulfur cinquefoil following prescribed fires were observed at Dancing Prairie (Lesica and Martin 2003).

Annual bromes, cheatgrass, Japanese brome, soft brome (*B. mollis*), rattlesnake brome (*B. brizaeformis*), and cheat, are extensive within *S. spaldingii* range. These invasive weed species have been reported within *S. spaldingii* occurrences in Idaho at EORs ID 006, 009, 010, 012, 015, 018, in Oregon at EORs OR 016, 018, in Washington at EORs WA 005, 006, 011, 015, 016, 017, 044, 046, 060, 065, 066, 069, 070, 074, 076, 078, 079. Japanese brome quickly invaded disturbed areas where yellow starthistle was pulled within *S. spaldingii* subpopulations at Garden Creek Ranch (Hill et al. 2001; Hill and Fuchs 2002). Japanese brome, rattlesnake brome, and cheatgrass were present within long-term monitoring transects for *S. spaldingii* (Hill and Weddell 2003, Hill and Gray 2004) and other research plots at the Garden Creek Ranch (Menke 2003).

Two non-native weed species that are fairly recent invaders in *S. spaldingii* occurrences include bur chervil (*Anthriscus caucalis*) and ventenata. Bur chervil has only been reported within the *S. spaldingii* occurrence at Wawawai Canyon, Washington (EOR WA 009) and is considered a serious problem (Caplow 2002b). Ventenata has been reported within *S. spaldingii* occurrences in Idaho at EOR ID 006, 009, 012, and 018, in Oregon at EOR OR

019, and in Washington at EORs WA 044, 070, 071, 074, 076. *Ventenata* was present within long-term monitoring transects for *S. spaldingii* at the Garden Creek Ranch (Hill and Weddell 2003; Hill and Gray 2004).

Numerous EORs and reports indicate threats from invasive weed species to *S. spaldingii* occurrences or potential habitat, but do not specify which species are involved. *Silene spaldingii* occurrences with non-specified weed species were reported in Washington at EORs WA 006, 008, 012, 013, 014, 015, 019, 022, 024, 025, 032, 034, and 047 (Table 18A – Appendix B). At *S. spaldingii* locations in the Niarada area of Montana, i.e., EORs MT 003, 004, 005, 009, 010, much of the suitable habitat on private land is heavily grazed, and conversion of the native grasslands to vast tracts composed of five or six non-native weedy species has resulted (Schassberger 1988). Weddell and Lichthardt (1998) indicated that invasive non-native weed species are a widespread threat to Palouse and Canyon Grassland communities. Primary non-native invasive species in their study areas included annual bromes, yellow starthistle, *ventenata*, Kentucky bluegrass, St. Johnswort, and Canadian thistle. At a *S. spaldingii* site in Washington (EOR WA 013 - East Upper Steptoe), weeds invaded and dominated the site following a fire (Wentworth 1996).

Only a few studies looked specifically at the direct effect of invasive non-native weed species on *S. spaldingii*. Caplow (2002a) observed the greater the weed cover the lower the vigor (number of stems, branching, stem height, number of flowers per stem, phenology of flowers, and herbivore damage) of *S. spaldingii* plants at the Fairchild AFB, Washington. Menke (2003) observed that cheatgrass and yellow starthistle infestations did not appear to influence negatively the vigor of nearby *S. spaldingii* populations as indicated by similar plant height and comparable levels of flowering, fruit and seed set in invaded and uninvaded *S. spaldingii* habitat. In this study, although cheatgrass and yellow starthistle infestations occurred within *S. spaldingii* habitat, they did not occur within any *S. spaldingii* clusters; most infestations were more than 2 meters from any *S. spaldingii* clusters (Menke 2003; Menke and Muir 2003). At Garden Creek Ranch belt transects established in *S. spaldingii* clusters within a yellow starthistle infestation and a Kentucky bluegrass infestation in 1999 had no *S. spaldingii* plants in 2003 (J. Hill, *in litt.* 2003a). The lack of *S. spaldingii* plants in these transects may be related to the presence of dense weed infestations, but also could be related to prolonged dormancy and needs to be further investigated.

Many invasive non-native weed species currently not reported at specific *S. spaldingii* locations, occur in close proximity to known occurrences and have high potential to invade *S. spaldingii* habitat. At Garden Creek Ranch these species include rush skeletonweed, Russian knapweed, common crupina, oxeye daisy, bur chervil, and spotted knapweed (J. Hill, *in litt.* 2003b). At Kramer Prairie (EOR WA 006), yellow starthistle occurs on nearby south-facing slopes in close proximity to *S. spaldingii* populations on northerly slopes (Weddell and Lichthardt 1998); this species is known to invade northerly *S. spaldingii* slopes at Garden Creek Ranch (Hill and Gray 2000).

## Weeds Increasing Ecological Amplitude

Some invasive weeds expand their range over time due to local adaptations that can increase fitness and therefore density and dominance in new habitats (Parker et al. 2003). Cheatgrass has shown adaptive genetic variation corresponding to habitat (Ypsilantis 2003). In the last few decades, cheatgrass has spread beyond the sagebrush ecosystem, that it initially invaded, into salt desert, mountain brush and open conifer communities, especially the open Ponderosa pine types (Ypsilantis 2003) that are characteristic of habitat types for *S. spaldingii* at Turnbull NWR. Bur chervil (*Anthriscus caucalis*), a serious problem at a *S. spaldingii* location in Wawawai Canyon, Washington, may have undergone an adaptive genetic shift and expanded its ecologic amplitude from moist, shady sites usually beneath hawthorn trees to include shallow, rocky, south-facing slopes (Old 2000).

Yellow starthistle has shown genotypic response to habitat differences in California (Maddox 1981) and may be expanding its ecologic amplitude in the Canyon Grasslands of Idaho (Hill and Gray 2000). According to the literature, yellow starthistle is restricted to southerly slopes or level ground (Roché 1965; Callihan et al. 1989; Lass et al. 1999) where there is sufficient winter light to support rosette and taproot growth during the winter (Roché and Roché 1991; Roché et al. 1994). It was thought that northerly slopes, especially steep northerly slopes, do not receive enough winter light to support yellow starthistle due to the low angle of the sun in the winter (Roché et al. 1994). Initially, yellow starthistle at Garden Creek Ranch invaded only southerly slopes and did not appear to present a direct threat to the mesic Idaho fescue communities on northerly slopes that support *S. spaldingii*. However, in recent years, yellow starthistle at the Garden Creek Ranch is clearly invading the mesic Idaho fescue sites on north-facing slopes that support *S. spaldingii*. At seven different *S. spaldingii* subpopulations, numerous yellow starthistle infestations were mapped (Hill and Gray 2000; Hill et al. 2001). Two of these sites had slopes of at least 60% inclination. These infestations are currently successfully reproducing and persisting at these sites (Hill and Fuchs 2002, 2003). Long-term nested plot frequency monitoring in an Idaho fescue-prairie junegrass community at Garden Creek Ranch showed no yellow starthistle in 1993, but had several, small, scattered infestations in 1999 (Hill 2001b). Numerous yellow starthistle infestations were also located in other mesic Idaho fescue communities supporting *S. spaldingii* at the Garden Creek Ranch (Menke 2003). Whether yellow starthistle will seriously degrade or displace the mesic Idaho fescue habitats that support *S. spaldingii* is unknown at this time. Generally, more mesic, productive sites are thought to be less invisable than drier sites (Burke and Grime 1996). Sulfur cinquefoil is slower to invade cool slope exposures at Dancing Prairie in Montana (P. Lesica, *in litt.* 2003b).

## Major Contributing Factors for Weed Invasion

Disturbances to soil and vegetation, both natural (fire, soil slumps, animal burrowing and trailing, etc.) and anthropogenic (livestock grazing and trampling, cultivation, road-building, fire suppression activities, off-road recreational use, etc.) are major contributing factors to degradation of *S. spaldingii* habitat by weed invasion. Disturbances enhance the likelihood of weed invasion into native vegetation by affecting the availability of suitable microsites and reducing the ability of the native vegetation to compete (Fox and Fox 1986; Christensen

and Burrows 1986; Hobbs 1991; Hobbs and Huenneke 1992; Burke and Grime 1996; Lonsdale 1999; Keeley 2002).

Many of the weed species present in *S. spaldingii* habitat originated in Eurasia where they have evolved with agriculture and livestock grazing and are well adapted to grazing and soil disturbances (Mack 1986). Their superior competitive abilities allow them to take advantage of sudden releases of resources from disturbances at the expense of the slower-growing, native plants that characterize the native bunchgrass communities (Hobbs 1991; Hobbs and Huenneke 1992; Burke and Grime 1996).

The soil and vegetation disturbances associated with animals plays a significant role in weed establishment and spread (Hobbs and Huenneke 1992). Areas with open, bare ground are prime sites for aggressive weeds to establish (Burke and Grime 1996). Ground squirrel burrowing activity created microhabitats suitable for spotted knapweed colonization at Glacier National Park, Montana (Tyser and Key 1988). At Garden Creek Ranch, yellow starthistle seeds and infestations have been observed on the mounds of soil around animal dens (Hill et al. 1999). Rodent activity has created soil disturbance and bare open ground at several *S. spaldingii* populations (B. Benner, *in litt.* 1999; Caplow 2001; F. Caplow, *in litt.* 2003b; Hill and Gray 2004). Native ungulates and domestic livestock create soil and vegetation disturbances by trailing, digging, trampling and bedding activities (Hobbs and Huenneke 1992; Olson 1999a). Many *S. spaldingii* occurrences at Garden Creek Ranch show considerable disturbance from trailing of native ungulates, especially on steep slopes (Hill and Gray 2000).

The animals that create soil disturbances also serve as vectors for weed seed dispersal, often moving them into uninfested native vegetation. Animals can transport weed seeds in hooves, fur and feces (Mack 1986; Hobbs and Huenneke 1992; Olson 1999b). Ground squirrels at Glacier National Park were shown to carry seeds to their dens dispersing spotted knapweed seeds (Tyser and Key 1988). Native ungulates and domestic livestock also ingest weed seeds. Most of these seeds are destroyed in the gastrointestinal tract, however, 5-15% of leafy spurge and spotted knapweed seeds have been shown to pass through sheep, goats, and mule deer and some of those seeds remain viable (Lacey et al. 1992, Wallander et al. 1995; Olson et al. 1997). The leafy spurge seed passed through sheep and goats within five days (Lacey et al. 1992); however mule deer were still passing viable seeds on the 10<sup>th</sup> day (Wallander et al. 1995). Mule deer are common at many *S. spaldingii* locations at in the Canyon Grassland occurrences on Craig Mountain in Idaho (Hill and Gray 2000, 2004; Hill and Weddell 2003), and may be responsible for the distribution pattern of leafy spurge infestations at Garden Creek Ranch, which currently are few in number and widely separated (Hill and Gray 1999).

Domestic livestock grazing is a major contributing factor for weed invasion (see Livestock Grazing section that follows).

Fire is a natural disturbance that plays important ecological roles in regeneration, maintenance of species richness, nutrient recycling, ecosystem health and function; however, it can become problematic in the presence of invasive weed species. Fire can create

conditions that favor highly competitive, disturbance-adapted weed species and promote weed invasion (Christensen and Burrows 1986, Peters and Bunting 1994, Agee 1996; Asher et al. 1999; Keeley 2002; USDI 2000; Parks 2003).

Fire changes the composition and physical environment of plant communities and can alter competitive interactions among species (Melgoza et al. 1990; Defossé and Robberecht 1996). Fire reduces the cover, productivity, and competitive capacity of native bunchgrass vegetation (Daubenmire 1968c). Fire kills cryptogamic crust (Hill et al. 2003; Hill and Weddell 2003) that serves to protect communities from invading species (USDI 2000). The increase in open, bare ground following fire provides prime sites for aggressive weeds to establish (Burke and Grime 1996). Bare ground is also susceptible to increasing soil disturbance from erosion and trampling from grazing wildlife and/or livestock attracted to green-up following fire (Burke and Grime 1996). The reduction in native competition, exposure of bare ground, and increased soil disturbance associated with fire, favor establishment of invasive weeds.

The release of light and mineral nutrient resources following fire favors invasive weeds. Fire significantly increases levels of soil phosphorus, nitrate and ammonium (Hobbs and Atkins 1988), and the reduction in cover of native vegetation following fire results in increases in light. Invasive weed species show an enhanced ability to capture the elevated quantities of light and mineral nutrient resources compared to the native species (Bazzazz 1983; Hobbs and Huenneke 1992; Burke and Grime 1996). Invading species are well-equipped to behave opportunistically when resources are suddenly released at disturbed sites (Hobbs 1991). Invasions are greatly increased where different types of disturbances occur together, such as soil disturbance, vegetation disturbances, and disturbance from increase in nutrients (Hobbs and Atkins 1988; Burke and Grime 1996).

Fire results in a more xeric environment for plant establishment that can favor fast-growing competitive weed species over native bunchgrasses. The darkened bare soil absorbs more solar radiation and warms up quicker and earlier, causing higher evaporation rates compared to undisturbed communities (Daubenmire 1968c). Cheatgrass successfully competes with native species for the limited soil water at post-fire sites despite these plants being well established and able to reach deeper levels in the soils (Melgoza et al. 1990; Defossé and Robberecht 1996). Plants that can access moisture earlier and more efficiently, such as the annual brome species, greatly enhance their capability to exploit soil resources after fire and enhance their status in the community.

Many weed populations rapidly increase following fire (Asher 1998; Asher et al. 1999). Spotted knapweed increased following fire in grassland and forested habitats in eastern Washington and western Montana (Sheley et al. 1999b). Higher amounts of weeds have been documented following burns within *S. spaldingii* habitat. Two late-season wildfires in Idaho fescue grasslands at the Garden Creek Ranch killed cryptogamic crust and reduced bunchgrass cover with a concomitant increase in cover of bare ground (Hill et al. 2003; Hill and Weddell 2003; Menke 2003). Following an August 2000 fire within an Idaho fescue-bluebunch wheatgrass community, the basal cover, density, and frequency of Japanese brome and yellow starthistle were significantly greater in burned plots than in unburned plots (Hill

et al. 2003). At an Idaho fescue-rose community that supports a population of *S. spaldingii* at Garden Creek Ranch, hand-pulling control of yellow starthistle infestations showed annual declines; however, following a September 2001 fire, the density of yellow starthistle increased in half of these infestations (Hill and Fuchs 2003). Significant increases in non-native annual brome species were observed following a recent wildfire in long-term vegetation sampling plots established in Canyon Grasslands on Craig Mountain in the 1960s through 1980s by Dr. Ed Tisdale (Gray and Lichthardt 2003a).

In northeastern Oregon, annual bromes increased dramatically in FEID-AGSP habitat types following wildfires (Johnson 1998). At a *S. spaldingii* site in Washington (EOR WA 013 - East Upper Steptoe), weeds invaded and dominated the site following a fire (Wentworth 1996). At Dancing Prairie, recruitment of sulfur cinquefoil (*Potentilla recta*) was higher in prescribed spring and fall burn plots compared to control plots one-year following the burn, and density was greater in herbicide plots that were burned than those that were not (Lesica and Martin 2003).

Weed invasion is a prevailing, ongoing, and imminent threat to remaining *S. spaldingii* habitat. The major contributing factors for weed invasion, soil and vegetation disturbances, livestock grazing and fire, are expected to continue with increasing human activity. Although considerable habitat has been degraded, many areas are currently uninvaded or minimally invaded, but are highly susceptible to invasion. Action is needed to protect these areas from weed invasion and activities that can promote the establishment and spread of invasive weeds.

### **Herbicide Drift**

Herbicides used to control invasive weeds can drift into non-target areas and degrade *S. spaldingii* habitat and threaten *S. spaldingii* populations (Lichthardt 1997; J. Hill, *in litt.* 1999; J. Hustafa, personal communication 1998; J. Kagan, personal communication 1998; S. Riley, personal communication 1999). Herbicides can harm native bunchgrass communities by altering their composition, structure and function. Selective herbicides can effectively remove certain groups of plants and shift competitive balances in the plant community. Broadleaf herbicides used to control forbs such as yellow starthistle, have potential to negatively impact native perennial forbs, including *S. spaldingii* and other rare perennial forbs within *S. spaldingii* habitat, such as Palouse goldenweed (*Haplopappus liatrifolius*) and Palouse thistle (*Cirsium brevifolium*). Herbicides used to control annual bromes can negatively impact native bunchgrasses, particularly bunchgrass seedlings. Some herbicides kill all aboveground vegetation and can open up the community to weed invasion if desired vegetation is not reestablished (Lass et al. 1999).

Herbicide drift is a particular problem in the Palouse cropland area where agricultural fields are commonly sprayed aerially. Here, herbicides can further alter small remnants of *S. spaldingii* habitat that are already impacted by edge effects, and negatively impact the small populations of *S. spaldingii* they support. In Idaho, two *S. spaldingii* populations (EORs ID 002 – Genesee South and ID 005 – Upper Hatwai Creek) are particularly vulnerable to herbicide drift because of their close proximity to cropland (Lichthardt 1997). Numerous

EORs in the Washington Palouse Grasslands recorded herbicide spraying and/or drift as a threat to *S. spaldingii* (EORs WA 006, 008, 009, 011, 018, 025, 051) (WNHP 2003b).

Aerial spraying is also commonly used to control weeds in the rugged, inaccessible, terrain of Canyon Grasslands. Unpredictable wind currents and high temperatures that can volatilize herbicides, i.e., cause them to become suspended in the air, can carry herbicides into non-target areas (Bussan and Dyer 1999; USDI 2003). Depending on barometric pressure and wind conditions, volatilized herbicides can affect non-target areas long distances from the application site (Bussan and Dyer 1999). Native plants, including the rare plant *Haplopappus liatrisiformis*, demonstrated the effects of herbicides following an aerial spraying project ¼ mile away at the Garden Creek Ranch. These plants were in close vicinity to a known *S. spaldingii* population (J. Hill, *in litt.* 2003a). Three Canyon Grassland *S. spaldingii* populations in Idaho (EOR ID 006 - Garden Creek Ranch; EOR ID 014 - Salmon Oxbow; EOR ID 017 - Billy Creek North) have reported herbicide drift from aerial spraying as a threat to *S. spaldingii* (IDCDC 2003b).

The threat from herbicide drift is greater in areas with little active management to protect *S. spaldingii* and its habitat, such as privately owned land. The use of herbicides in vicinity of *S. spaldingii* also varies with government agencies. The BLM requires surveys for rare plants be conducted prior to spraying an area (USDI 2003); however the Idaho Fish and Game Department is not required to survey for rare plants before spraying an area (J. Lichthardt in G. Glenne, *in litt.* 2003b).

It is likely that herbicide drift will continue to be a threat to *S. spaldingii* and its habitat in both the cropland regions of the Palouse Grasslands and in the rugged, inaccessible regions of the Canyon Grasslands where aerial herbicide spraying is often the most economical and efficient means of weed control.

### **Livestock Grazing**

Livestock grazing has major negative effects on *S. spaldingii* and its habitat. Prolonged heavy grazing pressure from domestic livestock has resulted in major alterations of the structure, function and composition of the fescue bunchgrass communities that support *S. spaldingii* and promoted weed invasion (Schassberger 1988; Lorain 1991; Gamon 1991; Lichthardt 1997; B. Benner, *in litt.* 1999; 2003c). Selective grazing by cattle has decreased palatable native bunchgrasses while species of low palatability and preference, primarily non-native invasive weed species, have increased in abundance (Weaver 1958; Mitchell 1958; Daubenmire 1970; Mueggler and Stewart 1982). These bunchgrass communities did not evolve with large grazing ungulates comparable to the bison populations of the Great Plains (Daubenmire 1970, 1978; Mack and Thompson 1982; Tisdale 1961, 1983, 1986a). The caespitose nature of native bunchgrasses make them very susceptible to damage from prolonged and intense grazing pressure from domestic livestock (Mack and Thompson 1982; Mack 1986). Regions with no recent history of grazing are often dominated by plants that lack tolerance mechanisms to grazing (Mack and Thompson 1982). At disadvantage in competing for limited soil water and nutrients (Olson 1999b), the native bunchgrasses have declined in abundance with grazing and been replaced largely by non-native, winter annual

weed species, particularly those of Mediterranean and Eurasian origin like the annual bromes (*Bromus tectorum*, *Bromus japonicus*, *B. mollis*, and *B. brizaeformis*) and yellow starthistle (Tisdale 1961, 1986a; Daubenmire 1942, 1970). These species are particularly well adapted to the mild wet winters and hot, dry summers occurring in the tri-state area of *S. spaldingii* occurrence, and are extremely difficult to eradicate once established (Tisdale 1961, 1986a; Daubenmire 1942, 1970). Seeds of yellow starthistle can remain viable in the soil for up to 10 years (Callihan et al. 1989). Domestic livestock are likely vectors for introducing weeds into native vegetation, and the soil disturbance they create provides sites for weed establishment (Lichthardt 1997).

The negative effects of heavy grazing are often irreversible. Kentucky bluegrass, cheatgrass, and yellow starthistle are capable of assuming dominance as grazing pressure eliminates native perennial bunchgrasses (Daubenmire 1970; Olson 1999b). These invasive weed species create a new stable state that prevents the re-establishment of the climax bunchgrass community (Heady et al. 1992; Cox 1999; Tausch et al. 1993, 1995). Kentucky bluegrass can assume dominance in the more mesic grazed Idaho fescue-snowberry and Idaho fescue-rose habitat types that support *S. spaldingii* (Daubenmire 1970). Kentucky bluegrass has a high tolerance to grazing and a strong system of vegetative reproduction, producing a rhizomatous sward capable of excluding all other species (Daubenmire 1970; Tisdale 1986a). Cheatgrass can assume dominance in the drier grazed bluebunch wheatgrass-fescue and sagebrush-Idaho fescue habitat types that support *S. spaldingii* (Daubenmire 1970). Native bunchgrass seedlings cannot compete with the aggressive, fast-growing winter annual bromes (Daubenmire 1970; Hironaka 1991). The dominance of bunchgrass communities by weeds such as yellow starthistle, spotted knapweed, and leafy spurge may also represent new steady states (Olson 1999b). This new weed-dominated stable state cannot be rectified with removal of cattle or reducing their numbers (Olson 1999b). Removal of grazing did not improve degraded portions of Dancing Prairie, Montana (B. Martin, *in litt.* 2000). Heavy livestock grazing can also cause long-term soil changes in fescue grasslands. Johnston et al. (1970) observed reductions in percent of organic material, increases in soil temperature and decreases in percent soil moisture in heavily grazed fescue grasslands.

Livestock grazing also has direct negative effects on *S. spaldingii* populations. Livestock grazing is especially detrimental to *S. spaldingii* with late summer grazing or heavy grazing (Kagan 1989; R. Taylor, *in litt.* 2003a; G. Glenne, *in litt.* 2003a). Grazed *S. spaldingii* plants have little time and limited moisture resources for regrowth if grazed late in the season. Trampling and breakage of plants have been associated with livestock grazing where stocking rates are high or duration too long (B. Benner, *in litt.* 1995, 2003d). Trampling by livestock when the ground is wet in the early spring results in increased breakage of the caudex of *S. spaldingii* plants compared to trampling during drier periods (B. Benner in G. Glenne, *in litt.* 2003b). Trampling of livestock also has potential to endanger *S. spaldingii*'s pollinator, the ground-nesting bumblebee, *Bombus fervidus* (Sugden 1985; Lesica 1993; Lesica and Heidel 1996).

The risk of further habitat degradation from livestock grazing is high, especially in rugged terrain regions of *S. spaldingii* range where other land uses are prohibitive. Livestock grazing has been and will continue to be an important economic force in western North

America (Olson 1999b; Conner et al. 2001), and will continue to be a primary disturbance and major contributing factor for weed invasion.

### **Alteration of Fire Regimes**

Life histories of native plant species are often fine-tuned to a particular regime of fire frequency, intensity and seasonal distribution (Keeley 1986). Alterations of fire regimes, including fire suppression, increasing fire severities and frequencies, and out-of-season fires, have potential to degrade *S. spaldingii* habitat.

#### Fire Suppression

Fire suppression and habitat fragmentation have altered fire regimes within *S. spaldingii* range. In some fragments of the steppe and sagebrush steppe in Washington, it has been observed that deciduous shrubs and/or ponderosa pine have been increasing, particularly sites near forests or woody riparian areas, presumably due to landscape fragmentation and changes in fire regime (R. Crawford, *in litt.* 2003).

Community successional changes associated with fire suppression can threaten *S. spaldingii* in grasslands that are fire-maintained such as the rough fescue grasslands in northwestern Montana. In the absence of fire, community succession continues toward the potential natural community for the site, in most cases, a shrub- or tree-dominated habitat type. Fire is used to maintain the community at an early seral stage, typically a grass-dominated system (Conner et al 2001). In the absence of fire, *S. spaldingii* in these grasslands is threatened by woody encroachment. *Silene spaldingii* is unable to persist in the dense shade created by this vegetation (B. Benner, *in litt.* 1993). Fire played important roles in structuring the vegetation in intermontane valleys of northwestern Montana by preventing the establishment of woody vegetation (Koterba and Habeck 1971; Barrett and Arno 1982). In the absence of fire, *S. spaldingii* and its rough fescue habitat at Dancing Prairie Preserve, Montana, are threatened by invasion of ponderosa pine (Lesica 1997a, 1999; Lesica and Martin 2003).

Fire suppression at sites where large accumulations of plant litter suppress recruitment of *S. spaldingii* is a threat. Fire has a positive effect on recruitment of *S. spaldingii* at Dancing Prairie Preserve by reducing the build-up of large amounts of accumulated litter from the productive rough fescue grasslands that can inhibit *S. spaldingii* seedling establishment (Lesica 1999). Fire increased fecundity and recruitment of the native prairie species, *Silene regia*, by removing accumulated litter, vegetation cover, and increasing ground-level light (Menges 1995). Large litter buildup resulting from fire suppression can create more severe fires and cause more damage to the native plant community. Fire severity influences the ability of a species to compete, and maintain its presence in the community. The shorter the return interval between fire events, the less damaging and dramatic the changes in plant composition (Johnson 1998; Antos et al. 1983).

#### Increasing Fire Frequencies

Fires in areas dominated by fire-tolerant cheatgrass contribute to increased fuel loads and inflammability, resulting in more frequent and intense fires (Whisenant 1990; Peters and Bunting 1994; Keeley 2002). Prior to the arrival of white settlers, fire-return intervals in the sagebrush steppe probably varied between 60-110 years, but much of the region now burns at intervals of less than 5 years (Whisenant 1990). Human-caused fires are also increasing near urban areas and with recreational backcountry use.

Increasing fire frequencies threaten *S. spaldingii* and its habitat. *Silene spaldingii* is one of the few perennial native plants in the fescue bunchgrass communities that is late-maturing (Daubenmire 1970; Tisdale 1986a). Fires occurring during the normal fire season in summer and fall cause large losses of the seed crop (Hill and Weddell 2003). *Silene spaldingii* is a long-lived plant that produces relatively large amounts of seed (Lesica 1997a). It is adapted to historic fire frequencies and has persisted in spite of loss of seed crops to fire. However, if fire frequencies increase, more seed will be lost and long-term persistence of *S. spaldingii* may be threatened. Increasing fire frequencies may also be a problem for *S. spaldingii* recruitment, which has been shown to be infrequent and sporadic (Lesica 1997a). Short fire cycles can impede re-establishment of perennials (Antos et al. 1983), and may hinder *S. spaldingii* recruitment.

Increasing fire frequencies enhance weed invasion (Whisenant 1990; Peters and Bunting 1994; USDI 2000; Keeley 2002). When the natural fire regime is altered, even highly fire adapted native plant communities may be vulnerable to alien competition (Keeley 2002). The ubiquity of Eurasian annuals with superior ability to exploit disturbed areas has changed the consequences of fire in the Pacific Northwest Bunchgrasses biome (Weddell 2001). Increased fire frequency alters biotic and abiotic factors enough to change perennial-dominated communities to annual-dominated communities (Tausch et al. 1995). Increased fire frequencies may threaten *Bombus fervidus*, the ground-nesting pollinator of *S. spaldingii* (Lesica and Heidel 1996).

### Out-of-Season Fire

Pacific Northwest Bunchgrass communities have evolved with fire and have adaptations to survive fires in the normal fire season during summer and fall (Daubenmire 1968c; 1970; Tisdale 1986a). Growth cycles of the native bunchgrasses and the majority of perennial forbs within *S. spaldingii* communities follow closely the climate of the region (Daubenmire 1970; Tisdale 1986a). Seasonal activity patterns are especially important in the tri-state area where the lowest amounts of precipitation occur when temperatures are the highest (Weddell 2001). Many plants enter a dormant state during the summer drought period (Daubenmire 1970). Within the Idaho fescue-snowberry and Idaho fescue-rose communities that support *S. spaldingii*, the major bunchgrasses and approximately 1/3 of the native perennial forbs produce new foliage at the beginning of the rainy season and are photosynthetically active from autumn through spring (Daubenmire 1970, 1972).

Fires are least harmful to these plants when they are summer dormant and not actively growing (Daubenmire 1968c). Their aboveground tissues are not damaged by burning, and they regenerate readily from underground organs. Thus, summer and fall fires cause minimal

damage to many native species in *S. spaldingii* communities (Daubenmire 1968a; Menke 2003). Fires are most harmful when they are actively growing in late fall, winter and early spring (S. Bunting, personal communication, 1999). Spring and late fall burns harm cool-season, C3 grasses, like bluebunch wheatgrass and Idaho fescue that are adapted to a fire regime occurring during summer months (Towne and Owensby 1984; Steuter 1987; Ewing and Engle 1988). Burning is especially detrimental if it occurs when carbohydrate reserves in underground storage organs have been depleted to produce foliage or fruits (Wright and Klemmedson 1965; Daubenmire 1968c; Beardall and Sylvester 1976). Early spring fires are more likely to subdue native perennial bunchgrasses during their primary period of growth, diminishing their competitive capacity and thus allowing invasive weeds to expand rapidly (Daubenmire 1968c; Wright 1972). Out-of-season prescribed fires have a high probability of harming native bunchgrass communities that support *S. spaldingii*. Most prescribed fires are conducted during the moist conditions of early spring and late fall when the fires can be more easily controlled; this is particularly true in areas of steep terrain of the Canyon Grasslands (Hill 2001a).

The risk of further habitat degradation from alteration of fire regimes is high. Habitat fragmentation associated with urban and residential development and associated fire suppression activities is expected to increase in the future. Increasing fire frequency and severity is expected to increase as weed invasion increases. Safety issues involved with controlled burns, especially those in steep terrain, limit their use to moist periods during late fall and early spring.

#### **OVERUTILIZATION FOR COMMERCIAL, RECREATIONAL, SCIENTIFIC, OR EDUCATIONAL PURPOSES**

*Silene spaldingii* is not a human food source nor is it currently of commercial horticulture interest, and overutilization is not a threat at the present time. However, listing a species and publicizing its rarity can make it more susceptible to collection for scientific and/or commercial purposes. Some *S. spaldingii* populations are small enough that even limited collection could have adverse impacts.

#### **DISEASE AND PREDATION**

##### **Herbivory and Predation**

Herbivory and predation of *S. spaldingii* by native herbivores, i.e., deer, elk, rodents and insects, has been observed at many *S. spaldingii* locations across its range (Kagan 1989; Lesica 1988a; Youtie 1990; Gamon 1991; Heidel 1995; B. Benner, *in litt.* 1999; Hill and Gray 2000; Caplow 2001; Caplow 2002a; Hill and Weddell 2003; Hill and Gray 2004). These interactions likely occurred historically; however, the effects of these activities have become more significant as population size and distribution of this species decreases (USFWS 2001). Grazing herbivory and trampling by domestic livestock can be a significant threat to *S. spaldingii* populations as well. *Silene spaldingii* reproduces only by seed, any activity that affects reproduction, i.e. removal of flowers or seed capsules, or breakage of plants by trampling, can decrease reproductive output.

## **Native Herbivores**

Grazing of *S. spaldingii* by native herbivores can significantly reduce seed production and is considered a significant threat to *S. spaldingii* (Kagan 1989; Lesica 1993; Heidel 1995; B. Benner, *in litt.* 1999; Hill and Gray 2000, 2004; Hill and Weddell 2003). In 1995 at the Kramer Prairie site in Washington (EORs WA 006, 024), the vegetation in the area where *S. spaldingii* plants had occurred previously, had been heavily grazed, and no *S. spaldingii* plants were found (Wentworth 1996). Since livestock have been excluded from this site, it is likely deer were responsible for the herbivory. In Idaho at EOR ID 017 - Billy Creek North, several plants had been grazed so that no inflorescences were present (IDCDC 2003b). At Garden Creek Ranch (EOR ID 006), 62% of 453 stems monitored at 3 different *S. spaldingii* populations showed evidence of grazing herbivory. In long-term monitoring transects established within *S. spaldingii* populations at the Garden Creek Ranch, 50% of *S. spaldingii* stems had been grazed in 2002 and ~70% of stems had been grazed in 2003. Domestic livestock have been removed from the Garden Creek Ranch, including both BLM- and TNC-owned lands, since 1994 and elk and deer are common. Typically this type of herbivory occurs after seedheads have formed and results in the removal of the entire top portion of the plant and any inflorescence that may have been present. Often all stems of a multi-stemmed plant were grazed at similar height (Photos 11 and 12 – Appendix E). Since the majority of native ungulate herbivory occurred late in the growing season when precipitation is at its lowest, little or no regrowth occurred following grazing (Hill and Gray 2000, 2004; Hill and Weddell 2003).

## **Rodents**

Rodent activity appears to be a significant factor affecting survival of several *S. spaldingii* populations. Rodent runways, burrows, and holes have been reported at sites where *S. spaldingii* plants occurred previously (B. Benner, *in litt.* 1999; Caplow 2001; Hill and Gray 2004). At a BLM monitoring site in eastern Washington where numerous plants were marked with stakes and metal tags, many plants were either broken off or missing completely. Rodent burrows and runways were found where many of the plants had been (B. Benner, *in litt.* 1999). At Garden Creek Ranch, many previously recorded *S. spaldingii* plants were missing in monitoring transects where rodent runways occurred. Typically the runways were dug slightly into the soil surface and all vegetation, including cryptogamic crust, had been removed (Hill and Gray 2004). At Garden Creek Ranch pocket gophers are suspected of being responsible for 7% of monitored plants that were entirely missing, with only a hole in the ground remaining (Hill and Gray 2000). Rodent herbivory and activity, including holes at the base of plants, were observed at Fairchild AFB, Washington (EOR WA 044) (Caplow 2001). Rodent activity appears to vary from year to year. At Fairchild AFB, Washington, far less rodent activity was observed in 2002 than 2001 (Caplow 2001, 2002a). At Garden Creek Ranch, little activity was observed in 2002 but considerable activity was noted in 2003 (Hill and Weddell 2003; Hill and Gray 2004).

## **Insects**

Insect predation on flowers and fruits is also a significant threat to *S. spaldingii* populations (Kagan 1989; Gamon 1991; B. Benner, *in litt.* 1999; Hill and Gray 2000; Hill and Weddell 2003). Lesica (1988a) observed 2 cm-long caterpillars, presumably lepidopteran larvae, feeding on flowers of *S. spaldingii* at the Dancing Prairie Preserve (EORs MT 001, 011) in Montana. At Garden Creek Ranch (EOR ID 006), larvae were observed emerging from holes in the capsules, and many empty capsules with holes had larval or insect frass inside (Photos 13 and 14 – Appendix E). Approximately 30% of reproductive structures (buds, flowers, developing capsules, and mature seed capsules) on 70 monitored reproductive stems showed signs of insect herbivory (Hill and Gray 2000). Predation by noctuid larvae was a major factor in reducing the number of mature *S. spaldingii* fruits at Clear Lake Ridge (EORs OR 003-006, 015) in Oregon (R. Taylor, *in litt.* 2003a). A high percentage of seed heads were also destroyed by a seed weevil at Clear Lake Ridge (Kagan 1989; Youtie 1990). At a large *S. spaldingii* population in the Umatilla National Forest in Washington (EORs WA 049, 050, 056, 058), insect consumption of seeds resulted in numerous empty capsules (S. Riley, personal communication 1999). Insect damage to foliage of *S. spaldingii* plants has also been observed (Lichthardt 1997; Hill and Gray 2000, 2004; Hill and Weddell 2003). Heidel (1979) recorded 90% capsule penetration at the Kramer Prairie, Washington, location (EORs WA 006, 024) and 50% at the Cold Springs, Idaho, location (EOR ID 007) with corresponding predation of most seeds. She hypothesized the predation was due to a coleopteran, possibly a member of the Chrysomelidae.

### **Domestic Livestock**

Domestic livestock grazing threatens *S. spaldingii* in two major ways. The primary threat from livestock grazing is a threat to *S. spaldingii*'s habitat that alters the composition, structure and function of the native bunchgrass communities that support *S. spaldingii* and promotes weed invasion (this threat is discussed in the Habitat Destruction/Modification Threat section). A second threat from livestock grazing is direct consumption of *S. spaldingii* plants.

Grazing herbivory by domestic livestock represents a significant threat to *S. spaldingii* populations by damaging plants and reducing reproductive effort. Heavy grazing, especially in late summer, can destroy individuals and populations of *S. spaldingii* (Kagan 1989; Gamon 1991). Intensive grazing of *S. spaldingii* plants has been observed late in the season at Zumwalt Prairie in Oregon (Elseroad and Taylor 2002a). During a 10-day period in July, cattle grazed 75% of *S. spaldingii* flower and seed heads at the Zumwalt Prairie (G. Glenne, *in litt.* 2003a). *Silene spaldingii* is often heavily utilized late in the season when it may be one of the only green plants present. Droughty conditions can exacerbate the negative effects of livestock grazing (Heidel 1979; Schassberger 1988). Plants grazed late in the season have little time and available moisture for regrowth (B. Benner, *in litt.* 2003c). It is suspected that an Oregon site (EOR OR 002 - Darr Flat) was extirpated as a result of overgrazing (Schassberger 1988).

### **Disease**

No diseases are known for *S. spaldingii*.

## THE INADEQUACY OF EXISTING REGULATORY MECHANISMS

Fifty-two percent of *S. spaldingii* populations (50% of EORs) occur on private lands; this does not include the 12% of populations in which a private individual or corporation is a part-owner (IDCDC 2003b; ONHP 2003b; MNHP 2003b; WNHP 2003b; BCCDC 1999). *Silene spaldingii* populations on privately owned lands are not adequately protected by existing regulatory mechanisms. Although each of the four states and the Canadian province in which *S. spaldingii* occurs consider *S. spaldingii* as rare and imperiled (in addition, the state status is Endangered in Oregon and Threatened in Washington), no legislation provides protection or insures survival for *S. spaldingii* on private land. Therefore, land use is at the discretion of the landowner and may involve uses that could negatively impact or extirpate populations such as certain livestock grazing practices, herbicide use, conversion to crops, residential development, recreational off-road use, etc.

Federal agencies, such as BLM and U.S. Forest Service (USFS), have laws and regulations that address the protection of *S. spaldingii* on their land, i.e., the Federal Land Policy and Management Act and the National Forest Management Act. The U. S. Department of Defense at Fairchild Air Force Base, Washington, with the assistance of the Washington Natural Heritage Program, has developed a management plan for *S. spaldingii* on their land (Rush and Gamon 1999). A private conservation organization, The Nature Conservancy, whose goal it is to protect habitat and natural communities, owns and/or manages three of the largest *S. spaldingii* populations (Dancing Prairie, Zumwalt Prairie, and Garden Creek Ranch). Land exchanges and acquisition, ongoing surveys, research, long-term monitoring, management plans, weed control, implementation of sustainable grazing practices, and restoration efforts are in progress to protect *S. spaldingii* populations on these lands (Caplow 2001, 2002a; B. Benner, *in litt.* 2003a, 2003b, 2003c, 2003d; Lesica 1997a; Lesica 1993; Lesica and Heidel 1996; P. Lesica, *in litt.* 2003b, 2003c; Lichthardt and Gray 2002; Gray and Lichthardt 2003b; Hill 1998; Hill et al. 1999, 2001; Hill and Gray 2000, 2004; Hill and Fuchs 2002, 2003; Hill and Weddell 2003).

The regulatory and conservation actions taken by federal and state agencies and conservation organizations, though a step in the right direction, are inadequate to eliminate the major threats to *S. spaldingii*. Surveys, monitoring, evaluation and research all require considerable time and resources. Monitoring studies of species with prolonged dormancy such as *S. spaldingii* require longer periods of time to determine population size and therefore to assess management actions. Many of the threats are difficult to remedy, i.e. eradicating or controlling invasive, non-native plant species that are well adapted to the climate across *S. spaldingii* range and occur in rugged inaccessible terrain. Control of severe weed infestations and required restoration activities commonly exceed the market value of the land (Asher 1998). Although regulatory mechanisms and conservation actions have reduced a number of threats, they have not eliminated them, and they will likely remain threats for *S. spaldingii* in the future.

## **OTHER NATURAL OR MANMADE FACTORS AFFECTING *SILENE SPALDINGII* CONTINUED EXISTENCE**

Other natural or manmade factors negatively impacting the continued existence of *S. spaldingii* include: 1) habitat fragmentation and small populations and 2) prolonged drought and global warming.

### **Habitat Fragmentation and Small Populations**

Habitat fragmentation has resulted in loss of *S. spaldingii* habitat and reduced population size. Alteration of habitat on the remaining fragments can effectively reduce amount of suitable habitat for *S. spaldingii*. Populations of *S. spaldingii* on habitat fragments are isolated from each other by considerable distances and are subject to pollinator limitations, inbreeding depression and high risk of extirpation from chance environmental events. Small populations are subject to a number of genetic hazards that can threaten their long-term viability.

### **Habitat Fragments**

Site conditions on the habitat fragments are altered or modified as a result of the fragmentation process. The amount of edge habitat, the boundary zone around the fragment, increases and the interior habitat in the core of the fragment decreases. Species that are dependent on the interior habitat suffer while edge-dependent species, such as invasive species thrive (Defenders of Wildlife 2003). The increased edge habitat impacts the fragment's microclimate and in turn alters plant communities present on the fragment (Jules 1997; Major 2003; Defenders of Wildlife 2003). The edge habitat is typically drier than the interior habitat (Jules 1997; Briggs and Walters 1997). *Silene spaldingii* and the associated mesic fescue community, which are adapted to the interior habitat, may not be able to persist in the drier conditions in the edge habitat, and effective available habitat within the fragment is further reduced. These fragments may not be of sufficient size to support their natural biodiversity (Soulé 1987), and species diversity within the community typically decreases (Noss and Csuti 1994). The habitat fragments also suffer degradation from weed invasion, herbicide drift, livestock grazing, alteration of fire regimes (see the Destruction/Modification of Habitat Threat section of this report). Some small fragments may be altered to the point that they no longer have potential to support *Silene spaldingii* or its associated mesic fescue community.

### **Isolation and Pollination Limitation**

The majority of fragmentation that has occurred in *S. spaldingii* habitat has resulted from conversion to cropland, and remaining fragments tend to be small in size and isolated from each other by large expanses of agricultural fields (Lichthardt and Moseley 1997a, 1997b). Species movements are disrupted, including the movement of pollinators and seed dispersers (Hobbs and Huenneke 1992). The bumblebee, *Bombus fervidus*, a primary pollinator for *S. spaldingii* (Lesica 1993; Lesica and Heidel 1996) is critical for the persistence of *S. spaldingii*, which requires pollination by insects for sexual reproduction, maximum seed set,

maintenance of fitness, and population viability (Lesica 1993). Although *S. spaldingii* is primarily out-crossing, it is capable of selfing. Selfed progeny; however, suffer from loss of fitness due to inbreeding depression. Excluding *Bombus fervidus* from *S. spaldingii* plants resulted in significantly lower proportion of fruits matured, seeds per fruit, germination, seedling growth and seedling survival, and loss of fitness due to inbreeding depression was at least 97% (Lesica 1993).

Small populations of *S. spaldingii* located on isolated habitat remnants may be at high risk of pollinator reduction or exclusion. The small number of *S. spaldingii* plants in these populations do not provide enough nectar to attract the pollinator, and the surrounding agricultural fields do not support habitat for the pollinator which needs a continuous source of nectar provided by a diversity of plants flowering throughout the season (Biodiversity Legal Foundation 1995). Competition for the pollinator services of *Bombus fervidus* with other flowering species, such as St. Johnswort (Lesica and Heidel 1996), also has the potential to reduce pollination of *S. spaldingii* in small populations on isolated habitat fragments. Since plants are sedentary, proximity to nearest neighbor can be crucial to reproduction; if the nearest neighbor distance is greater than the foraging distance of a pollinating insect, local extirpation is very likely (Given 1994). The foraging distance for *Bombus fervidus* is not known; however; many small *S. spaldingii* populations are separated by considerable distances (Figures 5A through 8A – Appendix A), and are likely at high risk of pollinator exclusion, inbreeding depression and loss of fitness. Inbreeding depression has been noted in small populations of other species in fragmented landscapes (Menges 1995; Fischer and Matthies 1998; Kaye and Kuykendall 2001).

### **Small Populations**

Fragmentation reduces population size (Briggs and Walters 1997), and due to the small amount of habitat on fragments, populations cannot increase in size. Small populations, less than 100 individuals, are particularly threatened by variation in the environment, including random events from natural catastrophes, and evolutionary and genetic hazards (Menges 1991; Barrett and Kohn 1991).

Of the 66 known populations of *S. spaldingii*, 43 of them (65%) are small populations consisting of less than 100 individuals each (see Table 4 this report). It is likely that many of these small populations were part of larger populations prior to fragmentation. An estimation of the number of small populations resulting from fragmentation can be made by correlation with fragmentation levels in the regions in which they occur. The following table (Table 11) summarizes ownership and distribution within physiographic region of small populations of *S. spaldingii* (<100 plants each) (also see Tables 5A through 8A – Appendix B).

**Table 11 – Ownership and Distribution of Small *Silene spaldingii* Populations (<100 plants each).**

<b>Owner-ship</b>	<b>Idaho Palouse Grassland</b>	<b>Idaho Canyon Grassland</b>	<b>Montana Inter-mountain Valleys</b>	<b>Oregon Wallowa Plateau</b>	<b>Washington Palouse Grassland</b>	<b>Washington Channeled Scabland</b>	<b>Total</b>
<b>Private</b>	6	1	4	4	8	7	<b>30</b>
<b>State</b>		1			2	5	<b>8</b>
<b>Federal</b>		1				4	<b>5</b>
<b>Total</b>	<b>6</b>	<b>3</b>	<b>4</b>	<b>4</b>	<b>10</b>	<b>16</b>	<b>43</b>

Those small populations that are located in areas of high human-induced fragmentation include: 1) the 16 Palouse Grassland populations (6 in Idaho and 10 in Washington), 2) four Intermontane Valley populations in the Niarada area, 3) four Wallowa Plateau populations located in the Wallowa Valley area, and 4) at least 10 of the 16 populations in the Channeled Scabland of Washington. The 3 known populations in Canyon Grasslands of Idaho are located within minimally fragmented, extensive, contiguous habitat and are not likely the result of fragmentation.

Therefore, 34 of the 43 small populations, 79%, likely resulted from the fragmentation process and are located on isolated habitat fragments surrounded by agricultural fields and development. Furthermore, the majority of them, 30 of 43 (70%), are privately owned without regulations for their protection. Although these populations support a small proportion of the total known plants rangewide, they may represent a large portion of the genetic diversity within the species. They represent the only remaining genetic diversity from the Palouse Grasslands, a critically endangered ecosystem with only 1% of original habitat remaining (Noss. et al. 1995).

### **Genetic Consequences**

If these small populations are genetically isolated, i.e., no gene flow is occurring between them due to exclusion of pollinators and seed dispersers, they are especially vulnerable to evolutionary and genetic hazards that may hasten their extinction. Genetic drift, the loss of genetic variability as a result of chance, can result in large random changes in gene frequencies in small populations, making them vulnerable to extinction from random events (Barrett and Kohn 1991). The loss of genetic variability as a result of small population size is likely to reduce the ability of a population to respond adaptively to changing environmental conditions (Huenneke 1991; Menges 1991; Given 1994). The smaller the population, the more vulnerable it becomes to loss of genetic diversity in successive generations; loss of selectable variation can facilitate extinction (Given 1994). Species that have experienced severe reduction in population numbers owing to habitat destruction may be more susceptible to genetic stresses imposed by small population size than those that are naturally sparsely distributed and may be adjusted to close inbreeding (Barrett and Kohn 1991).

## Viability of Small Populations

The small populations of *S. spaldingii* in highly fragmented regions may be experiencing patterns of gene flow radically different from those typical of the species' evolutionary history (Huenneke 1991), and the viability of these remnant populations is very much in question. Although *S. spaldingii* plants are long-lived (Lesica 1997a), seed production and seedling establishment are necessary for long-term viability (Menges 1995).

Genetic and demographic processes interact to determine the viability of small populations occupying fragmented habitats (CPBR 2002). Fragmentation reduces population size and disrupts pattern of gene flow and demographic functions. The time-scale involved will depend upon the longevity of the plants concerned; in long-lived plants population turnover is very slow and it may take a long time for extinction to occur (Briggs and Walters 1997). Certain characteristics of *S. spaldingii* may buffer it against random environmental, demographic, and genetic events, such as indeterminate growth, metapopulation structure, partial selfing characteristics, and others may enhance its vulnerability, such as population isolation, specific mutualisms, and inbreeding depression (Menges 1991). Genetic links among metapopulations can favor population viability because the contribution of even one or a few migrants per generation can reverse the loss of genetic variation in small populations (Menges 1991). Movement of propagules among populations may allow persistence of metapopulations even when individual plant populations are not viable (Menges 1990).

Rates of gene flow as low as one individual per generation are sufficient to counteract the effects of genetic drift, even in very small populations (Mills and Allendorf 1996; Newman and Tallmon 2001). Gene flow varies substantially among species, populations, seasons, and even individual plants, however, gene flow is often sufficient to counteract genetic drift even if populations are physically isolated by hundreds to thousands of meters (Ellstrand 1992a, 1992b). Most pollen and seed are dispersed very close to the source, the mean distance varying from 1-20 meters (Linhart 1991; Ellstrand 1992b); however, some dispersal can occur over very long distances (Linhart 1991). *Silene spaldingii* seed may be carried long distances in the digestive tracts or on the fur/feathers of native ungulates, domestic livestock, small mammals, or birds. *Silene spaldingii* pollen may be dispersed considerable distances by *Bombus fervidus* because bumblebees are known to fly long distances (Linhart 1991).

Minimum population size needed for viability [minimum population viability (MPV)] is a controversial topic and one that perhaps has more application to animals than plants (Menges 1991). However, varying estimates of MPV have been made for plants. Estimates based on inbreeding depression have been used to recommend a general effective MVP of 50 individuals (Menges 1991). Given (1991) considered an effective population size of about 500 should lead to reasonable persistence times, and a minimum necessary to continue evolution and to sustain long-term adaptability for change.

Many factors influence viability. The viability of small *S. spaldingii* populations on isolated habitat fragments is unknown. Considerably more information is needed to make these predictions, including 1) the effect of fragmentation on ecological, genetic, and demographic

patterns, 2) potential gene flow among populations, i.e., foraging behavior of pollinators, seed dispersal mechanisms, 3) how life history traits and adaptability affect the impact of fragmentation and isolation, and 4) landscape attributes, such as remnant size, distance between remnants, and connectivity (CPBR 2002).

### **Prolonged Drought and Global Warming**

*Silene spaldingii* has adaptations that enable it to persist during dry periods, a long taproot and prolonged dormancy; however, prolonged drought may adversely affect *S. spaldingii* and contribute to extirpations of small populations. Perennial desert plants adapted to arid environments are negatively impacted by a prolonged and severe drought (Schultz and Ostler 1995). Climatic variation likely affects recruitment, survival of established plants, and flowering and fruit set of *S. spaldingii* (B. Benner, *in litt.* 1999). The proportion of plants exhibiting prolonged dormancy has been observed to be higher during extremely droughty years (Heidel 1995). At BLM monitoring sites in Washington, lower numbers of plants were recorded in droughty years, and the plants that were present senesced early in the flowering period had poorer fruit set (B. Benner, *in litt.* 1999). At Garden Creek Ranch in extremely dry years, it has been noted that many plants that produce flowers dry up and senesce before setting seed (J. Hill, *in litt.* 2003a). Drought conditions likely were associated with the decreased vigor, as measured by flower and fruit production, of *S. spaldingii* plants monitored in 1986 and 1988 at Wild Horse Island (Lesica 1988c) and at Dancing Prairie (Lesica 1988d). The drier conditions following fire at *S. spaldingii* sites at Garden Creek Ranch may be responsible for *S. spaldingii* ramets in burned areas being shorter and maturing a lower percentage of flowers to seed capsules (Menke and Muir 2003). Droughty conditions may exacerbate the negative effects of livestock grazing (Schassberger 1988) and promote weed invasion. Although prolonged dormancy may be an important life history strategy for *S. spaldingii* growing in semi-arid environments (Lesica 1997a), the length of time plants can remain dormant appears to be limited. In a long-term demographic study, Lesica (1997a) considered a plant to be dead if it had not reappeared at a known site within three years.

Global warming and major changes in climate regimes could have significant effects on *S. spaldingii* and its associated communities in the future. Survival will depend on the ability to adapt to changing climate, or migrate to areas that still maintain the conditions necessary for their survival. Plants are particularly vulnerable since they are relatively immobile (Wilson 1989). Changing rainfall patterns are anticipated, with some regions becoming drier and others wetter (Given 1994). Within *S. spaldingii* locations, decreases in precipitation could create more xeric conditions that would not support mesic grassland communities, and increases in precipitation may allow woody vegetation to establish. The effects of global warming will likely be greatest on plants that are sensitive to small changes in rainfall and temperature and cannot readily migrate (Given 1994). Survival may depend on the ability of species to migrate altitudinally; a vertical ascent of 500 meters could compensate for a 3°C rise in average temperature (Given 1994).

Changes in atmospheric gases associated with global warming could have major negative impacts on *S. spaldingii* and its habitat. Carbon dioxide levels, which influence structure and

composition of vegetation, have been increasing due to human activities (Given 1994). Increases in this gas have been shown to increase productivity of cheatgrass (Mayeux et al. 1994). The substantial increases in cheatgrass productivity could increase even further the number and severity of wildfires in the Great Basin (Smith et al. 1987).

## ECOSYSTEM CONSIDERATION

The mesic fescue bunchgrass communities that support *S. spaldingii* have high species diversity and support other rare and regionally endemic plant and animal species. Habitat loss and fragmentation, degradation from weed invasion, livestock grazing, herbicide drift, and changes in fire regime, are also threats to these species and the native bunchgrass communities that support them. Ecosystem oriented approaches to protection and recovery of *S. spaldingii* habitat would benefit these species, help prevent them from declining to threatened or endangered status, and support the goal of maintaining overall biodiversity.

Other rare plant species that are regionally endemic to the Palouse Grasslands and/or Canyon Grasslands of Idaho and/or Washington include *Haplopappus liatrisformis*, *Aster jessicae*, *Cirsium brevifolium*, *Trifolium plumosum*, *Calochortus nitidus*, and *C. macrocarpus* var. *maculosus*. In some cases, *H. liatrisformis* and *S. spaldingii* intermingle, but *S. spaldingii* appears to be restricted to moister aspects and microsites where shrubs are more prominent and forbs more abundant (Lichthardt 1997). *Aster jessicae* is not known to occur with *S. spaldingii* in Idaho; however at two sites in Washington (EORs WA 010, WA 025), all three species, *H. liatrisformis*, *A. jessicae*, and *S. spaldingii*, occur together. In Washington, other rare plants that occur within *S. spaldingii* habitats are *Rubus nigerrimus* and *Astragalus riparius* (F. Caplow, *in litt.* 2003b). *Polemonium pectinatum*, a local endemic occurring in the Crab, Coal and Wilson Creek area of the Channeled Scabland region, occurs in silty stream floodplains often near *S. spaldingii* habitat. This species is likely affected by trampling, soil disturbance, and encroachment of exotic species that can occur when livestock grazing involves too many animals or too long a duration (B. Benner, *in litt.* 1993, 1995, 1999, 2003d).

Rare animals in vicinity of known *S. spaldingii* sites include the Columbian sharp-tailed grouse (*Tympanuchus phasianellus columbianus*) at Tobacco Valley, Montana, and the Swanson Lakes Wildlife Area, Washington, the bald eagle (*Haliaeetus leucocephalus*) at Wild Horse Island in Flathead Lake, Montana (Schassberger 1988).

## CONSERVATION EFFORTS

Conservation efforts, including inventory, monitoring and research, and protection actions, have been completed or are ongoing at several *S. spaldingii* populations within each state where the species occurs.

### IDAHO

**Inventory:** Considerable inventory work has been conducted in the Canyon Grasslands at Craig Mountain. Extensive surveys by TNC, IDCDC and BLM have been ongoing at the 14,000-acre Garden Creek Ranch since the discovery of the species there in 1993 (J. Hill, *in litt.* 2003a; Hill and Gray 1999; Hill and Gray 2000; Hill et al. 2001; Lichthardt and Gray 2002). Further survey work was conducted at Garden Creek Ranch in conjunction with research for a Masters Thesis (Menke 2003). The current total plant count for Garden Creek Ranch represents the results of these surveys (J. Hill, *in litt.* 2003b). New locations discovered at Garden Creek Ranch and increases at known locations since listing include eight new locations [Lower Corral Creek (LCC) 86, 91, 193, 223, 225, 238, Cave Gulch (CG) 212, and Upper Corral Creek (UCC) 281/282] and a total of ~2800 plants. Other inventory work has been conducted on Craig Mountain by IDCDC (Mancuso and Moseley 1994; Lichthardt and Gray 2002; Gray and Lichthardt 2003b) and BLM. New sites resulting from this work since final listing include five new Craig Mountain occurrences (Snake River drainage) (EORs ID 009, 010, 015, 016, and 017) for a total of 492 plants, and the first occurrence in the Salmon River drainage, Salmon River Oxbow (EOR ID 014), for a total of 50 plants. Recent surveys (Gray and Lichthardt 2003b) by IDCDC on BLM parcels south of the Salmon River and in the Eagle Creek drainage on Craig Mountain located another new occurrence in the Salmon River drainage, Rice Creek (EOR ID 018), for a total of 50 plants (K. Gray, *in litt.* 2003a; Gray and Lichthardt 2003b).

Inventory work in the Palouse Grassland region of Idaho has continued since the listing of *S. spaldingii*. Three new occurrences were located in 2001 (EORs ID 011, 012, 013) (Lichthardt and Gray 2002) for a total of 29 plants. A grant proposal is being developed for conducting *S. spaldingii* surveys on Nez Perce Tribal property in the Camas Prairie region of Idaho (A. Sondenaar, *in litt.* 2003a).

**Monitoring/Research:** A number of monitoring and research projects have been conducted or are ongoing at Garden Creek Ranch and Craig Mountain including the following [LCC = Lower Corral Creek Study Area (Hill and Gray 1999)]:

#### Garden Creek Ranch:

- 1) A permanent population monitoring transect was established in LCC 68 by BLM (USDI 1998).
- 2) Genetic studies were conducted on *S. spaldingii* populations in LCC 56 (Baldwin and Brunfeldt 1995).

3) Pollination studies were conducted at *S. spaldingii* populations in LCC 68 (Lesica and Heidel 1996).

4) Mapping of *S. spaldingii* clusters and yellow starthistle weed patches has been done at seven *S. spaldingii* locations, LCC 51, LCC 56, LCC 63; LCC 65; LCC 69; LCC 196; LCC 272 (Hill and Gray 2000; Hill et al. 2001). All these locations are on BLM-owned land. Total population counts at two of these locations, LCC 51 and LCC 65, were made for five consecutive years. All yellow starthistle occurring at these two locations was manually controlled for four years of the project. Seeds of native grasses and forbs were collected at these sites and planted into areas where heavy infestations of yellow starthistle were manually controlled. At LCC 65, a permanent belt transect was established in 1999 within a *S. spaldingii* cluster with a dense yellow starthistle infestation to determine the effect of hand-pulling yellow starthistle (Hill and Gray 2000; Hill et al. 2001; Hill and Fuchs 2002, 2003).

5) Seven permanent belt transects for long-term demographic monitoring of *S. spaldingii* plants were established at burned and unburned *S. spaldingii* locations, LCC 51 and LCC65 [burned] and LCC 196 [unburned], and data collected in 2002 and 2003 (Hill and Weddell 2003; Hill and Gray 2004) to determine the population dynamics of *S. spaldingii* in Canyon Grasslands and the effect of fire on *S. spaldingii*. These locations are on BLM-owned land.

6) Permanent belt transects were established in two *S. spaldingii* clusters, one with a dense infestation of yellow starthistle and one with a dense infestation of Kentucky bluegrass in LCC 63, to determine the effect of competition from these invasive weed species on *S. spaldingii* over time (Hill et al. 2001). Data was collected on cover of the two weed species and locations of *S. spaldingii* plants in transect in 1999 and 2003 (J. Hill, *in litt.* 2003a). No weed control was done in these transects. These locations are on BLM-owned land.

7) As part of a Masters Thesis, monitoring data on *S. spaldingii* plants, communities, and weed species were collected for two years at numerous burned and unburned and invaded and uninvaded *S. spaldingii* locations (Menke 2003).

8) Research has been conducted using hyperspectral remote sensing (Lass 1999) with high spectral resolution and QuickBird satellite remote sensing with high spatial resolution to determine the ability of these technologies to detect yellow starthistle and habitat types that support *S. spaldingii* (Hill 2002a, 2002b).

9) IDCDC botanists established and collected data in 2001, 2002, and 2003 at long-term monitoring plots at Garden Creek Ranch, LCC 69, LCC 225, LCC 271 (Lichthardt and Gray 2002).

#### Other Craig Mountain Locations:

1) IDCDC botanists established and collected data in 2001, 2002, and 2003 at long-term monitoring plots at other Craig Mountain occurrences, EOR ID 009 (Madden Creek) and EOR ID 016 (Billy Creek South) (Lichthardt and Gray 2002; Gray and Lichthardt 2003b).

**Other Conservation Actions:** *Silene spaldingii* seeds at a number of Garden Creek Ranch locations (LCC51, LCC56, LCC72, LCC 69, LCC 196, and LCC 272) have been collected and sent to the Berry Botanic Garden for long-term storage and future restoration or re-colonization efforts.

BLM has released a number of yellow starthistle bio-control insects in infested areas surrounding *S. spaldingii* locations at Garden Creek Ranch and other Craig Mountain locations. Reductions in the cover of yellow starthistle have been documented where yellow starthistle bio-control insects are present (Danly 1999; Hill 2001b).

A draft management plan has been developed for Garden Creek Ranch in which priority locations (including all *S. spaldingii* locations) for protection are identified and protection methodology is outlined (Hill 1998). Completion date for the plan is unknown.

The Tri-State Coordinated Weed Management Area (CWMA) and the Joseph Plains CWMA are coordinating and conducting weed control activities in the vicinity of *S. spaldingii* locations in the Canyon Grasslands of the tri-state area of Idaho, Washington and Oregon.

### **MONTANA**

**Inventory:** The Montana Natural Heritage Program conducted survey work in the 2002 and 2003 field season to document the current status of all populations and systematically survey potential habitat for *S. spaldingii* on portions of the Kootenai National Forest area around Dancing Prairie and the Niarada area (MNHP 2001; S. Crispin, personal communication, 2003; S. Crispin, *in litt.* 2003). Considerable update information was obtained on known occurrences, although no new occurrences of *S. spaldingii* were located (C. Bjork in G. Glenne, *in litt.* 2003b). In 2002, a new occurrence was located at the Lost Trail NWR (L. Garner, personal communication, 2003; M. Mantas, *in litt.* 2003).

**Monitoring/Research:** Numerous monitoring and research studies have been conducted at the Dancing Prairie Preserve (EOR MT 001) and other *S. spaldingii* populations, EOR MT 002 (Wild Horse Island), and Lost Trail NWR:

- 1) *Silene spaldingii* plants were monitored in two belt transects at Wild Horse Island between 1986 and 1992 (Lesica 1988c; P. Lesica, *in litt.* 2003b).
- 2) *Silene spaldingii* plants were mapped in four permanent belt transects, and size and reproductive status have been recorded since 1987 to the present at Dancing Prairie Preserve (Lesica 1988d; Lesica 1997a; P. Lesica, *in litt.* 2003b).

- 3) Prescribed fall and spring fires were conducted within permanent monitoring plots at the Dancing Prairie Preserve to determine the effect of fire on *S. spaldingii* (Lesica 1991b; Lesica 1994; Lesica 1995; Lesica 1999).
- 4) Genetic studies were conducted on *S. spaldingii* populations at Dancing Prairie Preserve (Baldwin and Brunsfeld 1995).
- 5) Pollination studies were conducted at *S. spaldingii* populations at Dancing Prairie Preserve (Lesica 1991a; Lesica 1993; Lesica and Heidel 1996).
- 6) A single long-term demography plot was established during the 2003 field season at Lost Trail NWR. A second will probably be established in 2004 (P. Lesica, *in litt.* 2003b, 2003c).

**Other Conservation Actions:** A prescribed burn plan was initiated at Dancing Prairie in the spring of 2003 by burning 100 acres in *S. spaldingii* habitat. This plan prescribes burning of similar size patches every other year (P. Lesica, *in litt.* 2003b).

Spot-spraying of weeds (*Potentilla recta*, *Centaurea maculosa*, *Hieracium pratense*, *Hypericum perforatum*) from an ATV or backpack has been conducted at Dancing Prairie for the past nine years (P. Lesica, *in litt.* 2003b).

An experimental design study using a combination of spring burn, fall burn, no burn, picloram herbicide, no herbicide, was conducted at Dancing Prairie to determine the effects of fire, season of burn, and their interaction with herbicide on the recruitment and population growth of the non-native invasive species, sulfur cinquefoil (*Potentilla recta*), over a five-year period (Lesica and Martin 2003).

## OREGON

**Inventory:** TNC is in the process of completing inventory on the Zumwalt Prairie Preserve. Surveys are planned at the Preserve in summer of 2003 on TNC-owned lands. A complete inventory of the Clear Lake Ridge Preserve has been done (R. Taylor, *in litt.* 2003a). A grant proposal is being developed for conducting *S. spaldingii* surveys on Nez Perce Tribal property in Oregon (A. Sondena, *in litt.* 2003a). Surveys within the Canyon Grasslands of Oregon are being considered for the 2003 field season (J. Kagan, *in litt.* 2003a, 2003c).

**Monitoring/Research:** Fire and grazing research are priorities at TNC preserves at Zumwalt Prairie and Clear Lake Ridge (R. Taylor, *in litt.* 2003a). Completed and ongoing monitoring and research projects at *S. spaldingii* populations in Oregon include:

- 1) Permanent population monitoring plots were established and data collected at Clear Lake Ridge in 1990; these plots were recently re-sampled and will continue to be assessed (Youtie 1990; Elseroad and Taylor 2002b).

2) Coordination is ongoing between TNC and the U.S. Forest Service (USFS) regarding the feasibility of doing prescribed burning at Clear Lake Ridge Preserve (R. Taylor, *in litt.* 2003a) Elseroad and Taylor 2002b).

3) Genetic studies were conducted on *S. spaldingii* populations at Clear Lake Ridge (Baldwin and Brunsfeld 1995).

4) Pollination studies were conducted at *S. spaldingii* populations at Clear Lake Ridge (Lesica and Heidel 1996).

**Other Conservation Actions:** TNC is in the process of developing a complete management plan for the Zumwalt Preserve, which will include a section on *S. spaldingii*. One of the goals is to manage livestock to ensure no grazing occurs during flowering and fruiting of *S. spaldingii* (R. Taylor, *in litt.* 2003a).

Two of the four locations of *S. spaldingii* at the Clear Lake Ridge Preserve have been fenced and protected from livestock grazing (Youtie 1990).

TNC and other land managers in Wallowa County in the vicinity of Zumwalt Prairie have tested the capability of high resolution satellite remote sensing techniques (i.e., QuickBird Digital Globe) to map vegetation, however, the resulting map appears to distinguish only broad vegetation types, i.e. shrubs, grasses, etc. (R. Taylor, *in litt.* 2003b).

The Tri-County CWMA (Union, Wallow, Baker counties) is coordinating and conducting weed management activities in the vicinity of *S. spaldingii* locations in Wallowa County.

*Silene spaldingii* seeds were collected for the Berry Botanic Garden in 1989 at the Cats Back EOR OR 008 population.

## WASHINGTON

**Inventory:** Recent inventories of *S. spaldingii* populations since the time of listing include 2002 surveys (B. Benner, *in litt.* 2003a; 2003b) on BLM lands in the Channeled Scabland area. These surveys located one new occurrence of 145 plants (EOR WA 080) in Adams County (the only known extant occurrence in this county) and additional plants at known occurrences (EORs WA 033, 034, 035, 036, 037, 042, 052, 055, 065, 066) for a total of 843 plants. Additional inventory work since listing has been conducted at the Fairchild AFB (Caplow 2001). Recent inventory work on state land (Swanson Lake Wildlife Area) and federal land (Turnbull NWR) in the Channeled Scablands area located three new EORs for a total of 555 plants (Caplow 2002b; Weddell 2002). A grant proposal is being developed for conducting *S. spaldingii* surveys on Nez Perce Tribal in Washington (A. Sondenaar, *in litt.* 2003a).

**Monitoring/Research:** Monitoring and research projects have been completed and are ongoing at several *S. spaldingii* populations in Washington:

1) BLM has monitored individual *S. spaldingii* plants at 10 sites in Lincoln County since 1995, and is performing habitat monitoring on lands subject to livestock grazing (B. Benner, *in litt.* 1999; 2003d).

2) Population monitoring plots were established and data collected at Fairchild AFB (EOR WA 044) in 1994 by the Washington Natural Heritage Program. A management plan for *S. spaldingii* was developed (Rush and Gamon 1999), and population and habitat monitoring has occurred annually (Caplow 2001, 2002a). Monitoring will continue at Fairchild Air Force Base through 2005 (F. Caplow, *in litt.* 2003b).

3) Long-term monitoring transects at Kramer Prairie were established by Daubenmire in the 1950s although they were not set up specifically to monitor *S. spaldingii*. Weddell and Lichthardt (1998) repeated the vegetation sampling in 1998 using the methods described by Daubenmire (1970). Other transects have been recensused as well (B. Weddell, *in litt.* 2003c).

4) Three permanent belt transects were established in 2003 in the Lick Creek drainage of the Blue Mountains to monitor *S. spaldingii* demography. A single demography transect was established at Lamona in 2003 (P. Lesica, *in litt.* 2003b, 2003c).

**Other Conservation Actions:** BLM has grazing management practices that include assessments of livestock use, pasture rotation, monitoring of use and removal of livestock if use criteria are exceeded, and evaluation of plant community composition and population trends (B. Benner, *in litt.* 2003d).

A biological weed control program was implemented at Fairchild AFB in 1996 and four biocontrol agents were introduced to control Canada thistle, diffuse knapweed, and spotted knapweed (Rush and Gamon 1999; Caplow 2001).

# CONSERVATION RECOMMENDATIONS

## SUMMARY

Our conservation recommendations for *S. spaldingii* focus on protection of existing populations and habitat, and maintenance of potential habitat. Implementation of these actions will help to maintain or enhance viable populations and natural ecosystem functions and processes, and contribute to the long-term viability of *S. spaldingii*. Several *S. spaldingii* populations are recommended for protection with priority given to those that support large areas of suitable habitat and large populations, are key to maintaining the current genetic diversity, maintain the integrity of larger, potential metapopulations, and have an ownership type that enables realistic protection of the site. A yet undiscovered population within Canyon Grasslands is included in the list of populations recommended for protection because this area has high probability of occurrence of *S. spaldingii*, consists of a large contiguous, relatively unfragmented area of suitable habitat, possibly low current weed invasion, but high susceptibility to weed invasion.

Conservation recommendations are provided that will help reduce the most imminent and pervasive threats to *S. spaldingii* and its habitat. In order of priority, recommendations address the following issues: 1) Habitat Degradation from non-native invasive plants, and major contributing disturbance factors, livestock grazing and fire; recommendations provide guidelines for effective weed, livestock, and fire management and habitat restoration, 2) Inventory of Potential Unsurveyed Habitat; recommendations identify areas with immediate survey needs, 3) Habitat Fragmentation; recommendations help protect pollinators, provide suggestions to reduce further habitat fragmentation and protect small populations on isolated habitat fragments, retain genetic diversity of threatened small populations, and suggest areas that would allow protection of groups of small populations, 4) Monitoring; recommendations identify priority monitoring needs and provide suggestions of appropriate monitoring methodology for accurate determination of population size, trends, and geographic differentiation to assess threats and management activities and provide guidance for future conservation and management decisions, and 5) Reporting and Record-keeping; recommendations help to standardize and improve reporting and record-keeping across the four-state region of *S. spaldingii* occurrence.

## AREAS RECOMMENDED FOR PROTECTION

The size of an area has a pronounced effect on the viability of species and on ecological processes. Small areas are limited in the size of the populations and species diversity they can support (Given 1994; Briggs and Walters 1997; Noss et al. 1995). Maintenance of large areas of suitable habitat with multiple clusters of plants tends to conserve genetic diversity and metapopulation dynamics that allow for colonization and dispersal (Menges 1991). Population size has an effect on genetic integrity and viability of the species. Small populations are subject to pollinator limitations, inbreeding depression, and high risk of extirpation (Lesica 1993; Lesica and Heidel 1996; Given 1994; Briggs and Walters 1997). Large populations tend to maintain genetic diversity and are less vulnerable to extirpation than smaller populations (Barrett and Kohn 1991). Consequently, the areas recommended

for priority conservation for *S. spaldingii* emphasize larger populations and large, interconnected landscapes that maintain environmental regimes and processes and have adequate suitable habitat to allow for colonization and dispersal. These large populations, especially those occurring in relatively extensive natural areas, will be capable of supporting viable populations of pollinators as well (Lesica 1993).

Several *S. spaldingii* populations were prioritized so that protection and conservation efforts can be focused on those that have the greatest likelihood of sustaining viable populations of *S. spaldingii* and that maintain the current geographic distribution pattern and genetic diversity. The small amount of genetic data available for *S. spaldingii* (Baldwin and Brunsfeld 1995) generally supports the assumption that correspondence exists between genetic variation and geographic location (P. Lesica, *in litt.* 2003b). Criteria used to prioritize *S. spaldingii* populations fell into three basic categories and included: 1) features that relate to the viability of *S. spaldingii*, i.e., area of suitable habitat, size of population, habitat condition, and the degree of human-induced habitat fragmentation, 2) features that degrade *S. spaldingii* populations and reduce viability, i.e. threats from invasive weeds, livestock overgrazing and altered fire regimes, and 3) features that assist in the realistic protection of the site, i.e., ownership. Within each category a *S. spaldingii* population was given a number rating. Numbers in all categories were totaled for an overall conservation rank for each *S. spaldingii* population (Table 19A – Appendix B). Table 12 below lists the populations with the highest conservation rank and includes information on population size, nearby populations, physiographic region, and conservation rank.

Information on populations in close proximity is provided to indicate the potential for the recommended sites to become larger populations if intervening occurrences are located in the future. The four Craig Mountain, Idaho, populations, i.e. Garden Creek Ranch, Madden/Captain John Creeks, Camp/Billy Creeks, and Red Bird, occur within a few miles of each other, and the intervening area has suitable, unsurveyed habitat that has potential to support other occurrences of *S. spaldingii*. Lichthardt and Gray (2002) recommend all known populations on Craig Mountain be considered as subpopulations of one large population. In Washington, suitable unsurveyed habitat exists between the two Swanson Lake populations and between the Crab Creek and Rocky Ford populations (B. Benner, *in litt.* 2003b; Caplow 2002b; Weddell 2002).

The twenty-one populations recommended for protection include nine in the large population category (500 or more plants), nine in the 100-499 population category, two in the <100 population category, and a yet undiscovered site in Oregon Canyon Grasslands (see following section). The recommended populations contain 93% of the known plants and 30% of the known populations rangewide. These populations likely contain the majority of known genetic diversity in *S. spaldingii* since they are distributed over the entire known range of *S. spaldingii*, including the northern and southern groupings identified as being genetically different in genetic studies (Baldwin and Brunsfeld 1995).

**Table 12 - *Silene spaldingii* Populations Recommended for Protection.**

Population Name	Population Number	EOR	Population Size	Nearby Populations (plants)	Physio-graphic Region	Owner-ship	Conser-vation Rank
<b>IDAHO</b>							
Garden Creek Ranch	5	006	3995	ID-1,7,11 (506)	Canyon Grassland	TNC/Federal	26
Madden/Captain John Creeks	7	009, 010	272	ID-1,5,11 (4229)	Canyon Grassland	State	21
Camp/Billy Creeks	11	015, 016, 017	220	ID-1,5,7 (4281)	Canyon Grassland	Federal	21
<b>MONTANA</b>							
Dancing Prairie	1	001, 011	10,023	MT-2, BC-1 (223)	Intermountain Valley	TNC	26
Lost Trail NWR	8	012	350	-	Intermontane Valley	Federal	23
Wild Horse Island	2	002	200	-	Intermontane Valley	State	21
<b>OREGON</b>							
Zumwalt Prairie	8	022-027	1721	OR-2,4,6 (422)	Wallowa Plateau	TNC	24
Clear Lake Ridge	2	003-006, 015	401	OR-5 (15)	Wallowa Plateau	TNC	20
Crow Creek	3	013-014, 016-020	844	-	Wallowa Plateau	Federal	20
Wallowa Lake	1	001,010, 011	513	-	Wallowa Plateau	Private/Federal	20
Oregon Canyon Grassland	No known plants; conservation rank similar to Garden Creek Ranch but with lower weed threat						
<b>WASHINGTON</b>							
Coal Creek	24	048	500	WA-14,18,23 (492)	Channeled Scablands	Federal	26
Twin Lakes	27	052-054, 065-068,073	627	WA-22,36,37 (759)	Channeled Scabland	Private/Federal	23
Swanson Lake	37	079	488	WA-22,27,36 (899)	Channeled Scablands	State	23
Swanson Lake	36	078	52	WA-22,27,37 (1335)	Channeled Scabland	State	22
Crab Creek	18	030,055	162	WA-14,23,24 (830)	Channeled Scablands	Federal	22
Rocky Ford	23	046	300	WA-14,18,24 (692)	Channeled Scablands	Federal	22
Fairchild AFB	21	044	66	WA-19 (3)	Channeled Scablands	Federal	21
Kramer Prairie	1	006 024	200	WA-3,5,6 (32)	Palouse Grassland	State	21
Blue Mountains	25	049,050, 056,058	997	-	Palouse Grassland	Federal	21
Miller Ranch	20	033-037, 042,047	540	WA-15,16,17 (244)	Channeled Scablands	Federal	21

Inaccessibility and habitat degradation of several of these recommended *S. spaldingii* populations make protection of these areas difficult, but should not be considered strikes against them as areas for conservation. Some of the best remaining populations of *S. spaldingii* are located in rugged terrain, i.e., Canyon Grasslands and portions of the Channeled Scabland. The inaccessibility of these sites is the major reason the habitat has suffered little human-induced fragmentation and that large populations in extensive suitable habitat remain, and inaccessibility will continue to protect these areas from development and habitat fragmentation in the future. Degraded habitats that still retain high richness of native species should not be dismissed as important conservation areas (Noss et al. 1995).

### **Oregon Canyon Grasslands Population**

An undiscovered population is included in the areas recommended for protection for the following reasons. Of all the areas where *S. spaldingii* is known to occur, the Canyon Grasslands have the least human-induced fragmentation and habitat loss. They represent a large, intact, contiguous habitat for *S. spaldingii* with abundant unoccupied suitable habitat for colonization and dispersal. The majority is under federal or state ownership and portions are within an established CWMA (Tri-State CWMA).

The Canyon Grassland *S. spaldingii* populations on Craig Mountain, Idaho, have a severe invasive weed problem. Large infestations of yellow starthistle currently exist on the drier, southerly slopes, and yellow starthistle is invading the northerly aspects that support *S. spaldingii* (Hill and Gray 1999, 2000; Hill et al. 1999, 2001; Hill and Fuchs 2002, 2003; Lichthardt and Gray 2002; Gray and Lichthardt 2003b). If another Canyon Grassland *S. spaldingii* location existed with less yellow starthistle or other invasive weed problems, it would be of very high conservation value. Management could be focused on prevention and early detection that is far easier and more economical than the weed control and restoration needed at Craig Mountain. Therefore, it would be more defensible and more realistically protected with limited recovery money.

It is highly likely *S. spaldingii* exists in Oregon Canyon Grasslands. The steep slopes of the Imnaha, Grande Ronde, and Snake Rivers of Oregon are within Canyon Grassland area (Tisdale 1986a; Johnson and Simon 1987) (Figure 9A – Appendix A), and support large areas of the Idaho fescue-prairie junegrass (low elevation) habitat type (Johnson and Simon 1987) in which many *S. spaldingii* populations are known to occur in Canyon Grasslands in Idaho (J. Hill, *in litt.* 2003a, 2003b; Menke 2003; Lichthardt and Gray 2002; Gray and Lichthardt 2003b; IDCDC 2003b). It is likely these areas support large populations of *S. spaldingii* similar to those found in the Idaho Canyon Grasslands on Craig Mountain. Historical and recent evidence exists to support this hypothesis. *Silene spaldingii* was collected from Canyon Grasslands in Oregon historically (S. Yelton, *in litt.* 2003; P. Holmgren, *in litt.* 2003). In a letter to S. Tucker in 1903, C.V. Piper states “Mr. Cusick gets this plant on the Imnaha River in northeastern Oregon”, and this was the “only place in which the plant is known to grow”. A potential recent *S. spaldingii* occurrence was located in the 2003 field season on the Nez Perce Precious Lands Preserve in Joseph Canyon, a large tributary of the Grande Ronde River in Oregon (A. Sondenaar, *in litt.* 2003b). The collection was post-bloom and positive identification was not possible.

The weed threat in Oregon Canyon Grasslands is less than that in Idaho or Washington Canyon Grasslands. Typically, the public lands west of the Snake River (i.e., Oregon), are not yet significantly infested with many invasive weeds, including yellow starthistle, however, this area is highly susceptible. The primary area for prevention and early detection within the Tri-State Demonstration Weed Management Area (DWMA) were portions of Oregon that currently had low levels of invasive weed infestations (USDI 1996a). Location of this population, if it indeed does exist, is of primary conservation concern.

## **RECOMMENDATIONS FOR HABITAT DEGRADATION**

The following habitat management actions are recommended to eliminate or control the major causes of habitat degradation within *S. spaldingii* habitat and protect essential habitat within the priority areas recommended for protection.

### **Weed Management**

Weed invasion is the major cause of *S. spaldingii* habitat degradation. Effective weed management focuses on 1) education and awareness, 2) inventory, 3) early detection and prevention, 4) Integrated Weed Management (IWM), 5) monitoring and evaluation, 6) restoration and/or rehabilitation, and 7) cooperation and partnerships (USDI 1994; National Strategy for Invasive Plant Management 1994; USDI 1996a; USDI 1996b; Idaho State Department of Agriculture (ISDA) 1999; USDA/USDI 2000; USDI 2003). These strategies for effective weed management, which are outlined in the afore-mentioned publications, are recommended for weed management at known *S. spaldingii* occurrences and potential habitat and are summarized in the following sections.

#### Inventory

Baseline information is needed at *S. spaldingii* occurrences regarding the weed species present, the locations and density of infestations, acreage infested, soil conditions, the plant community, and proximity to *S. spaldingii* plants. This baseline data will provide site-specific information to guide weed management decisions and assess control activities. Areas supporting potential habitat for *S. spaldingii* should be surveyed prior to weed control treatments to determine the presence of *S. spaldingii*. Surveys should be conducted by qualified personnel during flowering to ensure accurate identification (USDI 2003).

#### Prevention and Early Detection

Prevention, early detection and eradication of small weed infestations are the most practical, economical, and effective means of weed management (USDI 1994; National Strategy for Invasive Plant Management 1994; USDI 1996a, 1996b; ISDA 1999; USDA/USDI 2000). Prevention includes controlling the spread of weed seed from various sources, including: 1) contaminated seed, feed grain, hay mulch, 2) movement of animals with viable weed seed in hair, hooves and digestive tracts, 3) on tires or undercarriages of vehicles, and 4) gravel or roadfill contaminated with weed seed (USDI 1996a, 1996b).

Early detection of small, recently established infestations (satellites) and prompt control helps to ensure that new invader weed species do not become established and begin to spread. Small satellite infestations present the greatest threat because they increase in size at a faster rate than larger infestations (Moody and Mack 1988). Weed control is greatly increased by destroying up to even 30% of these satellite populations (Moody and Mack 1988). Small, satellite infestations are relatively easy to control due to their small areal size. Larger infestations require more time and resources to control and can degrade the site to a point that is difficult or impossible to restore.

Periodic, systematic weed surveys are needed for early detection of small satellite infestations, and, in light of the current ubiquitous nature of invasive weed species and the seriousness of the threat, these surveys are likely needed indefinitely. Frequent weed surveys, every 3-5 years, are recommended at priority *S. spaldingii* populations (Table 11). Ground surveys are time-consuming, difficult, and expensive, particularly in inaccessible, rugged terrain like the Canyon Grasslands. The use of remote sensing for weed detection, which offers wide coverage and quick results, has potential to increase the feasibility of conducting periodic weed surveys on an indefinite basis (Lass 1999). Two advanced remote sensing technologies, hyperspectral imagery and high-resolution, QuickBird (Digital Globe) satellite imagery, have been tested for weed detection in Canyon Grasslands along the Snake River and Salmon River. However, presently this technology is not sufficiently advanced to detect small satellite infestations of a specific weed species (Hill 2002a; Hill 2002b; R. Taylor, *in litt.* 2003b).

### Weed Control / Integrated Weed Management

Integrated Weed Management (IWM) uses site-specific information to make decisions about weed control treatments and is based on the fact that combined strategies work more effectively than a single strategy (USDI 1996a). The IWM approach is recommended to control invasive weeds at *S. spaldingii* sites using the best combinations of the following methods: 1) Manual Control, 2) Chemical Control, and 3) Biological Control. Sheley and Petroff (1999) summarize the current understanding of appropriate management for specific weed species.

#### Manual Control

Hand-pulling of weeds can be effective for controlling small infestations and causing minimal harm to *S. spaldingii* and the associated native plant community. At a *S. spaldingii* site at the Garden Creek Ranch, small satellite yellow starthistle infestations were hand-pulled and showed decreases in density, ranging from 39 to 100%, the following year (Hill and Fuchs 2002). Following initial chemical control on a ten-acre infestation of diffuse knapweed, hand-pulling missed or returning plants two or three times a year provided effective control for this species (Youtie 1997). Hand-pulling is a particularly useful method for sites that have scattered weed plants intermixed within *S. spaldingii* plants. Manual methods require annual treatment for as long as any weed plant is present or viable seed exists in the soil. Although yellow starthistle seeds can remain viable in the soil for up to 10

years (Callihan et al. 1989), the majority of the seed germinates or is lost to predation or decay within the first few years (Joley et al. 1992). Thus, most of the effort in controlling this species by hand-pulling will be during the first few years.

### Chemical Control

The sensitivity of *S. spaldingii* to herbicides is not known, however, many of the native plant species in communities that support *S. spaldingii* can be damaged by herbicide use, and there is a high likelihood herbicides would have adverse effects on *S. spaldingii* as well (see Threats section). Although application of a non-residual herbicide early in the season, before *S. spaldingii* emerges, will probably do little harm to *S. spaldingii* plants (P. Lesica, *in litt.* 2003b), it will likely be detrimental to the major bunchgrasses (or/and bunchgrass seedlings) and many native perennial forbs in *S. spaldingii* communities that are photosynthetically active from late fall through mid-July (Daubenmire 1970). The effect of herbicides on the native plant community that supports *S. spaldingii* should also be considered when utilizing chemical weed control.

Limited and carefully controlled application of herbicides may be used in the vicinity of known populations of *S. spaldingii*. Methods of herbicide application, such as wicking, that affects only the targeted weed species, have little potential to negatively impact *S. spaldingii* or native vegetation in the community. Guidelines for herbicide use near known populations of *S. spaldingii* on BLM lands managed by the Cottonwood Field Office, Cottonwood, Idaho, have been developed and approved by USFWS (USDI 2003). These guidelines include: 1) no boom-applicator spraying within 50 feet of *S. spaldingii* plants, 2) wick, wiping, and backpack or handpump spot spraying applications may occur from 5-50 feet of *S. spaldingii* plants, 3) only manual control may be used within three feet of *S. spaldingii* plants, 4) no aerial application within 500 feet of *S. spaldingii* plants, and 5) Tordon, a persistent herbicide, will not be used within 50 feet of *S. spaldingii* plants. The guidelines for aerial spraying may not be sufficient. In Canyon Grasslands, unpredictable wind currents and high temperatures can volatilize herbicides and carry them further than 500 feet (J. Hill, *in litt.* 2003a).

Distribution of weed infestations within *S. spaldingii* habitat often consist of small dense patches that are disjunct from *S. spaldingii* clusters (Hill and Gray 2000; Menke 2003). These patches lend themselves well to spot chemical control. Combination of a non-selective herbicide like Round-up, used to eliminate fall-germinating annual weeds like cheatgrass and yellow starthistle, with a residual broadleaf herbicide for longer control of broadleaf weeds, would be an effective form of chemical control for these infestations. Grasses could be seeded immediately and desirable broadleaf species could be incorporated once grasses are established (Jacob et al. 1999).

## Biological Control

The aim of biological control is not to eradicate the target weed, but rather to reduce its dominance to a more acceptable level. This method is cost-effective and self-perpetuating but its effects are neither immediate nor always adequate (Wilson and McCaffrey 1999). *Chrysolina* beetles cause drastic reductions in abundance of St. Johnswort (Tisdale 1976; Hill et al. 2001); however, fluctuations in population size of both St. Johnswort and the *Chrysolina* beetles occur. During its peak growth period in the fluctuating cycle with the *Chrysolina* beetles, St. Johnswort can reach large infestation sizes at some *S. spaldingii* locations (Hill and Gray 2000). Other forms of weed control may be necessary at these times to reduce St. Johnswort's competition with *S. spaldingii* for pollination services (Lesica and Heidel 1996) and soil and nutrient resources.

Bio-control insects appear to have some degree of success in controlling yellow starthistle infestations that threaten *S. spaldingii* habitat in Canyon Grasslands (Danly 1999; Hill et al. 2001; Hill 2001b; Hill and Fuchs 2002, 2003). Introductions of bio-control insects for control of weed species known to occur in *S. spaldingii* habitat, i.e., yellow starthistle, leafy spurge, spotted knapweed, rush skeltonweed, and diffuse knapweed, is recommended. The use of domestic sheep or goats as biological control agents for weeds in *S. spaldingii* habitat is not appropriate due to their preference for forb species and problems with disease transmission to big horn sheep (Schommer 2002).

## Restoration/Rehabilitation

Restoration or rehabilitation is a crucial step after control or eradication of invasive weeds; without it, areas are subject to reinvasion. Vacancies created by control of one invasive weed will immediately be filled by other invasive species (Hobbs 1991; National Strategy for Invasive Plant Management 1994; Wilson and McCaffrey 1999; Lass et al. 1999; Hill et al. 2001; Hill and Fuchs 2002, 2003). If considerable cover of native bunchgrass remains following weed control, natural recovery with native vegetation filling in the space opened by removal of weeds can occur (Monsen and McArthur 1995; Wilson and McCaffrey 1999). Sites where weeds and native species are intermixed are especially challenging because it is difficult to remove the weed competition and interplant additional species (Monsen and McArthur 1995). Infestations of invasive weed species within *S. spaldingii* sites at Garden Creek Ranch often occur as disjunct dense patches separate from *S. spaldingii* clusters (Menke 2003; Menke and Muir 2003); however, infestations also occur within *S. spaldingii* clusters, often as scattered plants but occasionally as dense infestations. Weed control and restoration within these *S. spaldingii* clusters have been difficult and little success has been attained (Hill and Gray 2000; Hill et al. 2001; Hill and Fuchs 2002, 2003).

The use of native species is particularly appropriate for restoration of *S. spaldingii* sites. *Silene spaldingii* appears to be confined to late-successional, good condition native bunchgrass communities, and is not found in communities with high proportions of non-native species (Schassberger 1988; Kagan 1989; Gamon 1991; Lorain 1991; Lichthardt 1997). The use of native species will restore the biological and genetic diversity and ecological integrity of the site. Genetic diversity of local site-adapted species will provide

the adaptive plasticity to change genetically in response to environmental fluctuations specific to the site, and will serve as a buffer against uniform susceptibility to pathogens and insects (Belnap 1995; Linhart 1995; Fairbanks and Andersen 1995). Local site-adapted native species are genetically appropriate for the site, whereas non-local native species can result in genetic pollution and loss of local genomes (Linhart 1995). Non-native species may overpower and displace resident native species (Belnap 1995; Kuykendall 2003). Therefore, restoration with local, site-adapted native species is recommended for *S. spaldingii* sites where weed control has created openings susceptible to reinvasion.

Many constraints to using native seeds exist, however, including cost, availability, and establishment, particularly in areas where considerable amounts of aggressive invasive weeds are present and precipitation is low and unpredictable (Allen 1995; Roundy et al 1997). Current availability of native seed is limited and often costly (Belnap 1995; Monsen and McArthur 1995; Roundy et al. 1997; Bermant and Spackeen 1997; Boyer 2003; Gisler et al. 2003; Kendig 2003; Darris 2003).

There has been a recent trend in rangeland revegetation projects from using introduced grass species for forage production and soil conservation objectives to more emphasis on ecological goals and utilization of native grass species to increase biodiversity and maintain ecological integrity of the native ecosystem (Roundy et al. 1997; Richards et al. 1997; Darris 2003). Consequently, supplies of native seed and knowledge regarding their utilization are increasing.

Public land management agencies, like the USFS and BLM, have initiated programs to collect and use native plant species in habitat restoration and revegetation projects (Erickson 2003; Kendig 2003). The USFS in Umatilla National Forest in Blue Mountains of eastern Oregon and Washington built a seed bank of local seed which is used to reseed species back into specific areas from which they were collected to maintain genetic integrity (Kaiser et al. 2003).

The nursery industry and university plant materials centers have evolved to meet the demand for restoration projects using native species, providing seed and planting stock and facilities for native seed increaser projects (Hill 2000), developing cultural practices to propagate native seed, and incorporating considerations of genetic appropriateness of native seed stock (Monsen and McArthur 1995; Munda and Smith 1995; Bermant and Spackeen 1997; Darris 2003; Kuykendall 2003). Partnerships between the nursery industry, public agencies, non-profits, and local communities have formed to meet the need of native seed for restoration projects (Boyer 2003). Information databases have developed to provide information on native species and access to economical sources of native plant materials from local genetic neighborhoods (Gisler et al. 2003).

Restoration with local, site-adapted native species is recommended for restoring *S. spaldingii* occurrences, however, in view of current constraints of using native seeds and depending on conditions of the site to be restored, rehabilitation projects may be more appropriate in some cases. Sites with considerable disturbances or alteration of site conditions, i.e., loss of topsoil, may require revegetation using more competitive, quickly-establishing native

cultivars. The use of these species could be used initially to stabilize the soil or recapture the site from weeds and subsequently be replaced by more locally adapted native species (Roundy et al. 1997).

Restoration of *S. spaldingii* sites may be more costly than rehabilitation to implement initially, but will help to ensure viability of *S. spaldingii* by maintaining biodiversity and ecological integrity of the site. Restoration will also be more stable and require less maintenance in the long run (Allen 1995).

#### Coordinated Weed Management Area (CWMA)

The development of CWMA's at priority *S. spaldingii* populations where they do not currently exist is needed. CWMA's facilitate cooperation among federal, state, county land managers and private landowners by combining resources and obtaining funding to accomplish the common goal of preventing the spread of weeds into and within the CWMA (USDI 1996a; USDA/USDI 2000). Weed control at *S. spaldingii* locations within CWMA's should be a priority for CWMA's.

### **Livestock Management**

Livestock grazing threatens *S. spaldingii* and its habitat by altering the native plant community and promoting weed invasion, dispersing weeds, and reducing reproductive effort through herbivory and trampling (see Livestock Grazing section in Threats section of this report).

Livestock grazing should be reduced or eliminated at priority *S. spaldingii* populations. If this is not possible, reduce intensity, duration, and adjust seasonality to avoid active growth periods of *S. spaldingii* (B. Benner, *in litt.* 1999, 2003c, 2003d). Avoid consecutive years of use to allow rest years from grazing. Rotate grazing so that the pasture is not grazed at the same time each year (unless it is dormant season grazing) (B. Benner, *in litt.* 2003c). If grazing occurs during the growing season, monitor the area and remove livestock when utilization criteria are reached on key areas that get "normal" use (B. Benner, *in litt.* 2003c). Where they do not currently exist, implement grazing management plans that include assessments of livestock use, plant community composition, and population trends (B. Benner, *in litt.* 2003c). Consider the construction of fences to exclude livestock from *S. spaldingii* populations; ongoing inspection and maintenance of fences would be necessary. All grazing allotments should be surveyed for *S. spaldingii* and the management agencies should commit to a deadline (J. Hill, *in litt.* 2003c)

### **Fire Management**

Fire is beneficial in preventing woody encroachment at some *S. spaldingii* sites such as rough fescue grasslands in northwestern Montana (Lesica 1999); however, the majority of *S. spaldingii* locations occur within areas where low summer moisture cannot support trees and grasslands are the climax vegetation (Daubenmire 1968a, 1970, 1972; Tisdale 1986a). In productive rough fescue grasslands that accumulate large amounts of litter, fire can enhance

*S. spaldingii* recruitment (Lesica 1995, 1999); however, fire may be less beneficial in Idaho fescue grasslands that produce less litter. Fire, both spring and fall prescribed fires and fires occurring during the normal fire season do not appear to negatively affect established *S. spaldingii* plants (Lesica 1999; Menke 2003; Hill and Fuchs 2003; P. Lesica, *in litt.* 2003b).

However, fire has the potential to promote the establishment and spread of invasive weeds in *S. spaldingii* habitat (Christensen and Burrows 1986; Peters and Bunting 1994; Agee 1996; Asher 1998; Asher et al. 1999; Keeley 2002; USDI 2000). Prescribed spring and fall fires at Dancing Prairie increased recruitment of sulfur cinquefoil (Lesica and Martin 2003). Wildfires at Garden Creek Ranch increased yellow starthistle and Japanese brome in Idaho fescue communities (Lichthardt and Gray 2003; Hill et al. 2003; Hill and Fuchs 2003). Fires in areas dominated by fire-tolerant cheatgrass contribute to increased fuel loads and inflammability, resulting in more frequent and intense fires that further enhance invasion (Whisenant 1990; Peters and Bunting 1994; Keeley 2002; USDI 2000). Prescribed fires conducted in early spring and late fall have potential to harm native bunchgrasses and many native perennial forbs that are photosynthetically active during these time periods (S. Bunting, personal communication, 1999; Daubenmire 1968c; Wright 1972). High fire frequencies or out-of-season fires may harm ground-nesting *S. spaldingii* pollinators like *Bombus fervidus* (Lesica and Heidel 1996). (Also see other threats associated with fire in the Threats Section).

The effects of fire, both natural and prescribed, on the mesic fescue communities that support *S. spaldingii*, have not been thoroughly investigated and need further research (see Research Needs section of this report). Use of fire should mimic as closely as possible natural fire frequencies and seasonality to which the associated plant community is adapted (Holdgate 1986). *Silene spaldingii* and associated plant communities are well suited to naturally occurring late season fire (Menke 2003). Use of fire at *S. spaldingii* populations should be carefully evaluated if any invasive weeds occur at or near the site. Pre-burn and post-burn inventory and monitoring of *S. spaldingii* plants, weed infestations, and the associated plant community should be conducted where fire is being considered as a management tool. Studies should involve pre-fire vegetation monitoring conducted during the optimum growth period for the plant community, and long-term (at least 5 years) annual post-fire vegetation monitoring of the plant community and associated *S. spaldingii* plants. Long-term monitoring may be necessary to determine fire treatment effects on vegetative trend, mortality of native bunchgrasses and increases in weed species (Bunting et al. 1987; Johnson 1998; Dyer 2003). Long-term, pre-fire monitoring (at least 3 years) of *S. spaldingii* plants is recommended to verify age class of rosettes prior to application of fire. Weed infestations within *S. spaldingii* habitat to be burned should be mapped, and type of species present, their cover or density, and areal extent of infestation recorded. Studies should include comparison with similar unburned areas with similar extent of weed infestations. Long-term, pre- and post-fire monitoring should be incorporated in prescribed fire plans and budgeting (J. White, *in litt.* 2003).

## INVENTORY RECOMMENDATIONS

Potential unsurveyed habitat for *S. spaldingii* exists in all physiographic regions across its range. Areas in the Canyon Grasslands and portions of the Channeled Scablands have high conservation value due to their extensive, contiguous, unfragmented nature, high quality habitat, current low degree of weed invasion, and low threat from conversion to cropland and residential development. Continued surveys for additional *S. spaldingii* occurrences are recommended within the areas listed in the Potential Unsurveyed Habitat in the Distribution and Abundance section of this report.

Immediate surveys are recommended in the Oregon Canyon Grasslands because this area has high potential for *S. spaldingii* occurrences, limited human-induced fragmentation, large contiguous potential habitat, less weed infestation than Idaho or Washington Canyon Grasslands, yet is highly susceptible to invasion (USDI 1996a). Weed management at any discovered *S. spaldingii* populations in this area could be focused on prevention, early detection, and control of small satellite infestations, which is far easier and more economical than the extensive weed control and restoration needed at other Canyon Grassland *S. spaldingii* locations.

Many of known *S. spaldingii* occurrences were surveyed in the late 1980s and early 1990s and have not been re-visited. These sites need to be revisited to determine the current status of the populations (G. Glenne, *in litt.* 2003b).

## **RECOMMENDATIONS FOR HABITAT FRAGMENTATION**

Small *S. spaldingii* populations located in fragmented habitat are highly threatened by pollination limitation, inbreeding depression, and genetic and evolutionary hazards associated with small population size (Lesica 1993; Lesica and Heidel 1996; Barrett and Kohn 1991). Viability of these small *S. spaldingii* populations is a major concern (G. Glenne, *in litt.* 2003b; B. Benner, *in litt.* 2003c; F. Caplow in G. Glenne, *in litt.* 2003b). Of the known 66 populations of *S. spaldingii*, 43 of them (65%) are small populations (<100 plants each). The majority of these populations, ~34 of 43 (~79%), of them are located on isolated habitat fragments resulting from agricultural and urban/residential development, and the majority of them, 30 of 43 (70%), are privately owned without regulations for their protection. Although these populations support a small proportion of the total known plants, they may represent a large portion of the genetic diversity within the species. They represent the only remaining genetic diversity from the Palouse Grasslands, a critically endangered region with less than 1% of its original habitat remaining (Noss et al. 1995).

State or federal land management agencies and private conservation organizations should pursue land acquisition or exchange, or conservation easements of privately owned *S. spaldingii* populations to provide permanent protection to these sites. Two *S. spaldingii* locations are obvious priorities, Kramer Prairie, Washington, and Wallowa Lake, Oregon:

1. Kramer Prairie: Privately owned populations, i.e., Wawawai Eyebrow (EOR WA 011), Steptoe Canyon (EORs WA 012, 013) and Upper Wawawai (EOR WA 009), occur in close proximity to the Kramer Prairie population (EORs WA 006, 024) that is a recommended site for protection (Table 12) (Figure 7A – Appendix A). Genetic

links may exist between them that will favor viability of these small populations. This is one of the few Palouse Prairie *S. spaldingii* locations with that potential. Genetic links among metapopulations can favor population viability because the contribution of even one or a few migrants per generation can reverse the loss of genetic variation in small populations (Menges 1991). Movement of propagules among populations may allow persistence of metapopulations even when individual plant populations are not viable (Menges 1990). All three of these privately-owned populations are currently under the Volunteer Registry Program and have received some degree of protection in the past. Access restrictions may be necessary at Kramer Prairie to ensure that public use and research studies are compatible with maintenance of the *S. spaldingii* population that occurs at the site (B. Weddell, *in litt.* 2003c).

2. Wallowa Lake: The Wallowa Lake population (EOR OR 001, 010, 011) is relatively large (>500 plants), but is primarily under private ownership. Recreational development and further habitat fragmentation are an imminent threat at this site.

Private landowners of *S. spaldingii* populations should be contacted and made aware of the high conservation values of their property. They should be informed of options for voluntary protection, conservation easements, or donation of the property to a land trust, and the tax benefits of such agreements (Weddell and Lichthardt 1998). They should also be informed of opportunities through private landowner incentive programs, such as the USFWS Partners for Fish and Wildlife Program and the Private Stewardship Grants Program (USFWS 2004), and the Natural Resource Conservation Service (NRCS) Grassland Reserve Program and Wildlife Habitat Incentives Program (NRCS 2004).

Encourage protection of remaining native bunchgrass habitat by informing the public of the high conservation value of *S. spaldingii* and associated native bunchgrass communities. Cooperation with partners whose goals are to protect habitat that supports *S. spaldingii*, such as the Palouse Land Trust and Palouse Prairie Foundation (PLT 2004; PPF 2004), is encouraged. Help to revise government policies that provide incentive for private landowners to convert grassland to cropland and/or retain marginal cropland instead of converting it back to grassland (Conner et al. 2001).

Collect *S. spaldingii* seeds from locations where it is in immediate danger of destruction. Maintain this genetic diversity in botanical gardens or long-term seed storage to provide insurance against catastrophic events and facilitate the possibility of reintroduction in the future when appropriate habitat becomes available (Barrett and Kohn 1991; Holsinger and Gottlieb 1991; Guerrant 1997; Way 2003). Seeds could either be grown at a regional botanical gardens such as the University of Idaho Arboretum in Moscow, Idaho, or the Palouse Prairie Preserve in Pullman, Washington, or placed in a long-term storage facility like the Berry Botanic Garden in Portland, Oregon, a seed bank set up for the purpose of conserving the rare flora of the Pacific Northwest (Guerrant 1997). Currently, seed from two *S. spaldingii* populations are stored at Berry Botanic Garden, i.e., Garden Creek Ranch (EOR ID 006) and Cats Back (EOR OR- 008).

## MONITORING RECOMMENDATIONS

Monitoring is essential for determining population trends, reproductive success, habitat condition, and response to disturbances and management activities. Long-term monitoring is essential for determining the size of *S. spaldingii* populations and the degree of recruitment and mortality. Since rosettes can be produced by both seedlings and adult plants and the distinction between them is difficult (Hill and Weddell 2003), long-term monitoring is necessary to determine the life stage of individual plants. Without long-term demographic studies prior to introduction of management activities, it is difficult to determine whether management is enhancing or maintaining the population.

The presence of prolonged dormancy and the variation in the proportion of dormant plants between years complicates determination of the size of *S. spaldingii* populations and makes it difficult to determine population trends or assess management. Surveys may be hampered in years when large proportions of populations are dormant during the growing season and may be erroneously reported to be extirpated (Lesica and Steele 1994).

Monitoring of *S. spaldingii* populations should follow a standardized protocol that allows for: 1) comparisons between sites, management, threatening activities, etc, 2) site-specific differences, 3) evaluation of long-term response to local variability in habitat or climate, and 4) habitat assessment, i.e. information on plant community, invasive weed species, herbivory, and weather (to correlate demographic features with weather data). Techniques for monitoring plants may need to differ according to the site, but the habitat assessment variables should be consistently measured across all sites (G. Glenne, *in litt.* 2003b). An example of a monitoring protocol for *S. spaldingii* has been developed by Lichthardt and Gray (2003).

Monitoring trend rigorously requires estimating density in sample plots and comparing between two time periods or over many time periods (repeated measures analysis) (P. Lesica, *in litt.* 2003b). Trend monitoring of a plant with prolonged dormancy can be dealt with in two ways, i.e., repeated measures analysis (Lesica and Steele 1994; Lesica and Steele 1996; Lesica and Steele 1997) and mark-recapture methods (Alexander et al. 1997; Shefferson et al. 2001; P. Lesica, *in litt.* 2003b). An alternative approach to monitoring trend of *S. spaldingii* populations is monitoring of the associated plant community, including non-native invasive species (F. Caplow in G. Glenne, *in litt.* 2003b).

Long-term monitoring of *S. spaldingii* populations and associated plant communities should be conducted at priority conservation areas (Table 11). *Silene spaldingii* populations that currently have long-term monitoring transects in place at priority conservation sites include Dancing Prairie (Montana), Lost Trail NWR (Montana), Wild Horse Island (Montana), Garden Creek Ranch (Idaho), Coal Creek (Washington), and Blue Mountains (Washington).

At project sites where planned management has the potential to cause some change to *S. spaldingii* or its associated community, monitoring of *S. spaldingii* and the associated native plant community should be established and data collected (at the appropriate time of the

year) prior to project implementation. Post-management monitoring should also be done for the number of years that changes might be expected from this management activity.

Consider conducting monitoring *S. spaldingii* twice a year, once in the early part of the growing season prior to blooming and once in July or August when the plants are flowering and seeding. Early season is a better time to locate rosettes and get a more accurate count of stemmed plants. A considerable proportion of plants can disappear or not be visible later in the growing season. Later in the season many of the rosettes have dried up and stemmed plants may be overlooked due to herbivory or desiccation. In monitoring transects at Garden Creek Ranch, 5% of stemmed plants and 10% of rosettes present in June were absent in August, and 28% of remaining stems and 67% of remaining rosettes had senesced by August (Hill and Weddell 2003). A later visit in July or August when the plants are blooming can be used to document reproductive effort, effects of herbivory, etc.

### **REPORTING AND RECORD-KEEPING RECOMMENDATIONS**

There is a need for standardization in reporting information at *S. spaldingii* sites. It is recommended that everyone doing inventory work for *S. spaldingii* become familiar with the plant community keys for habitat typing and their use, and to use the appropriate keys for the areas in which they are working. It is recommended that the Daubenmire (1970) plant community key be used for habitat-typing Canyon Grasslands with a snowberry and/or rose component and the Johnson and Simon (1987) or Tisdale (1986a) plant community keys be used for habitat-typing Canyon Grasslands without a shrub component (J. Hill, *in litt.* 2003a). Standardization is needed for how *S. spaldingii* plants are counted. Some EORs recorded “stems”, others reported “plants”, and others reported “plant clumps” (IDCDC 2003b; MNHP 2003b; ONHP 2003b; WNHP 2003b). The majority of EORs counted all visible plants at the site while others were estimated from walking random transects and counting visible plants (IDCDC 2003b; MNHP 2003b; ONHP 2003b; WNHP 2003b; P. Lesica, *in litt.* 2003a, 2003b). What is meant by the size of an occurrence should be more adequately explained and consistently applied across all regions. All EORs should include aspect, slope, elevation, etc.

The development of rangewide Element Occurrence (EO) and EO Rank specifications for *S. spaldingii* is needed. This will allow for more consistent reporting and record-keeping between states. There may be fewer, larger EOs than how we currently think about EOs for *S. spaldingii*, and this will be relevant for the species recovery criteria (F. Caplow, *in litt.* 2003b). The development of a biologically- or ecologically-defined population criteria for *S. spaldingii* is needed. It is essential to determine the pollen dispersal range by understanding more about the pollinators and their potential ranges (F. Caplow, *in litt.* 2003b).

It is recommended that people from government agencies, NHP/CDC programs, and those most active in inventory and monitoring form a recovery team (F. Caplow, *in litt.* 2003b), and that annual reports documenting conservation actions and results, status of priority populations, and evaluation of conservation activities are prepared (J. Lichthardt, *in litt.* 2003c).

## RESEARCH NEEDS

A number of questions about *S. spaldingii* and its relationship to the biotic and abiotic elements of its environment need addressing. Information obtained from research on these topics will enable more effective protection and conservation of *S. spaldingii*. In order of priority, research studies that address the following issues are recommended: 1) Non-native invasive plants, and major contributing disturbance factors, livestock grazing and fire; this research will help to lessen the imminent and pervasive threat from weed invasion and help to determine appropriate use of fire and livestock grazing within *S. spaldingii* habitat, 2) Habitat fragmentation, gene flow (pollination and dispersal), population viability, and other genetic factors; this research will help to elucidate metapopulation dynamics, population parameters, a genetic basis for population conservation prioritization, and viability of small *S. spaldingii* populations on isolated fragments of habitat, 3) Life history and population trend; this research is critical to understanding of the biology and ecology of the species and to determine response to threats and management activities, and 4) Predictive habitat models; this research would help direct further searches in unsurveyed potential habitat.

### NON-NATIVE INVASIVE PLANTS

- Research is needed to determine the effectiveness of different weed control techniques (i.e., spraying, hand pulling, mowing, etc.) within and adjacent to *S. spaldingii* populations. Since chemical control is often the preferred method of weed control in inaccessible areas because of its residual control, speed of application and reduced cost, it is necessary to determine the response of *S. spaldingii* to various types and strengths of herbicides.
- Research to determine the effects of weed invasion on *S. spaldingii* populations is needed. Declines of *S. spaldingii* habitat due to degradation by weed invasion need to be documented. Competition studies are needed to determine whether *S. spaldingii* seedlings can successfully establish in invaded areas and whether established populations of *S. spaldingii* can be displaced by invasive weed species. Research is also needed to determine the effect of different invasive weed species and different weed densities on *S. spaldingii* viability, i.e., can *S. spaldingii* successfully reproduce within areas infested with low-density infestations of certain weed species?
- Research is needed to develop additional biological control agents for invasive weeds affecting *S. spaldingii* and its habitat. Research to determine the effectiveness of biological control agents on invasive weeds within *S. spaldingii* habitat is also needed.
- Research is needed to determine how to successfully restore *S. spaldingii* habitat. Little information is available regarding the restoration of degraded mesic fescue bunchgrass communities. Restoration is essential to return habitat degraded by weed invasion or livestock grazing to its original structure, composition and function.

- Research is needed to determine whether the mesic fescue communities that support *S. spaldingii* are more resistant to weed invasion than the drier surrounding bluebunch wheatgrass communities. At Garden Creek Ranch, the drier southerly slopes that support bluebunch wheatgrass are very seriously degraded by weed invasion, however the mesic fescue communities are considerably less invaded (Hill and Gray 1999). In Montana, long-term data suggest that weeds, particularly sulfur cinquefoil, invade sparsely vegetated warm slopes more quickly than cool slopes. Cool slopes are eventually invaded but at a slower rate (Lesica 1997b). This research has important implications for viability of *S. spaldingii* in regions where mesic fescue grasslands are intermixed with drier bluebunch wheatgrass communities that may be seriously degraded by weed invasion.
- Further research studying the feasibility of using remote sensing technology to detect weed infestations is needed. Periodic weed surveys are necessary for prevention and early detection; however, ground surveys are difficult, time-consuming and costly, especially in areas of rugged, inaccessible terrain. Remote sensing methods offer the capability of rapid, complete and frequent coverage of an area on a periodic basis (Lass 1999).
- Research is needed to determine whether yellow starthistle is expanding its ecological amplitude and evolving to adapt to the low winter light levels present on northerly slopes that support *S. spaldingii* in Canyon Grasslands (Hill and Gray 1999, 2000; Hill 2001b). This research has important implications for viability of *S. spaldingii* in regions where mesic fescue grasslands are intermixed with drier bluebunch wheatgrass communities that may be seriously degraded by weed invasion.
- Research is needed to determine what factors are responsible for dispersal of invasive weed seed within *S. spaldingii* habitat and ways to reduce this dispersal.

## **LIVESTOCK GRAZING AND HERBIVORY**

- Research is needed to determine at what level livestock grazing is compatible with *S. spaldingii* habitat integrity. Livestock grazing can alter native bunchgrass communities and promote the establishment and spread of weeds (Mack 1986; Tisdale 1961, 1986a). Pacific Northwest Bunchgrasses are poorly adapted to grazing (Mack 1986; Tisdale 1961, 1986a) and their seedlings cannot compete with invasive weeds (Harris 1967; Hironaka 1989). Herbivory and trampling by livestock can cause considerable damage to *S. spaldingii* and reduce reproductive effort (Kagan 1989; Gamon 1991; Elseroad and Taylor 2002a; B. Benner, *in litt.* 1999; 2003d). Information is needed on how differences in duration, intensity, and seasonality of grazing can impact *S. spaldingii* habitat, weed invasion, and reproductive output of *S. spaldingii* in different regions. Studies should include comparison with similar ungrazed areas.

- Research investigating selection for *S. spaldingii* by domestic livestock and native ungulates is needed. Issues needing clarification include: 1) the degree to which livestock and native ungulates select for *S. spaldingii*, 2) if and how selection changes through the growing season, and 3) preferential selection of certain *S. spaldingii* plants, i.e., those that are taller, support more flowers/seedheads, or occur in groups, etc. (B. Benner, *in litt.* 2003d; B. Weddell, *in litt.* 2003a).
- Research to identify insect and small mammal predators and determine the type and degree of damage they cause is needed. Predation on *S. spaldingii* seeds occurs by a number of different small mammal and insect species, most have not been identified or the degree of damage monitored or assessed.

## **FIRE**

- Further research investigating the effect of fire, both naturally occurring and prescribed fires, on *S. spaldingii* and the associated plant community is needed. Although fire, both prescribed spring and fall burns, and wildfire does not appear to negatively affect established *S. spaldingii* plants (Lesica 1999; Hill and Weddell 2003, Menke 2003), it can have negative effects on the associated native plant community. Fire can promote weed invasion (Hill et al. 2003; Hill and Fuchs 2003; Gray and Lichthardt 2003a; Christensen and Burrows 1986; Asher 1998; Asher et al. 1999; USDI 2000). Early spring and late fall fires have potential to harm *S. spaldingii* communities in the tri-state area because they are not compatible with the seasonal activity patterns of the major bunchgrasses and many of the perennial forbs that are photosynthetically active from autumn through early summer (Daubenmire 1942, 1970; Tisdale 1986a). Studies should involve pre-fire vegetation monitoring conducted during the optimum growth period for the plant community, and long-term (at least 5 years) annual post-fire vegetation monitoring of the plant community and associated *S. spaldingii* plants. Long-term pre-fire monitoring (at least 3 years) of *S. spaldingii* plants is recommended to verify age class of rosettes prior to application of fire. Weed infestations within the *S. spaldingii* habitat to be burned should be mapped and type, density, and areal extent of infestation recorded pre-fire and five years post-fire. Studies should include comparison with similar unburned areas.
- Research is needed to determine the effect of litter on *S. spaldingii* recruitment in Idaho fescue grasslands of the tri-state area of *S. spaldingii* occurrence. Fire increased *S. spaldingii* recruitment in highly productive rough fescue grasslands that accumulate large amount of litter (Lesica 1991b, 1995, 1999; P. Lesica, *in litt.* 2003 b), but the effects of fire may not have similar effects in the less productive Idaho fescue grasslands in Idaho, Oregon and Washington. In addition, the warmer, winter-high-precipitation climates that characterize Idaho fescue grasslands in the tri-state area may have different effects on decomposition rates of litter that accumulates in these areas compared to the cooler, summer-high precipitation climate that characterizes the Dancing Prairie area in northwestern Montana (Climate Diagrams – Appendix C). There are also some questions as to whether fire is necessary for *S. spaldingii* viability in rough fescue grasslands (B. Martin, *in litt.* 2000; P. Lesica, *in*

*litt.* 2003b) and whether fire increased *S. spaldingii* recruitment. All rosettes in the fire study at Dancing Prairie were counted as new recruits although no long-term monitoring was done on fire plots prior to burning (Lesica 1991b, 1995, 1999). Research conducted at Garden Creek Ranch indicates rosettes produced by seedlings and rosettes produced by older, established plants are difficult to distinguish without long-term monitoring.

- Research is needed to determine whether woody encroachment is occurring in *S. spaldingii* habitat in the tri-state area and whether that encroachment presents a threat to *S. spaldingii* or its habitat (B. Weddell, *in litt.* 2003b). Increases in deciduous shrubs and ponderosa pine have been observed on steppe and shrub-steppe fragments (R. Crawford, *in litt.* 2003) in the tri-state area of *S. spaldingii* occurrence. However, limited summer soil moisture has been shown to prevent trees from becoming established in these zones (Daubenmire 1968a, 1970, 1972). Oscillations in vegetation composition within steppe regions may be due to climatic fluctuations, changes in fire regime, or response to overgrazing (Daubenmire 1978; Tisdale 1986a; Johnson and Simon 1987; R. Crawford, *in litt.* 2003).
- Research is needed to determine if fire is an effective method for controlling snowberry and rose in Idaho fescue-snowberry and Idaho fescue-rose habitat types that support *S. spaldingii*. Snowberry and rose are known to sprout readily and often increase in vegetative cover following fire (Fire Effects 2003).

#### **POLLINATION AND SEED DISPERSAL**

- Research on the natural history of *Bombus fervidus* within *S. spaldingii* range is needed. Information on location of nests, queen over-wintering sites, preferred habitats, resources besides *S. spaldingii*, competition with invasive species for pollinator service would be helpful (P. Lesica, *in litt.* 2003b). This information will be helpful in developing management plans that will protect this important pollinator.
- Research to determine the effects of fire, livestock grazing, and pesticide application activities on *Bombus fervidus* is needed. Since a beneficial and necessary connection exists between *S. spaldingii*, information on the effects of these potentially harmful activities on this ground-nesting insect is essential.
- Research to determine the possibility of night pollinators for *S. spaldingii*, specifically noctuid moths, is needed. Many of these mostly night-pollinating moths are known to pollinate as well as predate seeds of other species of *Silene* (Brantjes 1978; Pettersson 1992; Proctor et al. 1996; Young and Gravitz 2002; Westbrook 2003; Kephart 2003; Reynolds 2003), including the rare *Silene douglasii* var. *oraria* (Kephart 2003). Some visits by these noctuid moths to *S. spaldingii* have been observed (Lesica and Heidel 1996). Further pollination research during nighttime hours may show these moths to be significant pollinators.

- Research investigating foraging behavior of *S. spaldingii* pollinators is needed. Determination of the foraging territory of pollinators of *S. spaldingii* will provide information on gene flow, help to develop ecological criteria for defining *S. spaldingii* populations, and help to determine viability of small populations located on isolated habitat fragments.
- Research investigating seed dispersal of *S. spaldingii* is needed, in particular, the mechanism of dispersal, agent of dispersal, and potential distance of dispersal. Research is needed to determine if *S. spaldingii* seeds can survive passage through the gastrointestinal tract of potential agents of dispersal such as domestic livestock, native ungulates, small mammals and birds. Determination of seed dispersal distances will provide information on gene flow, help to develop ecological criteria for defining *S. spaldingii* populations, and help to determine viability of small populations located on isolated habitat fragments.

### **FRAGMENTATION AND VIABILITY**

- Research investigating population viability of small populations on isolated fragments of habitat is needed. This research is particularly important since the majority (65%) of known *S. spaldingii* populations are small in size (<100 plants in size), and many of them are located on small, isolated fragments of habitat where they are subject to pollination limitation, inbreeding depression, and genetic consequences that place them at high risk of extirpation (Barrett and Kohn 1991).
- Research is needed to determine the effects of anthropogenic versus natural fragmentation of *S. spaldingii* populations. *Silene spaldingii* populations are fragmented by anthropogenic factors, i.e., conversion to croplands, urban development, but also appear to be fragmented naturally by their close association with edaphic and topographic features. Research is needed to determine at what distances naturally-fragmented subpopulations are able to interact. Information obtained from this research will help clarify the genetic system present in *S. spaldingii* and its ability to survive in fragmented situations. Populations that occur naturally in small and sparsely distributed populations may possess genetic systems adjusted to close inbreeding in contrast to species whose numbers have been recently reduced due to habitat destruction and are more susceptible to the genetic consequences imposed by small population size (Barrett and Kohn 1991).

### **GENETICS**

- Further research investigating the genetic analyses of geographic differentiation is needed as a basis for future conservation and management decisions. There may be important traits differentiating *S. spaldingii* populations, i.e., flowering times, plant size and number of stems produced, etc. Differences could be environmentally induced, but could also have a genetic basis (P. Lesica, *in litt.* 2003b). It is important to know whether these differences may have adaptive value and whether the mixing of gene pools from different populations will increase or decrease successful

establishment and long-term survival (Lesica and Allendorf 1999). Previous research has shown that little genetic differentiation occurs among populations (Baldwin and Brunsfeld 1995). This could indicate very recent separation or occasional long-distance dispersal. Further molecular genetic studies with larger sample sizes, are necessary to be able to consider genetic conservation when prioritizing populations for conservation (P. Lesica, *in litt.* 2003b).

### **LIFE HISTORY, DEMOGRAPHY AND POPULATION TREND**

- Further research investigating *S. spaldingii* demography in the tri-state area of Oregon, Idaho, and Washington is needed. In a long-term demographic study using elasticity analysis of matrix models at Dancing Prairie, Montana, Lesica (1997a) showed that prolonged dormancy is an important demographic feature of *S. spaldingii*. Similar research is needed in the tri-state area to determine the role of prolonged dormancy in the life cycle of *S. spaldingii* in this area. This research will help to ascertain whether recovery objectives are valid and to specify management and population goals. The information acquired would help in determining the actual population sizes and better identify the causes of population declines.
- Research the causes and effects of prolonged dormancy. There is evidence that prolonged dormancy is weather related with high dormancy occurring in summers preceded by a wet summer and dry fall (Lesica 1997a). A model with these two variables accounts for greater than 50% of the variation in the annual proportion of dormant plants (P. Lesica, *in litt.* 2003b). Prolonged dormancy may be related to longevity and/or reproduction, and ultimately the long-term viability of *S. spaldingii*. Information is needed on what drives the prolonged dormancy process and its role in the life cycle of the individual plant. Remaining below ground in a given year or emerging but remaining in the rosette stage might increase the probability that an individual survives and/or is able to reproduce in the following year (B. Weddell, *in litt.* 2003a)
- Conduct research to enable field botanists to determine the life stage of a rosette. Distinguishing between rosettes produced by seedlings and those produced by older plants is difficult. This problem complicates the identification of life stages and degree of recruitment in demography and fire studies. Further research is suggested on leaf surface features and root and caudex development (Hill and Weddell 2003; J. Hill, *in litt.* 2003a).
- Research to assess trends in *S. spaldingii* populations is needed. Population trends are critical to assess response to threats and management activities; however trend is difficult to assess due to prolonged dormancy. Sporadic counts are not accurate. Fifteen years of demographic monitoring suggest that the Dancing Prairie population is stable; there was no trend in density of *S. spaldingii* plants in demographic plots between 1989 and 2000 (P. Lesica, *in litt.* 2003b). Considerable annual demographic data is necessary from a particular site to assess trend. Counts could be analyzed by

regression analysis testing for a slope significantly different from zero (P. Lesica, *in litt.* 2003b).

- Research potential mycorrhizal associations for *S. spaldingii*. Symbiotic mycorrhizae may allow *S. spaldingii* to accumulate energy when in a dormant state (B. Weddell, *in litt.* 2003a).

### **HABITAT**

- Considerable unsurveyed potential habitat for *S. spaldingii* has been identified within different physiographic regions of its range. Research to develop predictive habitat assessment models for each region that utilizes associated species, soil information, aspect, elevation, etc. would be useful to direct survey work (C. Bjork in G. Glenne, *in litt.* 2003b).

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