

numbers, vigor and size (George et al. 2013). It is likely that native perennial grasses were present in and prior to 2011 throughout much of the area, but were severely diminished in size and vigor, making them difficult to detect. Perennial grasses (both native and introduced) are the only palatable, green grass at TomKat Ranch during some times of year, making them a targeted forage for livestock and susceptible to being grazed at a frequency that does not allow for adequate shoot and root regeneration and seed set. Hence we are likely documenting an increase in distribution of native perennial grasses as well as an increase in detectability of existing stands.

Timing of grazing has been highlighted as the most important aspect in promoting native grass restoration (Menke 1992, George et al. 2013). In the grazing plan described here, the timing of grazing was varied so that the same fields were not grazed during the same phenological period every year. Grazing was not specifically timed to promote native perennial grasses across the whole area but all pastures should have received rest during native grass seed production at least once every two years. We hypothesize that this rest facilitated perennial grass recovery and establishment even in the absence of careful timing in any single year.

Our results suggest that changing grazing practices was associated with the expansion and increased detectability of native grasses at TomKat Ranch. We need to further understand the effects of season, frequency and duration, and intensity of grazing for native grass restoration in California. It is likely that the grazing effects will depend on local site conditions and weather patterns and therefore grazing management must take an adaptive approach as we learn and respond to observation(s). Furthermore, we recognize one shortcoming of the information presented here is a lack of specific grazing management data. We recommend grazing managers keep accurate records of their grazing management so that we may further learn and understand grazing effects.

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Russian Olive Fruit Production in Shelterbelt and Riparian Populations in Montana

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Russian olive (*Elaeagnus angustifolia*) became a common ornamental plant in the southwestern United States in the early part of the 20th century and escaped cultivation in all southwestern U.S. states by the early 1950s (Stannard et al. 2002). Russian olive was introduced in the 1930s in the Great Plains of the U.S. for soil conservation. Few native trees are found in open, windswept areas of the northern Great Plains, and planted Russian olive windbreak populations provide shelter to humans and livestock. However, Russian olive is invasive in riparian areas throughout the western United States (Nagler et al. 2011). Riparian populations of Russian olive prevent

Table 1. Features that distinguish shelterbelt from riparian Russian olive populations.

Population type	Shelterbelt	Riparian
<i>Population maintenance</i>	Non-invasive, extirpation due to removal, wind, or age-related mortality	Invasive, extirpation due to removal, flooding, or ice-scouring
<i>Age structure</i>	Even-aged	Multiple ages
<i>Desirable characteristics</i>	Soil stabilization, wildlife habitat	Wildlife habitat
<i>Undesirable characteristics</i>	Propagule supply for riparian areas	Propagule supply for further invasion, prohibits recreational use, prevents livestock use, reduced functional diversity of avian community
<i>Distance between populations</i>	15 to 100s of kms	Small, generally <5kms
<i># trees per population</i>	Tens to hundreds	One to thousands
<i>Fruiting frequency</i>	Less variable	More variable

recreational and agricultural use of riparian areas and threaten native populations of cottonwood and willow trees (Lesica and Miles 2001). These nitrogen-fixing trees have the potential for cascading, ecosystem-wide negative impacts, including altered beaver population dynamics, restructured food webs in invaded watersheds, and altered nitrogen cycling in both local terrestrial and aquatic habitats (Lesica and Miles 1999, Pearce and Smith 2001, Mineau et al. 2011). The effectiveness and cost of invasive species control depends on the spatial distribution of populations (Richardson and Bond 1991, Epanchin-Neill and Hastings 2010) and persistence of biological control organisms is deeply affected by resource availability (De Clerck-Floate and Bouchier 2000). Therefore, differences between shelterbelt and invasive riparian populations may affect approaches to population removal and control of expansion. Here we summarize the different demographic characteristics of desirable shelterbelt populations and invasive riparian populations of this species in order to make predictions about costs and efficacy of control.

Planted shelterbelt populations of Russian olive are even-aged and rarely spread into immediately adjacent croplands and rangelands (Stannard et al. 2002). Invasive riparian populations are spreading along riparian corridors; few limits to expansion of these populations have been found (Lesica and Miles 1999, Nagler et al. 2010). Less well-studied are the populations of Russian olive that do not fit into the planted-windbreak invasive-riparian dichotomy: planted riparian populations and opportunist populations in drainages and wet pastures. Planted populations along rivers share most characteristics with invasive riparian populations (Table 1) and often blend with them. Opportunist populations are usually small and isolated.

Russian olive has prolific fruit production, with thousands of oblong fruits per tree late in the fall (Katz and Shafroth 2003). These fruits either remain on the trees over winter or are dropped. Fruits may be vertebrate- or water-dispersed and seeds are long-lived. Trees initiate fruit production after four years of growth but do not reliably produce fruit until they are ten years of age (Lesica and Miles 1999). Trees also stump- and root-sprout after fire, mechanical injury, or other disturbances.

Planted shelterbelt populations are relatively small (tens to hundreds of trees) and isolated, as plantings tend to be near farmsteads and settlements (Table 1). These populations are even-aged, as they are generally planted in a single-entry occurrence with little natural regeneration. While Russian olive trees in these populations rarely expand beyond planted population boundaries (Stannard et al. 2002), the distance of seed movement from these populations is unknown. There are no published data on the spatial distribution of shelterbelt populations in Montana.

Riparian populations are large, connected (sometimes isolated), with mixed age structure (Table 1). A spatial data set of Russian olive patches along the Yellowstone River (Combs and Potter 2011) reveals no stretches with Russian olive trees absent at the 5 km scale between Fairview and Miles City, MT (> 200 km). In contrast, Lesica and Miles (1999) mapped the number of Russian olive trees in 5 km segments along the Milk River in Montana. They found long stretches of river without any mature trees (≥ 25 km), however, around settlements tree numbers increased and remained well above zero (-5–100) for long distances (≥ 70 km). The Milk River is a tributary of the Missouri River with a median volume of 20,000 acre-feet, and has anthropogenically altered hydrology due to damming (USBR 2012). The Yellowstone River has a median volume of over 2.2 million acre-feet (Graumlich et al. 2002), and is undammed except for diversion to irrigation.

The available data suggest that shelterbelt and riparian populations of Russian olive have very different spatial structure, with shelterbelt populations tending towards small and isolated, while riparian populations are generally larger and more connected, depending on river size and waterflow characteristics.

Trees in three shelterbelt and three riparian populations near Sidney, MT have been monitored since 2009. Fruiting trees were marked in 2009 and checked every following year for fruiting (yes or no, per individual tree). Distance between shelterbelt locations range from 6.8 to 27.8 km and from 4.6 to 37.3 km between riparian locations. The shelterbelt and riparian locations are interspersed along a north-south gradient.

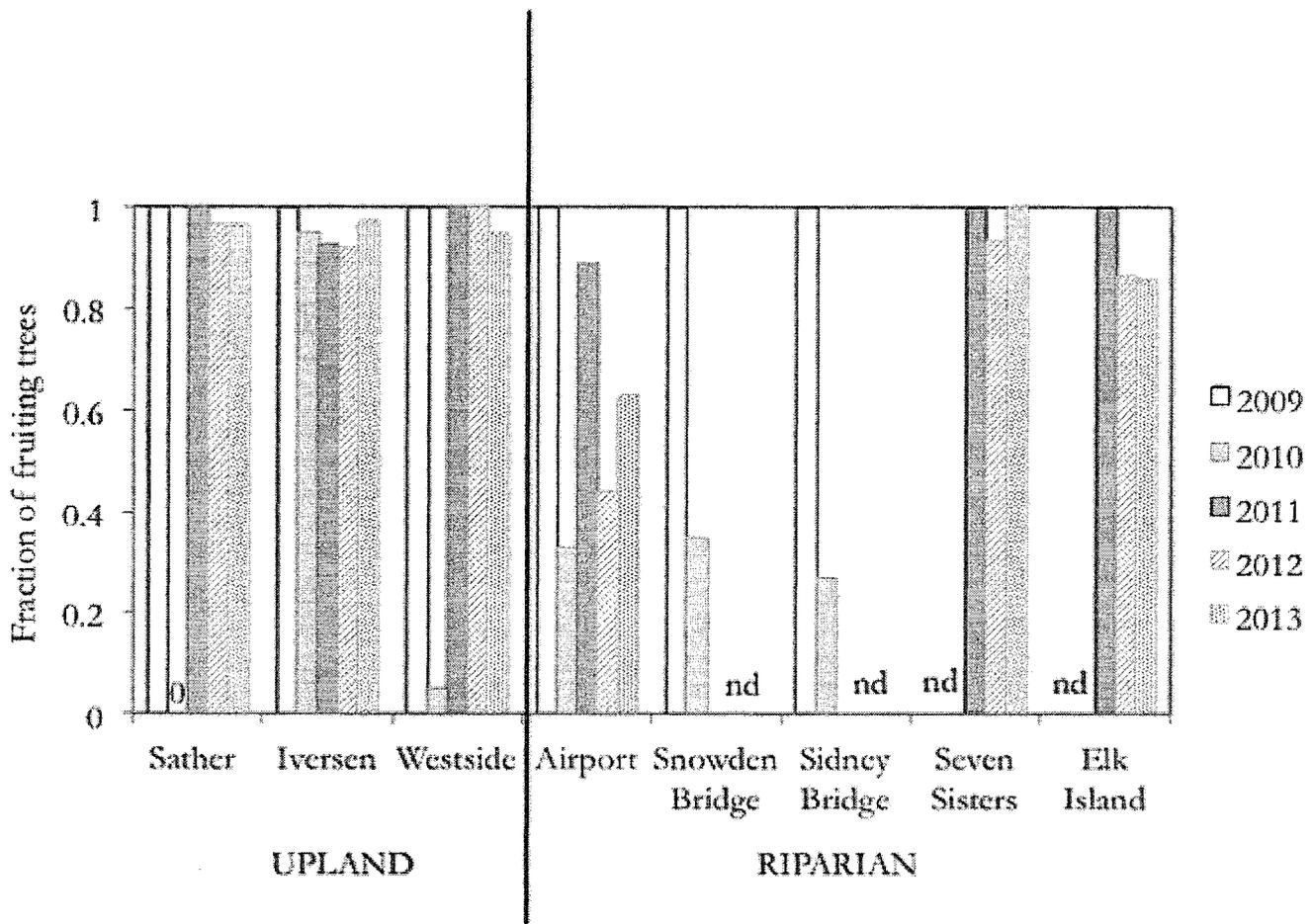


Figure 1. Percent of Russian olive trees fruiting each fall (2009–2013) in five riparian and three upland populations near Sidney, MT (nd = no data). Snowden Bridge and Sidney Bridge populations were extirpated in spring 2011 due to flooding. Elk Island and Seven Sisters populations were added in 2011. Percentage in the initial year (column in black outline) is always 100, as only fruiting trees were marked for later monitoring.

Our sample sizes are trees that comprise three shelterbelt Russian olive populations (Iversen $N = 40$, Sather Dam $N = 29$, and Westside School $N = 60$) and a subset of trees within three riparian populations (Airport, $N = 27$; Elk Island, $N = 15$; Seven Sisters, $N = 15$). At the beginning of our monitoring project, we followed trees in two other riparian populations (Snowden Bridge $N = 17$; Sidney Bridge $N = 15$) that were extirpated by flooding in spring of 2011, at which time we added Elk Island and Seven Sisters for monitoring. All riparian populations are located along the Yellowstone River, except Snowden Bridge that is along the Missouri River north of Sidney, MT. The years in which riparian populations established are unknown. For shelterbelt populations, the Sather Dam population was planted in the 1960s with a follow-up planting in 1989, and the Iversen population was planted in 1989. The planting date of Westside School is unknown but trees are of uniform size, and fall within the size range of the other two shelterbelt populations.

Figure 1 shows the percentage of marked trees that fruited in following years. Shelterbelt populations ranged

from zero (Sather Dam in 2010) to 100 (all populations in 2012; Sather Dam and Westside School in 2011). Riparian populations ranged from 27 (Sidney Bridge in 2010) to 100 (Seven Sisters in 2013). These percentages were not statistically different in a repeated measures model using JMP 10.0.2 (SAS Institute, Cary NC; $F_{3,15} = 2.92$, $p = 0.08$).

The costs of limiting or eradicating invasions are balanced with the benefits of control, and both depend on the biology and ecology of the invasive species. Biological control is an option when mechanical control is prohibitively expensive for land managers. Cost per hectare of removal of Russian olive in 2011 was 18 person hours plus US\$450 in herbicide and fuel costs (Espeland et al. in press), or, 0.01 hours and US\$0.34 per tree. Yearly reentry controlling re-sprouts and emerging seedlings costs an average of 0.4 person hours and US\$232 in herbicide per hectare. Others have noted that costs of control increase when seed banks are long-lived (e.g. Epanchin-Niell and Hastings 2010), and controlling the propagule supply is a significant fraction of the cost of eradicating Russian olive at a single site.

The type of spread of invasive species affects our ability to control them in a cost-effective manner. High rates of spread increase the benefit of control, and species that spread linearly are less damaging than those that spread radially (e.g. Epanchin-Niell and Hastings 2010). Shelterbelt populations of Russian olive appear not to spread at all, while spread of riparian populations is linear along riparian corridors. The rate of spread of Russian olive is slow because of the four-year delay from emergence to seed set (Lesica and Miles 1999) and is further slowed by the fact that individual trees, even those as old as 25 years or more, do not reliably produce fruit every year (this paper). In addition, our data show that the amount of seed set can vary greatly among populations, with some populations setting zero seeds in some years. Therefore, when deciding how to spend dollars allocated to invasive species control, Russian olive may be low priority due to its slow spread compared to other species.

Because the benefits of control may not justify the high cost of mechanical removal, biological control may be an effective choice for limiting the spread of Russian olive. Ten herbivore biological control agents for Russian olive (found in Bean et al. 2008) are under consideration. Our data show that resource collapse for fruit-feeding biological control agents is possible. This may affect agent persistence within populations of Russian olive trees, particularly when coupled with limited dispersal ability. We may exploit differences in shelterbelt and riparian populations to increase biological control agent persistence and efficacy, but we need more detailed information on the dispersal abilities and life-history or behavioral characteristics of candidate agents as well as phenological, in addition to demographic, data on Russian olive tree populations.

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