Desert Centre
Osoyoos

SER-BC

PROCEEDINGS

ECOSYSTEM AT RISK: ANTELOPE BRUSH RESTORATION

Edited by:
Robert Seaton

MARCH 28-30, 2003, OSOYOOS, BRITISH COLUMBIA, CANADA

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ECOSYSTEM AT RISK: ANTELOPE BRUSH RESTORATION

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THE SOCIETY FOR ECOLOGICAL RESTORATION, BC CHAPTER

THE DESERT CENTRE

CANADIAN WILDLIFE SERVICE

THE NATURE TRUST OF BRITISH COLUMBIA

MARCH 28-30, 2003, OSOYOOS, BRITISH COLUMBIA, CANADA

EDITED BY
ROBERT SEATON
BRINKMAN & ASSOCIATES REFORESTATION LTD
520 SHARPE ST, NEW WESTMINSTER B.C. CANADA

2003

PREFACE

This proceedings results from a conference held in Osoyoos B.C. on March 28th to 30th 2003. Organized by The B.C. Chapter of the Society for Ecological Restoration and The Desert Centre, with assistance from the Canadian Wildlife Service and The Nature Trust of British Columbia, the conference was focussed on the current state of the art in restoration and management of Antelope Brush (*Purshia tridentata*) ecosystems.

The Antelope Brush ecosystem of southern B.C. is a provincially red listed ecosystem, and is the habitat of a number of listed species. Threatened by landscape changes resulting both from deliberate conversion to other species, and from changes in ecological process brought about by fire suppression and other management actions, the ecosystem is the focus of ongoing restoration efforts.

The papers in this proceedings cover a range of approaches to the Antelope Brush ecosystem, including descriptive and analytic ecological studies, reports on specific restoration projects, overviews of restoration and recovery planning processes, and comparisons with recovery efforts in other ecosystems.

A number of people undertook key rolls in the organization of this conference. Particular thanks are given to the following individuals:

From SER-BC: Dave Polster, Don Eastman, Lee Schaeffer, John Parminter, and Patty Thomas
From The Desert Centre: Joanne Muirhead

Special thanks must be given to the authors of the papers for their exemplary efforts under tight time deadlines.

Front Cover Photo : Antelope brush burning during prescribed fire, Mar 30 2000, McIntyre Rd, east of Vaseux Lake.

Photo Credit: Jim Mottishaw, Forest Protection Officer, Penticton Fire Zone

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Experimental Ecological Restoration in the South Okanagan Shrub-Steppe

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ABSTRACT

A five-year experimental restoration project aimed at the recruitment of natural species and reduction of invasive species was initiated in Purshia tridentata habitat of the South Okanagan in 1998. After one growing season natural grasses had established as a result of both hayseeding and broadcast seeding and manual and chemical weed control methods were also effective. Soil condition and soil type influenced the cover of broadcast seeded natural grasses, while the broadcast seeding rate and addition of native VAM did not. Natural grass species responded differently to the seeding and soil amendment treatments. Manual control for Centaurea diffusa was effective the year of treatment and the year following, and chemical control was effective for one year, after which results were confounded by site changes. The most effective weed control method for C. diffusa appeared to be the removal of livestock.

INTRODUCTION

Restoration experiments undertaken at the Osoyoos Desert Site, in the South Okanagan, British Columbia, were aimed at either the recruitment of natural grass species that were missing or in low amounts (active restoration) or reducing the number or amount of non-native species that had invaded the site (passive restoration). The goal of the experiments was to provide restoration techniques to local community groups with little funding, however the data obtained over the five years also provides valuable information for large-scale development projects.

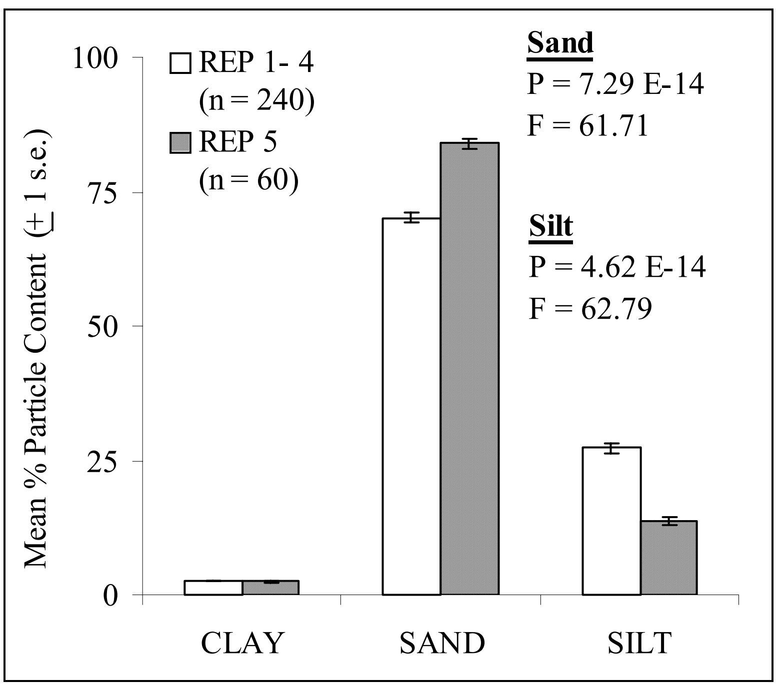
The *Purshia tridentata - Hesperostipa comata* (Antelope-brush - Needle and Thread grass) plant community is restricted to the South Okanagan and it is habitat for over 100 rare plants, 300 rare invertebrates, 29 provincially listed Red - and Blue-listed vertebrate species and four species of management concern for the South Okanagan Conservation Strategy (Scudder 1994, 1996, Bryan 1996, CDC 2002). The Osoyoos Desert Society acquired the lease to 50-ha of shrub steppe that contained remnants of the *P. tridentata* system. The *P. tridentata* shrub steppe is susceptible to livestock grazing, typically the diversity and cover of natural plant species are reduced and the amount of bare ground and cover of invasive non-indigenous species increased (1996 Antelope-brush Symposium unpublished). Historically, cattle grazed the Osoyoos Desert Centre site annually, between March and June. Approximately 40 head of cattle were removed from the site in late May 1998, prior to the Osoyoos Desert Society assuming the lease on June 1, 1998.

This paper discusses the results of 8, four active and four passive experiments conducted on the Osoyoos Desert Centre site between 1998 and 2002. Early site conditions and a description of experimental plot layout and baseline sampling are included in Atwood (1996) and Atwood and Osoyoos Desert Society 2000, respectively.

METHODS

Soil texture data that was collected from the 100-m^2 plots in 1998 identified differences in soil texture between the replications Reps 1 to 4 contained significantly more silt and significantly less sand than Rep 5 (Figure 1). As a result, for many of the experiments, data from Reps 1 to 4 were analysed separately from data collected from Rep 5.

Figure 1: The average percent of clay, sand, and silt particles in Reps 1 to 4 and Rep 5.



ACTIVE RESTORATION EXPERIMENTS

EFFECT OF NATIVE BUNCHGRASS HAYSEEDING ON SPECIES RECRUITMENT

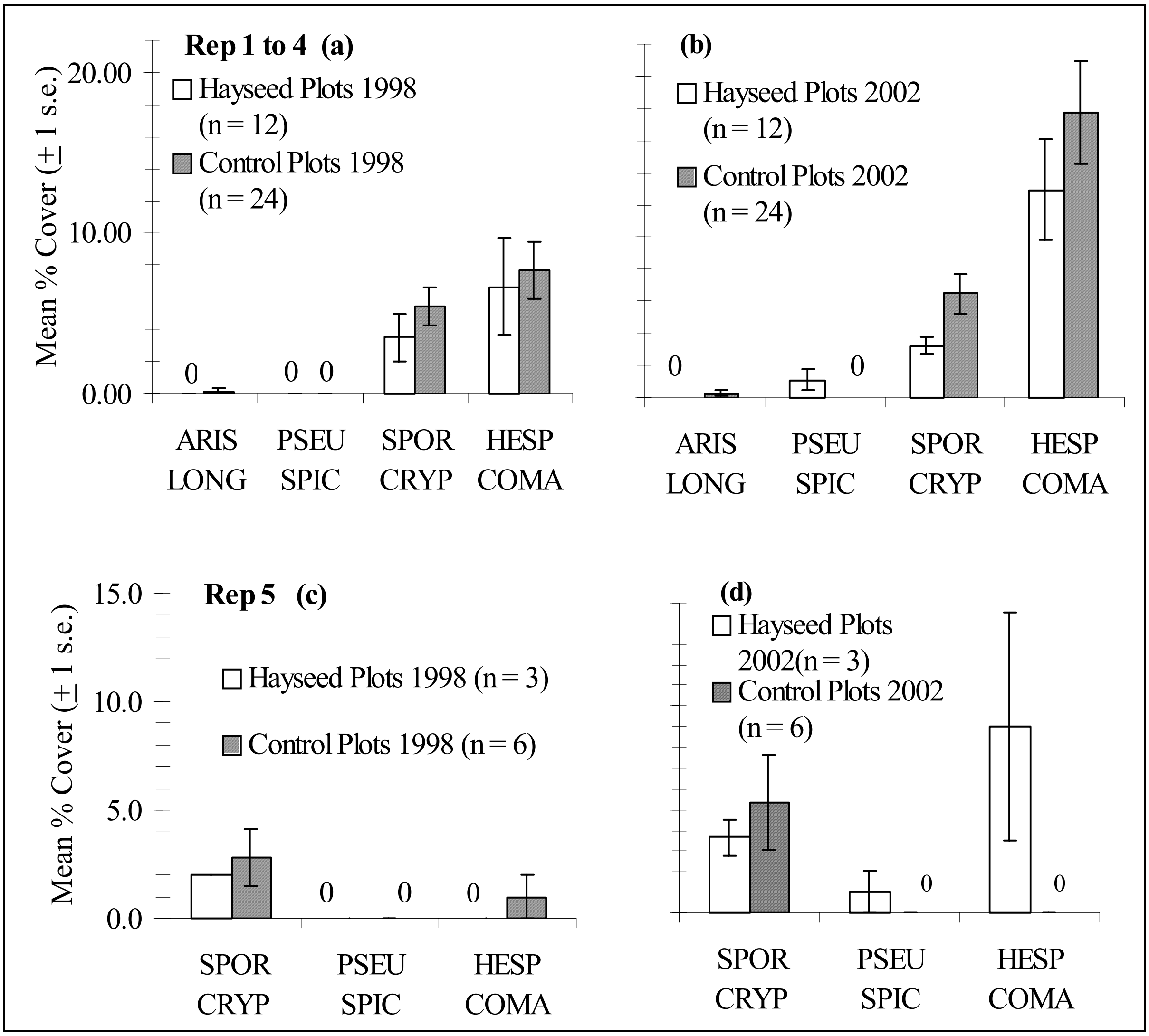
Method

The hayseeding experiment was initiated on two plots per replication (10 plots total) in September 1998. As seed matured, seed heads and plant stalks were cut from four native bunchgrasses; *Aristida longiseta* (red three-awn), *Hesperostipa comata* (needle and thread grass), *Sporobolus cryptandrus* (sand dropseed), and *Pseudoroegneria spicata* (bluebunch wheatgrass). The plant material was collected from natural shrub-steppe communities within the South Okanagan Basin Ecosection. Approximately 200 litres of plant material (50 litres from each species) was distributed evenly over each 100-m^2 plot.

Results

The hayseed material added one new native grass species to the hayseed plots in Reps 1 to 4 and two new species to the hayseed plots in Rep 5. *P. spicata* was not recorded in the hayseed plots in 1998 but in 2002 *P. spicata* accounted for 1.08% $\pm$ 0.65% of the cover in the Reps 1 to 4 hayseed plots (Figure 2b) and 1.0% $\pm$ 1.0% of the cover in Rep 5 (Figure 2d). *H. comata* was also recorded in Rep 5 and it was not found in the area before hayseed treatment. *H. comata* cover averaged of 9.0% $\pm$ 5.51% in the Rep 5 hayseed plots (Figure 2d).

Figure 2: The mean percent cover of four native grasses, *A. longiseta* (ARIS LONG), *P. spicata* (PSEU SPIC), *S. cryptandrus* (SPOR CRYP), and *H. comata* (HESP COMA), in the hayseed plots and control plots (no hayseed treatment) in in1998, before hayseed treatment and in 2002, four years after treatment in Rep 1 to 4 and Rep 5.



The hayseed material did not significantly increase the percent cover of *S. cryptandrus* in Reps 1 to 4 and Rep 5 or the cover of *H. comata* in Reps 1 to 4 as compared to the control plots ($P \geq 0.5$) (Figure 2a and c). The average cover of *H. comata* in the hayseed plots in Reps 1 – 4 did increase by 94% between 1998 and 2002 (6.67% ± 3.02% in 1998 and 12.92 ± 3.92% in 2002), however the average cover of *H. comata* in the control plots also increased substantially over the five years ($P > 0.05$) (Figure 2).

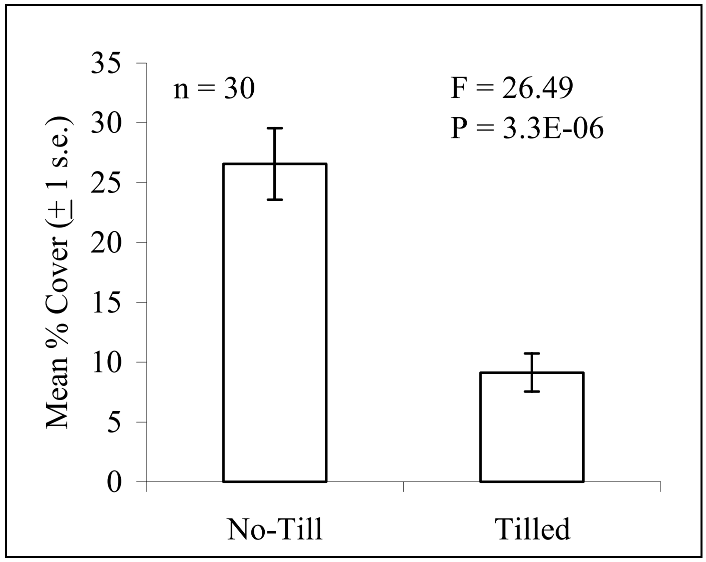
BROADCAST SEEDING OF NATURAL GRASSES: DIFFERENCE IN NATIVE SPECIES ESTABLISHMENT WITH LEVEL OF SOIL DISTURBANCE (TILLED VERSUS NON-TILLED PLOTS) AND TWO SEEDING RATES (28 KG/HA VERSUS 41 KG/HA),

Method

Broadcast seeding experiments were initiated in October 2000. 100-m^2 plots were double split, producing four 25-m^2 subplots. One-half of the plots were tilled to mimic soil disturbance that would be associated with development projects. Shrubs remained, but existing herbaceous vegetation was cut and removed from the plot before tilling and the soil was packed after tilling. Standing herbaceous vegetation was also cut and removed from the no-till plots. Non-native species remaining in the plots were spot treated with the herbicide glyphosate (Roundup), applied at the full label rate.

The seed mix consisted of four perennial native bunchgrasses (*A. longiseta, H. comata, S. cryptandrus,* and *P. spicata*) and one annual agronomic *Lolium multiflorum* (annual ryegrass). The native grasses were combined evenly in the mix (25% live seed per species) and seed rates were 28 kg / ha (1027 seeds / m^2) and 41 kg/ha (1504 seeds / m^2). Application rates were adjusted to account for the germination rate of the collected seed. Each seed rate was broadcast on one-half of the 100-m^2 plot and the soil was rolled after seeding.

Figure 3: The mean percent cover of seeded natural grasses in disturbed soils (Tilled) and non-disturbed soils (No-Till).



Results

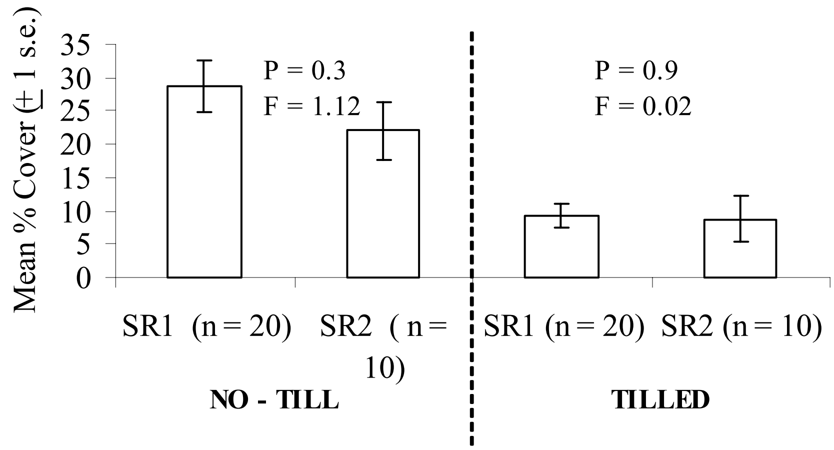
Native grass establishment in disturbed (Tilled) and undisturbed (No-Till) soils

There was a significant difference in the cover of seeded native grasses between tilled and non-tilled plots (df = 1; F = 26.49; P = 3.3E-06). After one growing season, cover of seeded grasses in tilled plots averaged 9.13% ± 1.59 compared to 26.57% ± 2.99 in no-till plots (Figure 3). Because of the difference in cover that resulted from the soil preparation technique the effect of seeding rate was analysed separately for tilled and non-tilled plots.

Effect of Seed Rate on native grass establishment

After one growing season, the seed rate did not affect seed mix establishment (No-till plots: df = 1; F = 1.12; P = 0.3: Tilled plots: df = 1; F = 0.2; P = 0.9). Plots with undisturbed soils (No-Till) seeded with native grassed at 28 kg/ha (SR1) averaged 28.8% ± 3.89% cover by June 2001 compared to an average cover of 22.1 ± 4.37% in undisturbed soil plots seeded at 41 kg/ha (SR2). The results were similar in plots with disturbed soils, with the cover from SR1 9.3% ± 1.79% and SR2 averaging 8.8% ± 3.32% (Figure 4).

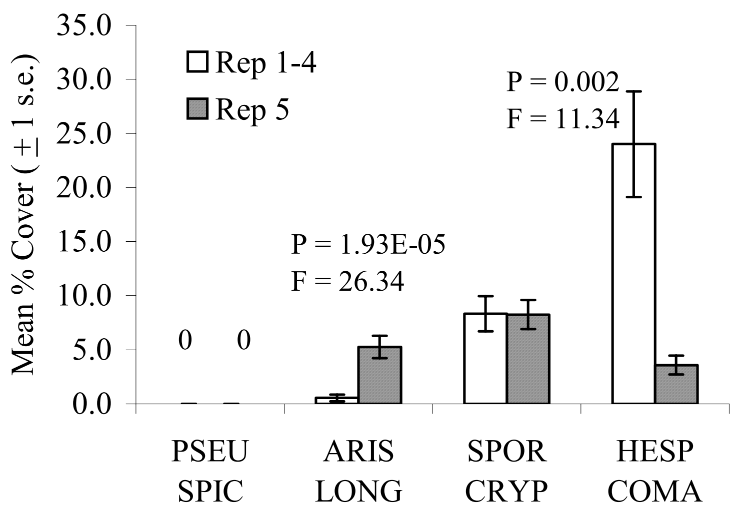
Figure 4: The mean percent cover of natural grasses seeded at rate 1 - 28 kg/ha (SR1) and rate 2 - 41 kg/ha (SR2) in disturbed soils (Tilled) and non-disturbed soils (No-Till) after one growing season.



After one growing season there were significant differences in the percent cover of the individual grass species seeded in Reps 1 and 4 and those seeded in Rep 5 (Figure 5). *P. spicata* was not recorded in any seeded plot on the site. Conversely, *S. cryptandrus* was found equally in Reps 1 to 4 and Rep 5.

However, *A. longiseta* and *H. comata* responded differently in the different replications. The average cover of *A. longiseta* was significantly higher in the sandy soils of Rep 5 (df = 1; F = 26.34; P = 1.93E-05) and the average cover of *H. comata* was significantly higher in Reps 1 to 4, which contained soils with a higher silt content (df=1; F = 11.34; P = 0.002).

Figure 5: Mean percent cover of *P. spicata* (PSEU SPIC), *A. longiseta* (ARIS LONG), *S. cryptandrus* (SPOR CRYP), and *H. comata* (HESP COMA) in Reps 1 to 4 (silty soils) and Rep 5 (sandy soils) after one growing season.



EFFECT OF THE ADDITION OF NATIVE VESICULAR ARBUSCULAR MYCORRHIZAE (VAM) FUNGI ON NATURAL GRASS SPECIES ESTABLISHMENT

Method

One 100-m^2 plots was divided into four 25-m^2 subplots and two treatments (Nurse plant inoculant and Soil-Root inoculant) and two control plots (no inoculant) were randomly established in each plot. The experimental plots were tilled, inoculated or not then seeded with the native grass seed mix at 28 kg/ha. The experiments were installed in the fall of 2000 and percent cover data for the seeded native grasses were collected in June 2001 and 2002.

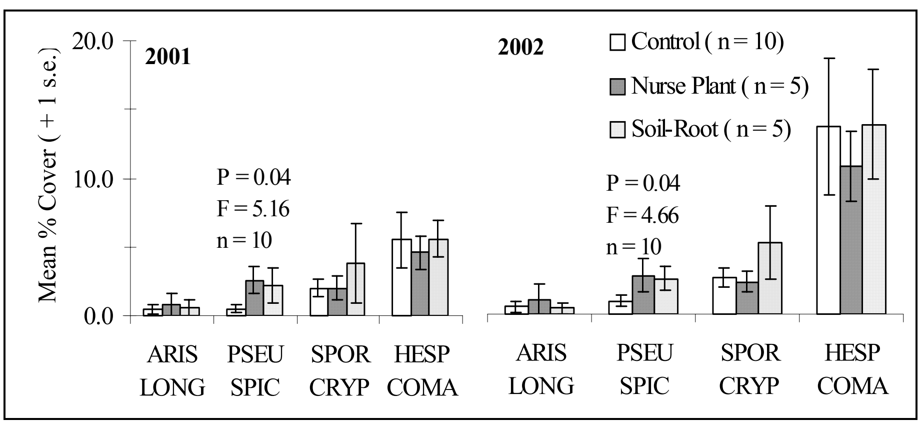
Native VAM was produced on site for the inoculant treatments. Mature bunchgrasses on the site where tested and found to be colonized with vesicular arbuscular mycorrhizae hyphae, ranging from an average percent colonization of 36.19% $\pm$ 2.28% for *A. longiseta,* 38.91% $\pm$ 1.69% for *S. cryptandrus,* 53.98% $\pm$ 1.13% for *H. comata*, and 55.4% $\pm$ 4.08% for *P. spicata*. Mature bunchgrasses from the site were used as nurse plants (Nurse plant treatment) and also used to produce VAM beds that were harvested for the Soil-Root treatment. The harvested material was added to trenches in the experimental plots, spaced 15-cm apart.

Results

After two growing seasons, there was no difference in the overall percent cover of the natural bunchgrass seed mix in the control, nurse, or soil-root inoculant plots (df = 2; F = 0.2; P = 0.8). Average cover of the seed mix ranged from 15.8% + 2.52% in the Nurse plant treatment, to 16.6% + 4.85% in the control plots and 20.4% + 4.57% in the Soil-Root treatment.

In 2001, there was a significantly higher average cover of *P. spicata* in the plots inoculated with VAM and this relationship was still evident in 2002 (n = 10, 2001: df = 1; F = 5.16; P = 0.04: 2002: F = 4.66; P = 0.04). Cover of the native grasses did not change significantly between 2001 and 2002 except for *H. comata*, which more than doubled in the year (Figure 6).

Figure 6: The mean percent cover of the seeded native grasses in the plots that did not receive VAM inoculant (Control), plots that were planted with VAM inoculated with mature bunchgrasses (Nurse plants), and plots that received a mixture of VAM inoculated roots and soil (Soil-Root). Data from the Nurse plant and Soil-Root treatments for *P. spicata* (PSEU SPIC) were combined and analysed for differences with the average cover of *P. spicata* in the control plots.



PASSIVE RESTORATION EXPERIMENTS

EFFECT OF SOLARIZATION ON WEED RE-ESTABLISHMENT

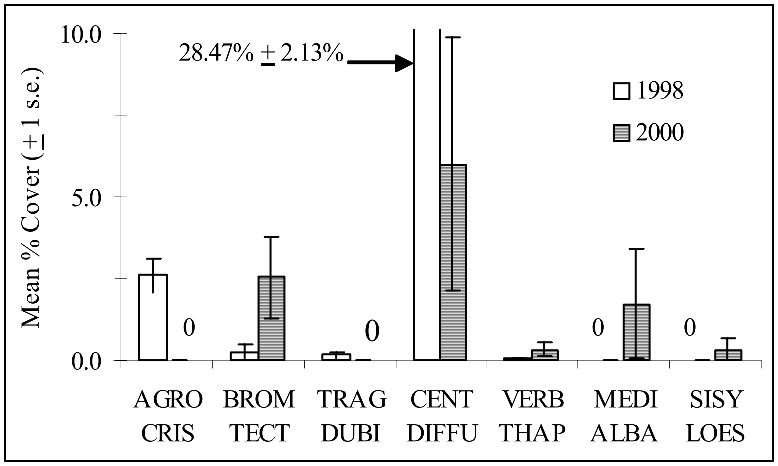
Method

Ten 100-m^2 plots were covered with transparent polyethylene sheets in 1998 and the plastic was removed from 5 of the plots in April 2000. Vegetation data were collected from the plots before the plastic was put down and following its removal (June 2000 and 2002).

Results

Five weed species were recorded in the solarization plots in 1998, before treatment and in June 2000, two months after the plastic was removed three of the species as well as two new weeds were found in the plots (Figure 10). The average cover of *C. diffusa*, and *Verbascum thapsus* (mullein) was greatly reduced from the 1998 level, but it was evident solarization had not killed the seeds. *Bromus tectorum* (cheatgrass) was the third species evident in the plot in June 2000, however seed from it and two new weed species that were found, *Meliotus alba* (sweet white clover) and *Sisymbrium loeselii* (Loesel's tumble mustard), likely moved into the plots between April and June. *Agropyron cristatum* (crested wheatgrass) and Tragopogon dubius (Yellow salsify) were recorded in the solarization plots in 1998 but were not evident in 2000.

Figure 7: The mean percent cover of weed species in the solarization plots in 1998, before treatment and in June 2000, two months after the removal of the polyethylene sheets.



EFFECT OF MANUAL AND CHEMICAL CONTROL OF *C. DIFFUSA*

Methods

One 100-m^2 plot per replication was randomly chosen for the manual weed control and two plots were treated chemically. The manual control of *C. diffusa* experiment was to determine the most effective time to hand weed *C. diffusa*, and whether weed density was related to the timing of the manual control. The first hand pulling was scheduled for early May, after which monthly treatments were scheduled if weed density was ≥ 25% of the original *C. diffusa* cover. The experiment for the chemical control of *C. diffusa* was implemented in 2000. *C. diffusa* plants in two 100-m^2 plots per replication (10 plots) were spot sprayed with an over-the-counter broad-leaf herbicide, Killex, at the recommended label rate of 1.85 kg active ingredient per hectare in May 2000. Killex, an over- the-counter combination of 2,4-D, mecoprop, and dicamba was used in the chemical control experiment to determine if adequate control of *C. diffusa* could be obtained using a less expensive broad-leaf herbicide with less residual than the commonly used Tordon 22K (picloram).

Results

Manual control of C. diffusa

The May 12th hand weeding (manual treatment) of *C. diffusa* drastically reduced the average cover in the plots between 1999 and 2002 and there was a significant difference in the average cover between the manual treatment plots and control plots (no-treatment) in 1999 and 2000 as a result of the one hand weeding in May 1999 (df = 1:1999: F = 18.07; P = 0.0002; 2002: F = 13.29; P = 0.0009). However, *C. diffusa* was decreasing across the site and by 2001 there was no difference in the cover of *C. diffusa* between the plots that received treatment and those that did not (Figure 8).

Chemical control of C. diffusa

Chemical control with Killex reduced C. diffusa cover from an average of 30.33% ± 2.74% in 2000 to 0.33% ± 0.21% in 2001. However, in 2001 there was no difference in the average cover of C. diffusa between plots treated with Killex and control plots (df = 1; F = 0.09; P = 0.8). Over the five years of the methods research project, *C. diffusa* decreased rapidly across the site. Over the long-term, the removal of livestock appears to have been as effective as either manual or chemical control

Figure 9: Mean percent cover of C. diffusa (CENT DIFF) in control plots and plots chemically treated with Killex in 2000 (after data collection) and in 2001 and 2002.

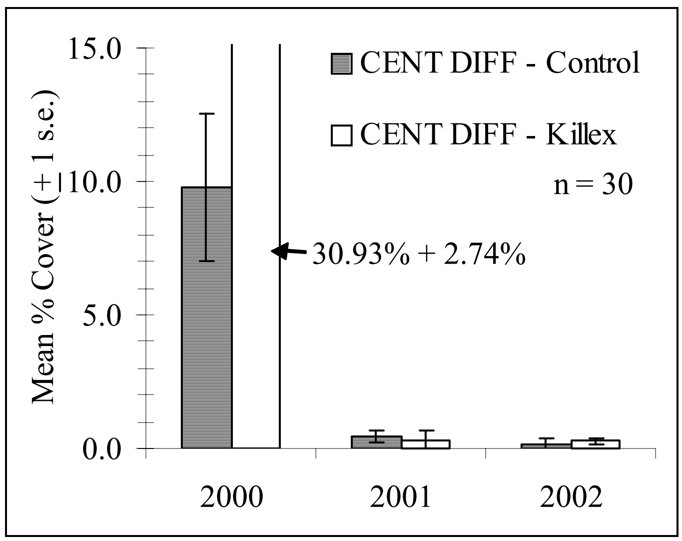
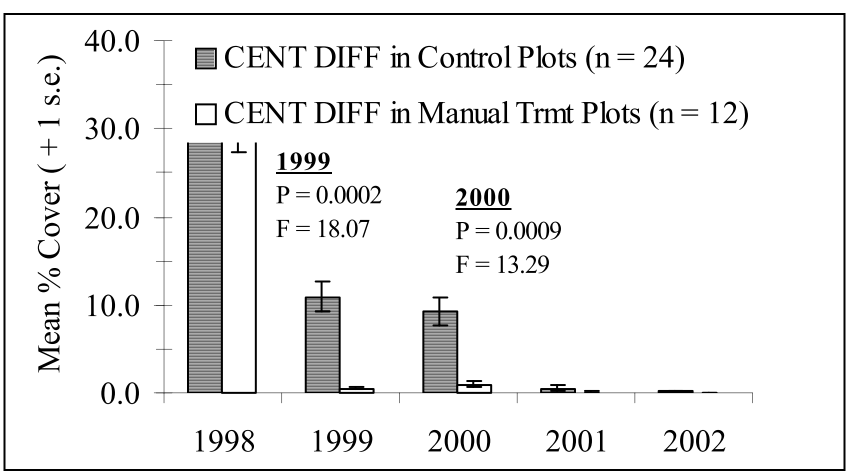


Figure 8: The mean percent cover of C. diffusa (CENT DIFF) in the plots scheduled for hand-weeding (Manual Trmt Plots) and plots that would not receive weed control treatment (Control Plots) in 1998, the year before treatment, in 1999 after one manual weed treatment, and in 2000, 2001 and 2002



RESPONSE OF THE SHRUB-STEPPE COMMUNITIES TO THE REMOVAL OF LIVESTOCK GRAZING

Method

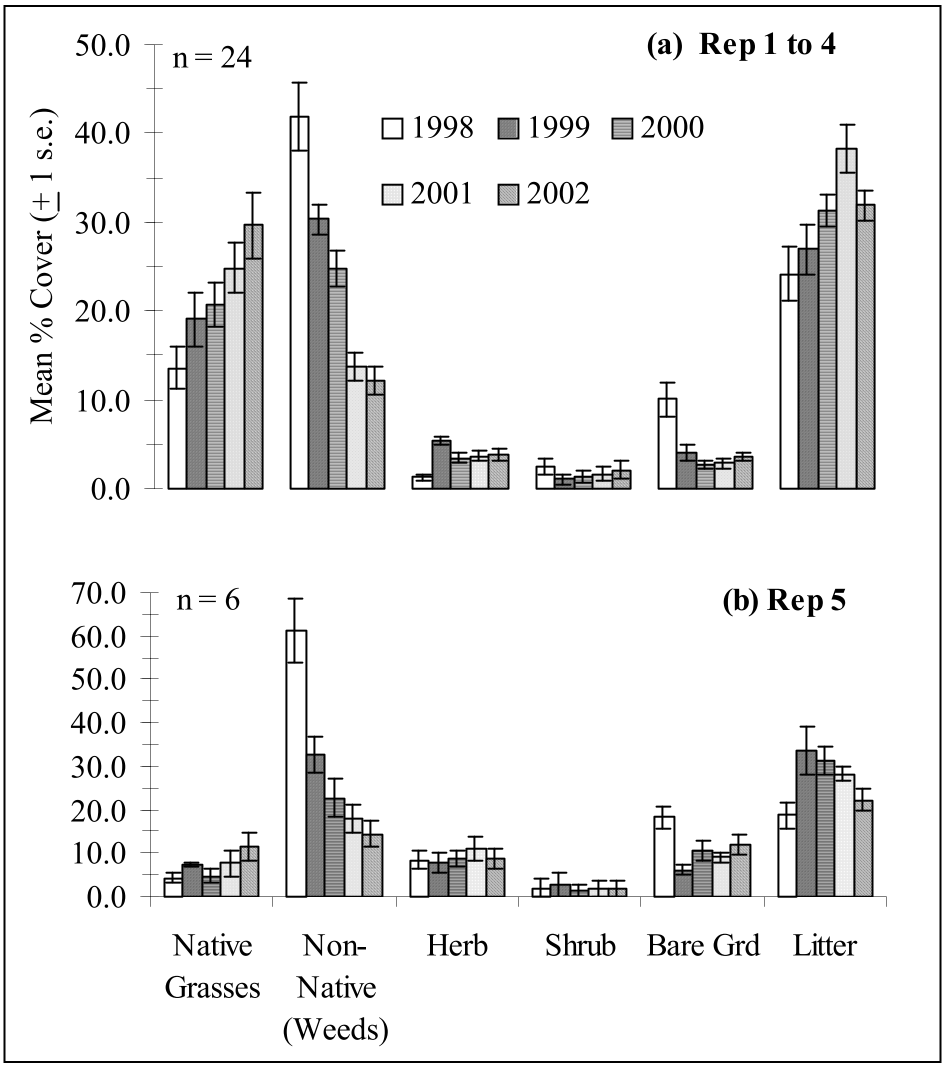
In 1998, two randomly chosen 100-m^2 plots in each replication were established as control plots (10 plots). Species identity and percent cover data were collected annually, in June, from 1998 to 2002. The control plots were monitored to document changes in the plant community without livestock grazing or restoration activity.

Results

The removal of livestock had a marked effect on vegetation components across the site. Although there was no change in the average cover of shrubs during the five years, herb and native grass cover increased significantly (df = 1; Herb: F = 10.51; P = 0.002: Native Grass: F = 5.85; P = 0.04) and non-native (weed) cover decreased significantly (df = 1: Weeds Reps 1 – 4:F = 51.92; P = 4.48E-09: Rep 5: F = 34.33; P = 0.0002). (Figure 10).

The largest reduction in weed cover occurred between 1998 and 1999, the year following the removal of cattle. Weed cover dropped 71% in Reps 1 to 4 and 77% in Rep 5 over the four years. *C. diffusa*, was the dominant weed on site in 1998 but in 2002, *Agropyron cristatum* (crested wheatgrass), which had been seeded by the former lessee, was the dominant non-native species.

Figure 10: Mean percent cover of native grasses, non-natives, herbs, shrubs and bare ground and litter between 1998, the year livestock was removed from the site, and 2002 for (a) Reps 1 to 4 and (b) Rep 5.



DISCUSSION

Native Species Recruitment

On the Osoyoos Desert Centre research site, locally collected natural grasses established as a result of both hayseeding and broadcast seeding, although after the first growing season, soil condition, disturbed versus undisturbed, and soil type, high sand content (Rep 5) versus higher silt content (Rep 1 to 4), coupled with ecological preferences of the grass species had more of an effect on overall cover of seeded natural grasses than the seed rate. Broadcast seeding was more effective than hayseeding on undisturbed soils and seed rate did not affect establishment. Rates of greater than 1000 s/m^2 are thought to be high (Jacobs *et al*. 1999) and it appears applications that exceed that amount are unnecessary. The average cover of 28% that was reported on site would indicate that roughly 290 of the 1027 seeds planted germinated. Further work is required to determine if the level of plant establishment is a reflection of the carrying capacity of the local soils, given their low moisture and nutrient availability (Wicklow-Howard 1994) or the result of self-induced seed dormancy, which has limited germination in the harsh environmental conditions (Halvorson 1989, Allan *et al* 1994, O'Keefe 1996).

The hayseed appeared to repress one of the most common grass species on the site, *S. cryptandrus.* The average cover of *S. cryptandrus* fell slightly in Rep 1 to 4 over the four years as compared to an 18% increase in cover in the control plots. *S. cryptandrus* cover increased in Rep 5, although the control plots increased at a higher rate (90% vs. 84%) (Figure 2). When the hayseeding experiment was initiated, Rep 5 contained twice the amount of bare ground as Reps 1 to 4 (Figure 10). Light availability was likely higher in Rep 5, even with the hayseed cover. Sabo *et al* (1979) reported germination of *S. cryptandrus* increased with light availability.

In contrast, the hayseed cover enhanced *P. spicata* and *H. comata* establishment (Figure 2). *H. comata* only established in the sandy soils of Rep 5 when covered by hayseed mulch and *P. spicata*, which was not found in the research plots before seeding, only established in areas that received the hayseed mulch or VAM inoculant (Figure 2 and Figure 6). All of the seeded grasses are mycorrhizal (Trappe 1981) and VAM is particularly critical for the establishment of warm season grasses (Clapperton and Ryan 2001), which would include *A. longiseta* and *S. cryptandrus.* To date, VAM colonization levels that will improve grass establishment are unknown.

Species establishment was also influenced by soil type. *A. longiseta* had higher establishment in sandier soils (Rep 5), while *H. comata* did best in siltier soils (Rep 1 to 4) (Figure 5). The percent cover of *H. comata* was almost 7 times higher in Rep 1 to 4 than in Rep 5. *A. longiseta* and *H. comata* are both promoted as drought tolerant species and yet the limited establishment of *A. longiseta* and preferential establishment of *H. comata* suggests they were affected by the droughty conditions experienced in the South Okanagan over the past four years. Weaver (1968) did find that *A. longiseta* decreased in extended droughts. It is also possible the high heat requirement for *A. longiseta* germination was not met prior to the June data collection (Evans and Tisdale 1972).

Weed Control

Solarization was not an effective weed control method for the primary weeds on the Osoyoos Desert site. *C. diffusa* and *V. thapsus* germinated readily following the removal of the plastic, indicating the 75^{o}C recorded under the plastic during treatment was not sufficient to kill the seeds. In addition, solarization resembles broadcast herbicide treatment, exposing large expanses of bare soil after treatment. Revegetating solarized areas with only locally collected native species will also require a consistent and long-term weed control program.

Manual and chemical control of *C. diffusa* did reduce the weed component, however across the site, livestock removal was the most effective weed control measure. In one-year weed cover in the plots monitored for the effect of the removal of livestock dropped an average of 74% (71% in Rep 1-4 and 77% in Rep 5) and over five years there was a significant increase in native grass and herb cover. The rapid decline of *C. diffusa* is puzzling since the species is known to have an extensive and long-lived seed bank. Reduced soil disturbance is a factor, because *C. diffusa* did germinate in the tilled plots. However, Clements *et al* (unpublished) also found few viable knapweed seeds on the site, which may be the successful result of *Sphenoptera jugoslavica* (biological control agent – beetle), which occurs throughout the area. By 2002, *A. cristatum* was the dominant non-native species on the site.

CONCLUSION

The South Okanagan has experienced serious drought conditions over the past four years and revegetating with natural perennial grass species that typically have low seed viability and slow establishment rates is challenging. The experiments conducted on the Osoyoos Desert site demonstrated that seeding technique and seed mix composition must be considered and matched to local environmental conditions, although some techniques can aid the establishment of species in environments they would not normally colonize.

ACKNOWLEDGMENTS

We would like to acknowledge the Habitat Conservation Trust Fund, Environment Canada, Human Resources Development Commission, and NSERC for funding the project. Jordan Carbery was invaluable as field supervisor and primary data collector and the project would have been impossible without him. We would also like to acknowledge Shannon Berch, Ministry of Forests Research Branch and Sharmin Gamiet for their assistance with the VAM experiment and members of the Osoyoos Desert Society Science Committee for their assistance in the early stages of the project. Committee members included Dr. P. Krannitz (Environment Canada), Judy Millar (Ministry of Water, Land, and Air Protection, and Lisa Scott (Eco-Matters Consulting).

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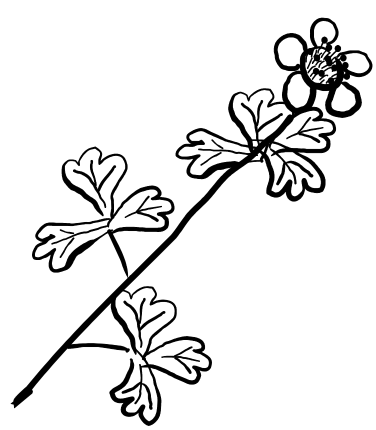

Status and Importance of the Antelope-Brush - Needle-and-Thread Grass Plant Community in the South Okanagan Valley, British Columbia

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ABSTRACT

The low elevation natural habitats of the south Okanagan valley make up one of Canada's four most endangered ecosystems. The Antelope-brush – Needle-and-Thread Grass plant community is a major part of this ecosystem and important for biodiversity conservation in British Columbia and Canada. This report provides information on biodiversity values for this plant community and summarizes historic and current ecosystem mapping and land ownership. The world distribution of this plant community includes the south Okanagan valley, mainly south of Penticton, eastern Washington and Oregon and western Idaho. It is Red listed in British Columbia and Globally Imperiled due to limited world distribution and substantial decreases in area related to urbanization and agricultural development. Much of the remaining area is in early seral condition due to years of heavy grazing by livestock. The Antelope-brush plant community supports 88 provincially listed Species at Risk: 33 invertebrates, 32 vertebrates and 23 plants. Fifty-eight are Red listed and 30 are Blue listed. Seventeen of these taxa are also federally listed by COSEWIC: 1 Extirpated, 5 Endangered, 4 Threatened and 8 Special Concern. Three of these federally listed species are found nowhere else in Canada. In addition, 104 of British Columbia's rare invertebrates (most not currently listed) are confined to south Okanagan Antelope-brush. Sixty-one percent of this plant community has been altered by development, leaving only 3898 hectares. Losses continue at an average of 90 hectares (2%) annually. Eighty-seven percent (3390 hectares) of the extant Antelope-brush – Needle-and-Thread Grass plant community occurs on Indian Reserve (58%) or private land (29%), including The Nature Trust's land. Only 509 hectares (13%) occurs on Federal and Provincial Government lands. Seven hundred and six hectares are protected by the Crown and The Nature Trust of BC. This is 18% of the extant community but only 7% of the historical area. It is clear that stewardship programs with First Nations and private land owners are critical for the conservation and restoration of this plant community.

INTRODUCTION

Six antelope-brush plant communities are now tentatively recognized by the Ministry of Forests for the South Okanagan area (Lloyd, 2002). The *Purshia tridentata-Hesperostipa comata* (Antelope-brush – Needle-and-Thread Grass) plant community is the largest of the six. It is characterized by a shrub layer dominated by *Purshia tridentata* (antelope-brush) with *Chrysothamnus nauseosus* (rabbit-brush) and a herb layer dominated by *Hesperostipa comata* (needle-and-thread grass) with *Opuntia fragilis* (brittle pricklypear cactus), *Sporobolus cryptandrus* (sand dropseed), *Pseudoroegneria spicata* (bluebunch wheatgrass) and *Koeleria macrantha* (junegrass). It occurs in eastern Oregon and Washington, western Idaho and south central British Columbia (Lea and Flynn in prep.). Within Canada, it occurs only in the South Okanagan Valley of British Columbia (Figure 1). It is important for biodiversity conservation, both provincially and nationally. This community is provincially Red Listed (BC Conservation Data Centre, 2002) and Globally Imperiled (NatureServe Explorer 2002). It supports high numbers of federally and provincially listed species at risk (Schluter et al. 1995.) This plant community and the species that depend on it are threatened by urban and agricultural development (Schluter et al. 1995). Lea 2001 estimated less than 50% of historic antelope-brush in the south Okanagan remained in 1995 and losses have continued. Most remaining sites have been disturbed by intensive cattle grazing leaving most of the community in early seral stages with reduced cover and forb diversity, affecting habitat quality for wildlife (Lea 1995, 2001). Due to these threats and high biodiversity values in Antelope-brush and adjacent plant communities Schluter et al. 1995 identified this area of the South Okanagan as one of the four most endangered ecosystems in Canada. This report provides information on the importance of the Antelope-brush - Needle-and-Thread Grass community to species at risk conservation in British Columbia and Canada. It also quantifies habitat loss over time and ownership, including conservation lands, of remaining parcels.

METHODS

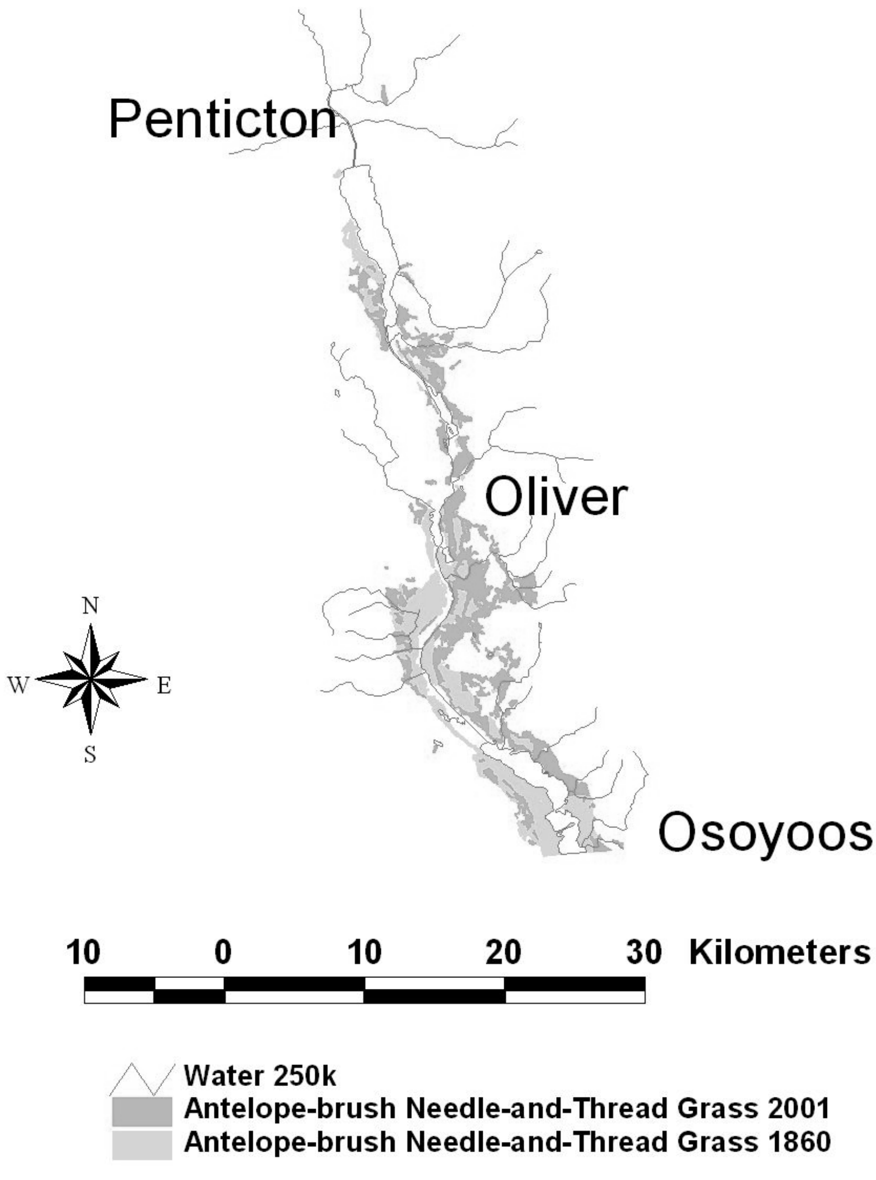
A list of species at risk was modified from Dyer 2001 and Scudder 1996 through consultation with species experts and updated with recent COSEWIC changes. Arcview 3.0 was used to quantify the area of antelope-brush community, based on digital ecosystem maps, for 1860 and 1938 (Lea in prep.) and 1995 (Lea and Maxwell 1995). Lea and Maxwell 1995 was updated to 2001 for this project using a combination of digital ortho photos (BC Ministry of Forests, 1995) and field checking selected polygons. Arcview 3.0 and Microsoft Excel 97 were used to identify and quantify land ownership using the above ecosystem maps and digital, land ownership maps (Steeves 2001).

RESULTS

Species at Risk

Eighty-eight species at risk occur in the south Okanagan Antelope-brush – Needle-and-Thread Grass plant community: 33 invertebrates, 32 vertebrates and 23 plants. Fifty-eight taxa are provincially Red Listed and 30 are Blue Listed. Eighteen of these taxa are also federally listed: 1 Extirpated, 5 Endangered, 4 Threatened and 8 Special Concern. Most invertebrates are not considered for listing by the Conservation Data Centre so have no formal status. One hundred and four species of British Columbia's rare invertebrates are confined to south Okanagan Antelope-brush. Seventy-two of these rare invertebrates are found, within Canada, only in British Columbia's south Okanagan Antelope-brush (G. G. E. Scudder 1996).

Figure 1:
South Okanagan
Antelope-brush - Needle-and-Thread Grass
Historic and Current Distribution



Habitat Loss

The area (hectares) of Antelope-brush – Needle-and-Thread Grass plant community was summarized for 4 years (Figure 2). There were 10,053 hectares of Antelope-brush community in 1860, 7425 hectares (74%) in 1938 and 4438 hectares (44%) in 1995. By 2001, 3386 hectares (39%) remained.

Figure 2: Area of Antelope-brush Plant Community by Year

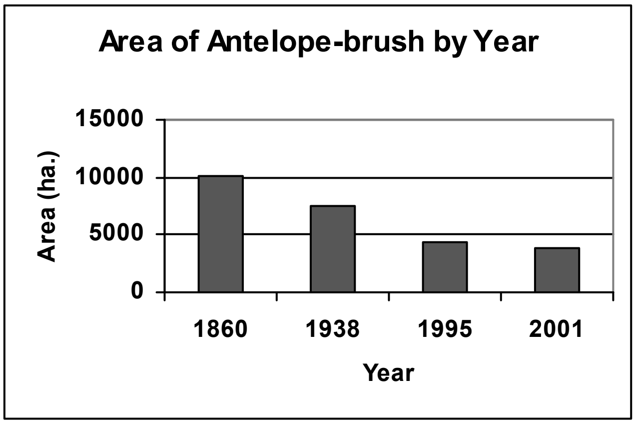
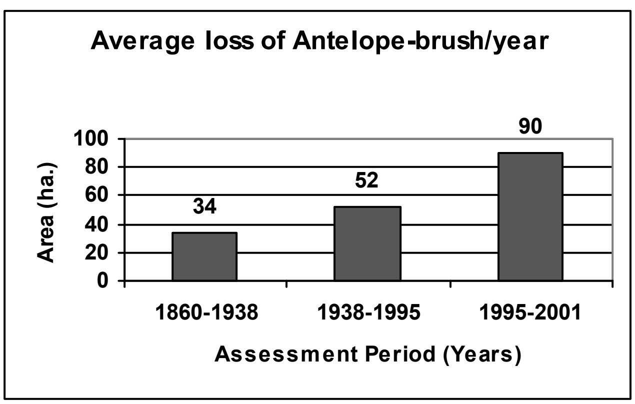


Figure 3 Annual Average loss of Antelope-brush



Rates of Habitat Loss

The average rate of habitat loss per year was calculated for three time periods (Figure 3). Rates of loss averaged 34 hectares per year from 1860 to 1938, 52 hectares per year from 1938 to 1995 and 90 hectares per year from 1995 to 2001. Rates of habitat loss have approximately doubled over the last few years and are now over 2% per year.

Ownership of Antelope-brush

The area of extant Antelope-brush – Needle-and-Thread Grass plant community in 2001 was summarized for four land ownership categories. Most of the remaining community occurs on Indian Reserve (2314 hectares, 59%) with 1076 hectares (28%) on Private Land, 382 hectares (10%) on Provincial Crown land and 127 hectares (3%) on Federal Crown land.

Habitat Conservation

Seven hundred and six hectares are protected by The Nature Trust of BC (343 hectares), Provincial Parks, Ecological Reserves and the South Okanagan Wildlife Management Area (236 hectares) and the Vaseux Bighorn National Wildlife Area (127 ha). Conservation lands represent 18% of the extant community but only 7% of the historical area.

DISCUSSION

This report confirms and quantifies the importance of South Okanagan Antelope-brush communities to biodiversity conservation in British Columbia and Canada. The Antelope-brush – Needle-and-Thread Grass plant community is provincially Red listed and Globally Imperiled. It provides habitat for 88 provincially listed species at risk; 17 are also federally listed. This is one of the highest densities of listed species for any habitat in British Columbia and far higher than most entire forest districts (Conservation Data Centre tracking lists 2002). We also confirm the substantial and increasing threat to South

Okanagan Antelope-brush plant communities and the species dependent on them. Sixty-one percent of the Antelope-brush – Needle-and-Thread Grass plant community was lost to urban and agricultural development by 2001 and declines continue at over 2% per year. Only 7% of the historic area has been protected and much of the remaining habitat is fragmented and in early seral condition, requiring restoration. The majority of the remaining habitat occurs on private land and Indian Reserve, making stewardship programs critically important for conservation planners. Although the future appears bleak for Antelope-brush, several positive steps have been taken recently due to increased awareness and diverse partnerships. Most of the habitat protection has taken place in the last decade, including substantial acquisitions by The Nature Trust of BC. The federal government manages a large tract of antelope-brush in the Vaseux Bighorn National Wildlife Area. The provincial government has conserved over half of the Antelope-brush it controls and is considering additional patches. In addition, it proposed this plant community for inclusion as Identified Wildlife under the Forest and Range Practices Act in 2003. This will encourage restoration of a more natural seral stage distribution on provincial Crown land. The SOS Stewardship Program continues landowners contact and the Osoyoos Desert Centre continues education, both aimed at increased awareness, conservation and restoration. Finally, the South Okanagan-Similkameen Conservation Program, formed in 2000, continues to promote an ecosystem approach to conservation through its many partners.

ACKNOWLEDGMENTS

We appreciate the assistance of the following people, who contributed to portions of this report: Robb Bennett, Christine Bishop, Rob Cannings, Syd Cannings, Adolf Ceska, Marta Donovan, George Douglas, Leigh Dyer, Aaron Enquist, Tom Ethier, Kelsey Furk, Crispin Guppy, Ron Hall, Caroline Jackson, Jan Kirkby, Walt Klenner, Robert Lincoln, Leah Ramsay, Dennis St. John, Geoff Scudder, Gail Smith, Chris Steeves, John Surgenor.

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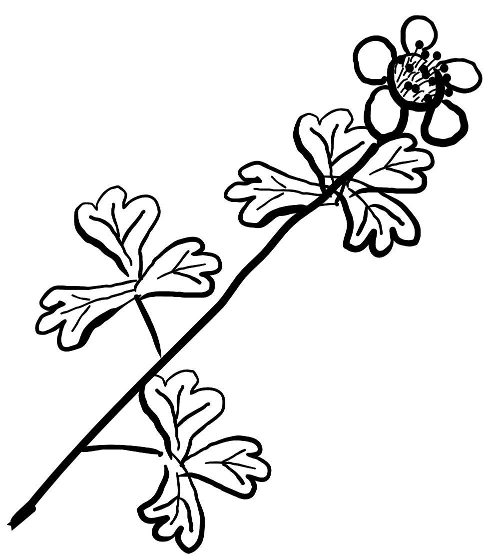

ANTELOPE BRUSH – BLUEBUNCH WHEATGRASS in the East Kootenay-Rocky Mountain Trench region of British Columbia

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ABSTRACT

Antelope brush – Bluebunch wheatgrass is a red-listed plant community in the East Kootenay-Rocky Mountain Trench region of British Columbia. It has declined with the landscape changes associated with the settlement era and remains in a threatened position today. This community requires natural disturbances in its ecology, and their loss is indicated among the causative factors in this decline. Legal protection mechanisms include parks and protected areas, range reference area exclosures, range use plan objectives under the Forest Practices Code, and the establishment of Wildlife Habitat Areas under the provincial Identified Wildlife Management Strategy. Within the context of this strategy, this account discusses the community, outlines provisions for the landscape level and for General Wildlife Measures, and recommends management actions under Additional Management. Many of the sites for this community occur on private land, so stewardship will also be an important component of a more comprehensive recovery.

PLANT COMMUNITY INFORMATION

INTRODUCTION

This plant community, known as the red-listed Purshia tridentata- Pseudoroegneria spicata, occurs as a shrub-steppe which features these two species plus Idaho fescue and balsamroot. It occurs in the Cranbrook vicinity of the southern East Kootenay-Rocky Mountain Trench. Sometimes the community is set in an open savannah physiognomic type. The antelope brush – bluebunch wheatgrass community is adapted to/ conditioned by a natural disturbance regime so preservation efforts will require a management restoration component. Establishment of Wildlife Habitat Areas under the Identified Wildlife Management Strategy may be one mechanism which allows these active components to occur in combination with setting aside the community from activities which may threaten it.

DESCRIPTION

Purshia tridentata- Pseudoroegneria spicata is a dry shrub-steppe grassland community (Erickson 2002, Meidinger 2002), which is rare in late seral stages with a natural fire cycle (Conservation Data Centre n.d.). These remnant stands of antelope brush (Purshia tridentata) and bluebunch wheatgrass (Pseudoroegneria spicata), often found on crests and upper slopes. Sites are also sometimes in open, savannah settings of Ponderosa pine (Pinus ponderosa) and Douglas-fir. Saskatoon (Amelanchier alnifolia), Idaho fescue (Festuca idahoensis), and arrowleaf balsamroot (Balsamorhiza saggitata) are also key species. These sites have a diverse herbaceous flora.

Antelope brush typically has a canopy cover of 15 to 35%, bluebunch wheatgrass 10 to 45%, saskatoon 2-10%, Idaho fescue 1 to 30%, and arrowleaf balsamroot 2 to 25%.

Rough fescue (Festuca campestris) or kinnickinnick (Arctostaphylos uva-ursi) are dominant on some sites, and others may have shared abundance (e.g. 5 to 10% cover) or patches of Columbia needlegrass (Achnatherum nelsonii), Sandberg's bluegrass (Poa secunda), pasture sage (Artemisia frigida), shining arnica (Arnica fulgens), or invading Kentucky bluegrass (Poa pratensis). The presence of hairy golden-aster (Heterotheca villosa) and stiff needlegrass (Achnatherum occidentalis ssp. pubescens) were diagnostic in the Nelson Forest Region site series flowchart (Braumandl and Curran 1992), and Holboell's rockcress (Arabis holboellii) was important in the original data (Meidinger 2002).

Other herbs typically present with a low cover (<5%) include Junegrass (Koeleria macrantha), slender hawksbeard (Crepis atribarba), timber milkvetch (Astragalus miser), yarrow (Achillea millefolium), death camas (Zigadenus venenosus), old man's whiskers (Geum triflorum), graceful cinquefoil (Potentilla gracilis), fern-leaved desert parsley (Lomatium triternatum), brown-eyed Susan (Gaillardia aristata), tufted phlox (Phlox caespitosa), mariposa lily (Calochortus macrocarpus), dwarf goldenrod (Solidago spathulata); and both blue-eyed Mary (Collinsia parviflora) and prairie crocus (Anemone patens) in spring.

Generally there is no moss/ lichen layer. Occasionally, sites have a high cover of lichens (Cladonia spp. up to 20%) or mosses (Tortula ruralis up to 10%).

DISTRIBUTION

British Columbia

In British Columbia, this plant community is restricted to valley bottoms and lower slopes (700 to 1200 m elevation) in the southern Rocky Mountain Trench. It occurs south of Canal Flats, is bounded on the west by St. Mary River and on the east by Baynes Lake, and extends to the border at Tobacco Plains. See Figure 1.

South from B.C. it extends east of the Cascades to the Klamath, North Coast, and Sierra ranges in California, across Washington, Oregon and Idaho into Montana, Wyoming, Nevada and New Mexico (Zlatnik et al. 1991a, NatureServe Explorer 2001).

These areas have been mapped as the PPdh2/00 and IDFdm2/02 biogeoclimatic/ ecosystem units (Conservation Data Centre n.d., Braumandl and Curran 1992)

NATURAL DISTURBANCE REGIME

Periodic fire, grazing and browsing, and insect outbreaks are among the historic natural disturbances for this community (University of Wyoming, n.d., Johnson, Don pers. comm. Research Scientist, Ag. Can., Lethbridge, Forest Practices Code 1995, Youtie et al. 1988, Rondeau 2001) Collectively, these disturbances would keep stands open and provide renewal or replacement opportunities where growth or vigour was stagnated due to plant density, bunchgrass litter and pine needle accumulations, or competition. Renewal would be provided by a frequent fire regime, such as the 5 to 25 year frequency required to maintain the *Pinus ponderosa/ Purshia tridentata* habitat type- *Agropyron spicatum* phase in a treeless state (Arno 1979, Fischer and Clayton, 1983).

This community is part of broader fire-maintained ecosystems, which have been

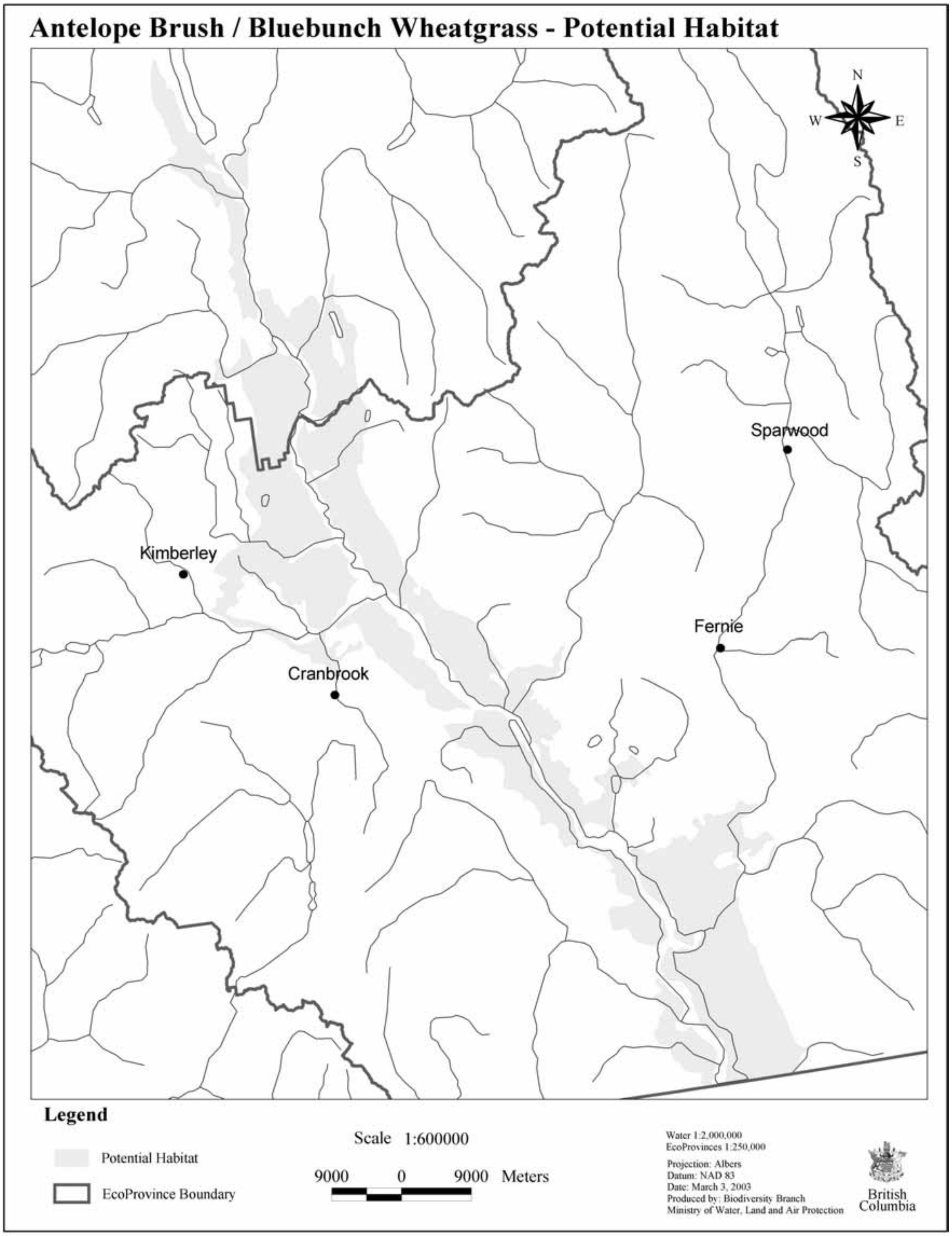

Figure 1: Potential range of Antelope Brush – Bluebunch Wheatgrass habitat

subject to fire suppression and consequent forest encroachment and ingrowth (Gayton 1996, Hardy and Arno 1996). In addition, the key species of the community still have susceptibilities to higher burn intensities in different seasons (Thomson 1988, Zlatnik 1999a, b).

The species of this community are generally adapted to and resilient to disturbance. An exception is the susceptibility of bluebunch wheatgrass and Idaho fescue to spring defoliation by herbivores (McLean and Marchand 1968). Conditioning of the vegetation by native ungulates is part of the natural ecosystem processes of this community. The subzone variant area supports large populations of ungulates and is important as winter range. Current typical composition reflects the influence of grazing/ browsing pressure, with more dominance by antelope brush and balsamroot, and less by Idaho fescue and bluebunch wheatgrass (Erickson pers. obs., University of Saskatchewan n.d., Youtie et al. 1988). In addition, these latter two bunchgrasses most likely have exchanged dominance on late seral sites. This community has been replaced by grazing pressure on early seral sites, with conversion to pussytoe species, needlegrasses and weedy forbs, and invading species such as cheatgrass (McLean and Marchand 1968, Erickson pers. obs.). Sometimes, however, the tough, arching stems of bitterbrush provide mini-refugia which protect the late-seral species (Gayton, D. pers. comm.).

For the most part, the form of antelope brush differs when compared with shrubs in the south Okanagan valley (Erickson pers. obs.). The smaller and less-upright form and presumably younger top-growth may suggest historic disturbances, more severe winter temperatures, effects or possibly a genetic difference in the Trench populations (pers. obs., Gayton, D. pers. comm.). A negative feedback mechanism should be noted, in which the old growth bitterbrush plants are killed in the event of a fire, due to the level of fuel accumulation in their structure and in the protective zone they provide (Gayton pers. comm.).

Many sites currently have considerable exposure of bare mineral soil. The extent to which this represents the natural condition (i.e. due to natural erosion or hoof action by native ungulates) is unknown.

FRAGILITY

Moderately fragile due to the dry climate and the effects of coarse soils on plant growth, ameliorated by the presence of underlying calcareous bedrock and the site stability influences of the coarse soils. Classic studies by McLean and Marchand (1968) in related habitats indicate the long period of recovery required from an early seral stage. Many sites may be stalled in a state with Kentucky bluegrass, needlegrass, or cheatgrass (*Bromus tectorum*) dominance, and may require management treatments for recovery (Westoby et al. 1989).

SITE CONDITIONS

Climatically, these are relatively hot dry subzones for plant growth. This community occurs on coarse textured, glacio-fluvial terraces or colluvial materials over calcareous bedrock. These latter occurrences are considered unique in their combination of moisture and nutrient conditions (Conservation Data Centre n.d.). Three common slope position occurrences are level, valley-bottom sites; warm-aspect, upper slopes (10-40%); and crests. These sites have been assigned to xeric moisture and medium to rich nutrient classes (Braumandl and Curran 1992). Soils vary from sandy and poor on the terraces to loamy and very rich on the slopes. They are classified variously, but melanization is a major soil process. Humus forms are usually Rhizomulls, but may be less well-developed (Moders or Mors) on poor sites.

Important features

- Late seral condition
- Large ponderosa pine and wildlife trees
- Important and preferred forage species for ungulates

CONSERVATION AND MANAGEMENT

STATUS

Purshia tridentata- Pseudoroegneria spicata is on the provincial *Red List* in British Columbia. It is ranked S2 by the Conservation Data Centre (2002). Its global status is not known but similar communities are ranked G3 (NatureServe n.d.).

TRENDS

Identified as declining, with remaining occurrences estimated at between 21 and 100 (Meidinger et al. 2002). The plant community has been replaced with weedy, seral species on many sites, and some sites have been lost to development. There is not a complete inventory of occurrences of this plant community, but at least 17 plots have been described (Erickson 2002, Meidinger 2002). Terrestrial Ecosystem Mapping (TEM) summaries indicate 710 ha mapped as this community in the Premier-Diorite project area, but the state of the understory is unknown.

THREATS

Threats include livestock and wildlife grazing/ browsing impact, fire suppression leading to forest encroachment; urban development, and noxious weed invasion. Habitat has been lost to urbanization, impoundments, golf course development and intensive agriculture. Fire suppression, soil exposure, reductions in plant cover, and the lack of prescribed burning led to forest encroachment. Outdoor recreation (e.g. trail bikes), livestock grazing and wildlife grazing/ browsing can impact soil exposure, stability, plant vigour and composition. Noxious weeds can invade the community with soil disturbance.

LEGAL PROTECTION AND HABITAT CONSERVATION

There is thought to be no legal protection for plant communities except for those within protected areas and parks.

This community occurs in several small protected areas, including Kikomun Creek Park, Premier Ridge and Sheep Mountain Wildlife Management Areas. It may also occur within Premier Lake Provincial Park and recently acquired conservation properties.

Several range reference areas (RRAs) have this community, including Skookumchuk, Old Premier Ridge, Gold Creek, Bagley's Pasture and Bull River. Others, such as Premier Ridge, Pickering Hills and Standard Hill, are currently in earlier seral stages, but have the site potential to develop this community over time. These long-term monitoring exclosures are considered too small in size (2 or 3 hectares) for plant community conservation, with the exception of Skookumchuk, which has 104 ha under protection (Gayton, D. pers. comm.).

The Forest Practices Code provided for the development and use of Range Use Plans in managing livestock operations. Within the plan, the district manager can specify the desired plant community to be maintained or achieved.

IDENTIFIED WILDLIFE PROVISIONS

Landscape level recommendations

Maintain and restore grassland and open savannah.

Control forest ingrowth and encroachment.

Restore and maintain plant composition of this community.

Establish range reference area exclosures at occurrences of this community and monitor changes in understory composition.

Wildlife habitat area

Objective

Restore and maintain known occurrences. Provide for a diversity of conditions which allow for the processes of litter accumulation and renewal. Where possible, contribute to the biodiversity of adjacent managed areas by selecting contiguous sites.

Feature

Establish WHAs at known occurrences of this community. Establish recovery WHAs in areas where high quality occurrences cannot be found and the key species of the community are present, in small patches. Minimize composition of introduced, especially weedy, species in area selection.

Size

The size of the WHA should be based on the extent of the plant community occurrence. WHAs will generally be between 5 and 20 ha when the community occurs in relatively pure composition. WHAs may be larger, up to 50 ha, when the community has a patchy distribution or when recovery is the main objective. WHAs may be up to 100 ha when plant community occurs in a complex with other at-risk plant communities.

Design

The WHA should encompass the entire extent of the occurrence plus a 100 m surrounding the perimeter of the community. When occurrences are narrow, such as along ridge tops, include 200 m surrounding perimeter.

General wildlife measures

Objectives

1. Maintain and restore antelope brush, bluebunch wheatgrass, Idaho fescue, rough fescue and balsamroot cover; cycles of litter and light intensity natural fire renewal. Increase cover and diversity of other native species (e.g. forbs, rough fescue) and maintain open savannah structure (e.g. <15% cover) of older (e.g. >150 year old) Ponderosa pine and Douglas-fir trees where they are present.
2. Maintain shrub-steppe/ grassland structure and processes.
3. Condition and restore sites from forest encroachment and ingrowth with prescribed fire treatments.

Measures

Access

- Avoid or minimize access. Proponents to mitigate to prevent damage.

Range

- Set the following species as the Desired Plant Community: shrub-steppe between 15 and 30%

canopy cover of antelope brush; herb layer dominated by >5%, preferably >15% cover each, of at least two of the following: bluebunch wheatgrass, Idaho fescue, rough fescue or arrow-leaved balsamroot. A composition with 19% each of co-dominating saskatoon, pinegrass or other herb layer species is acceptable

- Manage to maintain and increase the species named above as the Desired Plant Community.
- Maintain a diversity of specific community types (e.g. fescue types).
- Exclude livestock if temporarily required for restoration treatments under Additional Management, or to address problems identified in monitoring results.
- Plan livestock grazing to allow recovery and avoid impacts to soil surfaces and species composition.
- Avoid grazing impacts, such as shrub browsing, increasing the composition of introduced species, and bare soil exposure/compaction.
- Prevent, report and control invasions of noxious weeds.

Recreation

- Do not develop recreation access or structures.

Silviculture

- Do not harvest or salvage except to support restoration measures with silviculture treatments.
- Retain widely-spaced, large, older trees and snags.

ADDITIONAL MANAGEMENT CONSIDERATIONS

Restoration

Introduce prescribed fire. Plan to accommodate natural fire. Condition stands for prescribed fire treatments and manually control conifer encroachment where this cannot be achieved with fire prescriptions.

Reduce or eliminate invasive species and re-establish native species. Use only locally collected seed from native species (not cultivars) in any seeding which is required.

Actively manage to restore and maintain this community, emulating effects of natural fire regime, with restoration silviculture treatments and light intensity, prescribed burns in fall (Thomson 1988, Zlatnik 1999a, b). Where necessary, combine with preparatory silviculture treatments, such limbing to prevent surface fires from crowning. This prescription is compromise with species susceptibilities and the difficulties of a spring burn window before the onset of bunchgrass growth. Burns should be able to be carried out under a regular burn plan, plus species-level monitoring, without the need for a specific site management plan. Light to moderate grazing/ browsing and periodic renewal are necessary as part of the disturbance regime for this community, but higher levels can cause the loss of the community through competition-mediated shifts in composition and species invasions (McLean and Marchand 1968, Ross 1997).

Monitor wildlife impact on plant composition, vigour and soil; manage in order to avoid impacts if necessary.

Avoid linear or extensive soil disturbances. Access concerns are centred around any concentrating effect they may have on livestock or wildlife distribution, and on access corridors serving for the spread of noxious weeds and other invasive plants (e.g. cheatgrass).

Private land stewardship will be an important component of the conservation of this community as many sites occur on private land. Consider private land with stewardship agreements and protected Crown land when planning habitat contiguity for wildlife habitat areas.

INFORMATION NEEDS

1. Monitor recovery trends in relation to site factors and restorations treatments, and for the relationship between specific community types currently encompassed within this community.
2. Inventory to clarify the extent of this community in protected areas and any set-asides.
3. Classification review which considers related communities identified in other work (Erickson 2002b).

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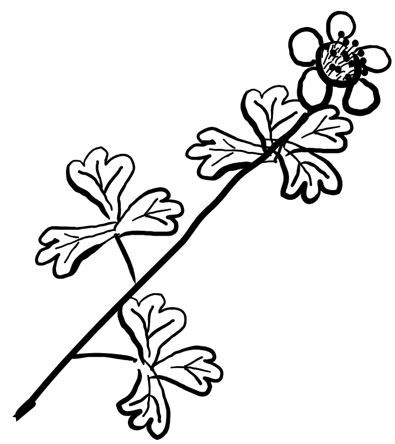

Art and Science: the Zen of Species at Risk Recovery

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ABSTRACT

The antelope-brush area of the south Okanagan is a hotbed of nationally unique species. It is well known that failure to act could likely result in many human-assisted extinction events. This paper provides an overview of recovery planning and the broad context within which it is done. Some lessons learned from the past are shared. Thoughts on planning are presented. Suggestions are made for future improvement.

INTRODUCTION

The purposes of this brief paper are to paint a picture of the broad context in which recovery planning for species at risk is operating, provide a bit of history, share some thoughts and lessons learned, and to take a peek into the future. Zen is about meditation (thinking) and enlightenment. My goals are to provide a bit of enlightenment and provide some food for thought.

The thoughts I share below are those of a practitioner. They have evolved as a result of work as a biologist over the past 30 years, many years of experience with different kinds of planning, 15 years of experience as a public service manager and many years of experience as a statutory decision maker.

Recovery is defined as "the act of bringing a species back from the risk of extinction to a self-sustainable population level, able to withstand stochastic events and other environmental variables (National Recovery Working Group 2003).

Recovery planning is only one step along the way of preventing species at risk from becoming extinct and helping them to survive into the foreseeable future. The larger and general process within which recovery planning is done, is as follows:

- Step One – decide which species and ecosystems are of conservation concern,
- Step Two – decide what should be done to recover species and ecosystems in greatest peril,
- Step Three – act on the decisions made in step two,
- Step Four – monitor and adapt.

HISTORICAL SYNOPSIS

In Canada, step one (decide what is at risk) has been partly underway at the national level for some 25 years through the activities of the Committee on the Status of Endangered Wildlife in Canada (COSEWIC). COSEWIC has conducted close to 600 status assessments and there are presently more than 400 species on the national species at risk list. Similar and complementary work at the provincial level has been underway in British Columbia for many years through the British Columbia Conservation Data Centre, presently housed in the provincial Ministry of Sustainable Resource Management. Recent passage of the federal Species at Risk Act provides legal status for the critical and independent technical work of COSEWIC and provides for its continuance as an advisor to government. Elected government will properly make the ultimate decision on which species will receive legal designation under the Species at Risk Act (SARA). The national-provincial Accord for the Protection of Species at Risk has also resulted in fairly comprehensive and reasonably accurate preliminary national status assessments for a range of plant and animal groups. So there has been a lot of work on deciding which species are at risk but I am not aware of a comparable national framework for deciding which ecosystems are at risk. Fortunately most provincial governments and some non-profit organizations have for many years been working on the topic of endangered ecosystems, independent of the formal national framework for species. Our understanding of which organisms and ecosystems are at risk has increased dramatically in recent years and will continue to increase.

Step two (recovery decisions) has also been underway for some years but recently has been more aggressively pursued. Recovery planning has been around for at least 15 years (probably longer) and has become an accepted conservation tool in countries around the globe; including Canada, United States of America and Australia. Of course actual recovery actions for some species predate the era of formalized recovery planning. Conservation practitioners have long had an international network that has grown far more efficient with the advent of digital communications technology. We continue to learn from each other and continue to explore better ways to confront the reality of biodiversity loss from human activity.

The USA has been very active in producing recovery plans, while in Canada the number of completed plans is substantially less. Because this tool has not been extensively used in BC or in Canada, we have relatively few people who have hands on experience in crafting the tool. Passage of the federal Species at Risk Act has provided legal impetus for more timely production of decisions for recovery of nationally Endangered, Threatened and Extirpated species. The good sense of building and documenting recovery decisions was recognized provincially prior to the passage of SARA and has resulted in some recent recovery documents dealing with, for example, Mountain Caribou (Mountain Caribou Technical Advisory Committee 2002 and Garry Oak ecosystem (Garry Oak Ecosystems Recovery Team 2002)

Step three (act on the recovery decisions) is also not new, although in the past most actions were taken in an *ad hoc* or opportunistic manner outside the more structured recovery process that is before us today. Step three consists of the many things that need to be done to implement a plan.

Step four (monitor and adapt) is a key step that is usually identified in plans but is often not undertaken as effectively as it perhaps should be. This comment should not be construed as criticism; indeed I am well aware, from extensive personal experience, of the multitude of real challenges and constraints that conservation practitioners face in an institutional environment of annual budgets and frequent policy changes by elected officials.

A RECENT EVENT

The single most important recent event that bears on recovery efforts and ecosystem restoration is the passage of SARA in 2002; after a multi-year and often contentious legislative process that is unprecedented in Canadian history. I heartily commend the numerous people who had the vision and tenacity to bring this vital piece of conservation legislation to fruition. The purposes of SARA, as stated in the Act, are:

“to prevent Canadian indigenous species, subspecies and distinct populations of wildlife from becoming extirpated or extinct, to provide for the recovery of endangered or threatened species, to encourage the management of other species to prevent them from becoming at risk".

Prior to SARA, recovery planning and restoration activity was undertaken as a matter of administrative policy and management practice. It was not unusual for funding interruptions to cause problems for completion of needed work. Now there is legal footing that will strongly encourage the appropriation of funds to do that work which is legally required. This new legal footing also specifies some things that must be done after the technical and social decisions are made as to which species are of highest conservation concern.

A particularly salubrious component of SARA is the requirement to place recovery strategies and recovery action plans on a public registry. Having these statements of intent open for full public scrutiny will ensure that they do not collect dust on a shelf. Many concerned and interested people will be watching to see that strategies and actions are manifested on the ground. Pointed questions will be asked of government officials and politicians when action is not forthcoming.

Other key provisions of SARA are:

- Prohibitions that protect listed threatened and endangered species and their critical habitat,
- Provisions for compensation to ensure fairness following the imposition of the critical habitat prohibitions,
- Consistency with Aboriginal and treaty rights and respect for the authority of provincial governments.

Prior to SARA the standard best practices for recovery planning called for the creation of a single document that included descriptive summary information (background), goals and objectives, strategies, and a preliminary action plan of specific items to implement the goals, objectives and strategies. While there is merit in continuing to capture this material in one document; SARA does not call for the preparation of traditional recovery plans. Instead it requires the preparation of recovery strategies and action plans.

The act does not prohibit the creation of recovery plans so in some cases we may continue to see the preparation of recovery plans built as two part documents that meet the legal requirements for both a recovery strategy and one or more action plans. It will be interesting to see how various organizations with lead roles in recovery activities respond to this new legislation. One thing we can be certain of is that we will collectively continue to learn how to produce better direction documents to guide recovery efforts. We must avoid the temptation to create a 'one size fits all' solution. There certainly are topics that need to be addressed for most species and there is value in having documents that are generally similar in structure and content to improve understanding and ease of use. But one size never fits all and we must maintain a reasonable degree of flexibility rather than getting dragged down by bureaucratic restrictions.

SARA articulates a number of legal requirements for a recovery strategy. The act calls for the following content items:

- A description of the species and its needs,
- Identification of threats to the species and its habitat; plus the broad strategy to be taken to address those threats,
- A species' critical habitat and examples of activities that are likely to result in its destruction,
- A schedule of studies to identify critical habitat, where available information is inadequate,
- A statement of the population and distribution objectives to assist

recovery; plus a general description of the research and management activities need to meet those objectives,

- Whether or not additional information abut the species is needed,
- When one or more action plans will be completed, and
- Anything else prescribed by regulation.

For action plans, SARA prescribes the following content requirements:

- Critical habitat and examples of activities that are likely to result in its destruction,
- Proposed measures to protect the critical habitat,
- Which portions of the critical habitat has not been protected,
- Measures to be taken to implement the recovery strategy and an indication of when these measures are to take place,
- Methods to monitor recovery and long term viability,
- Evaluation of socio-economic costs of the action plan and the benefits from its implementation, and
- Anything else prescribed by regulation.

The basic components of recovery plans (also known as strategies and action plans) have been around for some time and what started as fairly crude instruments are now becoming more sophisticated in their structure and content. Still, they will never be any better than the constraints of available information, time and money. A number of long and detailed documents have been created for high profile species that are relatively well known with respect to information. This may cause some unrealistic expectations about future documents that will be dealing with the great majority of species at risk. These lesser known organisms all too often have little available information except that they are known to have very restricted distributions and we have a general sense of the habitats they live in. Vascular plants living in compact populations on a rare substrate can at least be counted with relative ease, but good luck in coming up with easy or cheap population information for small insects and their naturally fluctuating populations. So what this means is that plans for the less known species will be shorter but that is a good thing in my view.

SOME LESSONS LEARNED

Collectively we have learned and demonstrated that we can:

- Produce verbose documents that we variously call plans, strategies or action plans. Sometimes the content is not congruent with the title (an example of this is the 'Strategy' for recovery of Mountain Caribou in BC, which has all the appearances and content of a traditional recovery plan).
- Produce documents that call for the expenditure of funds that may be far removed from being within the bounds of reality (an example of this is a USA recovery plan for a butterfly subspecies with a price tag of approximately 15 million dollars),
- Produce recovery plans within time frames that vary from a few weeks to 15 years,
- Identify which species and ecosystems are at risk,
- Take actions to assist in species at risk recovery without having strategy documents and action plans approved at the national level,

An especially important and expensive lesson was learned by an experienced conservation and restoration practitioner in the USA (Zentner 2001). It is worth briefly describing here so practitioners in British Columbia do not walk into the same pit of pain. The scenario is one that all of us could easily find ourselves in. The practitioner relied on his extensive experience and the best scientific information to conclude that a species of frog, protected under federal and state legislation, simply could not be in a piece of 'trash' habitat that had been very heavily impacted by human activity. Not only did the frog turn out to be present there, but the practitioner then committed another error. He graciously helped the frogs by moving them out of harms way (the trashed habitat was destined for development) into adjacent preserved and restored habitats. The practitioner and his client ended up pleading guilty to taking an endangered species and paid fines of $75,000.

The first important lesson to be learned from this example is to keep an open mind about what habitats species at risk will actually use. Do not discount the potential importance of degraded or otherwise 'trash' habitats without thorough investigation. The second important lesson is to be aware of all legal prohibitions related to species at risk. Taking and 'possessing' a species at risk, even for a short time and common sense reason, can be a contravention of pertinent legislation. Be aware that SARA has prohibitions against the capture or take of a listed wildlife species and that the act does allow for this prohibition to apply on non-federal lands.

A local example of 'trash' habitats that are important for species at risk are the habitats used by the Dione Copper (*Lycaena dione*), a red-listed butterfly in British Columbia and which is presently before COSEWIC for a status determination. As recently as June 2002, published information said that this butterfly was in dire straits and dependent on one wetland in a Cranbrook urban park. Subsequently two people secured a bit of funding to venture forth and see if this was in fact the case (Kondla and Nicholson 2002a, b).

Some preparatory literature review and open-minded fieldwork revealed that the butterfly is not at all a wetland species in British Columbia. It appears to be surviving nicely at about 15 sites as diverse as the weedy parts of an urban park, backyards, cattle pastures, roadside ditches, gravel pits etc. ('trash' habitats). Of course I still think that the butterfly is a legitimate species at risk by virtue of it now being known from less than .03% of the provincial land base. It is also worth noting that this butterfly depends on the presence of weedy dock (*Rumex*) species. The message here is that people engaged in ecosystem restoration should walk into projects with their eyes and minds wide open about both the presence of species at risk and what they might inadvertently do to said species by trying to repair a 'trashed' habitat.

SOME PLANNING THOUGHTS

Recovery planning has been and will be conducted within the constraints of institutional resources, competing workload priorities and the particular process used to create the planning documents. There will never be enough money, time and people to move recovery planning along as some would like. This reality is further compounded by the need to have many of the same people who are involved in the planning to also be involved in the implementation. Hard decisions about what gets done first and what gets done later will continue to be made. A traditional and effective process model for creating plans is to strike a team of people with a facilitator/scribe and have them develop the best plan they can within the usual time, workload and money constraints. The downside to this approach is that it can be very time consuming as the participants learn to work with each other, learn to make the planning decisions and have the sometimes lengthy discussions and negotiations needed to arrive at a reasonable level of consensus on key decisions. The upside to this approach is that the plan content benefits from the enhanced scope and depth of knowledge that a team approach yields. Another distinct advantage is a greater level of understanding of the final product and a greater sense of plan ownership by the participants. A more efficient approach is to have someone create the plan with minimal involvement, usually only review and comment involvement, by those who would normally be members of a planning team. The advantage to this approach is that the planning documents can be created in very short timelines. The disadvantage is that there is less ownership of the plan that is produced.

Planning is the art and science of making decisions about the future. Decision-making requires information and consideration of options as part of identifying the preferred decision. Planning and the plans that result from the process can encompass any geographic scope, timeline or content suite. Plans can be comprehensive without including a stifling amount of detail. When charting initial strategic direction, a lot of detail in fact hinders and detracts from the planning process. No plan will ever answer all questions or deal with all issues or solve all problems or specify the huge number of subsequent actions needed to implement the plan. Circumstances change, as do plans and their initial implementation action items. Actions and schedules need to be periodically reviewed and modified as necessary. In some cases even the strategic direction will need to be reconsidered and modified.

Plans cannot be static documents because decision-making is a continuous process. The broader biological, physical, social and institutional environment is also in continuous flux. Change is the only constant. Consequently the plans must be fluid in response. Plans should be realistic and practical. Plans are not detailed ‘wish lists’ of everything under the sun that could conceivably help implement the plan through an infinite future. Realistic plans get implemented, while long wish lists stay parked on the shelf. Plans are not research papers or reports. They serve a fundamentally different purpose. Plans point to the future through the best decisions that can be made today.

No plans are created nor exist in isolation from other plans and every plan is part of a decision-making hierarchy. The planning and decision-making hierarchy is often described through words like normative, strategic, tactical, operational, action. These concepts apply regardless of geographic scale or plan content with respect to single species, multi-species, landscape or ecosystem plans. Regardless of where one starts in the overall hierarchy, the normal practice is to begin with more general decisions and then work into more detailed decisions. Hence practitioners start with one or more goals. Goals are general statements of intent. Following on the goals we have objectives, which are more detailed but constructed in relation to the goals. In an ideal world we would state some objectives that are explicit in terms of geography or quantitative in terms of numbers. In the real world we frequently find ourselves in positions where the information to develop such objectives is lacking and we then must resort to more general or qualitative objectives. In such cases, subsequent action planning usually identifies information gaps that need to be bridged, so that the plan can be revised in light of the new information and improved understanding.

My experiences related to planning are probably similar to those of many other people. There are no magic lines that can be drawn between what is an objective vs. a strategy vs. an action item. No two people will have the same thoughts on where to draw the conceptual lines. This will continue to create challenges to and provide a source of vigorous debate to those people who are tasked with creating the plans. We can only seek guidance from the context of the planning, legal requirements, planning conventions and a liberal dose of common sense. At the end of the day, the people who write a plan must decide what does and does not go into a document called a plan. Some people will like the result and other people will not like the result. It is completely unreasonable to think that all content of any plan will ever be agreeable to everyone. Planning is a creative process and hence it is as much an art form as it is a science. Science itself is not a monolithic concept; there are many sciences just as there are many arts. Success in conservation, recovery and restoration will be enhanced to the extent that biologists and other technical experts apply a number of sciences and arts in going about their work in a strategic and flexible manner.

It is easy to lose sight of the forest by looking at the individual trees. It is also easy to get lost in the potential complexity and sheer volume of things that could potentially be done to recover a species at risk. One way to deal with this and retain focus on the fundamentals is to realize that there are only three fundamental strategies to recover a species at risk: remove the threats, manage the threats, and increase the population. We should be mindful of the reality of natural events, including extinction, some of which are beyond our control and ability to deal with through recovery or restoration activities.

We have an evolving set of guidelines and a draft content template for recovery plans in Canada (National Recovery Working Group 2003). This is very helpful but nobody should think that the work of creating a recovery plan is simply a ‘paint by numbers’ exercise of putting some words and numbers in a template. Planning requires a good deal of sober thought and rethinking. This is where the concept of Zen surfaces again. Planning is not a linear mechanical process; to be done well it requires some lateral thinking and frequent looping back and forth in plan content to ensure that the dots are appropriately connected to form a coherent picture.

THE ROAD AHEAD

Tremendous progress has been made in the area of species at risk recovery. But the job will never be done and there is ample opportunity for future improvement in the following areas:

- More concise recovery documents that dwell more on decisions and direction but less on description and discussion. Documents that are 70 to 180 pages long, with 8 page executive summaries are not overly helpful to very busy people who are being asked to implement the plans,
- Faster turn around of planning documents. One recently completed plan was begun around 1988 and not completed and published until 2002. Of course there are good reasons for such a long timeline but in the more legalistic world of SARA; we likely want to create the planning documents in much shorter timelines,
- Coordination, tracking and monitoring of recovery efforts identified in recovery strategies and action plans. Although work is underway, we have not yet constructed and tested the full range of institutional arrangements and administrative processes that will efficiently and effectively handle the new era of legal imperatives for species at risk,
- Restoration of areas infested by weeds or otherwise degraded by human activity, We should continue to aggressively conserve the best and restore the rest.
- Individual projects will get done in accordance with their time and dollar

allocations. But nothing will ever be perfect and we will continue to learn from both successes and errors; the road ahead has many potholes and rough spots. If we work together we can get around or over them. We should sometimes stop our journey down the road, long enough to fill in some of the potholes and smooth the rough spots. Failure to confront problems usually only makes them more difficult to deal with in the future.

A CLOSING THOUGHT

For those of you who are restoration practitioners, I offer the following suggestion: If you want funding for restoration work, pay attention to species at risk activities, both provincially and federally. If restoration is a reasonable requirement for a particular species at risk, this need should be identified in the recovery planning documents and should help immensely in securing scarce dollars to do such work.

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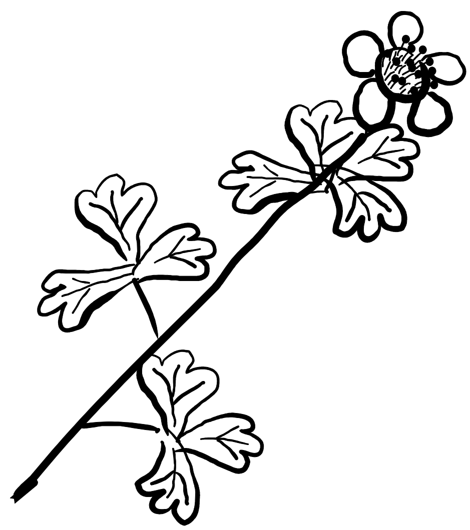

Fire effects and antelope-brush: fire not as detrimental as might be expected

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ABSTRACT

The use of fire as a restoration tool in antelope-brush dominated habitat is controversial. Antelope-brush itself is not fire dependent and can be killed by wildfire. In addition, seeds do not germinate because of heat, only few adult shrubs resprout, and wildfire often removes rodents necessary for seed dispersal in this species. However, antelope-brush will re-establish after a wildfire. We contrasted antelope-brush mortality following wildfire versus prescribed burn at Haynes lease Ecoreserve (wildfire: 1993), Oliver Ranch Road (wildfire: 1998), McIntyre Creek Road (prescribed burn: 2000), and Inkameep Provincial Park (2000). Only a few antelope-brush shrubs remained alive after the wildfires, but the prescribed burn resulted in almost 50% survival. It took almost 10 years before antelope-brush shrubs were again noticeable at Haynes lease Ecoreserve. We also studied understory plant re-establishment at Haynes lease Ecoreserve and Oliver Ranch Road. At Haynes lease Ecoreserve, the summer wildfire had some unexpected benefits: it eliminated downy brome, an invasive annual grass, for two years. Downy brome is well known to competitively exclude the establishment of other plants because it is so efficient at taking up early season moisture. The two-year window without downy brome facilitated seedling establishment of needle-and-thread grass, especially because 1993 was a wetter than average summer. By four years after the fire, cover of needle-and-thread grass had increased 7-fold. Oliver Ranch Road also had fewer downy brome plants in more severely burned plots, but needle-and-thread grass did not increase in response. Prescribed burning is being initiated in the south Okanagan to reduce fuel loading, to prevent wildfire, and to restore savannah-like attributes to ponderosa pine habitats for species at risk. Prefire preparations such as pruning of lower dead branches of selected antelope-brush individuals will further enhance the ability of antelope-brush to survive.

INTRODUCTION

Conservation of antelope-brush in the south Okanagan is important not only because it is uncommon in Canada, but also because it supports a variety of uncommon and at-risk animals. For example, the Threatened Behr's hairstreak depends on antelope-brush, its seeds are gathered by at-risk small mammals such as the western harvest mouse and Great Basin pocket mouse, and its leaves are browsed by California bighorn sheep (Krannitz & Hicks 2000). Therefore we need to be careful in our management of antelope-brush ecosystems. Recently, we have added controlled fire to our management tools, and a close examination of the effects of fire on antelope-brush is necessary to assess fire's conservation value.

Fire can be both detrimental and beneficial to antelope-brush. Densities of antelope-brush are highest in areas that have not been burned for decades, but they become "decadent" with age, with increasingly amounts of dead wood, seed production declines and seedlings become uncommon (Clements & Young 2001). Antelope-brush are more likely to resprout after a fire if it occurs in a cooler season (fall or spring) than in the hottest part of the summer (Driscoll 1963; Driver 1983; Saveland & Bunting 1987). This is likely because of higher stem water content: an August fire did not kill antelope-brush shrubs that had high moisture content, whereas in October, after a dry period, a fire killed them all (Zlatnik 1999).

The objective of this study is to study the effect of both wildfire and prescribed fire on the antelope-brush ecosystem in the south Okanagan. Response of antelope-brush to fire in the south Okanagan might differ from the response in other parts of its range because antelope-brush grows much more quickly here. In a study from 11 different sites, tree ring counts showed that the largest individuals with stem diameters of almost 16 cm and heights of four meters were no older than 40 years of age (Krannitz & Hicks 2000). Antelope-brush in other parts of the range further south grow much more slowly. Individuals about 40 years of age are only about one meter in diameter in California and Nevada (Clements & Young 2001) and Oregon (McConnell & Smith 1971). Antelope-brush in the east Kootenays also grow slowly, with a 2 cm diameter stem being 30 years old (Krannitz pers. obs.). Because antelope-brush grows relatively quickly in the south Okanagan, perhaps it will be able to re-establish more readily after a wildfire.

METHODS

The effect of wildfire and prescribed fire on antelope-brush was compared quantitatively at three sites: Oliver Ranch Road (wildfire July 28, 1998), McIntyre Creek Road (prescribed burn March 29, 2000), and Inkameep Provincial Park (wildfire July 21, 2000). Live and dead antelope-brush shrubs were recorded within belt transects established at each site in August 2001. The transects were 2.8 m in width (2 X 1.4 m), with variable lengths. A shrub was considered "in" if any part of it could be touched from the transect with the 1.4 meter stick. A shrub was considered "live" if there were any green leaves present. It was difficult to distinguish between antelope-brush and big sagebrush especially in the burned areas. Any shrub that was suspect was counted as antelope-brush, thus potentially inflating the mortality figure. Antelope-brush re-establishment following a wildfire was also studied subjectively at Haynes lease Ecoreserve, which burned on July 9, 1993.

Antelope-brush understory re-establishment following wildfire was studied at Haynes lease Ecoreserve and at Oliver Ranch Road. At Haynes lease Ecoreserve 298 Daubenmire (1959) (20 X 50 cm in size) permanent plots were established in September of 1994 over a range of burn severity, based on the effect of the fire on adjacent antelope-brush: Intense burn severity with no shrub remains, Moderate burn severity with a standing skeleton of blackened antelope-brush stems, Light burn severity with standing dead but not blackened antelope-brush stems, Lighter than light burn severity with resprouting (on the grazed side, and not included in this current study), and Unburned. The same burn severity scale was used at the

Vaseux Lake burn, but the site was much smaller, with only 88 plots.

Within the plots understory plant species were identified, percent cover was estimated and density was recorded for each species in each plot. At Haynes lease Ecoreserve the same plots were visited in September 1994, early June 1995, 1997 and 2002. At Oliver Ranch Road the same plots were visited in September 1998, early June 1999, 2000, 2001, and 2002.

RESULTS

Even without fire, 12% of antelope-brush shrubs were already dead at a control site opposite the Oliver Ranch Road wildfire (Figure 1). This would suggest that the site is "decadent". After the wildfire, 97.9% of antelope-brush shrubs perished (Figure 1), which was similar to the figure for the other wildfire at Inkameep provincial park (Figure 1). In contrast, the prescribed burn at McIntyre Creek Road resulted in 52.4% mortality (Figure 1).

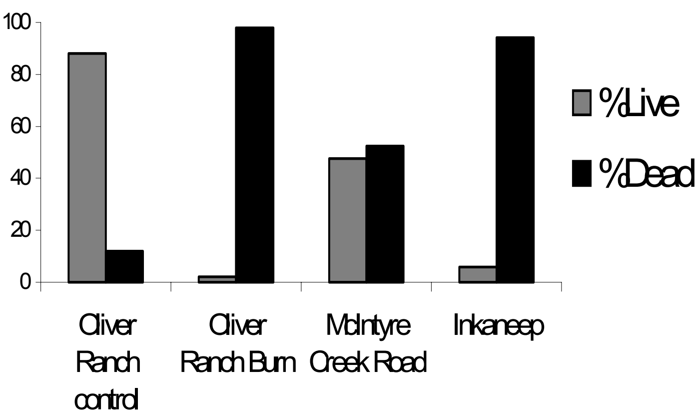

Figure 1: Percent dead and live antelope-brush shrubs at two wildfire sites (Oliver Ranch Road, Inkameep) and at Haynes lease one prescribed burn site (McIntyre Creek Road) and one control site (Oliver Ranch control). Total length of transect sampled was 174.5m (Oliver Ranch control), 236m (Oliver Ranch burn), 376.5m (McIntyre Creek Road) and 170.5m (Inkameep).

Figure 2: One year after the wildfire

Figure 3 : Nine years after the wildfire at Haynes lease

At Haynes lease Ecoreserve, understory species richness of plants found within the plots stayed the same since 1994 at 40 species. Some of the more common species, however, had an interesting and unexpected reaction to the 1993 wildfire. Downy brome, an alien invasive annual grass, was eliminated from the more severely burned areas of the site for two years post fire (Figure 4) which made it possible for seedlings of the native perennial grass needle-and-thread grass to establish and flourish (Figure 5). Plots that had downy brome had fewer individuals of needle-and-thread grass than plots that did not have any downy brome (Figure 6).

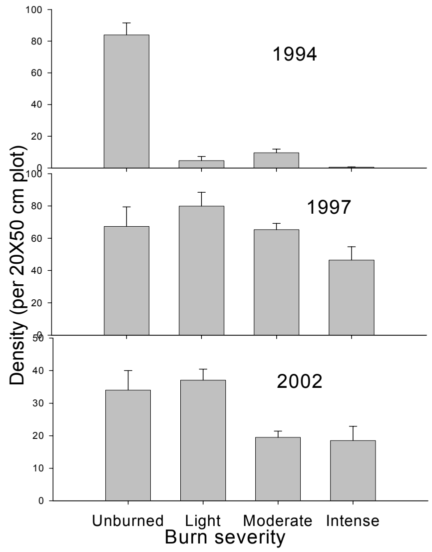

Figure 4: Number of downy brome per 20 X 50 cm plot at Haynes lease Ecoreserve in 1994, 1997, and 2002 at 4 different burn severities following a wildfire in 1993.

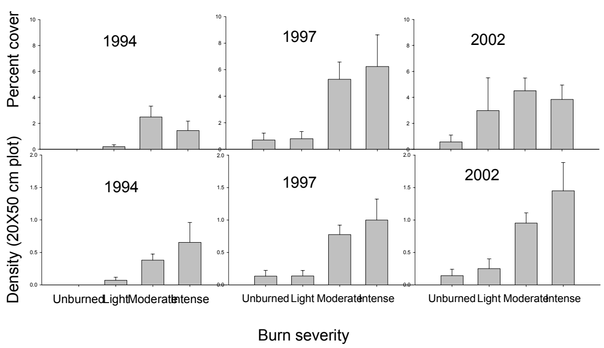

Figure 5: Density and percent cover of Stipa comata within 20 X 50cm plots at Haynes lease Ecoreserve in 1994, 1997, and 2002 at different burn severities following a wildfire in 1993.

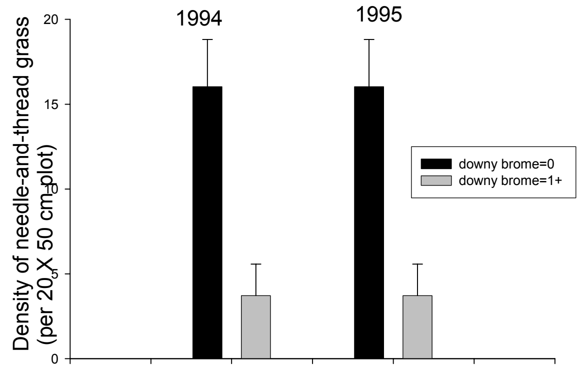

Figure 6: Number of needle-and-thread grass plants in plots at Haynes lease Ecoreserve without downy brome, as compared to plots with one or more of downy brome.

At the Oliver Ranch Road wildfire, downy brome density was also lower in the burned plots than in the unburned plots, and it was also a two year window (Figure 7).

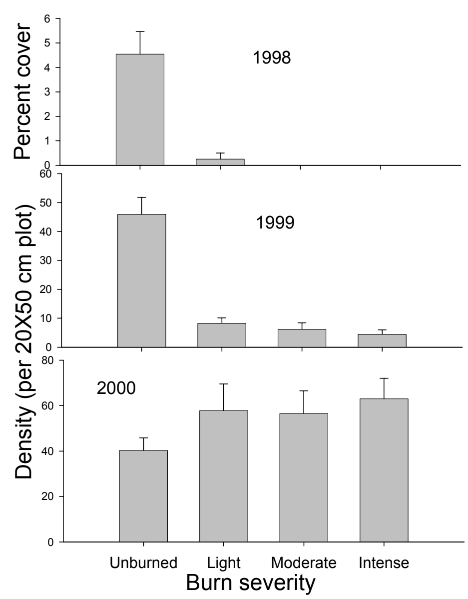

Figure 7: Density and percent cover of downy brome at Oliver Ranch Road in 1998, 1999, and 2000, following a wildfire in 1998.

However, needle-and-thread grass did not respond the same way as it did at Haynes lease

Ecoreserve (Figure 8). Instead of being more abundant in the more severely burned sites, it was less abundant.

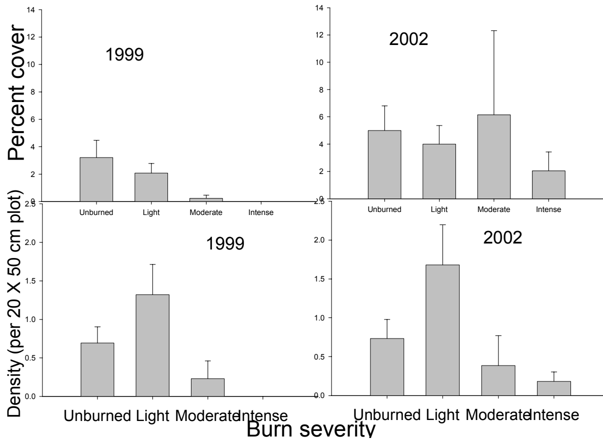

Figure 8: Density and percent cover of needle-and-thread grass at Oliver Ranch Road in 1999, and 2002, following a wildfire in 1998.

DISCUSSION

Even though the antelope-brush at our sites were young compared to individuals further south or in the Kootenays, they still appear to become "decadent" at approximately the same size, and begin to die off. Fire offers an opportunity for renewal, and prescribed burns provide a method for doing so while still retaining mature antelope-brush at a site. Resprouting and seedling establishment, two other components of antelope-brush response to fire, were not measured in this study, and are potentially fruitful areas for further restoration research. In Washington state resprouting did not occur after a prescribed burn in the fall because of low soil moisture, but did occur after a prescribed burn in the spring, when stem moisture was higher (Driver 1983). In contrast, seedling establishment was greatest following a fall prescribed burn, and non-existent following a spring burn (Driver 1983). If there is much dead wood on an antelope-brush shrub, it will burn completely (Figure 9). We are mitigating this effect in the current prescribed burns planned for areas with antelope-brush by working with the SOSCP Outreach Team to enlist volunteers to prune lower branches of some of the antelope-brush shrubs to ensure their survival.

There are many reasons to prevent wildfire in a developed area such as the south Okanagan valley, and prescribed fires will help reduce fuel loading and the possibility of wildfire. However, our data suggest that when wildfire does occur and eliminates antelope-brush, within a decade antelope-brush will re-establish at sites in the south Okanagan, though not yet at pre-fire densities. This falls within the range of 1- 32 years (mean of 13 years) for re-establishment in California (Nord 1965). Recently, competition from downy brome appears to prevent antelope-brush establishment following wildfire in the United States (Young et al. 1972; Updike et al. 1990). Unlike wildfire, prescribed burns in Oregon and California permit and facilitate antelope-brush seedling establishment with similar growth rates as in the south Okanagan (Martin 1983).

Figure 9: A mature antelope-brush bursts into flame during a prescribed burn.

Downy brome prevents native perennial seedling establishment by sequestering limited early spring moisture (Melgoza et al. 1990). Our data suggest that the more severe parts of the wildfire at Haynes lease Ecoreserve reduced downy brome densities enough for the perennial needle-and-thread grass to establish. Though we did not collect data on downy brome seed banks in the soil, it is well known that downy brome seed are not viable for more than a year (Crist & Friese 1993; Wicks 1997). Because both the Haynes lease Ecoreserve and Oliver Ranch Road wildfires occurred after downy brome seed production, the more severe fires probably burned up that year's crop of seeds. This has also been observed elsewhere (West & Hassan 1985; Claire Deleo personal communication). However, in the United States, fire in shrub-steppe habitats is synonymous with downy brome, and some shrub-steppe habitats previously with 80-year fire cycles now are dominated by downy brome, with annual fire cycles (Knick 1999). Our hypothesis, therefore, is that these fires in the United States are now at such a low severity, that they no longer kill downy brome seeds.

The differing effect of wildfire on needle-and-thread grass is curious. It is thought that needle grasses in general are sensitive to fire (Wright & Klemmedson 1965; Busso et al. 1993; Pelaez et al. 1997). Our results from the Oliver Ranch Road burn were consistent with this view, in that needle-and-thread grass was less abundant in the plots burned more severely. At Haynes lease Ecoreserve, it may be that the increased abundance with burn severity came from seed germination and establishment. July and August of 1993 were much wetter than average: total precipitation at Summerland was 120 mm, whereas the 86 year average from 1908 to 1994 was a total of 53.7mm (Environment Canada unpublished data). It is possible that 1993 provided unusually ideal conditions for seed germination and seedling establishment, whereas 1998 did not. Follow-up research from this project includes competition studies with downy brome and interaction with soil moisture.

In conclusion, there is some need for renewal in antelope-brush ecosystems that in a controlled fashion might be best served by prescribed burns, with prefire pruning of dead branches of the antelope-brush individuals to be retained. These cooler fires might also prevent the loss of other important attributes of the ecosystem such as at-risk snakes and rodents that would at that time be safely in their burrows and hibernacula.

ACKNOWLEDGEMENTS

We thank Bruce Bennett, Randi Mulder, Malcolm Martin, Lisa Scott, Dennis St. John, Judy Millar, Saskia Arnesen, Natalie Griller, and Jeremy Kroeger for assistance in the field. Operational funds from Environment Canada and BC Ministry of Forests funded the work.

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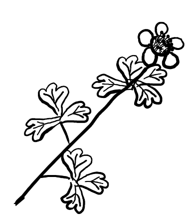

Using Bunchgrass Plugs to Restore Degraded Rangeland

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ABSTRACT

In dry forests of North America, fire suppression has facilitated a change from open stands to closed canopy, mesic stands of shade-tolerant species. In the East Kootenay Valley of British Columbia, this trend is evidenced by the ingrowth and encroachment of low-density veteran forests by younger age classes of interior conifer species. Within these stands, shading caused by conifer species as the stand moves toward a late-seral state has favoured the invasion of pinegrass, a shade-tolerant plant. Competition from pinegrass, combined with decreased light may result in the exclusion of native bunchgrasses from fire-maintained plant communities. The goal of this experiment was to assess the efficacy of transplanting native bunchgrass plugs (bluebunch wheatgrass and Richardson's needlegrass) for facilitating ecosystem restoration using different plant species, seasons of planting and pinegrass removal. Plugs with surrounding pinegrass removed had a significantly greater survival rate than those with pinegrass present. There was also a significant species effect, as a greater percentage of bluebunch wheatgrass plugs survived overall compared to Richardson's needlegrass. There was a significant season by species effect, due to greater survival of fall-planted bluebunch wheatgrass plugs compared to Richardson's needlegrass. A greater number of Richardson's needlegrass plugs survived when planted in the spring. Plug growth as measured by change in tiller number was significantly affected by pinegrass removal and choice of species in the fall planting season. Richardson's needlegrass and bluebunch wheatgrass plugs are both good candidates for restoration of recently thinned forests, if planted at optimal times.

INTRODUCTION

In fire-maintained forests of North America, fire suppression has facilitated a change from open, dry stands to closed canopy, mesic stands of shade-tolerant and fire sensitive species (Cooper 1960, Arno and Gruell 1986, Lunan and Habeck 1973, Habeck 1990, Arno et al. 1995, Gayton 1997, Arno et al. 2000). In the Rocky Mountain Trench of British Columbia (BC), this trend is evidenced by the ingrowth and encroachment of low-density veteran forests by younger age classes of interior conifer species, including *Pseudotsuga menziesii* var. *glauca* (Douglas-fir) and *Pinus contorta* var. *latifolia* (lodgepole pine) (Gayton 1997). Within ingrown stands, shading caused by the invasion of conifer species as the stand moves toward a late-seral or climax state, has favoured the invasion of mesophytic shrubs and herbs (Lunan and Habeck 1973). These late seral species can successfully out-compete mid-seral species (including native bunchgrasses) that are intolerant of the new conditions, including low light (Tilman 1988).

For example, Calamagrostis rubescens (pinegrass), a dominant rhizomatous species of northern inland forests (Franklin and Dyrness 1973), is often abundant under dense conifer canopies (Steele and Geier-Hayes 1993). In combination, decreased light and increased competition from pinegrass may limit the existence and distribution of native bunchgrasses that were once common prior to ingrowth. .

The Invermere Forest District, in the East Kootenay Valley of British Columbia (BC), initiated an intensive bunchgrass seeding program in 1994 for the purpose of facilitating rangeland rehabilitation, road reclamation and ecosystem restoration. Success of seeding trials was low due to poor germination. This is common of restoration seeding projects in arid and semi-arid rangeland, where projects fail because of the lack of moisture required for successful germination (Grantz et al. 1998). Restoration projects also fail because there is often a lack of species-rich grasslands to act as a propagule emigration source (Davies et al. 1999). Due to limited success of seeding, the use of transplants has been increasing since the 1980s for rangeland restoration. Several restoration techniques have been evaluated for successful establishment, in both a research and applied land management context (Bainbridge et al. 1995, Grantz et al. 1998, Davies et al. 1999, Ewing 2002, Mulligan 2002).

The Invermere Forest District proposed the use of bunchgrass transplants for the purpose of restoration in 1997. Native seed (local to the area), including Festuca campestris (rough fescue), P. spicata, and S. richardsonii, was grown into 'bunchgrass plugs' under greenhouse conditions for subsequent planting on degraded sites. All plugs were grown from seed at the Skimikin Nursery in Tappen, BC.

This research was designed to assess the feasibility of using two species (P. spicata and S. richardsonii) for use in restoring ingrown forests. In this context, bunchgrass plugs are specifically being used to restore bunchgrasses to thinned ingrown forests where they were once common in grasslands and open forests. The goal of this experiment was to assess the success of transplanting native bunchgrass plugs. We hypothesized that there was no difference between the survival and vigour of the two species, that there was no competition effect of pinegrass and there was no difference in survival of vigour of the plugs when planted in different seasons (fall and spring).

METHODS

Study Area and Experimental Design

Planting trials were conducted at 3 blocks (North, South and Zehnder), all located within 100 km of each other within the Rocky Mountain Trench of BC. Average annual precipitation is 384.5 mm, with May and June being the wettest months. This project was initiated in 2001 when summer rainfall was 35% of the long-term average during the growing season (May-September).

All blocks were uniform in vegetation type IDF biogeoclimatic zone (Braumandl and Curran 1992), abundant in pinegrass, level (i.e. no slope) and lacking ingrowth to give full light conditions. Bunchgrasses are the dominant grass species in open, dry zonal IDF, although ingrown and encroached areas are lacking these species (Page 2002). Current commercial uses of these areas include cattle grazing. Soils at all 3 blocks were characterized by Orthic Eutric Brunisols.

Each experimental block was 6.0 by 7.2 m in dimension (43.2m^2). Bunchgrass plugs were systematically planted 60 cm apart in the fall (8-9 October, 2000) and in the spring (8-11 May, 2001). The treatments were S. richardsonii-C. rubescens removal, S. richardsonii-no C. rubescens removal, P. spicata-C. rubescens removal and P. spicata-no C. rubescens removal. C. rubescens was removed using glyphosate, a systemic, translocated, non-residual herbicide. Glyphosate was applied directly to pinegrass plants by wiping the herbicide (7g/L concentration) onto leaves with a cloth. During treatment, transplants had a glass jar placed over them to prevent exposure to glyphosate. Herbicide treatments had a significant ($p<0.001$) effect on the cover of pinegrass at all blocks in both seasons, reducing pinegrass by an average of 8% (13% - 3%). There was no supplementary watering provided to plugs to mimic natural field conditions. 40 plugs of each species were planted at each of the 3 blocks in October 2000, with 20 randomly assigned to each of the 4 treatments, for a total of 240 plugs at each block. In the spring, 32 additional plugs were planted at each of 2 blocks, with 8 plugs randomly assigned to each treatment. Samples sizes were reduced in the spring due to a limited supply of plugs. Plant height, tiller number, and basal area were assessed for each plug at the time of establishment. Plugs planted in the fall were subsequently examined on 8 May, 2001 for overwinter survival Both spring and fall planted plugs were monitored for survival, height, basal area, number of tillers and number of inflorescences during September 2001. As this study was intended to test the operational success of transplanting plugs, sites were not protected from grazing during the establishment trials.

STATISTICAL ANALYSES

To meet the assumptions of analysis of variance, all data were tested for normality using univariate procedures in SAS (SAS 1999), with no transformations necessary. The effect of season, species and pinegrass removal on plug survival was tested using a split-plot design, with season of planting as the main plot factor. Survival percentages for each treatment combination were used as observations in the model (i.e. 24 observations with no subsampling). The effect of bunchgrass size (i.e. initial tiller number, height and basal area) on survival was also tested using a one-way ANOVA. For this analysis, each species was examined separately as the two species have inherently different tiller numbers (i.e. S. richardsonii generally has a greater number of tillers than P. spicata).

Survival was lower than expected for S. richardsonii plugs planted in the fall, and also variable across blocks, seasons and species (Table 1), resulting in an unbalanced number of remaining plugs among treatment groups.

An ANOVA using a split-plot design, identical to the survival analysis, was conducted on the surviving plugs to determine the effect of planting season, pinegrass removal, and plug species on growth characteristics (i.e. tiller numbers). Where higher level block interactions were detected (2-way or 3-way), blocks that were responding differently were isolated and analyzed separately. Due to unequal sample sizes among treatment combinations based on variable survival, treatment effects were further analyzed by season using a fully randomized block design (i.e. seasons were analyzed separately). This was done to isolate possible pinegrass effects within each season.

Table 1. Survival of S. richardsonii and P. spicata plugs in 2001 across 3 blocks, 4 treatments and 2 seasons. No plugs were planted at the south block in the spring.

Season	Block	Species	Survival %[1]		
			Pinegrass Removed	With Pinegrass	Combined
Fall[1]	North	*S. richardsonii*	35	25	30
		P. spicata	95	95	95
	South	*S. richardsonii*	15	10	12.5
		P. spicata	55	53	54
	Zehnder	*S. richardsonii*	25	10	17.5
		P. spicata	100	90	95
	Total	*S. richardsonii*	**25**	**15**	**20**
		P. spicata	**83**	**79**	**81**
Spring[2]	North	*S. richardsonii*	78	67	73
		P. spicata	50	38	44
	Zehnder	*S. richardsonii*	63	63	63
		P. spicata	50	37	44
	Total	*S. richardsonii*	**71**	**65**	**68**
		P. spicata	**50**	**38**	**44**

[1] n=20 per treatment combination at each block.
[2] n=8 per treatment combination at each block.

Although basal area and plant height were measured on each transplant, there was no evidence of a change in basal area (or the change was too small to analyze) and grazing effects confounded any changes in height. As a result, these latter variables were dropped from the analysis. All results were considered significant at $p<0.10$, unless noted otherwise.

RESULTS

Plug Survival

Pinegrass removal effected ($p=0.02$) the survival of bunchgrass plugs. Plugs with the surrounding pinegrass removed had a significantly greater survival rate than those plugs with no pinegrass removed (Fig. 1). There were also significant species ($p=0.002$) and season by species effects ($p<0.0001$), as a greater percentage of P. spicata plugs survived overall compared to S. richardsonii plugs (Table 1). The season by species interaction was due to greater survival of fall-planted *P. spicata* plugs compared to *S. richardsonii*. Conversely, a greater number of *S. richardsonii* plugs survived when planted in the spring than *P. spicata* (Table 1).

Additional analysis indicated the size of the bunchgrass plug, as determined by initial tiller number, basal area and height, generally had a significant effect (minimum $p<0.10$) on the survival of both species. These results indicate larger plugs had a greater likelihood of survival during the first year after establishment under the conditions of this investigation.

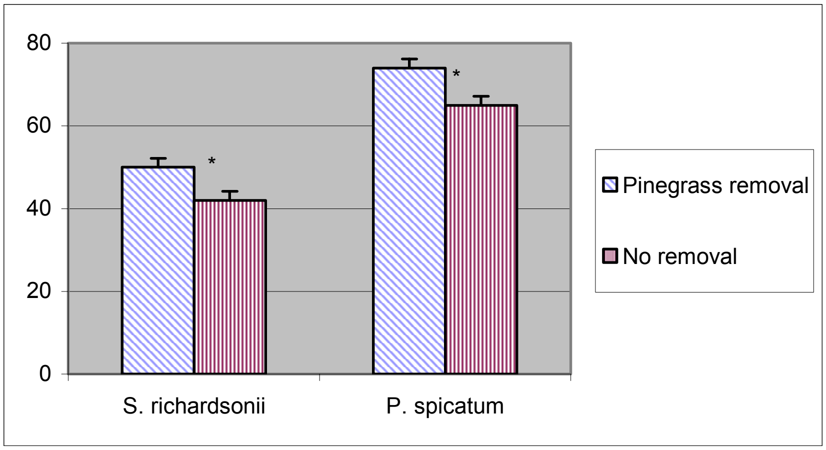

Figure 1. Effect of pinegrass removal on survival (%) of two species of bunchgrass plugs in 2001. Within a species, an (*) indicates a significant difference (p<0.1). Data includes both planting seasons.

Bunchgrass Plug Growth

Changes in tiller number among the remaining live plants was also effected by the treatments. A significant season by species interaction was evident (p=0.0003). S. richardsonii plugs that managed to survive fall planting lost fewer tillers compared to those planted in the spring (p=0.04). Similarly, P. spicata plugs that survived spring planting lost fewer tillers than those surviving the fall planting (p=0.002). These results are in contrast to the survival data outlined earlier.

When fall and spring planted plugs were analyzed separately, significant interactions between block and pinegrass, as well as block, pinegrass, and species, were found (p<0.01). Further examination indicated that plugs at the South block were behaving significantly different than at either the North or Zehnder block. The inconsistent results are likely due to initial differences in pinegrass cover among blocks (6% at the south block compared to 23.5% and 15% at the North and Zehnder block, respectively), and a greater level of grazing at the South block (15% of plugs were grazed, versus 2% at the North and Zehnder block). As a result, the South block was analyzed independently.

When the North and Zehnder block were combined in analysis there was a significant pinegrass effect (p=0.02) and a significant species effect (p=0.05) on the fall planted plugs. Bunchgrass plugs in the pinegrass removal treatment lost significantly fewer tillers than those with no pinegrass removed. Between species, *P. spicata* plugs lost a greater number of tillers overall (63%) versus *S. richardsonii* plugs (10%) (p=0.05). There was also a pinegrass by species interaction (p=0.07), which was largely due to the positive effect of pinegrass removal on tiller numbers in fall planted *S. richardsonii* plugs (Fig. 2). When the fall planted South block was analyzed in isolation, there was a significant species by pinegrass interaction (p<0.10), with *S. richardsonii* plugs losing a larger number of tillers when the adjacent pinegrass was removed.

Among the spring planting treatments, there was a significant block (p=0.08) and species effect (p=0.03). Plugs in the Zehnder block lost fewer tillers (10%) across both species than those in the North block (44%). Additionally, *S. richardsonii* plugs in both blocks lost a greater number of tillers than plugs of *P. spicata*. The absence of higher level interactions (e.g. block by main treatment effects) indicated treatments within these 2 blocks affected response similarly. There were no significant pinegrass treatment effects within the spring planting treatment.

Although inflorescence data were too variable to detect differences, all the plugs that did produce seedheads were planted in the fall (32 of 240) rather than spring. Two of these were *S. richardsonii* plugs while 30 were *P. spicata*.

DISCUSSION

Overall 56% of the plugs planted survived the first year of growth. Davies et al. (1999) observed 19% survival of grass plugs over three years. Further monitoring is needed to monitor long-term survival. Fall planted plugs were generally more likely to survive under the conditions of this study. Despite this, the favourable survival of S. richardsonii with spring planting complements successful spring planting trials completed by the Invermere Forest District using this species. Preliminary field studies completed by range ecologists in the Invermere Forest District showed that survivorship when transplanting S. richardsonii was 94% without grazing and 50% with grazing. All plugs were planted in the spring (May 21) at the same location. In that same trial, however, survivorship of P. spicata plugs was considerably lower (3.6%), reinforcing the results found here that this species is not adapted to spring planting. The current study also found survival of *S. richardsonii* to be much lower than *P. spicata*, particularly in the fall planting treatment (Table 1). Overall, the results observed here indicate survival can be optimized by planting *P. spicata* plugs in the fall and *S. richardsonii* in the spring. Abnormally dry weather conditions combined with other stresses may have contributed to the seasonal and intraspecific variation found in the 2001 growing season. Separate trials and continued monitoring of transplanted plugs at these 3 blocks are needed to broaden the temporal scope of inference.

Differential survival rates for spring and fall planting may be related to the biology of the 2 species. *P. spicata* initiates growth in the early spring, as early as the third week of February (Willms et al. 1979). Thus, planting this species in the spring (i.e. May) will shorten its growing season considerably. Parsons et al. (1971) found *P. spicata* required 51 days for the completion of reproductive development, while *Stipa comata* (needle and thread grass) required only 18 days. Rapid growth of *S. comata* grass appears to be related to an increase in temperature (Parsons et al. 1971), which implies this species behaves similar to a C_4 (warm season) rather than a C_3 (cool season) species. *Stipa* species have been reported to behave similar to C_4 species, growing well in relatively hot and dry climates (Gurevitch 1986). This may be related to anatomical and morphological characteristics associated with drought tolerance. Rolled leaves and prolonged metabolic activity after the onset of dry conditions both contribute to superior drought tolerance of this C_3 species (Gurevitch 1986) relative to other C_3 species. The growth of *P. spicata*, a C_3 species, appears to be unaffected by temperature (Willms et al. 1979). Planting *S. richardsonii* in the fall may decrease its chance of survival due to low temperatures, while planting *P. spicata* in the spring may decrease its chance of survival due to a shortened growing season.

Although season of planting had a clear effect on survival, the effect on growth is not

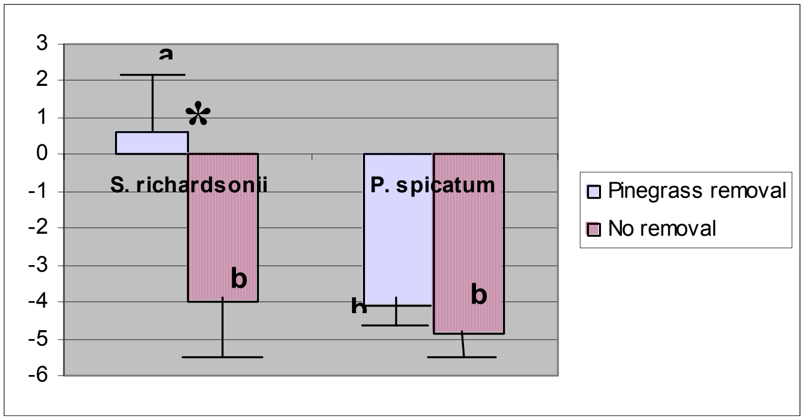

Figure 2. **Change in tiller number during the 2001 growing season for each of 2 species and 2 levels of pinegrass at 2 blocks following fall planting. Within a species, an (*) indicates a significant difference (p<0.1). Among all treatments, means with different letters differ significantly (p<0.05).**

as clear. While *S. richardsonii* plugs performed better than fall planted *P. spicata* plugs, the opposite relationship was true with spring planting. This is likely a result of selection for strong growth traits, as plugs that survived in sub-optimal planting conditions will have traits that predispose them to superior growth, resulting in a bias within the surviving plugs towards greater tiller increases (or fewer tiller losses).

These results are further supported by the established selection bias within plugs that survived planting trials in favour of larger individuals. Studies examining inter-specific competitive responses of grass plugs in North American grasslands have found that initial plant size confers a competitive advantage over other species (Wilson 1994, Gerry and Wilson 1995, Davies et al. 1999). Davies et al. (1999) found that initial size did not affect survival over all but did affect the survival of individual species. These findings are therefore consistent with the observations in this study and suggest that larger plugs should be used in restoration projects to maximize the potential for bunchgrass establishment and growth.

Another factor that affected the survival and vigour of plugs was pinegrass removal. Competition for limited resources may determine the presence, absence, or abundance of species in a community and determine their spatial arrangement (Pyke and Archer 1991). Pinegrass competition had an adverse impact on plug survival in both planting seasons (Fig. 1). Pinegrass is a rhizomatous species that initiates growth early in the spring (McLean 1979). Due to its shallow rooting habit and early emergence, pinegrass is a very effective competitor for the limited moisture found in dry forest stands. For example, Peterson (1988) noted that pinegrass competition had a negative impact on ponderosa pine seedling stemwood, foliage and root weight. Studies have shown early emerging species continually increase their ability to capture resources at the expense of later emergers, and in doing so, increase their physical zone of influence (Ross and Harper 1972). Therefore, when transplants are grown in the presence of early growing neighbours, their growth and establishment is compromised (e.g. Ross and Harper 1972, Wilson 1994, Gerry and Wilson 1995, Davies et al. 1999, Peltzer and Wilson 2001).

It appears that although pinegrass competition affected plug survival in both seasons it had a greater impact on *S. richardsonii* rather than *P. spicata* growth

(Fig.2). The lack of an effect of pinegrass removal on change in tiller numbers in the spring planting treatment could be due to abiotic conditions at the time of planting. Transplant shock and the lack of moisture may have limited spring growth rather than pinegrass competition.

Drought during 2000 and 2001 may also have affected plug survival and growth in this investigation. The year prior to planting (2000) was unusually dry (~45% of normal, May-September) as was the year of planting (~35% of normal, May-September). Precipitation was greater at the Zehnder block during the growing season, however, and may be responsible for the better plug growth at this site within the spring planting treatment.

Although inflorescence production was limited in this study, plugs of *P. spicata*, particularly those planted in the fall, did exhibit considerable seedhead production. This response is important as it represents an important recovery mechanism (rebuilding the soil seedbank) for this key bunchgrass, thereby increasing the likelihood for additional increases in this species.

CONCLUSIONS

S. richardsonii and *P. spicata* plugs are both good candidates for restoration of recently restored forests and are even able to establish during drought conditions. Proper planting season, the use of larger plugs and removal of competition should increase the chance of plug survival and improve growth.

Factors such as initiation of growth, tolerance of drought, grazing, initial plug size and time needed for development all need to be considered when planning a restoration strategy.

It is recommended that plugs be monitored for several years to ensure these results are not anomalous. This will allow for a more comprehensive evaluation of the effect of pinegrass competition on plug survival and growth

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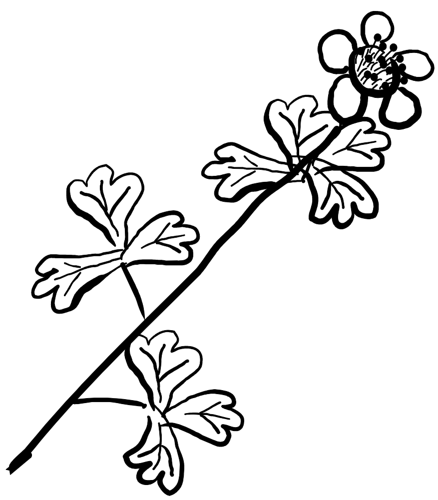

Garry Oak Ecosystem Restoration: Implications for Antelope Brush Restoration

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ABSTRACT

Similarities between the restoration of Garry Oak ecosystems and Antelope Brush ecosystems are greater than their differences. In both cases, urban and agricultural encroachment, modifications to natural disturbance regimes and the incursion by non-native invasive species have severely degraded the natural ecosystems. Restoration work in both environments needs to address these issues to be successful. Urban and agricultural encroachment has resulted in habitat fragmentation and loss of gene pool continuity. This may have a profound effect on the preservation of species at risk and the ability of these ecosystems to respond to other stresses. Control of natural and anthropogenic fire has significantly altered the species composition and the stand structure in these ecosystems. Non-native invasive species can significantly alter ecosystem processes and can modify species composition. Changes in nutrient flows associated with invasive legumes and grasses can alter the dynamics of succession as well as nutrient cycling in these ecosystems. In addition to the similarities between these ecosystems, differences in land management between the Antelope Brush and the Garry Oak ecosystems has had a profound effect on ecological processes and conditions. Livestock grazing is an integral part of much of the remaining Antelope Brush ecosystem while livestock grazing is not a significant factor in the current degradation of the Garry Oak ecosystems. All of these factors influence the manner in which ecological restoration is conducted in these ecosystems. This paper explores the similarities and differences in approach to the restoration of these complex ecosystems.

INTRODUCTION

Ecological restoration has been defined by the Society for Ecological Restoration as follows:

> *Ecological restoration is the process of assisting the recovery of an ecosystem that has been degraded, damaged or destroyed."* (Society for Ecological Restoration, 2002)

This definition provides a foundation for the development of restoration works as it incorporates the concept of helping in the recovery of degraded ecosystems. The best solution to degraded ecosystems is to help them along the natural recovery pathway. This solution works well in both the Garry Oak ecosystems and the Antelope Brush ecosystems that are the topic of this paper. However, in both cases, the natural recovery pathway has been modified by the incursion of alien invasive species as well as native invasive species that have been assisted in their invasion by modification of key ecological processes, notably fire. Restoration work in these ecosystems requires that the ecological impacts of invasive species be addressed to be successful.

Effective restoration plans consist of a **description** of the current state of the ecosystem to be restored; a **comparison** of this to a reference ecosystem; **interpretation** of the ecological processes that require adjustment to allow the ecosystem to regain the natural recovery trajectory; and the **application** of this information in the formulation of specific restoration activities. In addition to these fundamental aspects of restoration plans, some measure of monitoring and maintenance needs to be applied at restoration sites. Active management of the Garry Oak and Antelope Brush ecosystems is essential to the effective restoration of these ecosystems as both have suffered from the modification of natural disturbance regimes.

This paper provides an overview of Garry Oak ecosystem restoration and the various aspects that must be considered for successful restoration, recognizing that restoration in this ecosystem is a complex process that is only marginally understood. For this reason, an adaptive management approach (Murray and Jones, 2002) is applied in the development of restoration strategies for this ecosystem. Adaptive management incorporates the concept of a feedback loop in the framework so that knowledge gained during the process of restoration may be applied to improve the restoration work that is undertaken. By incorporating an adaptive management approach in the restoration works that are undertaken, knowledge will be gained of the outcomes of treatments. This knowledge can then be applied in future restoration efforts.

The knowledge and understanding that has been gained in the restoration of Garry Oak ecosystems can be applied, using an adaptive management approach, to the restoration works undertaken in the Antelope Brush ecosystems and vice-a-versa. By sharing our restoration experiences in these ecosystems the work of restoration will become more precise and effective.

MATERIALS AND METHODS

Site information for the Garry Oak ecosystem restoration plan (Somenos Garry Oak ecosystems) that is used as an example in this paper has been collected using the Terrestrial Ecosystem Mapping (TEM) methodology and from assessments of the example site using the methodology outlined in "Describing Ecosystems in the Field" (Luttmerding et al, 1990). Other information has been gained from a review of the literature.

RESULTS AND DISCUSSION

The first step in the development of effective restoration plans is an accurate description of the site to be restored. The use of TEM methodology allows an integration of landforms, soils and vegetation to be presented in the context of the current condition of the ecosystem. Detailed TEM work can provide the basis for defining management prescriptions. The Somenos Garry Oak ecosystems (SGO) can be classified on the basis of Garry Oak and conifer cover (or lack thereof) as well as the cover of shrubs in the understory or herbaceous vegetation.

Classification of the vegetation on a site where restoration is proposed must be developed to suit the planned restoration works. Often, simple or modified physiognomic classification schemes work well as the structural changes that arise from the modification of ecosystem processes can be captured with such systems. Table 1 provides a simple classification system that can be applied to the SGO ecosystems to form the basis for restoration treatments.

Table 1 Physiognomic Classification System for Somenos Garry Oak Ecosystems

Open meadow
Shrub lands
Invasive species dominated
Native species dominated
Savannahs (< 10% tree cover)
Herbaceous
Shrub dominated
Invasive species dominated
Native species dominated
Woodlands (10 – 50 % tree cover)
Deciduous (Garry Oak)
Mixed (Garry Oak/conifer)
Conifer

The classification system presented in Table 1 provides a distinction between native species dominated shrub lands and those dominated by invasive species, allowing differences in treatments between these two types. Clearly this is important in addressing stands dominated by alien invasive species such as Scotch broom (*Cytisus scoparius* L. Link), but how can we determine treatments for sites with native invasive species such as snowberry (*Symphoricarpos albus* (L.) Blake) using such a classification system? Maintaining the invasive status in the classification system for only alien invasive species and allowing native species such as snowberry to be included with the native classification, whether or not they are invasive in the context of these ecosystems is one way of addressing this issue. In addition to snowberry, Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) is a native invader that moves into deep soiled Garry Oak ecosystems in the absence of fire.

The use of specific ecosystem mapping allows the development of specific restoration treatments within the framework of the classification system presented in Table 1. Polygons that reflect vegetation condition that are indicative of encroachment and displacement of native ecosystems can be slated for specific treatments that reverse these trends. However, the end goal of the treatments needs to be identified.

The identification of reference ecosystems can be difficult in both Garry Oak ecosystems and Antelope Brush ecosystems as in both cases, sites where human influences have not been felt do not exist. Fire has been prevented as much as possible in both of these ecosystems for the past 100 or more years. In addition, livestock grazing has influenced the floristic composition of both of these ecosystems. In the absence of extant pre-contact ecosystems with which to compare the ecosystems proposed for restoration, some form of re-constructed ecosystem must serve.

Historic records indicate that the deep-soiled Garry Oak ecosystems were an open savannah with scattered meadow areas and Douglas-fir on the north slopes and in moist stream valleys. The understory was composed of herbaceous species, some of which (*Camassia* spp. and *Fritillaria* spp.) were harvested as a staple in the native diets. Harvesting of the bulbs of these plants required digging in the soil. Thus, in addition to burning, the soils were disturbed by digging (Turner in Boyd, 1999). However, alien invasive species were not present in significant numbers (if at all) at the time of these ecosystem treatments. While re-establishing the former disturbance regimes (fire and bulb digging) may serve to reverse the observed trends of native species invading these ecosystems, with the current densities of alien invasive species present these treatments may well open the site to invasion by alien species. The same may be true for the Antelope Brush ecosystem where re-introduction of fire may lead to establishment of species such as cheatgrass (*Bromus tectorum* L.) and knapweeds (*Centaurea* spp.). A strategy that deals with the re-introduction of pre-contact disturbance regimes while minimizing the potential for alien invasive species establishment is required.

Understanding the mechanisms that the alien invasive species employ to establish and spread through these ecosystems is essential to development of strategies for management of them. Specific autecological investigations can provide the information needed to develop such strategies. Murray and Pinkham (2002) present a list of the 10 worst invasive species in Garry Oak ecosystems. This list was derived from discussions with Garry Oak ecosystem experts in knowledge engineering workshops held for the purpose. The rankings are based on significance of impact; difficult of control or management; and urgency of control or management. These species are listed, from "worst" to "least worst" [1] in Table 2 (Polster, 2002).

Table 2: Ranking of 10 Worst Alien Invasive Species in Garry Oak Ecosystems

Common Name	Scientific Name
Orchard grass	*Dactylis glomerata* L.
Scotch broom	*Cytisus scoparius* (L.) Link
Gorse	*Ulex europaeus* L.
English ivy	*Hedera helix* L.
Velvet-grass	*Holcus lanatus* L.
Spurge-laurel	*Daphne laureola* L.
English hawthorn	*Crataegus monogyna* Jacq.
Sweet vernalgrass	*Anthoxanthum odoratum* L.
Himalayan blackberry	*Rubus discolor* Weihe & Nees
Hedgehog dogtail grass	*Cynosurus echinatus* L.

Restoration strategies that help to decrease the vigour of the alien invasive species while minimizing the impact on the native species in the community. Since little is know of the effects of various treatments on both the desired native species and the undesired alien invasive species, the design of treatments must incorporate some measure of research.

[1] The application of human value judgment terms such as "worst" and "least worst" are used for convenience only and do not imply a judgment on the absolute value of the plants biologically.

For instance, establishment of test plots that document the before and after floristics of the site being treated will allow comparisons to be made. Similarly, incorporating several treatments in side by side comparisons with nested plots within the treatment plots can provide information on the response of the ecosystem to the treatment regime.

Burning is probably the most important ecological process in both the Antelope Brush and the Garry Oak ecosystems. However, some alien invasive species respond positively to single burn treatments. Scotch broom banks seeds that readily sprout following fire (Turner in Boyd, 1999). Although this attribute may be considered a problem, it can be exploited to the benefit of the ecosystem. Burning twice within one growing season will allow the seeds of the broom to germinate initially while the young germinants will be killed by the second fire.

Interpretation of the ecological attributes of the community being restored allows effective strategies to be developed. Understanding the specific autecological characteristics of the alien species that might invade sites where restoration treatments are undertaken can allow the restoration treatments to be modified so that the invasive species are eliminated or at least their growth and reproduction is restricted. Cheatgrass is a significant problem in the Antelope Brush ecosystems as it displaces many native species (Carpenter and Murray, 1999). This species is a winter annual, germinating in the fall with the onset of fall rains. It grows vigorously in the spring, often robbing moisture from other plants. It may increase the likelihood of fire as it matures and dries in June rather than later in the summer for many of the native perennial grasses. Seed set is typically by mid-June (West, 1983). Treatments that might be effective for this species would accentuate the differences between this species and the native perennial species. For instance, burning in the late winter when the cheatgrass has started to grow but before the perennial grasses have initiated growth may be particularly effective at reducing cheatgrass populations while at the same time releasing nutrient bound in the dead materials from growth the previous season.

The primary factors that must be taken into consideration in the formulation of restoration strategies for both the Garry Oak ecosystems and the Antelope Brush ecosystems are fire and invasive species and the interaction between these. The successful implementation of both fire and invasive species management rests on the knowledge of how these elements will operate in the restoration environment. For instance, re-introduction of fire in an ecosystem where fire once played a role but has not for many years could be disastrous as the fuel loadings may be much greater than when fire was regularly applied. Similarly, the occurrence of the highly flammable alien invasive species cheatgrass in the Antelope Brush ecosystems may cause fires to behave unpredictably. In both cases, some accommodation must be made to allow fire to be re-introduced safely.

Details of the use of fire in the restoration of the Somenos Garry Oak Ecosystems that address the following factors can be developed:

- Increased fuel loading due to a lack of fire and the invasion of the ecosystems by woody species;
- The response of native and non-native invasive species to the re-introduction of fire;
- The effect of fire on species at risk that may be present; and

- The incorporation of an adaptive management approach to the design of restoration activities so that improvements in the treatments maybe achieved in future years.

Elements that must be considered in the development of detailed strategies to address invasive species in the restoration of the Somenos Garry Oak Ecosystems are:

- The presence of increased biomass associated with invasive species must be addressed in any restoration prescription;
- The role of invasive species in changing the nutrient dynamics of Garry Oak ecosystems must be accounted for in the design of restoration works;
- The competitive effects, both above and below ground, of invasive species must be considered in the design of restoration programs; and
- The response of seed banked species to proposed restoration treatments.

The first step in the design of an initial strategy for restoration of the Somenos Garry Oak Ecosystems is to document in detail the floristics of the areas where treatments are to be undertaken. It is recommended that a minimum of 10 sample plots be established in each of the polygons where treatments are proposed. These can serve as baseline measures against which the treatment effects can be determined. Ten sample plots is believed to provide sufficient coverage to accommodate the within polygon diversity and thus be able to be used for statistical comparisons following treatments (Andrew MacDougall, 2002 pers com).

Sampling the proposed treatment plots must be undertaken prior to treatment yet late enough in the season that the maximum number of species can be identified. Sampling in late April or early May is expected to be best as this will include winter annuals while allowing bulb plants and other perennials to start to emerge. The timing for sampling in the Antelope Brush ecosystems needs to be determined on the basis of the proposed treatments in these ecosystems and the floristics of the communities being treated.

The initial treatments within the Somenos Garry Oak Ecosystems must reflect the uncertainty of any restoration activities in these communities. Therefore only a portion of one community will be treated. Garry Oak stands where snowberry has established due to the lack of fire have been selected as the initial community for treatment. Six polygons from the TEM mapping are suggested for treatment (21, 23, 24, 25, 27 and 28). A Treatment design is planned that includes:

- a control (untreated) area;
- an area where a single mowing of snowberry is conducted;
- an area where a single mowing is combined with a single burning;
- an area where a single mowing is combined with two periods of burning; and
- replanting a portion of the area that has been mown and burned twice with native grasses and forbs.

Timing of the treatments can have a major influence on the outcome. The proposed schedule for the 2003 season is to mow the brush in three plots where mowing is immediately after the baseline sampling in late April or early May. The initial burning would be conducted once the majority of the "wildflowers" were through flowering and seeds had been set. Traditionally burning by First Nations was conduced after the harvest of the camas which occurred after flowering (B. Beckwith, pers com). It is expected that this first burning would occur in June. The second burning would be scheduled for late summer or early fall, just before the onset of winter rains. Replanting would occur after the second burn.

Monitoring the response to the treatments is an essential part of the restoration process. Assessment of the plots is planned for immediately preceding the first burn and again just before the second burn. Additional plot monitoring is scheduled for the late winter to see what in the way of winter annuals appears in the different treatment plots and again in late April or early May to compliment the baseline sampling that is planned for before the treatments. Additional monitoring may be scheduled if the results of the planned monitoring programs indicate a need. The results from the monitoring will be used to develop the treatments for the year after the treatments.

Responses of invasive species as well as native species to the various treatments will be used in the formulation of further treatments. Eventually, once a desired steady state has been reached routine management activities will be conducted to maintain the ecosystems in the desired state. For the Somenos Garry Oak Ecosystems, this could consist of alternating patterns of burning and camas harvest, possibly involving local First Nations groups.

CONCLUSIONS

Restoration of ecosystems where major ecological processes (fire) have been precluded for many decades and where invasive species have been introduced is far more difficult than just leaving the site alone and allowing natural recovery. Active removal of invasive species as well as the re-introduction of fire at suitable intervals can form the foundations of the restoration program, but the details of the work will have to be developed with experience from the site specific treatments. Using an adaptive management approach to the restoration of both the Garry Oak and the Antelope Brush ecosystems can provide a suitable pathway in the face of uncertainty.

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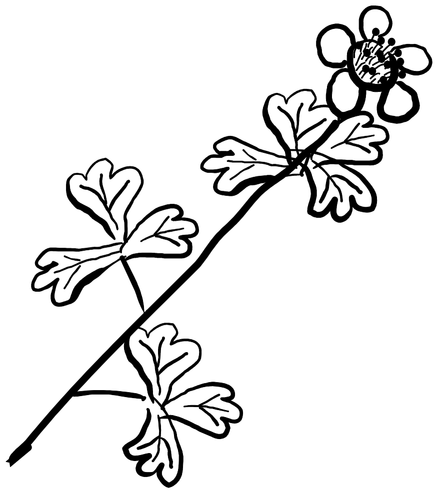

Action for Antelope-brush: a South Okanagan Stewardship Project

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ABSTRACT

An Antelope-brush Stewardship Project was conducted in the South Okanagan from 1999-2002. The South Okanagan-Similkameen Stewardship Program spearheaded the project. The aim of the project was to coordinate with other local groups and organizations, to raise awareness of the endangered plant community in a multi-faceted, cooperative manner. During the first phase of the project, informative materials were developed, including a two-page fact sheet on the antelope-brush habitat, a wall poster and a portable display board. More recently, a six-page colour fact sheet focusing on butterflies of the antelope-brush community and a butterfly checklist were completed.

Initially, the focus of the project was to raise general awareness of this endangered plant community through media releases, slide show presentations and distribution of the fact sheet and other literature. During the second phase of the project, terrestrial ecosystem mapping data was used for ground-truthing remaining antelope-brush habitat on privately owned land. Current condition of the habitat and potential threats were determined. Results were utilized to identify properties consisting of, or located within, a continuous band of antelope-brush habitat at least 20 hectares in size. Based on this information, a landowner contact program was initiated which was comprised of an initial mail-out, followed by phone contact and site visits upon request. Landowners were encouraged to understand both the value and fragility of this plant community. They were also advised of conservation options including habitat protection, restoration and securement. A site visit by a biologist provided an opportunity for landowners to learn more about the plants and wildlife occurring on their properties, as well as management options to enhance or restore the antelope-brush habitat.

In the future, we intend on promoting habitat sensitive development where appropriate and will continue to pursue higher levels of stewardship commitment. We also hope to explore opportunities to collaborate on an antelope-brush strategy involving both private and Crown lands.

INTRODUCTION

Although less than six percent of British Columbia is privately owned, this small percentage of land tends to coincide with the richest regions for biodiversity (Penn 1996). In the South Okanagan-Similkameen area, a large percentage of habitat that is critically important for wildlife is on private land. The ecological fate of that habitat depends upon the voluntary stewardship of landowners (Sandborn 1996). Conservation of the many species and habitats at risk in the area must focus on private land stewardship, since it paves the way to securement of large tracts of important habitat. It is cost-effective and provides the next best alternative to acquisition.

From 1994 to the present, the South Okanagan-Similkameen (SOS) Stewardship

Program has been fostering conservation partnerships, raising awareness, demonstrating sustainable land management practices, and offering a spectrum of land securement options, in collaboration with land trusts. The goal of SOS Stewardship is to promote conservation stewardship with landowners and managers to achieve the preservation and enhancement of Red- and Blue-listed wildlife species, plant communities and their habitats on private lands in the program area. SOS Stewardship is a local, community-based organization, working in cooperation with both government and non-government organizations, and is integrated into the stewardship arm of the South Okanagan Similkameen Conservation Program (SOSCP). The Land Conservancy of BC (TLC) has taken the lead role under the stewardship arm of SOSCP and, as such, has been involved in many elements of SOS Stewardship's work.

Initially, the emphasis of stewardship activities was placed on private lands buffering conservation holdings and on other priority areas of grassland and riparian habitat. However in 1999, SOS Stewardship recognized the urgent need to raise awareness of the endangered antelope-brush habitat by targeting a broader spectrum of landowners and the public. Consequently, a three-year community stewardship project was initiated in the south Okanagan.

METHODS

The antelope-brush stewardship initiative, aptly named "Action for Antelope-brush", aimed to raise awareness of the plant community in a multifaceted and cooperative manner. SOS Stewardship coordinated with other local groups and projects in an attempt to facilitate partnership and inter-agency cooperation. Initially, the focus of the project was to reach out to a broad spectrum of the community and raise general awareness of this endangered plant community. However, the project advanced to include both a public awareness campaign and a landowner contact program.

During the first phase of the project, important marketing tools were produced, including a two-page fact sheet on the antelope-brush habitat, a wall poster and a portable display board. The fact sheet and poster were distributed to libraries, schools, conservation organizations and government offices throughout the South Okanagan. Later on in the project, a six-page colour fact sheet focusing on butterflies of the antelope-brush community and a butterfly checklist were completed. Butterflies were identified as an important component of the ecosystem and provided a way of capturing people's attention. To broaden the scope of the public awareness campaign, biologists presented slide shows to municipal government and local organizations such as naturalist clubs. Public interest in the project also resulted in a small, informal workshop with interested landowners, to discuss the flora and fauna of antelope-brush communities, enhancement techniques and long-term securement opportunities.

During the second phase of the project, we embarked on a landowner contact program specific to those tenure holders with significant tracts of antelope-brush habitat. The contact program aimed to encourage sustainable land management practices and instigate long-term habitat protection initiatives, as a complement to land acquisitions. Terrestrial ecosystem mapping (TEM) data was used to determine the approximate extent of antelope-brush habitat in the south Okanagan. The TEM data was overlaid with land status information to identify the privately owned parcels of land that support antelope-brush habitat. Due to the ongoing loss of habitat on privately owned land and the need to confirm the accuracy of the mapping at a relatively finite scale, the privately owned antelope-brush habitat was ground-truthed. The site checks additionally provided an opportunity to assess the current condition of the habitat and identify potential threats, such as invasion by noxious weeds. Results were utilized to identify properties consisting of, or located within, a continuous band of antelope-brush habitat at least 20 hectares in size. Based on this information, a landowner contact program was initiated.

An introductory letter and a copy of the antelope-brush fact sheet were mailed to priority landowners. The letter was followed by a phone call, which served the primary purpose of arranging a site visit or "walk-about". Site visits were conducted by a biologist who was familiar with local plants and wildlife and could identify important habitat features. The walk-about provided landowners with an opportunity to learn more about the important ecological value of their property, as well as management options to enhance or restore the antelope-brush habitat. They were also advised of conservation options including habitat protection, restoration and securement. Soon after the site visit, follow-up letters were sent to landowners to provide any additional information requested and to reinforce the positive experience of the visit. Long-term follow-up has been based on the landowner's individual needs. Contact has been maintained with landowners who indicated an interest in long-term securement or had other interests/concerns requiring additional information. Technical advice has been provided on enhancement and educational projects, including one demonstration site at a local vineyard. The stewardship program has also arranged for youth crews or summer workers to assist landowners with manual weed control in antelope-brush communities.

Local elementary and secondary schools were also enthusiastic about the project and requested presentations. Discussions ensued after the talks and two high schools were encouraged to embark on demonstration projects. Under the guidance of stewardship biologists, students at Osoyoos Secondary School developed and implemented an antelope-brush re-vegetation project. The students surveyed existing native plants, collected garbage, removed non-native plants, prepared the planting design and layout, and planted native shrubs, grasses and forbs. They completed their project by installing a permanent educational sign, which was jointly funded by the school and the SOS Stewardship Program. At South Okanagan Secondary School, biology students mapped the native plants, identified disturbances, examined soils and developed a stewardship plan for a parcel of Crown land adjacent to the school. The project expanded into an enhancement project (STAB – Save the Antelope-brush) being undertaken by the school's Environmental Club, with direction and technical advice being provided by stewardship biologists. The demonstration project is still underway, with plans to re-vegetate the site in spring 2003.

RESULTS AND DISCUSSION

The educational materials were well received and widely distributed (17 venues). Several locations have made requests for additional fact sheets.

The success of the public education component of 'Action for Antelope-brush' is clearly evident through the high school demonstration projects. Both the students and teachers responded enthusiastically to the presentations, and biology teachers are exploring opportunities to incorporate grassland ecology and antelope-brush conservation into the curriculum. The demonstration projects at both high schools have provided a hands-on opportunity for students in the area of stewardship and environmental biology that will have a lasting impact on the communities.

Throughout the duration of this community-based initiative, the local media was advised of the project and produced several articles. The antelope-brush project was also profiled in newsletters produced by several different organizations. Favourable media exposure appears to have increased the positive response to the project.

The landowner contact component is also deemed highly successful. A total of 114 landowners were contacted by mail, with 55 receiving a follow-up phone call. This resulted in 24 site visits with a biologist. These landowners represent an estimated 200 acres of antelope-brush habitat. Nearly half of the landowners that were visited requested information on long-term securement options. During the second year of the landowner contact program, two-thirds of the landowners expressed an interest in pursuing an antelope-brush re-planting project. Relationships with program participants have been strengthened by maintaining contact with the landowners and advising them of current and new conservation activities. This ongoing process has also helped to secure connections with the landowners and has undoubtedly resulted in greater commitments. Experience from several landowner contact programs has shown that once landowners become interested, they want to see the dialogue continue (Hilts and Mitchell 1994).

In the next phase of the project, we will continue to work with landowners to raise awareness of this endangered habitat and promote higher levels of stewardship commitment. Our approach will involve reaching out to new landowners, and fostering closer relationships with landowners who have expressed an interest in conserving the antelope-brush plant community in the long-term. To complement the landowner contact aspect of our program, we will continue to encourage and support community initiatives aimed at conserving antelope-brush habitat. Furthermore, we will explore opportunities to collaborate on a private-Crown land strategy for the preservation of this important habitat.

Although the above-noted stewardship efforts are effective, there is a need to further promote the value of habitat sensitive development to landowners, developers and land use decision-makers. Habitat sensitive development helps respect key natural features while minimizing disturbance to the landscape. An example of this alternate development approach is clustering development on the fringe of important habitat or areas of lesser conservation value. In this way, only a portion of the property is altered, while the remainder is retained in its natural state. It helps minimize the construction footprint on the land.

Local land use bylaws can include provisions for environmentally sensitive areas. It is encouraging to see that in some official community plans, the Regional District of Okanagan-Similkameen has designated important habitat within environmentally sensitive development permit areas (ESDPA) and that some guidelines are in place to support habitat sensitive development. However, on agricultural lands, in particular lands within the Agricultural Land Reserve, agricultural uses take precedent over other land uses or other values including important habitat and species at risk. Apart from promoting habitat sensitive development, there is very little we can do to protect important habitat such as antelope-brush where intensive agricultural uses such as vineyards are planned.

When an official community plan is under review, there are opportunities for public input in the planning process. Landowners can make a difference by encouraging the creation of environmentally sensitive development permit areas for important habitat and supporting guidelines. Please become involved. We can't afford to lose any more antelope-brush habitat.

ACKNOWLEDGEMENTS

We wish to thank Anthea Bryan for her review of this paper. Support for the project has been provided by the Habitat Conservation Trust Fund, Vancouver Foundation, The Real Estate Foundation of British Columbia, *TLC* The Land Conservancy of BC, the Okanagan Region Wildlife Heritage Fund Society and the Government of Canada Habitat Stewardship Program for Species at Risk.

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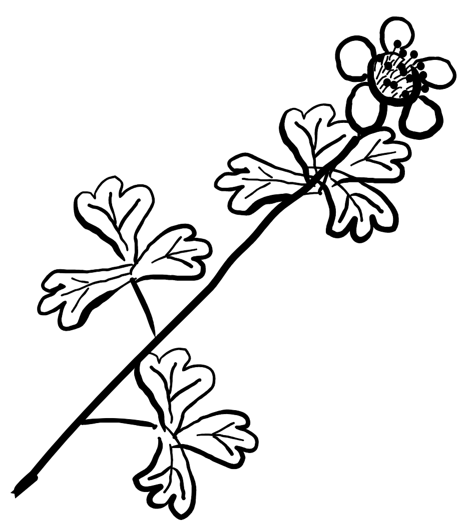

Great Antelope Brush Hunt and Lessons on Niche

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INTRODUCTION

A few years ago, members of the British Ecological Society were asked to list and rank the 10 most important ecological concepts. Importance was not defined, so could be most theoretical, practical, innovate, controversial, fundamental. The results were compiled and ranked into the top 50 concepts (Cherett 1990). The concept of "niche" ranked number 6. Number 5 was "competition," with which niche is intimately associated. Number 1 was the concept of ecosystem, which is also intimately related because species and species niches bind the living parts of ecosystems together. In this paper, we address the ecological niche of antelope brush (*Purshia tridentata*).

Restoration of any species requires an adequate understanding of its niche, that is, its functional role in an ecosystem. Niche theory provides a basic tool for understanding both physical and competitive environmental constraints that together define a species' role, including where it may live – "may" live, not necessarily "will" live, because environmental stochasticity, that is, pure chance, and historic accident, also have important influences.

In summer 2002 we evaluated the niche of antelope brush by searching for it across the Columbia Plateau in Washington State and the High Desert portion of the Great Basin in Oregon. We also conducted a literature search to identify habitats where it lives farther south, down to the edges of the hot deserts.

GEOGRAPHICAL PERSPECTIVE

Table 1 lists the locations, elevations, and community associated species where we found antelope brush. Obvious from that Table is that antelope brush has a broad geographic range and lives at a variety of elevations and sites from valley floors to high plateaus, and with a variety of associated species from bunchgrasses and sagebrush to open forest with coniferous trees like ponderosa and lodgepole pines, juniper, even deciduous trees such as white oak.

We did not follow antelope brush farther south – that comes this summer – but from ecology texts, most notably the classic work of Shelford (1963), we know we will find it, sporadically, rarely very abundantly, throughout the rest of the cold desert of the Great Basin in Nevada, Utah, and extreme western California, and even on the Colorado Plateau in northern Arizona and New Mexico, southwest Utah, and western Colorado. There, it lives at high elevation, because it has to – all the land is above 4000 feet in the cold desert portion of the Great Basin, or the Basin and Ranges as it is also called, and the Colorado Plateau is even higher yet. Where the lands drops down to the hot deserts of Mohave, Sonoran, Chihuahuan, antelope brush is gone, replaced by shrubs like creosote bush (*Larrea divaricata*) and black bush (*Coleogyne ramosissima*).

Table 1. Antelope brush sites examined in B.C., Washington and Oregon, summer 2002.

Location	Site	Topography	Elevation (ft)	Shading	Substrate	Co-dominants
Okanagan Valley	valley floor	flat/gently rolling	900-1500	none	till	bunchgrass, sagebrush
Okanagan Valley	valley floor	road cuts, disturbed	900-1200	none	till	none, or bunchgrass
Okanagan Valley	lower slopes	gentle to steep slopes	1200-3000	partial	rock, till, loess	bunchgrass in ponder/ D. Fir
Columbia Plateau along Colum. R.	aspect-shaded	steep slopes	800-900	partial	till	bunchgrass, sagebrush
Columbia Plateau benches above Columb. R.	plateau	flat/rolling	2500	partial		bunchgrass, white oak
Great Basin, Hart Mt. Antelope Rg.	high plateau	flat/gently rolling	5750	none		50% sagebrush, scattered juniper
Great Basin, Paulina Mts.	plateau	flat/rolling	4500-4800	partial		some to no bunchgrass some to no sagebrush
Great Basin, S side Blue Mts.	slopes	rolling	4000-4500	none		Ponder/planted lodgepl., bunchgrass sagebrush, scattered juniper
Great Basin, S side Strawbr. Mts.	slopes	rolling	4200			bunchgrass, sagebrush
Great Basin, N of Silver L.	plateau/slopes	rolling	4600	none		sagebrush, green/grey rabbitbr. no grass, bare soil

We had some good clues where to find antelope brush, at least on the Columbia Plateau, from coloured maps in a book by O'Connor and Wieda (2001). On both maps, one depicting historic (pre 1880s) and the other depicting current vegetation types in the Columbia Basin Ecoregion, you have to look hard to find it. When you do, you realize that both historically, and presently, antelope brush-dominated shrub-steppe was, and is, rare. It is found primarily on the periphery of the Plateau, and the only place it thrives out on the Plateau itself is sporadically along the Columbia River. And that provides the first clue about the niche of antelope brush – not so much where it is but where it isn't. It is not, and perhaps never was, on the wide-open, hot expanse of the Columbia Plateau, where it is hottest and driest.

SOIL MOISTURE AND INTERMONTANE PLANT FORMS

In the intermontane west, plant adaptations are dominated by responses to moisture deficiency – its low amounts, seasonal droughts, and high rates of evapotranspiration. Bunchgrasses dominate instead of more moisture-adapted rhizome grasses of the central prairies; leaves of shrubs are small and deciduous – there is a host of similar moisture-conserving devices (Archibold 1995). Schematically, the major groupings of plant taxa relate to soil moisture as shown in Fig. 1. Note that Fig. 1 relates to only the intermontane west. The ecological position of shrubs in other ecosystems, such as the arctic, is entirely different. Shrubs in the intermontane, with deeper roots than grasses, can withstand greater moisture deficiency especially through the growing season, so long as sufficient moisture is stored in the soil during the non-growing season. But among shrubs are physiological differences, of course, that result in different ecological niches. Fig. 2 depicts some of those differences, with the most xeric-tolerant shrubs being those that are salt-adapted, where osmotic pressure works against them. Antelope brush, no one would doubt, generally requires moister conditions than sagebrush – that is why so much of the open, exposed Columbia Plateau and cold desert grows sagebrush, not antelope brush.

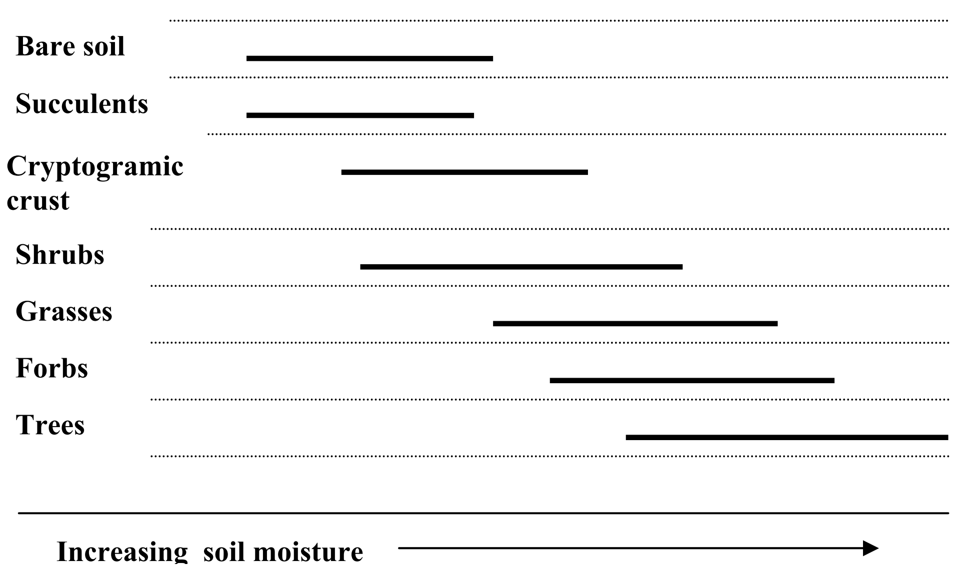

Figure 1. Plant-form relationships to soil moisture in the intermontane west.

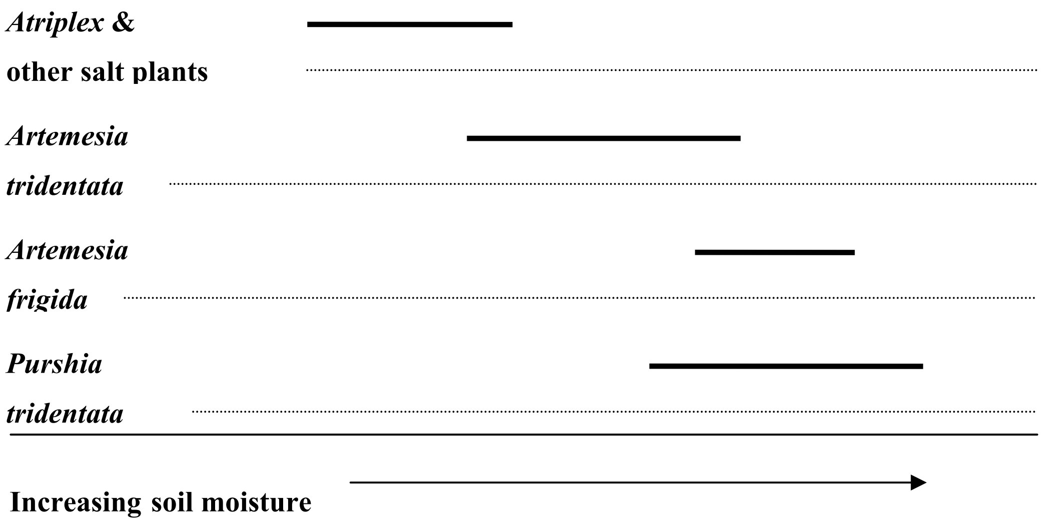

Figure 2. Soil moisture relationships among intermontane shrubs.

NICHE

The concept of "niche" has been defined in various ways (Schoener 1990), among which is a separation into both "fundamental" and "realized" niche. This bipartite way of thinking about niche provides considerable illumination when considering the ecological requirements of any species. Physical constraints set the breadth of the "fundamental niche," defined as "the total range of environmental conditions under which a species can survive" (Smith 1992). Among possible physical constraints, soil moisture is the limiting physical factor in most characteristically dry ecosystems and the most likely primary candidate for antelope brush. That is not to say that soil nutrients are unimportant. However, soil nutrients are not commonly limiting by themselves, as they normally are in tropical and tundra ecosystems (Brewer 1994).

In defining the niche of a species, there is a tendency to focus on physical constraints and forget the great importance of competition. Rarely, however, does a species live across the whole span of its fundamental niche. Instead, competition narrows the fundamental niche to a lesser "realized niche," "the portion of the fundamental niche space occupied by a population in the face of competition from populations of other species" (Smith 1992). Competition is especially relevant because the fundamental niches of species often overlap in an ecosystem.

Fig. 3 depicts the fundamental niche of antelope brush defined by soil moisture, and its realized niche constrained by its two great competitors, sagebrush and grasslands. The diagram is schematic because the fundamental niches of sagebrush and grasslands overlap in their moisture requirements. Antelope brush finds both its fundamental and realized niche within their competitive zones.

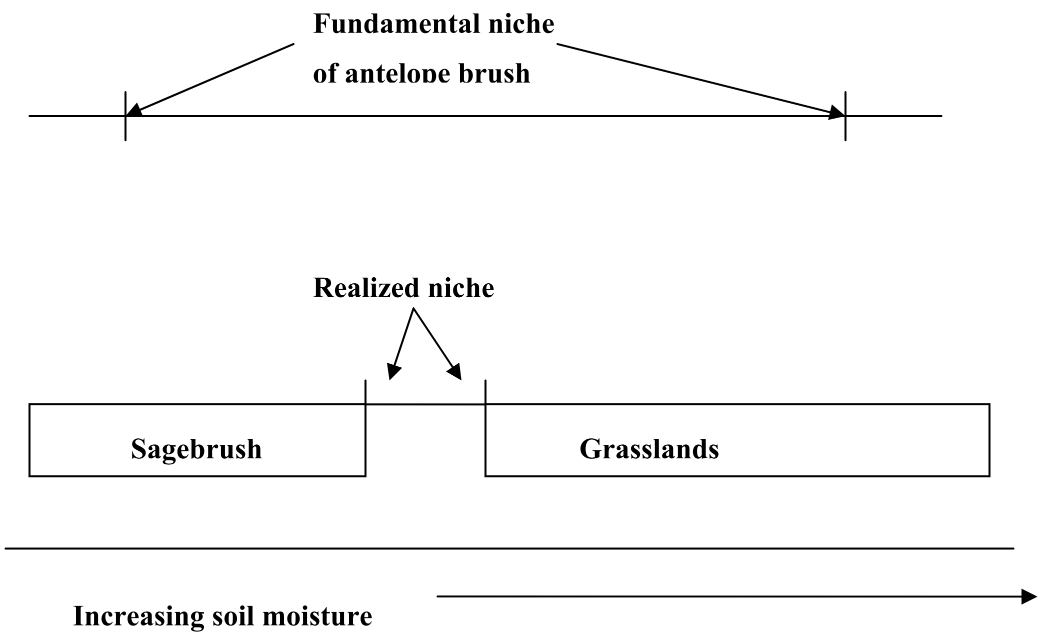

Figure 3. Schematic diagram to show the relationship between the fundamental niche of antelope brush (top line), which is constrained by extremes of soil moisture, and the realized niche, which is constrained by competition from sagebrush and grasslands. In reality, the moisture regimes for sagebrush and grasslands overlap, leaving the realized niche for antelope brush totally within their competitive zone.

BASIC SHRUB-STEPPE NICHE MODEL

Throughout the intermontane west, except in the hot deserts, a great ecological war is waged between grasslands and shrublands, the latter principally *Artemesia tridentata*. In Fig. 4 we describe this relationship. In Fig. 4a, abundance on the Y axis is the dependent variable, with soil moisture on the X axis

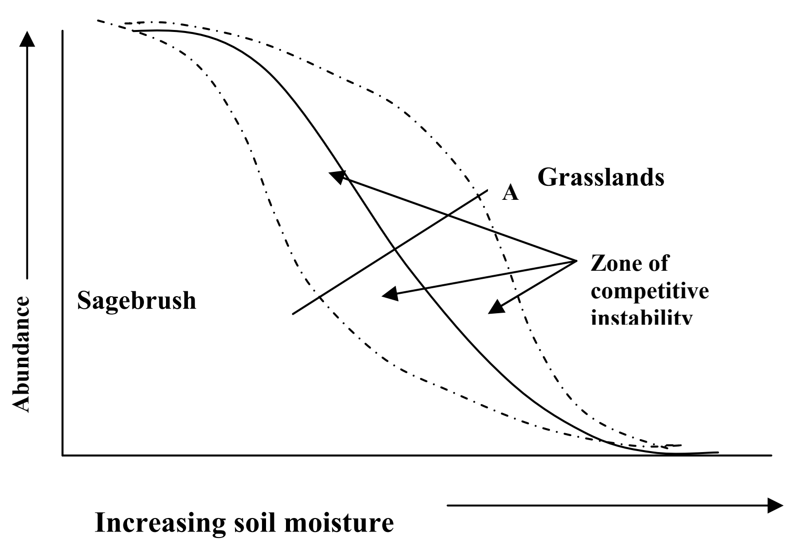

Figure 4a. Moisture relationship between sagebrush and grasslands showing fundamental niches of both, to left and right of dashed lines, respectively, and zone of competitive instability between the dashed lines, within which falls the realized niche of both.

as the independent variable. The solid curve depicts the mean separation position between sagebrush on the left side of the diagram and grasslands on the right. The dotted lines depict the zone of competitive instability, or what can be thought of as the "war zone," which represents the overlapping fundamental niche of both types of vegetation, where microsite, history, stochastic variables and disturbance features all influence which grows there. This zone is elliptical at both ends reflecting the reducing competition until one plant form takes over. The line labelled "A" transects this zone at its widest point, where the competitive advantages are approximately equal; this line serves as a reference for Fig. 4b.

Figure 4b shows the cross section through A. Viewing it requires flipping your mind obliquely through 90 degrees, (or lying on the floor and looking at the screen on an angle, or some such contortion). Anyhow, the Y axis represents competitive advantage and the X axis represents soil moisture again. The dotted vertical lines are the same bounds of the zone of competitive instability or the "war zone." The right-handed sigmoid curve represents the left margin of the fundamental niche of grassland, and everything to the right of it, beyond the war zone, represents soil moisture suitable for grasslands and not sagebrush. The left-handed curve represents the right margin of the fundamental niche of sagebrush, and everything to the left of it, beyond the war zone, represents soil moisture suitable for sagebrush and not grasslands.

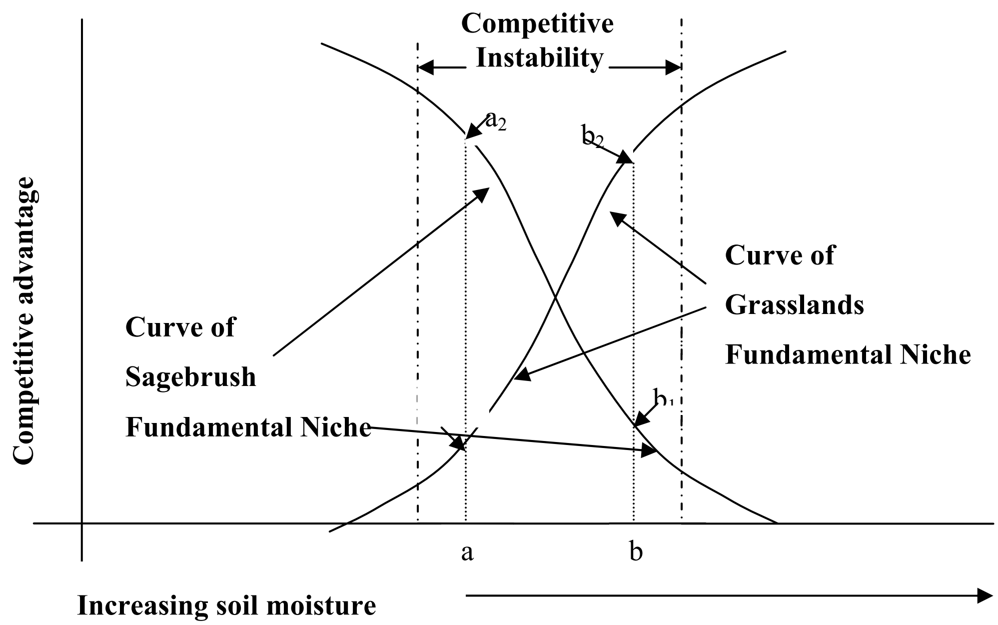

Figure 4b. Relative competitive advantage of sagebrush and grasslands related to moisture, corresponding with line A in Fig. 4a. The fundamental niche for each is described as a curve. Vertical dashed lines show the zone of competitive instability where competition constricts the fundamental niches into realized niches for both sagebrush and grasslands. Relative advantage at soil moisture level "a" is the lengths of line a – a1 for grasslands compared to line a1 – a2 for sagebrush. At moisture level "a," sagebrush has an advantage. Correspondingly, at greater moisture level "b", the relative advantage is the length of line b – b1 for sagebrush compared to b1 – b2 for grasslands. At moisture level b, grasslands have an advantage.

Within the war zone, vertical line "a" shows the expected ratio of grasslands to sagebrush at that level of soil moisture. Similarly, vertical line "b" shows the expected ratio of sagebrush to grasslands at that level of soil moisture. There is a likelihood of more sagebrush than grasslands at soil moisture level a, and the reverse at the greater soil moisture level b.

ANTELOPE BRUSH AND THE BASIC MODEL

In Fig. 5 we reproduce Figure 4a and 4b but bring in antelope brush at cross-section point B. Antelope brush generally requires more soil moisture than sagebrush and less

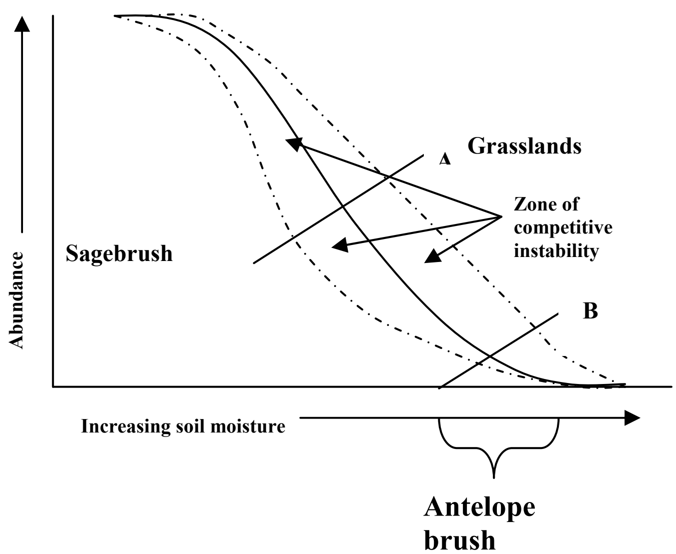

Figure 5a. Position of antelope brush, line B, situated at the moister end of the zone of competitive instability between sagebrush and grasslands.

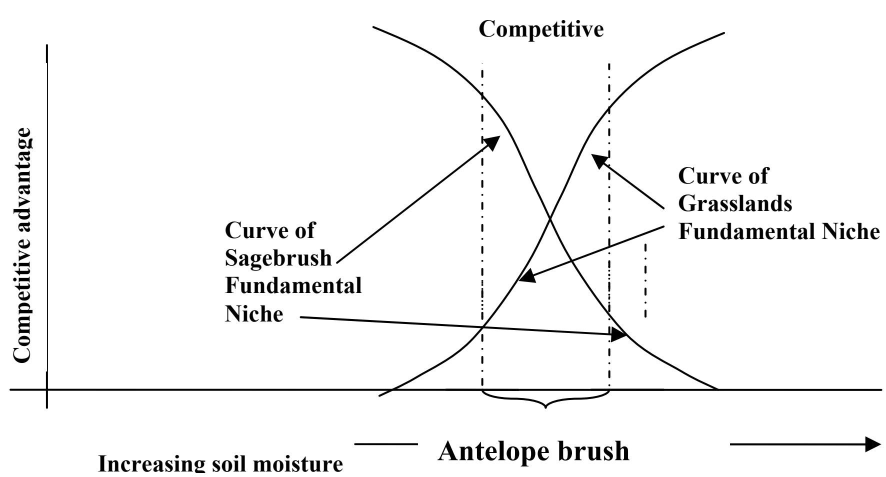

Figure 5b. Position of the fundamental niche of antelope brush, indicated by horizontal bracket on the X axis, corresponding to line B in Figure 5a. Curves are offset to the right, in comparison to Figure 4 b that is drawn through line A. Zone of competitive instability is narrower here as moisture allows grasses to tend to outcompete sagebrush. Antelope brush's entire fundamental niche falls within the competitive zones of sagebrush and grasslands.

than grasslands, but of course with overlap in the war zone, that is, where grasses tend to be favoured over sagebrush. In this very narrow zone of soil moisture, the realized niches of the three overlap.

So, in summary, antelope brush has a fundamental niche at the moister extreme of the war zone between sagebrush and grasslands, and there it competes based upon microsite, historic, stochastistic variables and disturbance features.

SOIL MOISTURE AND SITE PREFERENCES

Soil moisture, the primary determinant of antelope brush's fundamental niche, is a function of many things. Fig. 6 describes the main determinants: precipitation (both amount and seasonality), evapotranspiration (which depends on temperature, wind, plant cover), slope, aspect, soil. Focussing for a moment on soil, antelope brush is not particular, as long as that soil holds moisture reasonably well, which is a function of "fines" such as silt and clay, and as long as the soil can be still classes as either mesic or meso-xeric (not hydric). If there are enough fines even in moraine gravels, it will grow there, but it grows well in pure loess, such as exists in mountainside pockets in the south Okanagan even up to 1000 metres in elevation, and in very shallow soils among rocky crags. Soil salinity, however, is detrimental because of the difficulties it presents for moisture uptake by rootlets.

Within this fundamental niche, competitive advantage that defines the realized niche of antelope brush – increases its probability of winning in the war zone with grasses - include anything that weakens grasses, such as:

- cattle grazing and disturbance by hooves,
- bulldozers – it shows up in roadcuts,
- slope and aspect – it grows on steep slopes running along the Columbia

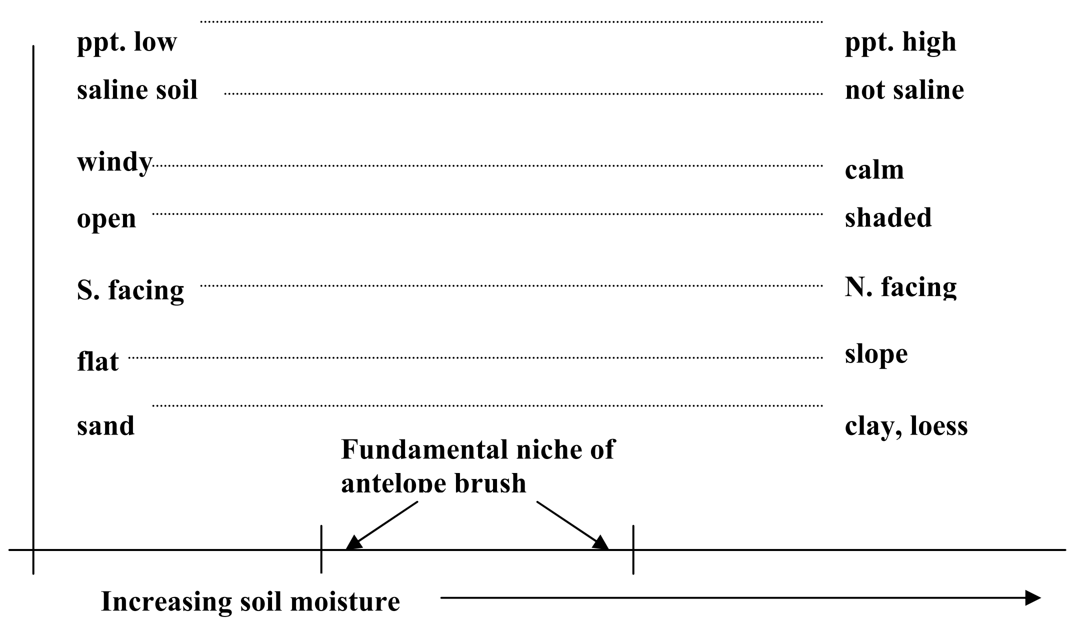

Figure 6. Some determinants of soil moisture influencing the fundamental niche of antelope brush.

River where for part of each day it is in shade,

- higher elevations around the edges of the Columbia Plateau where cooler temperatures increase precipitation and lower evapotranspiration, and result in open forest growth,
- partial shade, which weakens grasses, such as in the ponderosa or the lower ponderosa-Douglas fir biogeoclimatic zones,
- rocky places, where shallow soils dry out and stress grasses, but where deep roots of antelope brush may tap deeper moisture.

On the other hand, as environments favour grasses with more moisture, cooler temperatures and hence lower evapotranspiration, gradually antelope brush is out competed and falls out. That happens in the Okanagan Valley near Penticton – antelope brush is rare north of that.

Competitive advantage that increases the probability if antelope brush winning over sagebrush, resulting in less of a dead-heat competition, as explained, includes anything that knocks back sagebrush, such as:

- fire – but that is a complex topic, because both sagebrush and antelope brush are susceptible, favouring grasses, but antelope brush, unlike sagebrush can reproduce vegetatively and so recover better in some ecological conditions (Whitney 1989).
- heavy ungulate grazing – because antelope brush is favoured by a variety of species (Whitney 1989).

PLACE IN SPACE

The concept of niche is sometimes thought of in terms of multidimensional hypervolume contained within a suite of axes (Hutchison 1957). We have difficulty thinking in more than 3-dimensions, which leaves multidimensional hypervolume as only a vague concept. If antelope brush was considered in terms of only 3-dimensions,

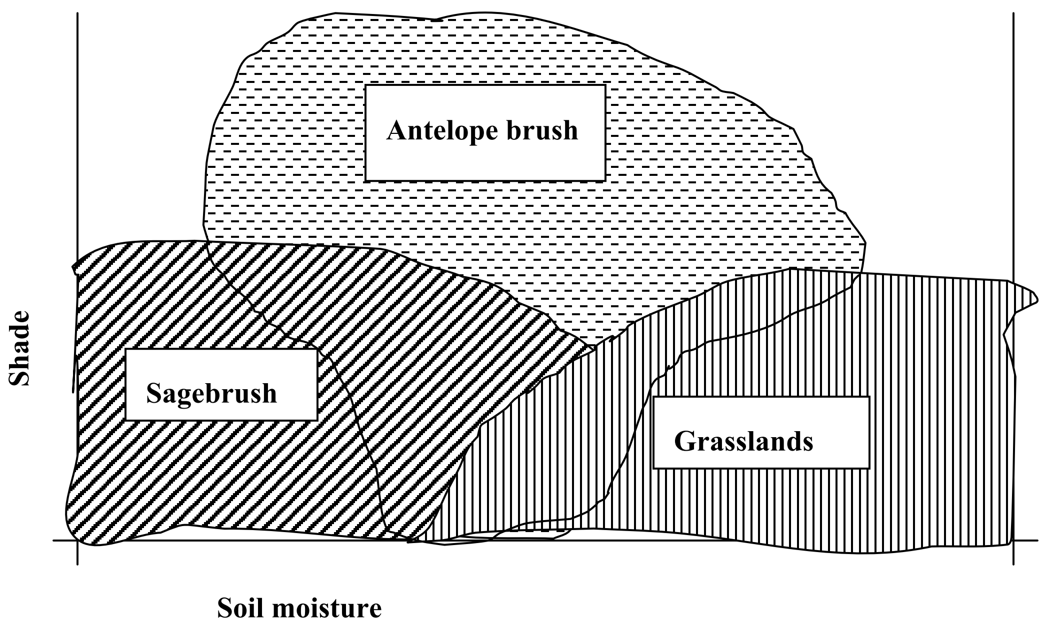

Figure 7. Ordination to show the position of antelope brush relative to sagebrush and grasslands when viewed within axes of soil moisture and shade.

however, one axis must be soil moisture, as discussed. Next in importance is likely shading, because where trees reduce sunlight on the forest floor, antelope brush is excluded. Shading, as forests become less open, may be most important in setting the upper altitudinal bounds of antelope brush on mountainsides throughout much of its range. Thirdly, temperature may form a third axis, relating directly to the plant's physiology, but any understanding of its role is confounded by the indirect effect of temperature on evapotranspiration and on tree growth and shading. So, we will ignore temperature and fall back on just 2 dimensions, soil moisture and shading. Fig. 7 shows the place of antelope brush in an ordination of just these two environmental variables, and relates it to sagebrush and grasslands. As is depicted, antelope brush has space in relatively mesic soil moisture conditions where shading restricts especially sagebrush – although in the South Okanagan, *Artemesia frigida* is capable of growing with antelope brush in more shaded conditions than is *Artemesia tridentata*.

ANTELOPE BRUSH AS TROPHIC CONDUIT

The concept of niche, defined as species role in an ecosystem, must include beneficiaries, that is the species to which it passes energy and nutrients in the next trophic level. Those plant features of antelope brush that are found attractive by other species include:

- seeds, which provide high-energy packets for small animals with the highest energy-to-biomass demands – great basin pocket mouse, yellow pine chipmunk, deer mouse, pocket gopher;
- roots, which as storage organs for water, nutrients and energy also provide high-quality forage – pocket gopher;
- bark, with nutritious inner bark and cambium – Nuttall's cottontail, white-tailed jackrabbit, black-tailed jackrabbit;
- buds and twigs, with their nutrient concentrations – mule deer, pronghorn, elk, white-tailed deer;
- foliage, with a bulk of low energy suitable for the same large mammals that eat its buds and twigs.

Lark sparrows and other birds nest in its branches, western meadowlarks, western and mountain bluebirds use it as perches, pallid bats forage for insects living around it.

Antelope brush is not an obligate for any vertebrate species, although it appears to be for the larva of at least one invertebrate, Behr's hairstreak (*Satyrium behrii*). *Artemesia* species provide most of the same energy and nutrient pathway over a much greater landmass in the intermontane west. However, being ecotonal between shrub-steppe and forests in its strongholds around the perimeter of the Columbia Plateau, High Desert and the southern Okanagan Valley, it is positioned to cater to a greater biodiversity than either the extensive sagebrush lands or grasslands that predominate across most of the shrub-steppe environments. In such an "ecological crossroads," antelope brush falls into a recently identified priority environment for conservation action (Spector 2002).

SAVING ANTELOPE BRUSH ECOSYSTEMS

We have tried to explain just where antelope brush fits ecologically in the physical and competitive dryland environments of the intermontane west. So what? Of what possible use is understanding these ecological relationships? We stated at the outset that "restoration of any species requires an adequate understanding of its niche, that is, its functional role in an ecosystem."

What can we do with this information, accepting, as we must, the complexity of basic antelope brush niche ecology depicted in the figures we have presented? Not model it, not even map it from remote sensing, because there is too much complexity, too many interacting influences, too many subtle relationships. Antelope brush is too ecologically squeezed between its two great adversaries, its claim on the land too tenuous, its existence too preconditioned by historic accident, human alterations of the land, and chance events.

Appropriate management of antelope brush ecosystems is where science should give way to philosophy. This is where we put aside scientific arrogance that we can manipulate and restore, and recognize that we will achieve much more success in perpetuating the antelope brush ecosystems if we spend available money on protecting what we have, not letting it go under vineyards, getting the ecological menace of livestock off it, leave the wildlife alone, let the land where it still grows heal, and hope it will.

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