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Plant community changes after the reduction of an invasive rangeland weed, diffuse knapweed, *Centaurea diffusa*

Andrea E.A. Stephens [a,*], Pam G. Krannitz [b], Judith H. Myers [a]

[a] *Department of Zoology, University of British Columbia, 6270 University Blvd., Vancouver, British Columbia, Canada V6T 1Z4*
[b] *Environment Canada, Canadian Wildlife Service, 5421 Robertson Rd., RR1, Delta, British Columbia, Canada V4K 3N2*

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ABSTRACT

The expected outcome of weed control in natural systems is that the decline of a dominant weed will result in an increase in diversity of the plant community but this has seldom been tested. Here we evaluate the response of the plant community following the decline of diffuse knapweed (*Centaurea diffusa*) in six different pastures at White Lake, BC, Canada over five years. This period followed the establishment, spread and high levels of attack by the introduced European weevil, *Larinus minutus*, as part of a biological control program. Knapweed declined immediately before and during the study period, but, contrary to expectations, the species richness and diversity of the rangeland plant community did not increase. The absolute cover of native and introduced forbs and grasses increased following knapweed decline, but only the introduced grasses showed a consistent increase in cover relative to the other life-forms. However, unlike in other studies, the native plants dominated the study site. We conclude that the changes in plant communities following successful biological control are variable among programs and that the impact of replacement species must be evaluated in assessing the success of ecological restoration programs that use biological control to manage an undesirable weed.

1. Introduction

Control of invasive weeds is often proposed as a key element of ecosystem restoration (Harms and Hiebert, 2006) because it is assumed that the removal of the target weed will be associated with an increase in the abundances of species previously suppressed by the weed and an increase in diversity of the plant community. Thus, reducing plant dominance in a community should promote diversity (Lesica and Hanna, 2004); an increase in native species diversity is often considered to be a measure of success in restoration ecology (Ruiz-Jaen and Aide, 2005). Biological control can be a cost-effective method of achieving long-term weed control in natural ecosystems and there are a number of cases where this has been successful in reducing weed population density (Myers and Bazely, 2003; Myers, 2008).

Ding et al. (2006) pointed out that, for most weed biological control programs, the majority of pre-release funding goes to host-specificity testing and far less is available for assessing the impact of the agent on the weed's performance. In the post-release period, priorities are either the assessment of non-target impacts (e.g. Louda et al., 2003; Paynter et al., 2004) and/or documentation of the response of the weed (e.g. Grevstad, 2006; Hoffmann and Moran, 1991; Supkoff et al., 1988). However, few studies quantify the effects of weed reduction on plant community composition or ecosystem processes despite the fact that the restoration of natural communities and/or natural ecosystem processes is often the stated goal of restoration projects, and thus the primary motivation for biological control projects (Denslow and D'Antonio, 2005). For example, less than 1% of weed biological control programs in Australia assessed the effect that the reduction of the weed through successful biological control had on the wider plant community (Thomas and Reid, 2007). Given the long history of interest in the invasion of exotic plants into plant communities (Darwin, 1859; Elton, 1958; Stohlgren et al., 2008), the lack of interest in what happens when a former community dominant declines, is notable.

Evaluations that have been done tend to show that plant community responses are highly site specific (Denslow and D'Antonio, 2005). For example, successful biological control of purple loosestrife (*Lythrum salicaria* L.) by *Galerucella* spp. (Blossey et al., 2001) resulted in one of two outcomes: either monotypic stands of purple loosestrife were replaced by a diverse wetland plant community or other invasive species such as common reed (*Phragmites australis* (Cav.) Trin. ex Steud.) or reed canary grass (*Phalaris arundinacea* L.) expanded as purple loosestrife declined. This suggests that, in some sites, additional restoration techniques may be needed, as well as biological control, to achieve goals of community restoration and ecosystem processes.

* Corresponding author.
E-mail address: stephens@zoology.ubc.ca (A.E.A. Stephens).

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Diffuse knapweed (*Centaurea diffusa* Lam., Asteraceae) and spotted knapweed (*Centaurea stoebe* L. subsp. *micranthos*) are Eurasian weeds that are prevalent in dry rangelands in many areas of western North America. They share many common attributes and biological control agents, but occur in slightly different habitats (Bourchier et al., 2002). Diffuse knapweed has colonized up to 1.4 million hectares of rangelands in western North America, from Washington to Michigan and British Columbia to New Mexico (Seastedt et al., 2005). It is unpalatable to livestock and is able to outcompete a range of native plant species (LeJeune and Seastedt, 2001; Seastedt et al., 2005).

A biological control program for diffuse knapweed was initiated in 1970. Over the next 20 years, 12 insect species were introduced for the control of this species and nine of these are now common throughout knapweed infested areas of British Columbia (Myers, 2007). The most common insect species in British Columbia include the two species of gallflies *Urophora affinis* Frauenfeld and *U. quadrifasciata* (Meigen) (Diptera: Tephritidae), the root boring beetles *Sphenoptera jugoslavica* Obenberger (Coleoptera: Buprestidae) and *Cyphocleonus achates* (Fåhraeus) (Coleoptera: Curculionidae), the root boring moth *Agapeta zoegana* (L.) (Lepidoptera: Cochylidae) and the weevil *Larinus minutus* Gyllenhal (Coleoptera: Curculionidae).

Larinus minutus was the most recent of these agents to be introduced and established. Following its establishment, diffuse knapweed has declined in British Columbia (Myers, 2007; Myers et al., 2009) and Colorado (Seastedt et al., 2003, 2005, 2007). Larvae develop in the flower-heads and prevent reproduction of the plant while the adults feed on the stems and leaves. Under some conditions the adult weevils can kill the host outright (Myers, 2007).

Here, we report changes that were measured in plant community composition following the decline of diffuse knapweed through biological control in six different pastures at White Lake, British Columbia, Canada. Prior to the initiation of this study in 2001, knapweed had already declined from peaks in abundance following the establishment of *L. minutus* in the late 1990s (Myers, 2007; Myers et al., 2009). We test whether the expected outcome of weed biological control in natural systems occurs, i.e. that native communities will recover, by monitoring the vegetation in a series of permanent quadrats.

2. Materials and methods

2.1. Study site

White Lake Basin (Okanagan Falls) (49°19.18N, 119°37.82W) belongs to the National Research Council's Dominion Radio Astrophysical Observatory and is managed by The Nature Trust of British Columbia in cooperation with a local rancher. The management goals for this area include restoration of native vegetation and sustainable grazing intensities. The vegetation type here is sagebrush (*Artemisia tridentata* Nutt. and *Artemisia tripartita* Rydb.) shrub-steppe. The elevation of the basin is 550 m and the average annual total precipitation for this region (1971–2000) is 333 mm. The average precipitation for the growing season (1st May–30th August) is approximately 135 mm based on data from the Penticton A weather station (WMO ID 71889) (Environment Canada, 2002) which classifies the habitat as semi-desert. Spring rainfall (March–May) showed little variation over the period 2001–05 but summer rainfall (May–August) was highly variable with below average rainfall occurring during the summers of 2002 and 2003 and above average rainfall in 2004. Total annual precipitation for 2001–2005 was 320, 197, 281, 427 and 303 mm respectively while growing season precipitation was 154, 82, 49, 183 and 128 mm for the same years.

The knapweed invasion in this area was associated with disturbance of the land that occurred with the installation of a radio telescope at the White Lake Observatory in the 1960s. The research quadrats were distributed over a distance of several kilometers through the White Lake basin, to the south and west of the Observatory. Both *Urophora* spp. were present in the area, *S. jugoslavica* was introduced in 1976 and *L. minutus* was introduced in this vicinity in 1991, 1995 and 2000. *Cyphocleonus achates* and *A. zoegana* were present in low numbers (Myers, 2007).

2.2. Monitoring methods

In the spring of 2001, 422 permanent quadrats (20×50 cm; 0.1 m^2) were established. The quadrats were spaced approximately 10 m apart along haphazardly placed transects. There were three to nine transects and 50–90 quadrats per pasture for six pastures of 100–200 ha (Table 1).

Surveys were conducted in late May to early June of 2001, 2002, 2003 and 2005. In 2004 complete sampling was not done due to personnel constraints, and only quadrats that had previously contained knapweed were sampled. As this is a non-random subset, only the complete data set was used for most analyses.

Cover of all plant species was visually estimated to the nearest 1% in each quadrat and total cover of each quadrat was calculated as the sum of percent cover of every plant species in the quadrat. Absolute cover was recorded but the totals do not reach 100% because there are substantial amounts of bare ground in these pastures. Relative cover (absolute cover of a plant species divided by total cover of all plant species) was calculated because substantial variation in rainfall influenced total plant cover between years with bare ground diminishing during wet years.

2.3. Statistical analyses

All statistical analyses, described below, were conducted using R (R Core Development Team, 2008). Both absolute and relative plant cover data were used to investigate how species diversity and the abundance of plant life-forms changed in these pasture communities during the period of knapweed decline.

2.3.1. Knapweed cover

Absolute knapweed cover in the pastures was determined by taking the average of all quadrats in each transect and then taking the average of the transects to give a figure for each pasture. Standard errors were calculated for each pasture from the averages of the transects.

2.3.2. Species richness

The number of introduced and native species found in each year in each of the six pastures was counted. χ^2 tests were conducted to determine whether there was a change in total, native or introduced species richness over time or across the different pastures.

2.3.3. Diversity indices

Shannon's H diversity index ($H = -\sum_{i=1}^{S} p_i * \ln p_i$, Spellerberg and Fedor, 2003; Begon et al., 1996) was calculated for all pastures

Table 1
The number of monitoring transects and quadrats in each pasture studied in the White Lake Basin, British Columbia, Canada.

Pasture	No. transects	No. plots
Park Rill North	9	90
Park Rill South	8	80
Set Aside	5	50
White Ranch South	3	67
White South	6	60
White North	8	75