Restoration Ecology
THE JOURNAL OF THE SOCIETY FOR ECOLOGICAL RESTORATION

OPINION ARTICLE

To what extent can and should revegetation serve as restoration?

Douglas A. Kelt[1,2], Peter L. Meserve[3]

Desertification and climate change are degrading large areas of arid and semiarid habitat in many regions. In response, many countries have implemented revegetation programs, commonly using *Atriplex nummularia*, native to Australia. Although not intrinsically targeted at habitat restoration, these programs aim to restore (stabilize) erosional processes and provide livestock forage (usually goats, sheep). Few investigators have assessed the utility of these novel habitats for native fauna. In a recent, extensive survey of small mammal communities in lands revegetated with *A. nummularia* in north-central Chile, we captured a single animal, the marsupial (*Thylamys elegans*). We also captured no birds in our live traps and saw numerous darkling beetles in traps. These striking results contrasted with data from a nearby ungrazed park with natural vegetation where 10 small mammal species are known with total abundances ranging from 15 to 80 animals in similar sampling efforts. These revegetated lands provide poor habitat for native mammals, and we argue that revegetation efforts should include proactive risk assessment and cost-benefit analyses in a structured decision-making framework. In addition, managers should target increased plant species diversity to support broader biological diversity in addition to the need to control erosion. Although our work has focused on revegetation with *Atriplex*, numerous other plant species have been intentionally introduced to arid lands, usually for economic reasons; many of these, like *Atriplex*, have unintended consequences for native biota.

Key words: ***Atriplex*, desertification, invasive species, plant introductions, small mammals**

Implications for Practice

- Introductions of non-native plants may have unanticipated, often negative, impacts on native biota.
- Such introductions should be preceded by both a risk assessment and a (environmental) cost-(financial) benefit analysis, likely within a structured decision-making context.
- Revegetation efforts should strive to retain floristic diversity for the benefit of native biota.

Introduction

Desertification is a global threat and a key risk in the face of climate change. Drylands comprise over 40% of the land area on earth (Millennium Ecosystem Assessment 2005); with 6–12 million km^2 (10–20% of existing drylands) facing desertification, this is one of the most challenging environmental threats with both local and global consequences (Millennium Ecosystem Assessment 2005, p. 7). Maintenance of vegetative cover is an important element of strategies to minimize the impacts of desertification. In many regions proactive vegetation and livestock management is being applied to mitigate or forestall the effects of anthropogenic desertification (western Asia and northern Africa, Mulas & Mulas 2004; temperate South America, Guevara et al. 2005, Meneses et al. 2012; Australia, Collard et al. 2011). These efforts do not comprise restoration in the traditional sense, but the goals are similar: to restore or retain fluvial and erosional stability, while simultaneously providing forage for livestock.

Arid and semiarid regions of north-central Chile have experienced extensive grazing and browsing since European colonization; combined with marginal cultivation and subsistence pastoralism, this has led to degradation of the native vegetation and extensive soil erosion (Bahre 1979; Núñez & Grosjean 2003; Casanova et al. 2013), with what Mulas and Mulas (2004, p. 17) deem “disastrous consequences”. To counter these threats, the Chilean government has promoted extensive planting of both native and exotic species of *Atriplex* (Executive Orders #701, 28 October 1974, and #2565, 3 April 1979; see www.leychile.cl for text of these laws). By the end of the twentieth century the University of Chile and three governmental agencies in Chile (the Corporación de Fomento de la Producción [CORFO], the Servicio Agrícola y Ganadero [SAG], and the Instituto Nacional de Investigaciones Agropecuarias [INIA]) collaborated to plant over 48,000 ha of land in Chile’s Coquimbo Region with *Atriplex*, primarily

Author contributions: DAK, PLM conceived of this project, and shared fully in data collection; DAK conducted analyses and wrote first draft; DAK, PLM edited the manuscript.

[1]Department of Wildlife, Fish, & Conservation Biology, University of California, One Shields Avenue, Davis, CA 95616, U.S.A.
[2]Address correspondence to D. A. Kelt, email dakelt@ucdavis.edu
[3]Department of Biological Sciences, University of Idaho, 875 Perimeter Drive, MS 3051, Moscow, ID 83844-3051, U.S.A.

doi: 10.1111/rec.12371

Atriplex nummularia (>90%, native to Australia), with lesser coverage by the smaller endemic *Atriplex repanda*, and minor coverage by another Australian native, *Acacia saligna* (Soto Alvarez 1998; Mulas & Mulas 2004). Unfortunately, while *A. repanda* is more palatable to livestock than *A. nummularia*, it performs poorly as pasturage and exhibits rapid mortality (R. Meneses 2015, INIA Coquimbo, personal communication). Consequently, revegetation has overwhelmingly emphasized the exotic *A. nummularia*. The resulting landscape in many regions approaches one reminiscent of an agricultural monoculture, with *Atriplex* planted in long, often contoured, rows and separated by 1–3 m (Fig. 1). Although the agricultural benefits of these plantings " … has given encouraging results" (Mulas & Mulas 2004, p. 21), the environmental impacts of such plantings have received limited investigation outside of Australia.

We recently surveyed *Atriplex* habitat in north-central Chile and here we compare our results with extensive data available from our long-term research at a nearby national park. We apply these data here to argue that revegetation in this region has serious ecological consequences, similar in scope to those reported for other plant species that have been introduced for short-term economic benefits without full consideration of longer-term environmental costs. These observations underscore the importance of cost-benefit analysis (e.g. Conser et al. 2015) and a structured decision-making protocol (e.g. Gregory et al. 2012) when considering such introductions. In addition, we argue for integration of more native vegetation and a sociocultural emphasis on biological diversity.

Methods

Fray Jorge

Since June 1989 we have monitored small mammals at replicated trapping grids in Parque Nacional Bosque Fray Jorge ("Fray Jorge" hereafter), located on the coast of north-central Chile about 100 km south of the city of La Serena. Fray Jorge is a UNESCO Biosphere Reserve and is about 10,000 ha in area (Armas et al. 2016). In spite of its name, over 97% of this park consists of drought-deciduous shrub habitat (matorral), which probably characterized much of the native flora in the region (Squeo et al. 2016), and our research there has focused on this habitat. Within this extensive matorral habitat, we regularly capture 3–4 "core" small mammal species (e.g. species that are always present; Milstead et al. 2007) and less regularly another 3–4 species that either favor more mesic habitats but spill over to our matorral sites or exhibit irruptive demographic responses to resource pulses triggered by high rainfall. Thus, the small mammal fauna here is diverse and, like the flora, representative of the regional fauna.

We sampled small mammals monthly on 20 75 × 75-m grids, including four replicates of each of four biotic treatments (exclusion of predators, lagomorphs, small mammals, and both lagomorphs and small mammals) and four control plots to which all species have free access. Here we use data only from four control plots. Our sampling methods have been presented in multiple papers (most recently by Gutiérrez et al. 2010; Kelt et al. 2013); briefly, we sampled small mammals over four consecutive nights on 5 × 5 trapping grids (15-m trap spacing) with two large Sherman-type live traps per station. Traps were baited with rolled oats near dusk and checked soon after sunrise. Unless conditions dictate otherwise, traps were left open until late afternoon for a second trap check to sample the diurnal degu (*Octodon degus*). Population size is estimated as the minimum number of animals known to be alive on a given survey (MNKA: Krebs 1966; Hilborn et al. 1976), although here we use the total number of animals captured in a given month for direct comparison with our sampling at El Tangue, because the latter comprises two single efforts that preclude calculation of MNKA (see below).

Fray Jorge has not been grazed or browsed since about 1941 and it contains a largely complete and intact fauna of native small mammal species; historically, the Chilean chinchilla (*Chinchilla lanigera*) may have occurred here, but if so, it has been extirpated locally. Squeo et al. (2016, p. 12) noted that the flora in Fray Jorge "constitutes a remnant of the natural vegetation that dominated the Coastal Desert before European colonization" and we consider this to be true for the small mammal assemblage as well.

Hacienda El Tangue

In March 2014, we established two live-trapping grids at each of two locations (hence, four grids total) on a large private ranch, Hacienda El Tangue ("El Tangue"; >45,000 ha). Our trapping sites were located about 25 km north of our sites at Fray Jorge (Fig. 2) and approximately 5 and 18 km from the coast. Habitat here comprises near-monocultural stands of *Atriplex*, primarily *Atriplex nummularia*, established in contoured rows (Fig. 1). El Tangue includes approximately 10,000 ha of *Atriplex* which were planted between 1983 and 2010; we do not know exactly when plants at our sampling sites were planted, but they were mature plants circa 1.5–2 m tall. Within this dominant background are occasional cacti or shrubs that have established since revegetation. Small mammals were sampled in March and September 2014, with methods identical to that in Fray Jorge except that logistical constraints dictated that we trapped only three nights. Given the high trappability of most species in this region (DAK, PLM, personal observation) and the results we obtained, we do not believe that the reduced sampling effort affected our conclusions (see the Results section).

As noted above, climatic conditions are similar at both sites, with two key exceptions. Sites in the park are just interior to a coastal range (Altos de Talinay) that rises to 660 m and intercepts coastal fog. It is likely that this provides some contributions to the water table at Fray Jorge that may not exist at El Tangue (Kummerow 1962, 1966; Squeo et al. 2004). As the vegetation at El Tangue has been impacted by extensive long-term grazing followed by revegetation with *Atriplex* it is not possible to assess the potential impact of this on vegetative complexity. In additon, one of our sites at El Tangue was located further inland such that morning fog likely dissipated sooner.

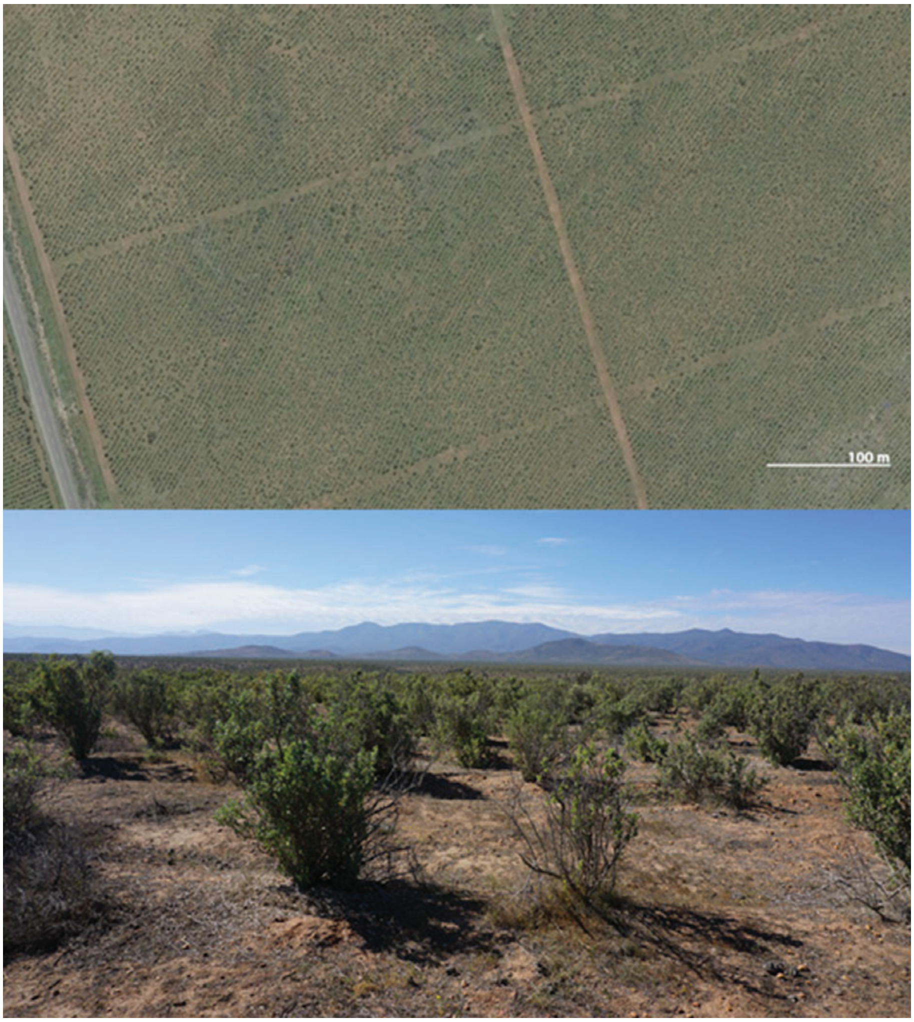

Figure 1. (Top) Satellite image of terrain within the Hacienda El Tangue just south and east of the coastal city of Tongoy, Chile, not far from our sampling sites. The road at the left is road D-440. The image shows *Atriplex nummularia* as dots planted in furrowed roads. Image downloaded 12 April 2015 from The Geo Web service of the Instituto Geografico Militar (http://www.igm.cl/). (Bottom) Habitat at Hacienda El Tangue where small mammals were sampled. The dominant shrub is exotic *A. nummularia*. Photo by P. L. Meserve.

Results

In 2014 small mammal numbers at Fray Jorge varied seasonally (Fig. 3) with smaller population sizes (15–30 animals per grid) in the austral winter and higher numbers (around 80 animals per grid) in spring and summer, reflecting annual recruitment. At El Tangue in the austral summer (March) of 2014, 600 trap-nights of effort across four grids resulted in no captures and no sprung traps, even though these were the same traps used in our monthly samples in Fray Jorge. When we applied an identical sampling effort in the austral winter (August) of 2014, we captured a single individual (the elegant mouse opossum, *Thylamys elegans*) on the final night of sampling at one grid; the other three grids yielded no captures.

Over the past quarter century of monthly efforts at Fray Jorge, we captured ≤ one animal on ≥ one control grid in only 18 of 309 months of effort; across all four control grids we have captured ≤ one animal only three times; hence, in probabilistic terms, the odds of capturing zero or one animal across a standard four-night effort on these grids is very low (3/309 = 0.0097). Moreover, of the 23 months in which we captured a single animal on a grid, seven of these occurrences were on the first night of effort, eight on the second, and three each on the third and fourth nights; these observations suggest that our three-night effort is unlikely to have led to a meaningful bias in documenting small mammal richness at El Tangue.

Discussion — What Is the Restorative Role of Revegetation?

Arid lands have been degraded in many regions, generally reflecting a lack of management foresight, but often involving efforts at revegetation that target one need—generally economic—at the real or potential cost of other ecosystem services. *Atriplex nummularia* has been planted in many arid lands

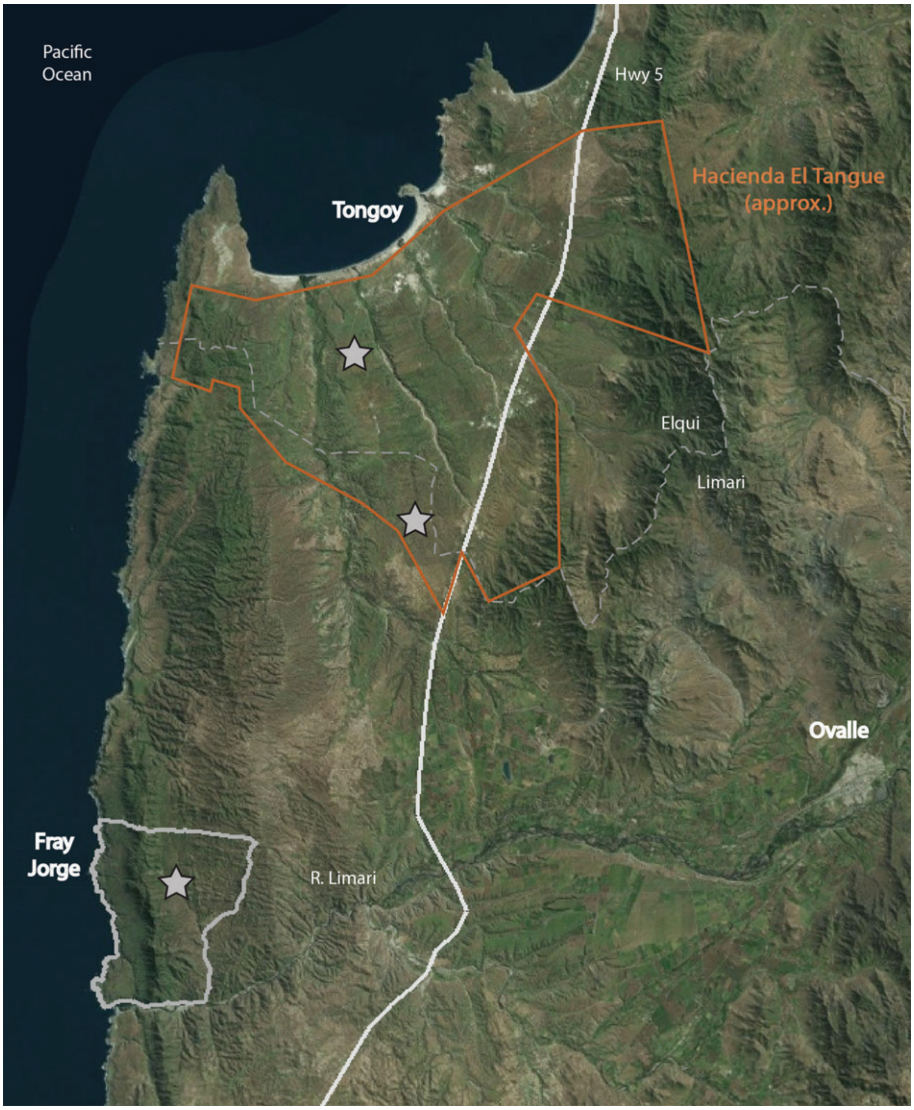

Figure 2. Map of study region in Coquimbo Province, Chile. The towns of Tongoy and Ovalle are indicated, as is Fray Jorge Forest National Park and the approximate borders of the Hacienda El Tangue. The solid white line is the Pan-American Highway (Hwy 5), and the dashed gray line is the boundary between Elqui and Limarí provinces. The Río Limarí discharges into the Pacific Ocean at the southern border of the park. Our three study sites are indicated with the gray stars.

primarily because livestock can tolerate this species and it grows well in stressful conditions. Small mammal numbers at Fray Jorge have fluctuated greatly over the 26 years of study, but they were markedly greater than at Hacienda El Tangue throughout the time of our sampling (Fig. 3). Indeed, small mammals in *Atriplex*-dominated habitat at El Tangue were virtually nonexistent. We consider it notable that the single animal we did capture was the insectivorous/carnivorous marsupial, *Thylamys*. Although this species is almost always present in Fray Jorge, it typically is a small element of the fauna there, and its numbers are always exceeded by those of sigmodontine rodents such as *Abrothrix* spp., or the caviomorph degu. The absence of these other species at El Tangue is remarkable. We also find it notable that we did not capture any birds (a common occurrence in Fray Jorge). Finally, we observed numerous darkling beetles (Coleoptera, *Eleodes* spp.) which, although common inhabitants of most aridlands, are observed rarely in Fray Jorge.

Although our samples are limited both in time and space, the almost complete absence of any sign of mammals agrees with our remarkably low trap success. Local agriculturalists told us that coruros (*Spalacopus cyanus*) occasionally invade these sites, and we did see abundant signs of their burrows, although our sampling areas evidently lacked any living animals; we saw no active burrows and never heard the characteristic trilling made by this species in response to intruders. We did see active burrows adjacent to a road circa 1 km from one of our grids.

Based on these observations, we conclude that *Atriplex* revegetation provides poor habitat for small mammals in this region,

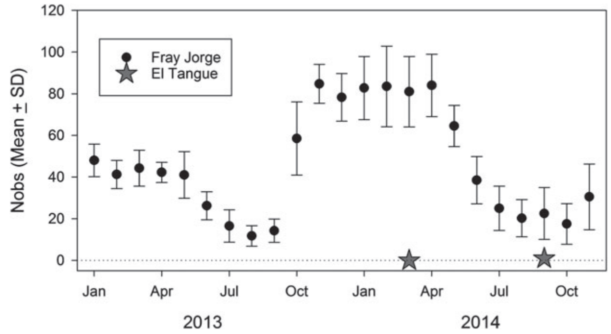

Figure 3. Number of individual small mammals (mean ± SD) captured in four control grids at Fray Jorge, and in two similar trapping efforts at Hacienda El Tangue.

other than occasional, irruptive, and transient populations of coruros. Further efforts should be pursued to confirm the generality of these results, both by sampling additional localities over longer time periods and especially by incorporating additional taxa, perhaps most notably birds and insects. Our sense, based on time spent on these plots, is that avian diversity is very low and actual numbers even lower. Given the limited vegetative diversity the same seems likely for arthropods.

It is important to emphasize that *Atriplex* was not planted here to restore habitat. Rather, it was planted to reduce erosion (e.g. control soil erosion resulting from overgrazing) and to provide forage for livestock; in this sense it has been modestly successful, although goats evidently forage on *Atriplex* only when no other food is available (personal communication from local agriculturalists). *Atriplex* grows well in this environment, and does so more rapidly than many native species. Therefore, barring a change in the environmental ethic of the people inhabiting this region (which in turn would require a change in economic conditions), this likely remains one of a limited set of options to offset the effects of overgrazing and climate-change-induced drought. According to Lailhacar (Lailhacar et al. 1989; Lailhacar & Torres 2002, both cited in Figueroa et al. 2004), *Atriplex* may enhance herbaceous species richness and cover when pruned and managed, by facilitating both organic matter and intercepting moisture that reduces water stress; this assumes an effort at pruning and management, but we observed no such benefits at our site. *Atriplex nummularia* also alters soil structure and composition, with higher pH and salinity beneath plants (Sharma & Tongway 1973; Yahiaoui et al. 2014), and increased levels of exchangeable sodium and organic matter at the surface (Sharma 1973), and at least some species of *Atriplex* have allelopathic effects on seed germination (Hamedanian et al. 2010). Hence, negative consequences of this revegetation are not limited to consumer groups. In the face of regional drought, overgrazing, and desertification, efforts to diversify the composition of species involved in such plantings may be necessary to allow native species to disperse across the landscape and thereby avoid demographic and genetic isolation in increasingly limited patches of high-quality habitat. That is, integrating restorative plantings into revegetation efforts would likely have diverse benefits.

Our results appear to paint a more negative picture of *Atriplex* revegetation than some other studies. Collard et al. (2011) reported lower bird and plant diversity in *Atriplex* revegetated areas relative to remnant native vegetation, but these were greater than in nearby pastures; moreover, some threatened bird species were found in *Atriplex* sites but not on pastures. Hence, while *Atriplex* habitat supported fewer bird and plant species than native vegetation, in the context of widespread regional conversion to agricultural lands these may provide "at least partial habitat for some native biota" (p. 37). More notably, one Australian lizard (*Tiliqua rugosa*) appears to benefit from saltbush plantings (Lancaster et al. 2012), at least at some life stages.

These observations underscore the need for further study on the impacts of *Atriplex* plantings on biodiversity. However, the results of our study also serve to highlight the importance of protected areas such as Fray Jorge, where source populations of the regional natural heritage are maintained through both good times and poor (Armas et al. 2016). Even in Chile, where about 19% of the country's land area is set aside in some form of natural park or reserve, such areas are established largely for visual aesthetic qualities and consequently often fail to adequately incorporate the diversity of natural systems throughout the country. Equally important is that these parks and reserves tend to be isolated within a sea of inhospitable landscape. Even the xeric-adapted matorral habitat of Fray Jorge is losing connectivity with similar habitat outside the park (Squeo et al. 2016), an observation that bodes poorly for small mammals that tend to be dependent on suitable habitat for dispersal movements.

Atriplex is by no means unique in exhibiting unexpected negative consequences of introduction. Some well-known groups (e.g. *Pinus* sp., *Eucalyptus* sp.) have been introduced globally for extractive purposes (e.g. Rejmánek & Richardson 2013). Tamarisk (*Tamarix* sp.) is a highly drought- and salt-tolerant tree that was introduced to the American Southwest and Great Plains, as well as Australia (Cooper 1963) for erosion control and as a windbreak (e.g. along railroad corridors; Zouhar 2003). However, tamarisk can reach deeper water, and survive on less water than most native species in this region, such that many natives plant species decline in the face of tamarisk expansion (Zavaleta 2000). Tamarisk also increases sedimentation in river channels, leading to increased flooding, narrowing of waterways, and expansion of the riparian corridor, albeit usually comprised solely or primarily of tamarisk (Zavaleta 2000). Recent studies document negative impacts on native cottonwoods (*Populus fremontii*) by disruption of both arbuscular mycorrhizal and ectomycorrhizal fungal associations (Meinhardt & Gehring 2012). In terms of ecosystem services, tamarisk invasion results in greatly reduced floristic diversity, and while this reduction in habitat heterogeneity may negatively impact avian diversity (Farley et al. 1994), in some cases it provides quality habitat for some bird species (Sogge et al. 2008).

Many invasive plants have escaped after being translocated for ornamental purposes (Reichard & White 2001; Lehan et al. 2013), whereas others have similar histories to that of *A. nummularia*, with initial introduction to address one concern—usually economic—only to lead to other, often unexpected concerns. Here we briefly outline three of these. In large parts of the Great Basin of North America, native shrubs (e.g. sagebrush [*Artemisia tridentata*], greasewood [*Sarcobatus*], and rabbitbrush [*Chrysothamnus*]) were removed both mechanically and with herbicide application because they are either physiologically challenging, toxic, or unpreferred browse for livestock, which preferentially consumed native perennial grasses and forbs. As rangelands were degraded by livestock foraging and disturbance, cheatgrass (*Bromus tectorum*) established and quickly dominated large areas, ultimately leading to extensive wildfires (the cheatgrass-wildfire cycle; Pellant 1990). To combat this, the more fire-resistant wheatgrasses (primarily *Agropyron desertorum* and *A. cristatum*) were introduced to outcompete cheatgrass; planted after wildfires as well as in "greenstrips" to provide firebreaks, wheatgrasses were considered "the lesser of two evils" (Pellant 1996, p. 7) because they resist burning and provide superior forage for livestock and some wildlife species (but not all; see Litt & Pearson 2013). However, both grass species form extensive monocultures and impact native shrub regrowth, leading to challenges for managers faced with species (e.g. Greater Sage Grouse, *Centrocercus urophasianus*) that require a more heterogeneous habitat for survival (Crawford et al. 2004). Efforts now are underway to revegetate these areas with a mix of native species (Huber-Sannwald & Pyke 2005; Davies et al. 2013; GBNPP 2016), indicating that active management can offset problematic invasions given sufficient effort.

Lehmann lovegrass (*Eragrostis lehmanniana*) and buffel grass (*Pennisetum* [*Cenchrus*] *ciliare*) are two other fire-adapted grasses introduced to North America at least in part as livestock forage. Both species have been documented to directly and indirectly reduce both floral and faunal diversity in the arid lands. The former occur because these species grow rapidly under favorable conditions and generate fire regimes that kill native plant species in the Sonoran Desert (Esque et al. 2006; McDonald & McPherson 2011, 2013) and in Australia (Butler & Fairfax 2003); this can transform rich desert scrub to depauperate grassland in the absence of any change in fire regimes (Olsson et al. 2012) and is associated with reduced growth by native species in Australia (Clarke et al. 2005). Direct impacts on wildlife vary but include changes in density of various avian foraging guilds, variable impacts on reproductive success by grassland birds, changes in composition and relative abundances (but not total abundance) of small mammals, and reduced body condition in adult desert tortoises (summarized by Steidl et al. 2013).

These examples reinforce our argument that revegetation efforts should be pursued with caution. The consequences of lost biodiversity often are not visible in the early stages of these introductions, and short-term socioeconomic gains may appear to exceed unknown long-term environmental consequences including loss of basic ecosystem processes (Dukes & Mooney 2004). Integration of some form of assessment of invasion risk assessment (Leung et al. 2012; Blackburn et al. 2014) or cost-benefit analysis (e.g. environmental cost versus socioeconomic benefit; Conser et al. 2015) should be integrated into a structured decision-making protocol when considering such introductions. In addition, we believe that land managers, both in Chile and elsewhere, should approach revegetation efforts from a more inclusive restoration perspective.

We close by noting that the benefit of natural landscapes may not be merely psychological. In tropical Africa, natural landscapes appear more resistant than human-altered environments to desertification induced by agriculture and climate change, through the engineering activities of mound-building termites (Bonachela et al. 2015); no such dynamics have been documented for temperate regions of the Southern Hemisphere, but conservative management should employ a bet-hedging strategy that favors protection and preservation of natural habitats and biota whenever possible. In the face of climate change and global efforts to revegetate denuded lands with exotic species such as *Atriplex*, further work is needed to confirm that these efforts will not have unexpected negative consequences for biodiversity as well as ecosystem services.

Acknowledgments

G. Gutiérrez (CEAZA, Centro de Estudias Zonas Áridas) helped us locate trapping sites, and C. Jorquera (CEAZA) helped access multiple papers on the use of *Atriplex* in revegetation. Dr. R. Meneses (Instituto de Investigaciones Agropecuarias INIA, Coquimbo, Chile) clarified the history of *Atriplex* revegetation in north-central Chile. W. Bryan Milstead distilled data from our long-term efforts at Fray Jorge. This article has benefitted greatly by the cogent insight from two anonymous reviewers, and by a highly constructive critique provided by the editor. We thank the leadership at the Hacienda El Tangue, particularly Srs. Tomas Cuevas and Rodrigo Perez, for granting free access to their lands, and the Corporación Nacional Forestal for hosting our long-term research at Fray Jorge. This research was funded in part by the National Science Foundation (DEB-1456729 to D.A.K. and P.L.M.) and by the University of California Davis and Department of Wildlife, Fish, & Conservation Biology.

LITERATURE CITED

Armas C, Gutiérrez JR, Kelt DA, Meserve PL (2016) Twenty-five years of research in the north-central Chilean arid zone: the Fray Jorge Long-Term Socio-Ecological Research (LTSER) site and Norte Chico. Journal of Arid Environments 126:1–6

Bahre CJ (1979) Destruction of the natural vegetation of north-central Chile. University of California Publications in Geography 23:1–117

Blackburn TM, Essl F, Evans T, Hulme PE, Jeschke JM, Kuehn I, et al. (2014) A unified classification of alien species based on the magnitude of their environmental impacts. PLoS Biology 12:e1001850

Bonachela JA, Pringle RM, Sheffer E, Coverdale TC, Guyton JA, Caylor KK, Levin SA, Tarnita CE (2015) Termite mounds can increase the robustness of dryland ecosystems to climatic change. Science 347:651–655

Butler DW, Fairfax RJ (2003) Buffel grass and fire in a Gidgee and Brigalow woodland: a case study from central Queensland. Ecological Management & Restoration 4:120–125

Casanova M, Salazar O, Seguel O, Luzio W (2013) Human-induced soil degradation in Chile. Pages 121–158. In: Casanova M, Salazar O, Sequel O, Luzio W, (eds) The soils of Chile. Springer, Dordrecht, The Netherlands

Clarke PJ, Latz PK, Albrecht DE (2005) Long-term changes in semi-arid vegetation: invasion of an exotic perennial grass has larger effects than rainfall variability. Journal of Vegetation Science 16:237–248

Collard SJ, Fisher AM, McKenna DJ (2011) Planted saltbush (*Atriplex nummularia*) and its value for birds in farming landscapes of the South Australian Murray Mallee. Ecological Management & Restoration 12:37–45

Conser C, Seebacher L, Fujino DW, Reichard S, DiTomaso JM (2015) The development of a plant risk evaluation (pre) tool for assessing the invasive potential of ornamental plants. PLoS One 10:e0121053

Cooper CF (1963) Tamarix as a potential noxious shrub. Journal of the Australian Institute of Agricultural Science 29:178–181

Crawford JA, Olson RA, West NE, Mosley JC, Schroeder MA, Whitson TD, Miller RF, Gregg MA, Boyd CS (2004) Ecology and management of sage-grouse and sage-grouse habitat. Journal of Range Management 57:12–19

Davies KW, Boyd CS, Nafus AM (2013) Restoring the sagebrush component in crested wheatgrass–dominated communities. Rangeland Ecology & Management 66:472–478

Dukes JS, Mooney HA (2004) Disruption of ecosystem processes in western North America by invasive species. Revista Chilena de Historia Natural 77:411–437

Esque TC, Schwalbe CR, Lissow JA, Haines DF, Foster D, Garnett MC (2006) Buffelgrass fuel loads in Saguaro National Park, Arizona, increase fire danger and threaten native species. Park Science 24:33–37

Farley GH, Ellis LM, Stuart JN, Scott NJ Jr (1994) Avian species richness in different-aged stands of riparian forest along the middle Rio Grande, New Mexico. Conservation Biology 8:1098–1108

Figueroa JA, Castro SA, Marquet PA, Jaksic FM (2004) Exotic plant invasions to the mediterranean region of Chile: causes, history and impacts. Revista Chilena de Historia Natural 77:465–483

GBNPP (2016) Great Basin Native Plant Project. http://www.greatbasinnpp.org/ (accessed 4 Mar 2016)

Gregory R, Failing L, Harstone M, Long G, McDaniels T, Ohlson D (2012) Structured decision making: a practical guide to environmental management choices. Chichester, United Kingdom, Wiley-Blackwell

Guevara JC, Allegretti LI, Paez JA, Estevez OR, Le Houérou HN, Silva Colomer JH (2005) Yield, nutritional value, and economic benefits of *Atriplex nummularia* plantation in marginal dryland areas for conventional forage crops. Arid Land Research and Management 19:327–340

Gutiérrez JR, Meserve PL, Kelt DA, Engilis J, Andrew Previtali MA, Milstead WB, Jaksic FM (2010) Long-term research in Bosque Fray Jorge National Park: twenty years studying the role of biotic and abiotic factors in a Chilean semiarid scrubland. Revista Chilena de Historia Natural 83:69–98

Hamedanian F, Jafari M, Dehdari S, Henteh A, Zare Chahouki MA (2010) The allelopathic effects of *Atriplex canescens* (Four wing saltbush) on seed germination of *Salsola rigida*. Desert 15:15–18

Hilborn R, Redfield JA, Krebs CJ (1976) On the reliability of enumeration for mark and recapture census of voles. Canadian Journal of Zoology 54:1019–1024

Huber-Sannwald E, Pyke DA (2005) Establishing native grasses in a big sagebrush-dominated site: an intermediate restoration step. Restoration Ecology 13:292–301

Kelt DA, Meserve PL, Gutiérrez JR, Milstead WB, Previtali MA (2013) Long-term monitoring of mammals in the face of biotic and abiotic influences at a semiarid site in north-central Chile. Ecology 94:977

Krebs CJ (1966) Demographic changes in fluctuating populations of *Microtus californicus*. Ecological Monographs 36:239–273

Kummerow J (1962) Mediciones cuantitativas de la neblina en el Parque Nacional Fray Jorge. Boletín Universidad de Chile 28:36–37

Kummerow J (1966) Aporte al conocimiento de las condiciones climáticas del bosque Fray Jorge. Universidad de Chile. Facultad de Agronomía, Estación Experimental Agronómica. Boletín Técnico 24:21–24

Lailhacar S, Torres C (2002) Influencia de los arbustos del género *Atriplex* L. con y sin corte de rebaje en el comportamiento del estrato herbáceo del secano árido. I: Efecto del arbusto promedio del género sobre las características cuantitativas del estrato herbáceo total. Avances en Producción Animal (Chile) 26:79–96

Lailhacar S, Carrasco A, Sadzawka MA, Castellaro G (1989) Effet de l'arbuste fourrager *Atriplex repanda* Phil. sur fertilité du sol et sur la strate herbace é associée. XVI Congrès International de Herbages, Nice, France

Lancaster ML, Gardner MG, Fitch AJ, Ansari TH, Smyth AK (2012) A direct benefit of native saltbush revegetation for an endemic lizard (Tiliqua rugosa) in southern Australia. Australian Journal of Zoology 60:192–198

Lehan NE, Murphy JR, Thorburn LP, Bradley BA (2013) Accidental introductions are an important source of invasive plants in the continental United States. American Journal of Botany 100:1287–1293

Leung B, Roura-Pascual N, Bacher S, Heikkila J, Brotons L, Burgman MA, et al. (2012) TEASIng apart alien species risk assessments: a framework for best practices. Ecology Letters 15:1475–1493

Litt AR, Pearson DE (2013) Non-native plants and wildlife in the Intermountain West. Wildlife Society Bulletin 37:517–526

McDonald CJ, McPherson GR (2011) Fire behavior characteristics of buffelgrass-fueled fires and native plant community composition in invaded patches. Journal of Arid Environments 75:1147–1154

McDonald CJ, McPherson GR (2013) Creating hotter fires in the Sonoran Desert: buffelgrass produces copious fuels and high fire temperatures. Fire Ecology 9:26–39

Meinhardt KA, Gehring CA (2012) Disrupting mycorrhizal mutualisms: a potential mechanism by which exotic tamarisk outcompetes native cottonwoods. Ecological Applications 22:532–549

Meneses R, Varela G, Flores H (2012) Evaluating the use of Atriplex nummularia hay on feed intake, growth, and carcass characteristics of creole kids. Chilean Journal of Agricultural Research 72:74–79

Millennium Ecosystem Assessment (2005) Ecosystems and human well-being: desertification synthesis. World Resources Institute, Washington D.C.

Milstead WB, Meserve PL, Campanella A, Previtali MA, Kelt DA, Gutiérrez JR (2007) Spatial ecology of small mammals in north-central Chile: role of precipitation and refuges. Journal of Mammalogy 88:1532–1538

Mulas M, Mulas G (2004)Pages 1–101. The strategic use of *Atriplex* and *Opuntia* to combat desertification. University of Sassari Desertification Research Group, Short and medium-term priority environmental action program (SMAP), Sassari, Italy

Núñez L, Grosjean M (2003) Biodiversity and human impact during the laast 11,000 years in north-central Chile. Pages 7–17. In: Bradshaw GA, Marquet PA (eds) How landscapes change: human disturbance and ecosystem fragmentaiton in the Americas. Springer-Verlag, Berlin Heidelberg, Germany

Olsson AD, Betancourt J, McClaran MP, Marsh SE (2012) Sonoran Desert Ecosystem transformation by a C4 grass without the grass/fire cycle. Diversity and Distributions 18:10–21

Pellant M (1990) The cheatgrass-wildire cycle—are there any solutions? Pages 11–17. In: McArthur ED, Romney EM, Smith SD, Tueller PT, compilers (eds). Proceedings-symposium on cheatgrass invasion, shrub die-off, and other aspects of shrub biology and management. General Technical Report INT-276, U.S. Department of Agriculture, Forest Service, Intermountain Research Station, Ogden, Utah

Pellant M (1996) Cheatgrass: the invader that won the west. Interior Columbia Basin Ecosystem Management Project, Bureau of Land Management. Idaho State Office, Boise, Idaho

Reichard SH, White P (2001) Horticulture as a pathway of invasive plant introductions in the United States: most invasive plants have been introduced for horticultural use by nurseries, botanical gardens, and individuals. Bioscience 51:103–113

Rejmánek M, Richardson DM (2013) Trees and shrubs as invasive alien species—2013 update of the global database. Diversity and Distributions 19:1093–1097

Sharma ML (1973) Soil physical and physico-chemical variability induced by *Atriplex nummularia*. Journal of Range Management 26:426–430

Sharma ML, Tongway DJ (1973) Plant induced soil salinity patterns in two saltbush (*Atriplex* spp.) communities. Journal of Range Management 26:121–125

Sogge MK, Sferra SJ, Paxton EH (2008) Tamarix as habitat for birds: implications for riparian restoration in the southwestern United States. Restoration Ecology 16:146–154

Soto Alvarez G (1998) *Atriplex nummularia*: alternativa contra la desertificacion. First Latin American International Union of Forest Research Organizations (IUFROLAT) Congress, Valdivia, Chile

Squeo FA, Pizarro-Araya J, Vega S (2004) Ecofisiología del bosque relicto de *Aextoxicon punctatum* (Olivillo) en el Parque Nacional Bosque Fray Jorge, Coquimbo, Chile. Pages 219–234. In: Squeo FA, Gutiérrez JR, Hernández IR (eds) Historia Natural del Parque Nacional Bosque Fray Jorge. Ediciones Universidad de La Serena, La Serena, Chile

Squeo FA, Loayza AP, López RP, Gutiérrez JR (2016) Vegetation of Bosque Fray Jorge National Park and its surrounding matrix in the Coastal Desert of north-central Chile. Journal of Arid Environments 126:12–22

Steidl RJ, Litt AR, Matter WJ (2013) Effects of plant invasions on wildlife in desert grasslands. Wildlife Society Bulletin 37:527–536

Yahiaoui FZ, Labani A, Terras M, Anteur D, Adda-Hanifi NN, Kerache G, Hadouche MI (2014) Study of the impact of three species of *Atriplex* (*Atriplex halimus*, *Atriplex nummularia* and *Atriplex canescens*) on the soil physico-chemical parameters in the steppe zone; case of the south west of Algeria. European Scientific Journal 10:142–150

Zavaleta E (2000) The economic value of controlling an invasive shrub. Ambio 29:462–467

Zouhar K (2003) *Tamarix* spp. In: Fire effects information system. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory (Producer), Missoula, Montana. http://www.fs.fed.us/database/feis/ (accessed 7 Jan 2016)

Coordinating Editor: David Pyke

Received: 6 August, 2015; First decision: 13 September, 2015; Revised: 14 March, 2016; Accepted: 14 March, 2016; First published online: 18 May, 2016