Long-term outcomes of a natural-processes approach to riparian restoration in a large regulated river: the Rio Grande Albuquerque Overbank Project after 16 years.

Running head: Natural-processes approach outcomes in riparian restoration.

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ABSTRACT

A demonstration project was initiated in 1998 in the Middle Rio Grande of New Mexico to test the efficacy of a natural-processes approach to large-river restoration. A point bar dominated by non-native Russian olive was mechanically treated by the removal of all vegetation, and a portion lowered sufficiently to allow overbank flooding during typical spring releases from the upstream dam. Side channels and small islands were engineered in the lowered bar to slow flood waters and aid sediment deposition, and create a complex landform mosaic. At the same time, a high-resolution vegetation-monitoring grid was installed. Releases were sufficient to initially flood the site in the spring of 1998. Over 10,000 cottonwoods/ha were established in the first growing season, but numbers varied based on the fluvial landforms. Zones that were sufficiently wetted or naturally formed behind large woody debris piles were most successful, while those on artificial fill sites or sites that were not lowered showed the lowest numbers. Over the ensuing 15 years, tree numbers declined through a process of self-thinning and beaver browsing, but woody natives continued to dominate 3.5:1. Natives also dominated a species-rich herbaceous layer, particularly on the lowered sites. We were surprised by the intensity of beaver herbivory and the unexpected incursion of new herbaceous invaders that was revealed by the long-term monitoring record. Yet, based on several criteria, the site continues to constitute a success with respect to the application of a natural processes approach to large-river restoration.

KEYWORDS: vegetation monitoring, dynamic patch mosaic, cottonwood (*Populus deltoides*), Russian olive (*Elaeagnus angustifolia*), saltcedar (*Tamarisk* spp.), Bosque, hydrology, exotic species management.

Introduction

Natural-processes approaches to riparian restoration have gained broader acceptance as tools for generating and maintaining native vegetation diversity, providing supporting habitat for fish and wildlife, and enhancing overall ecological services (Stanford et al. 1996; Poff et al. 1997; Molles et al. 1998; Stromberg 2001; Follstad Shah et al. 2007; Stromberg et al. 2007). A guiding principal is to restore a dynamic riverscape of shifting ecological communities on a changing fluvial geomorphic template driven by hydrological processes (Crawford et al. 1993; Hupp and Osterkamp 1996; Crawford et al. 1999; Stanford et al. 2004; Latterell et al. 2006; Weisberg al. 2013). That is, to foster a dynamic patch or habitat mosaic of vegetation succession intertwined with the evolution of fluvial surfaces in response to flooding and channel migration (e.g., a range of small pioneer bars of herbaceous vegetation and shrublands to mature forested wetland on large terraces). In addition, pursuing the reestablishment of a dynamic patch mosaic holds the potential for being the most cost-effective approach to restoration (Stanford et al 1996; Taylor and McDaniel 1998), by letting the river do the bulk of work in a way that leads to the long-term sustainability of a complex and relatively natural ecosystem.

This is particularly pertinent in the southwestern U.S. where regulated lowland river systems have been extensively invaded by woody, non-native Russian olive (*Elaeagnus angustifolia*) and saltcedar (*Tamarisk* spp.) that present a daunting task for restoration (Everitt 1998; Shafroth et al. 2002; Cooper et al. 2003; Freidman et al. 2005; Reynolds et al. 2010). The classical and still predominant approach is to remove the woody invaders with a combination of herbicide, stem cutting, or root plowing followed by pole plantings of native phreatophytes such as cottonwoods and willows. While effective in the short-term, sites often return to exotic–dominated communities because nothing has changed with respect to the hydro-geomorphic configuration of the site and the associated stream-flow regime (Hultine et al. 2010; Shafroth et al. 2005). Accordingly, restoration practitioners in the Southwest have been embracing the natural-processes approach to large rivers as potentially more cost-effective in establishing and sustaining a native-dominated, diverse, and productive riparian ecosystem (Shah et al. 2007).

Water, sediment, and seed availability along with the correct fluvial-geomorphic context are key factors in a natural-processes approach to restoration. In lowland rivers of the southwestern U.S., one of the crucial elements is creating the right environment for the reproduction of cottonwood (*Populus deltoides* and *P. fremontii*) and willow (*Salix gooddingii*, *S. amygdaloides*, and *S. exigua*) stands and doing so at the right time (Stromberg 1997; Bhattacharjee et al. 2006; Taylor et al. 2006; Bhattacharjee et al. 2008). This calls for sites that can be flooded during spring runoff which also have a substrate conducive to germination of the current seed crop, followed by rapid to moderate drawdown, but a sufficiently shallow water table to ensure establishment and sustainability of the young trees and shrubs (Taylor et al. 1999). The desire is also to create a degree of geomorphic complexity which leads to the development of a complex mosaic of communities, not just monotypic cottonwood and/or willow stands (e.g., emergent wetlands, meadows, etc.) (Weisberg et al. 2013). Lastly, we want to reinitiate the natural processes within current constraints so as to generate a cascade of positive effects in the reach as well as on-site processes that lead to a dynamic, sustainable river ecosystem in the long term.

To test the efficacy of this natural, process-based approach in a large regulated river, we initiated in 1998 a demonstration project known as the Albuquerque Overbank Project (AOP) located on a Russian olive-dominated point bar in the urban Albuquerque reach of the Middle Rio Grande in central New Mexico (Fig. 1). The Middle Rio Grande has been highly modified over the past century—channelized with flows controlled by a large dam along with extensive diversions for agriculture and drinking water (Figs. 2 & 3). Despite this, many of the essential elements of functionality and biodiversity are still extant, and can be used to advantage in restoration. There are still spring peak flows, albeit smaller, driven by snowpack (Fig. 4a). Later in summer there are sediment-laden flows fed by tributary washes (arroyos) during summer storms, and, in non-drought years, a more or less consistent base flow from year to year (Fig. 4b). This base flow supports dense vegetation on point bars and islands, and an extensive riverside cottonwood forest that is the largest in the Southwest. Our goal was to take advantage of those remaining attributes by clearing the site, physically lowering portions of the bar, and engineering channels and islands to create geomorphic heterogeneity that would facilitate overbank flooding along with sediment deposition. In the process we hoped to create a seedbed that took advantage of the seed availability of native cottonwoods and willows within the local riverscape for establishing native vegetation.

At the time of the manipulation, we installed a high-resolution monitoring grid for vegetation that has provided one of the longest records of vegetation change in a riparian zone in the Southwest (16 years to date). At that time, we recognized the need for a long-term monitoring platform to really understand the dynamics of the system beyond the typically limited window of restoration-project monitoring. Palmer et al. (2005) have proposed five criteria for evaluating ecologically successful river restoration: 1) a guiding image of the dynamic state (a dynamic, ecological endpoint identified); 2) improved ecosystems; 3) increased resilience; 4) no lasting harm; and 5) completed ecological assessment. Follstad Shah et al. (2007) followed up with specific recommendations for practitioners in the southwestern U.S. We report here on some of the insights and surprises this relatively long record has provided with respect to natural-processes restoration and how well we met the goals outlined by Palmer et al. (2005) and Follstad Shah et al. (2007).

Methods

Site

The AOP site is located on the west bank of an urban reach of the Middle Rio Grande through Albuquerque, in central New Mexico (see Fig. 1). The manipulation occurred on the lower end of a point bar where the river changes from south to south-west trending (Fig. 5). The river at this location had a shifting, sandy bed at approximately 2 m (6.6 ft) below the bank edge. The point bar as a whole forced a localized narrowing (75 m; 250 ft) and deepening of the river channel at the upper end which then opened up again at the lower end to approximately 125 m (410 ft). The restoration site was approximately 2.5 ha (6 acres) and averaged 50 m in width along a channel length of about 400 m.

Climatically, the AOP site is located in a semi-arid zone where annual precipitation ranges from 83.5 mm to 403 mm (3.29 to 15.8 in) with a mean of 219 mm (8.67 in) as reported at the Albuquerque International Airport (290234), due east approximately 5 km (3 miles). About 50% of the precipitation arrives during the four-month summer "monsoon" season (June-September). Precipitation varied widely during the project period (Fig. 6). Key events were the above-normal summer amounts at the initiation of the project in 1998 and 1999, followed by exceptionally low amounts in 2000 and 2003 (the latter part of the region-wide extreme drought of 2002-03). Following 2003, the summer and winter precipitation was at or above normal (with the exception of 2011). In 2013, summer precipitation was above normal, but came late in the growing season, after our monitoring period.

Hydrologically, Middle Rio Grande flows are determined by water passing through Cochiti Dam (approximately 50 km (30 mi) upstream), by discharges from a relatively small tributary (the perennial Jemez River), and by several large ephemeral washes that contribute sediment and water, primarily during the summer rainy season (see Fig. 4b). Flows in the Middle Rio Grande are also significantly affected by diversions for municipal drinking water and agriculture, but some of the diverted water may be returned to the river by agricultural drains and municipal waste water. With respect to the project period (1998-2013), river discharges were essentially normal in 1998 and 1999 with peak releases occurring in late spring following snowmelt and relatively steady summer base flows (see Fig. 4). But beginning in 2001, spring and summer releases began a downward trend in response to the regional drought conditions that continued through 2003. In 2004, spring runoff increased and peaked in 2005 along with good flows in 2007-08, but summer base flows remained low through 2013.

Treatment

In the winter of 1997-98, the stand on the southern end of the bar was mechanically removed by root plowing and the woody materials taken off site (Fig. 7). To set the stage for restart, a portion of the cleared bar was lowered sufficiently to allow overbank flooding during typical spring releases from the upstream dam. Side channels and small islands were engineered in the lowered bar to slow flood waters and aid sediment deposition, and to bring water to the back portion of the floodplain to create further complexity. Adjacent to and downstream of the lowered bar, an artificial bar at the same height was created from the fill from the lowering. In addition, where possible we left in place remnant, large woody debris (a few stumps and snags) to further aid sediment deposition.

In March 1998, mechanical treatment was initiated whereby the nearly monotypic, mature Russian olive stand was cleared and root-plowed with heavy equipment along approximately 200 m (700 ft) of a bar adjacent to the channel (Fig. 7). The vegetative material was shredded and disposed of off-site. Following clearing, bulldozers lowered a portion of the bar (5.9 ha; 2.4 ac) adjacent to the river by approximately 0.66 m (2 ft) to facilitate overbank flooding during spring runoff events, and channels were engineered to bring water to the back of the bar, adjacent to the forest. Additional channels and intervening island bars were excavated into the new bar to mimic conditions that might be found on a natural bar following flooding, and that would create possible microhabitats for successful cottonwood and native plant regeneration. Together the channels and islands are referred to as the Lowered Bar zone. Soil material from the lowering was deposited on a sand shoal downstream and adjacent to the lowered bar to create the Bar Fill zone, and the entire lowered area blended and leveled to a height approximately 0.5 m (1.5 ft) above the channel bed. An area of about 0.4 ha (1 acre) of cleared bar was not lowered and, hence, not subject to overbank flooding under normal circumstances. This is the High Bar zone, which served as a comparative site, representing the more standard restoration practices of removing exotics without flooding (but typically followed by cottonwood pole plantings). The total amount of material moved was approximately 6,100 m^3 (8,000 yds^3). All work was performed in dry weather and without entering the active channel, at a cost of approximately $5,000/acre (equipment and labor).

As the mechanical treatment did not remove Russian olive and other exotics in surrounding areas, limited re-sprouting as well as new recruitment did occur. In 2003, the Bureau of Reclamation, in cooperation with the City of Albuquerque and the U.S. Forest Service, conducted spot treatments (utilizing low-impact, selective herbicide application) of young Russian olive, saltcedar and Siberian elm (*Ulmus pumila*) to further control their regrowth or re-establishment. Smaller diameter trees were sprayed with Garlon 4 herbicide only, while larger diameter trees were cut down by hand and the stumps treated with Garlon 3A. All vegetative debris was chipped and disposed of off-site, but smaller, standing dead vegetation was left in place. In February 2005, a second treatment of exotics, primarily Russian olive, occurred.

Biological monitoring

To monitor tree recruitment and vegetation composition and abundance, a grid system of one-meter quadrats was established across the site at 12.5-m intervals (see Figure 2). The grid system was oriented in the cardinal directions and was installed from a random starting point using GPS at the same time as the topographic survey was conducted. A total of 153, 1 x 1-m quadrats were originally established across the site on this grid. In 1998 and 1999, all tree regeneration was recorded by height class and the percent cover of vegetation by major growth form estimated in 10%-cover classes (shrubs, grasses, and forbs). In 2000, the system was refined to estimate cover on a species basis for all plant species within the quadrats, and the tree counts expanded to a full census of all trees over 25 cm on the 12.5 x 12.5 grid squares. This sampling method was used in 2001, 2002, 2007 and 2013. The number of quadrats was reduced by attrition, with 124 quadrats remaining in 2013. In 2013, all of the remaining grid stakes were replaced with rebar when necessary, re-tagged and surveyed with a handheld GPS unit.

Two additional sets of monitoring lines were established in 2003 on a new bar (New Bar zone) created by sediment captured by a large snag below the manipulated bar. In 2004, these two lines were expanded into two five-meter interval grids of 17 sample points, for a total of 34 sample points within the new bar. The sampling grid design differs from the grid system used elsewhere within the AOP site. In 2003-2005, percent cover was estimated for all species within 1 x 1-m quadrats on the grid, along with counts of woody individuals by size classes. In 2013, the tree census and herbaceous species cover on the lower bar was completed following the aforementioned protocols. However, on the new bar the tree-sample grids were for 5 x 5-meter grid squares.

Specifically, the tree census data include the diameter at the root crown class and height class of each individual per species per square. In 2013, tree species were also each monitored for condition, live or dead. Individuals with signs of treatment or beaver herbivory were also tracked.

Herbaceous species canopy cover was estimated per species and as a combined group of graminoids and forbs in each 1 x 1-m quad. Estimates were to the nearest percent when covering more than 1%, and to the nearest fraction of a percent when covering less than 1%. Ground surface cover was estimated separately, using categories including soil, moss, lichen, gravel, rock, water, wood (larger than 2 inches in diameter), litter and/or the basal area of living vegetation. Stem and individual counts for trees and shrubs within size classes were completed within each quad. Quads were read to the north of the line and east of the rebar.

Vegetation sampling across all years occurred during the mid-to-late summer. In 1998 and 1999, there were multiple sampling dates each year, but from 2000 onward, sampling occurred only once during the growing season (late June to September). Voucher specimens were collected for all but the most common species, identified, and deposited in the University of New Mexico (UNM) Herbarium. Naming conventions follow the USDA PLANTS database (http://plants.usda.gov/). All data were entered into an MS Access database following data quality-control protocols.

For analysis, sample points were grouped into zones that correspond to the major geomorphic settings on the bar and restoration treatments (see Figures 2 and 5). In the High Bar zone, Russian olives were root plowed and root raked, and remnant older overstory cottonwood trees were preserved. The Lowered Bar zone represents the engineered areas where the bar height was lowered by a bulldozer to encourage overbank flooding and where channels and islands were constructed. The Bar Fill zone was the site that primarily received fill materials from the Lowered Bar zone upstream. Analysis consisted of T-tests and repeated measures ANOVA stratified by zones (SAS 2010). The GPS locations and classifications of all sample points are provided as Supplementary Material.

Results

Flooding and geomorphology

Overbank flooding across the lowered bar and bar fill zones occurred between 1998 and 2013, but the High Bar terrace was never flooded (although a lower depression along the northern edge became saturated at peak discharges). The first flood occurred from late May into early June 1998, with the peak discharge exceeding 4,000 cfs (113 cms) as measured at the Albuquerque gauge. Based on *post facto* backwater modeling of water and vegetation elevations, overbank flooding at the site likely occurred at between 2,750 and 3,000 cfs (79 and 85 cms). Regardless, this was below the design estimate of 5,000 cfs. The initial flooding lasted at least 10 days and inundated all of the constructed channel areas and bar surfaces (Fig. 8a). The site flooded again in 1999 on several occasions from late May into late June. By then, vegetation cover was already high enough to help stabilize the site and modify flood-water dispersal (Fig. 8b). There was no flooding in 2000, despite above-normal precipitation, but during the drought year of 2001, a two-day flood occurred in mid-May that inundated the entire constructed bar surface. Overbank flooding occurred again on the lowered bar in 2005, 2008, 2009, and 2010 leading to slow filling of the channel and buildup of the island bars plus extensive deposition at the distal end of the constructed back channel.

With respect to erosion, the first flood event (1998) removed a constructed island and a portion of the main constructed channel at the upper end of the site. There was also ongoing erosion along the entire bank that was accelerated during the 2001 flood. A large snag, consisting of a portion of a cottonwood stump and roots, approximately 6 x 3 ft, was originally on the bankline near the center of the New Bar area. It limited erosion where it was located, but with the successive floods its location shifted downstream. Estimated total sediment losses over the five-year period between 1998 and 2002 were between 3,360 m^3 and 4,000 m^3 (Ortiz et al. 2002). But while this material was being removed, deposition was occurring at the downstream end of the bar below the new point where the snag came to rest in the 2001 flood (Fig. 8c). Sediment deposition behind the snag created a new bar in this area and provided new habitat for vegetation establishment.

Tree establishment and growth

Following the initial overbank flooding of May 1998, there was a significant cottonwood germination event that generated a carpet of young seedlings by the end of the growing season. Following the 1999 flooding and growing season, there were more than 10,000/ha established trees, but establishment patterns differed across the bar zones (Fig. 9). Densities on the Lowered Bar were nearly seven times that found in the Bar Fill Zone, and little or no establishment occurred on the High Bar that was out of reach of the flood. The highest densities occurred in and along the constructed channels and, to a moderate degree, on the island between the channels. Lowest densities were in areas characterized by flooded deposits over the bar-fill materials. In subsequent years densities declined and by 2007 only 25% of the trees remained on the Lowered Bar and 5% in the Bar Fill Zone. There were limited increases on the High Bar that occurred mostly in saturated areas. In 2013 there was a small uptick in numbers in areas that had been flooded in the intervening years, but there was no direct sampling in that period to confirm the year of establishment (an age sampling of saplings indicated that some establishments had occurred between 2007 and 2013). Also during this period a new cohort of trees became established beyond the fill zone on the accrued sand deposits that formed behind large woody debris at the site's distal end (New Bar). The estimated density across these sites was 1,128/ha as of 2013.

With the exception of the High Bar and Bar Fill zones, exotic-tree densities were significantly lower through time than native trees. The prevalence on the High Bar is attributed primarily to resprouting remnant roots, and this may be the case in the other zones as well. Russian olive was the most abundant, but saltcedar, mulberry (*Morus alba*), and Siberian elm were also present. Densities and sizes of individuals continued to increase through 2002, when numbers were then twice that of 1999. In the fifth year, 2003, and again in 2005 spot retreatments reduced the numbers down to approximately the 1999 level. Numbers have risen again since 2007, but at a slower rate, and in 2013 densities on the Lowered Bar and High Bar were still at only about 35% of their 2002 levels and 54% in the Bar Fill zone. Introduced species had made some headway on the New Bar sites (320/ha), but cottonwoods still outnumbered them at a ratio of 3.5:1.

The stand structure of cottonwood through time mirrored its overall density trends on the site. On the Lowered Bar, the majority of the new establishments had reached the one-to-two-meter or more sapling stage at year 3 (2000) (Fig. 10). While overall numbers declined in the ensuing years through self-shading and beaver herbivory, the majority of the cohort continued to move into larger size classes. Ten years after establishment (2007), over 875 trees/ha were between three and five meters tall. But by 2013, there was little or no recruitment into large size classes. Instead there was a renewed peak in the one-to-two-meter class, reflecting possible new establishments but also the impact of beaver herbivory. Across all zones 47% of the living trees were browsed and resprouting (Fig. 11). Beaver also showed a preference for cottonwood although about 33% of the introduced species, mostly Russian olive, were also browsed.

Beavers have significantly impacted cottonwood growth and stand development. Nearly half of the established cottonwoods have been browsed by beaver, while only about a third of the exotics, mostly Russian olive, were. Many cottonwoods were killed outright, but most browsed trees are resprouting from multiple stems at the base (the majority also remain less than 2 m (6 ft) in height). Many larger trees are being felled preventing the development of full-canopied woodlands.

Vegetation diversity

With a few exceptions, native shrubs, graminoids (grass-like plants), and forbs have dominated the site with respect to species richness and cover. While community composition has shifted through time, cumulative native species richness has only continued to increase. By 2013, 148 native versus 26 introduced species had been recorded, and through the years the percentage of native species has remained more or less constant at about 85%. Overall, forbs accounted for about 50% of the species, followed by graminoids at 40%, with trees and shrubs making up the remainder. Native species richness was highest on the Lowered Bar with 40 or more species present in any given year except the drought year of 2013, and at about three times the number of introduced species (Fig 12). The other zones, while less rich, were still predominantly native in composition.

As with the trees, bar zones were important with respect to abundance of shrubs, grasses and forbs. Among shrubs, obligate and facultative wetland native species were the overwhelming dominants led by coyote willow (*Salix exigua*) and false willow (*Baccharis salicina*). They were particularly prevalent on the Lowered Bar, remembering that saltcedar and Russian olive were considered trees here (Fig. 13). Cover increased steadily, reaching a peak in 2007 and dipping in 2013. In the Bar Fill zone, cover also increased as coyote willow clones expanded from the bar edge into the main part of the zone.

A similar trend was seen among graminoids, with obligate and facultative wetland species such as sedges (*Carex emoryi*), common threesquare (*Schoenoplectus pungens*), inland saltgrass (*Distichlis spicata*), and vine mesquite grass (*Panicum obtusum*) dominant in the channels and islands of the Lowered Bar zone (Fig. 14a). Even in the usually depauperate Bar Fill zone, grass cover, led by inland saltgrass, increased through time approaching that of the other sites. Some introduced species also increased in cover and include redtop (*Agrostis gigantea*), Johnsongrass (*Sorghum halepense*), and bermudagrass (*Cynodon dactylon*). Beginning in 2007, Ravenna grass (*Saccharum ravennae*) has increased in cover significantly and came to dominate the northern portion of the Lowered Bar in 2013.

Forbs were abundant in the early years on the High Bar and Lowered Bar but significantly dropped off in cover by 2007 and 2013 (Fig 14b). Native perennials in particular have declined since 2007, e.g., horseweed (*Conyza canadensis*), western goldenrod (*Euthamia occidentalis*), and Canada germander (*Teucrium canadense*) declined 50-90% depending on location. Some species, such as smooth horsetail (*Equisetum laevigatum*), a native facultative wetland species, increased in cover. Introduced perennials such as yellow sweetclover (*Melilotus officinalis*) and field bindweed (*Convolvulus arvensis)* were variable depending on year and sampling zone, but in general they also declined from 2007 onward. Annuals, as might be expected, were even more variable. For example, prairie sunflower (*Helianthus petiolaris*) dominated several sites in 2001, but was found as only scattered individuals in 2007 and beyond. Rough cocklebur (*Xanthium strumarium*) abundance peaked in 2002 but was nearly absent by 2007. Common ragweed (*Ambrosia psilostachya*) was abundant in both 2001 and 2002 and was still common in the High Bar zone in 2007, but uncommon elsewhere. Kochia (*Kochia scoparia*) and Prickly Russian thistle (*Salsola tragus*), both introduced annual species, reached peaks in cover in different places and times: 2002 on the High Bar (15%), and in the Fill Zone in 2007 (31%), but were non-factors in other zones and years. By 2013, native and introduced species cover was not significantly different on the High Bar and Lowered Bar sites despite the greater number of native species. In contrast, on the New Bar, introduced species were particularly low in cover, but herbaceous cover declined overall as tree and shrub cover increased (Fig. 15). Regardless, even with declining abundance, forbs continued to account for 50% of the species richness, as they had in all years.

The construction of topographic features such channels and islands and overbank flooding led to increased vegetation patch diversity. Beginning with bare ground in 1998, over the years several early successional communities came and went. The site is now a fine-structured mosaic of young cottonwood stands, willow thickets, saltgrass meadows, and herbaceous wetlands. This composition and structural diversity was in striking contrast to the adjacent nearly uniform mature cottonwood forest that dominates the river corridor.

Discussion

Cottonwood forest development

After 16 years, cottonwoods and other native species continue to dominate on a restoration site designed to take advantage of the remaining natural-processes potential, particularly, overbank flooding, in a highly regulated river system. This is in keeping with other studies which show that given a sufficient flooding regime, cottonwood regeneration can be significant in western streams and rivers (Rood et al. 2003; Bunting et al. 2013). The site's physical heterogeneity led to differing residence times of flood waters on various surfaces and subsequent depths to groundwater following flood recession. Accordingly, tree and vegetation establishment was not uniform across the site. Slow drawdown times over a month or so are thought to be a critical element of successful cottonwood regeneration on flooded sites (Stromberg 1997; Bhattacharjee et al. 2006; Taylor et al. 2006). Accordingly, the highest cottonwood densities were on the Lowered Bar, particularly along the constructed channels and on the lower islands, areas with the longest residence time (two to three weeks). Sites with low tree numbers still had high native phreatophyte shrub and herbaceous cover reflecting sufficient moisture availability (willows and saltgrass).

In contrast, The Bar Fill zone composed of mechanically deposited sediments remained problematic. Even after 16 years, the majority of this zone was still barren and saltcedar was more prevalent here than elsewhere on the bar. Neither salinity nor water availability were an issue. While some cottonwood trees have become established along with expanding clonal willow shrublands from the river edge of the zone, the general depauperate nature of the zone suggests that how fill materials are handled may affect the restoration effort and should be considered carefully in a restoration plan. Mechanically mimicking how sediments are laid down by a river remains a challenge in need of more experimental attention.

The surprise came in the New Bar zone where sediment accumulated naturally behind large woody debris that was purposely left after the initial site clearing. Herbaceous communities quickly became established followed by willow shrublands and now tall cottonwood stands dominate these sites after only ten years (the trees in the New Bar zone are taller than those on the older Lower Bar zone). This was an unplanned outcome of the project, which bolsters the concept that having large woody debris available in large river systems is a key element for successful restoration (Abbe and Montgomery 1996; Montgomery and Piegay 2003), and supports the idea that the more natural the configuration can be made in restoration efforts the better.

Most of the decline in tree density can be attributed to self-thinning of dense stands and beaver herbivory. Beavers are endemic to the system, but because the river is now confined to a single channel rather than the historical braided, multichannel system, beavers no longer serve the same ecosystem engineering function of creating small dams on side channels that lead to backwater wetlands, ponds, etc. Rather, they are now primarily bank beavers living in dens along the river edge and then foraging onto the bar, acting primarily as consumers. We only began to track beaver herbivory quantitatively in 2013 in response to anecdotal accounts that beaver activity was increasing following the cessation of trapping in the early 2000's along the river corridor. While additional measures, mechanical and otherwise, could be taken to protect trees, the key question is what should be the final density of mature cottonwoods in a stand and whether those target densities can be met despite the beaver browsing. Provisional studies from mature 50 to 70-year-old forests in the reach estimate typical densities at between 200 and 400 stems/ha (Eichhorst et al. 2012), still below the current densities in the Lowered Bar Zone (800+/ha). Additional studies of stand structure and beaver herbivory rates are warranted to evaluate whether the cottonwoods will prevail as the restoration stands mature.

Both woody and herbaceous invasive species remain a chronic problem, but we would suggest it is less so using this natural-processes orientation than with other types of intensive intervention strategies. Russian olives and other introduced woody plants were steadily increasing after the manipulation, but at a slower rate than expected and natives remained dominant on all the sites except the un-flooded High Bar. Because densities were relatively low, the retreatment in 2003 was an inexpensive intervention compared to the initial treatment and much lower than the costs would have been to remove these invasives in the adjacent mature forests. A decade after retreatment, exotic species numbers remain 70% below the pre-treatment 2002 densities. As with beaver herbivory, the question is whether cottonwoods will out-compete the introduced species over the long term. Bunting et al. (2013) studied natural cottonwood recruitment following flooding for five years on an Arizona site and concluded the cottonwoods had a competitive advantage over saltcedar. Currently at AOP, cottonwood numbers suggest that it will be possible for the young stands to grow into a mature forest of some sort (sans beaver and other impacts), but introduced woody species, particularity Russian olive, may still come to dominate the understory. Russian olive remains a threat in this context because it does not require flooding for successful germination and establishment (Shafroth et al. 2005; Reynolds and Copper 2010). Hence, to help ensure long-term success of a restoration program, we would suggest timely and low-cost spot treatments be built into restoration monitoring and management plans.

Biodiversity trends

Aside from establishing cottonwoods and willows, geomorphic manipulation and overbank flooding significantly enhanced local plant biodiversity. The cumulative plant species richness of 132 species on the manipulated and flooded bar at AOP was far higher than that found in the adjacent forests where the average richness was 10 (+/- 5) species (Milford and Muldavin 2004). This is most likely due to the local patch diversity generated by the designed geomorphic complexity at the AOP site, as well as the continued overbank watering of the site and the maintenance of groundwater connectivity for many species. Furthermore, the flooded site has been persistently dominated by native trees, shrubs and forbs throughout the study period, suggesting that as succession proceeds and the site matures, natives will prevail.

On a community basis, the AOP site is developing into an intricate mosaic of young cottonwood stands, wetlands, wet meadows, saltgrass meadows, and open ground that is in keeping with the recommendations of Weisberg et al. (2013) that riparian restoration look beyond just establishing trees to building into the restoration process a diversity of community types on a diverse fluvial geomorphic template. Moreover, the high richness and productivity of the overbank flooding site indicates that merely clearing a site and following up with cottonwood pole planting without flooding will leave out the majority of the biodiversity potential in a restoration. Overall, this points to the potential contribution that flooded, restored sites can make to large-river ecosystem restoration, not only in terms of riparian forest regeneration, but also in overall biodiversity.

Efficacy of this natural-processes riparian restoration project

The creation of a diverse patch mosaic was a central tenet of the Middle Rio Grande Biological Management Plan (Crawford et al. 1993), and was the guiding principle for this demonstration project. Accordingly, we would argue that our outcomes with respect to diversity of stands and species richness met the first of the five restoration evaluation criteria of Palmer et al. (2005): the identification of an ecological endpoint. We also think that our outcomes met, in part, the next three criteria. The project lead to an improved ecosystem (#2), at least at a local scale, and continued education about the project among Rio Grande restoration practitioners has led to applications of the natural process restoration paradigm at larger scales in the Middle Rio Grande. Has resilience been increased (#3)? If greater biological diversity is assumed to impart resilience and stability (*sensu* Hooper et al. 2005), then the persistence of species-rich, native-dominated vegetation communities at AOP would suggest yes. But new exotic threats such as Ravenna grass and the impacts of altered beaver dynamics still put the site at risk. Yet, we do not believe we have done any lasting harm (#4). The removal of Russian olive did take away that species' berry production as forage for wildlife, particularly birds, but that has been replaced by a greater diversity of forage elements among native shrubs, grasses, and forbs.

With respect to the last criterion: is the ecological assessment complete (#5)? Our results point to the value of long-term monitoring in understanding riparian dynamics as stated by Follstad Shah et al. (2007) and Bunting et al. (2013). Yet, it is still only a 15-year record and surprises like the invasion of Ravenna grass and heightened beaver activity point to a need for continued monitoring and assessment. But even within this research-oriented restoration project, there is a significant challenge to maintaining the monitoring grid beyond the initial five-year monitoring period. Each of the later monitoring campaigns was funded on an *ad hoc* basis by agencies. A promising alternative is to engage citizen science initiatives to continue crucial elements of the monitoring protocol (Palmer et al. 2007). To that end, AOP has been adopted as a Bosque Ecosystem Monitoring Program (BEMP) site, an outdoor education program that engages students from elementary schools up to college in the monitoring of the Middle Rio Grande ecosystem (Eichorst et al. 2012). BEMP will serve a dual purpose of collecting the needed monitoring data on the AOP site on an on-going annual basis while giving students a sense of the value of ecological restoration.

AOP continues to provide an understanding of restoration possibilities using natural processes in a lowland arid river system. Our results suggest that the AOP site can be a useful model for other large-river riparian restoration in that it has an abundance of young native trees, grasses, and forbs, as well as a diversity of habitats — all initiated by site manipulation to allow for overbank flooding at current flow levels, followed by low-intensity management. Restoration success, though, may be dependent on the timing and duration of flooding, the design of the constructed floodplain, groundwater connectivity, soil conditions, the availability of seed sources, and the subsequent adaptive management strategies implemented, including retreatments as necessary. Accordingly, we would recommend that additional AOP-style restoration sites be initiated in other lowland rivers to further refine overbank flooding prescriptions. But overall, the success of AOP in itself is encouraging for the prospects of restoring many of the compositional, structural, and functional qualities of riverscapes on regulated large-river systems—a potential for bringing, as Roseman and DeBruyne (2015) proposed, a "renaissance of ecosystem integrity in North American large rivers."

Implications for Practice [box]

- Mechanically manipulating a site to take advantage of remaining flow and sediments in a regulated river can lead to the development of a near-natural riparian zone.
- Designs should encompass a diversity of fluvial geomorphic landforms but avoid constructing elements that do not mimic natural processes.
- Retreatment of invasive species at a periodic but manageable level may still be required.
- Long-term monitoring is a must to fully understand and adapt to changing conditions on a restoration site.

Acknowledgments

This project and its reporting are dedicated to the memory of Dr. Cliff Crawford (1932-2010), the original project leader, whose guiding example remains a force for restoration in the Middle

Rio Grande of New Mexico. Funding was provided by a grant from the U.S. Department of the Interior, Bureau of Reclamation, Albuquerque Area Office through the Southern Colorado Plateau Ecosystem Studies Unit, North Arizona University, Flagstaff, Arizona (Assistance Agreement No. R13AC40011). Additional in-kind support was provided by the Middle Rio Grande Conservancy District, the City of Albuquerque-Open Space Division, the U.S. Army Corps of Engineers, and the U.S. Fish and Wildlife Service.

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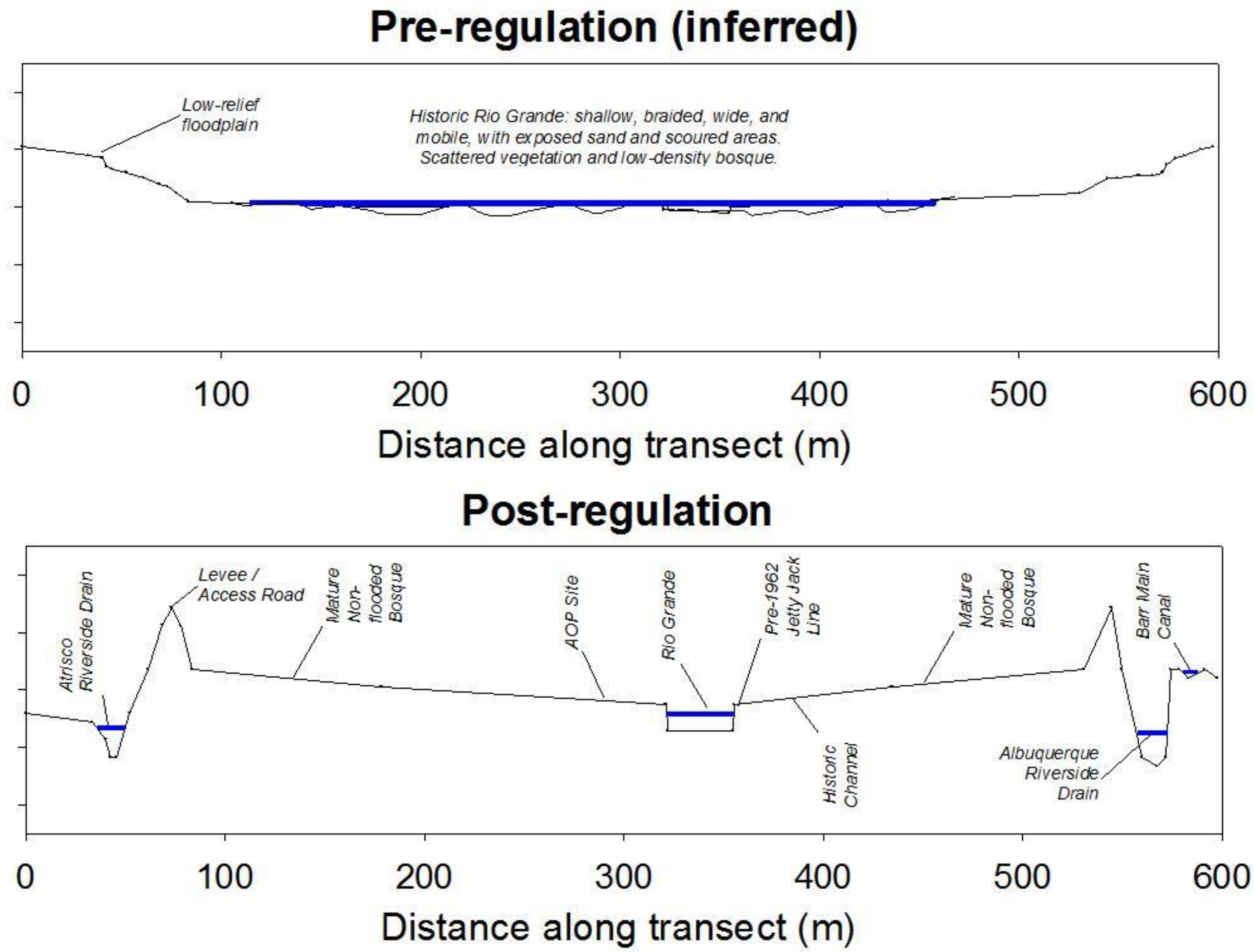
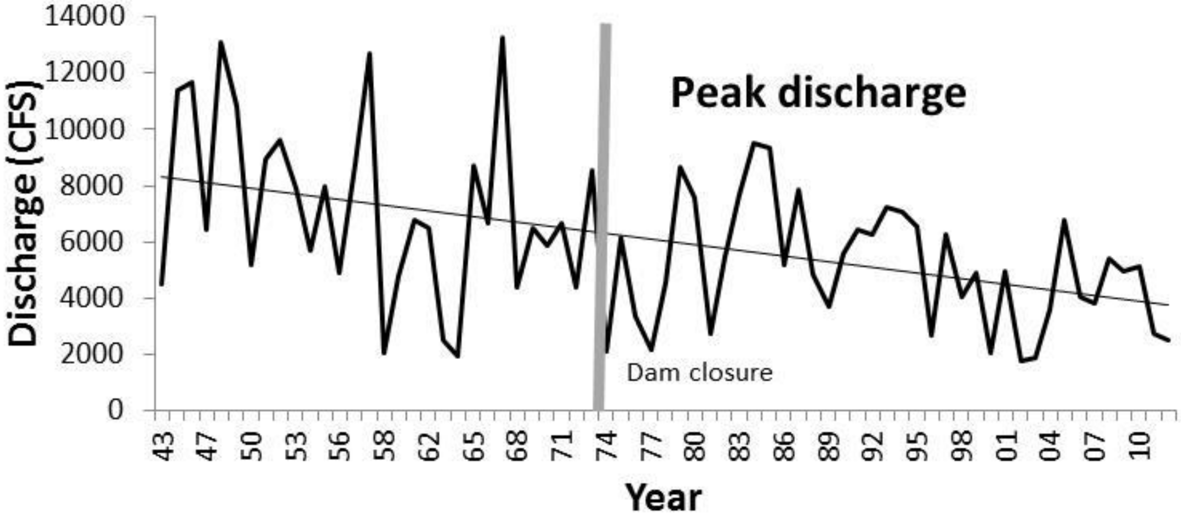


Figure 1. A schematic view of middle Rio Grande fluvial geomorphology pre- and post-regulation. Before regulation, the Rio Grande was a complex multi-channel braided system. After regulation, it was stabilized into a single-channel system bounded by levees (Ortiz, Shah, and Vinson (2002), University of New Mexico, personal communication).



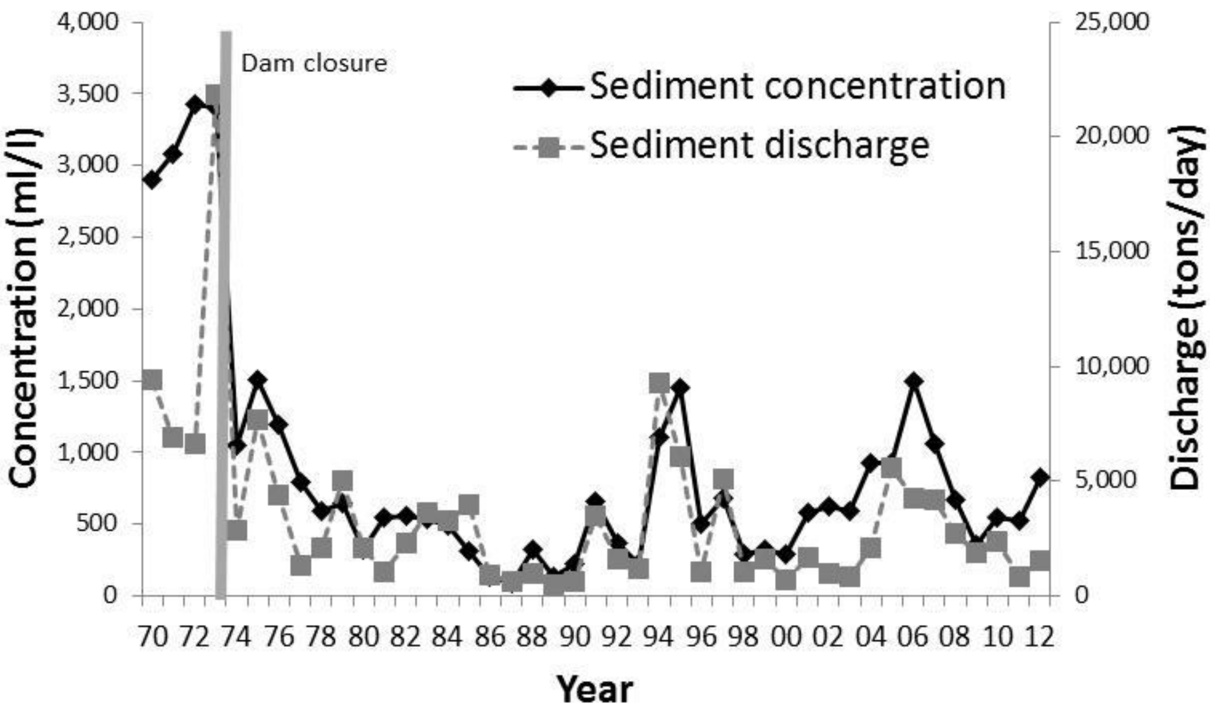


Figure 2. a) Annual peak discharges from 1943 to present at the Albuquerque gauge station (8330000), approximately 5 km (2 mi) upstream from the AOP site. Cochiti Dam was completed and closed in 1973 and peak discharges have declined since (Y= -66.375x + 8363.1, R^2 = 0.2195); b) sediment concentration and discharge since 1970 showing the steep declines since dam closure.

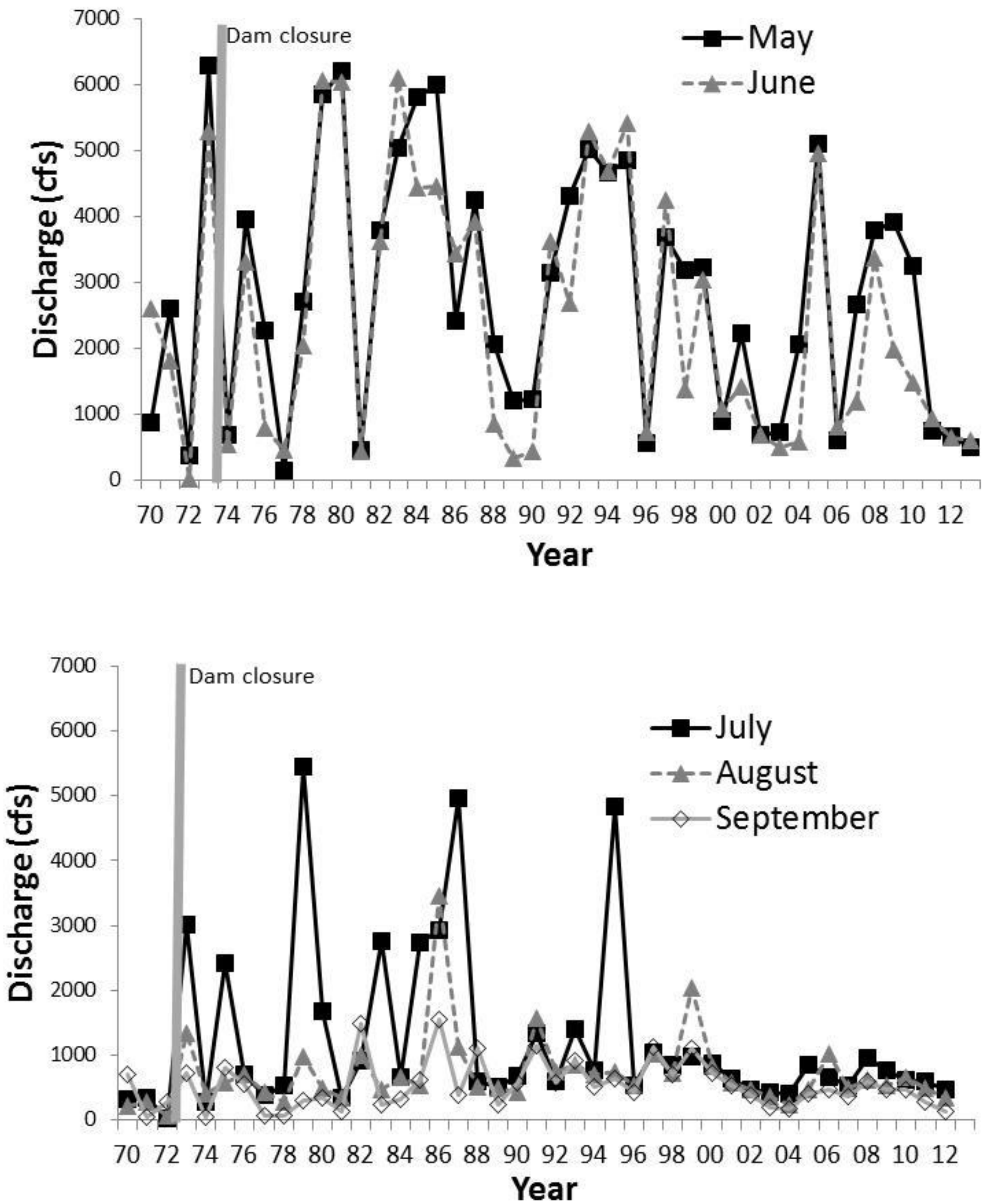
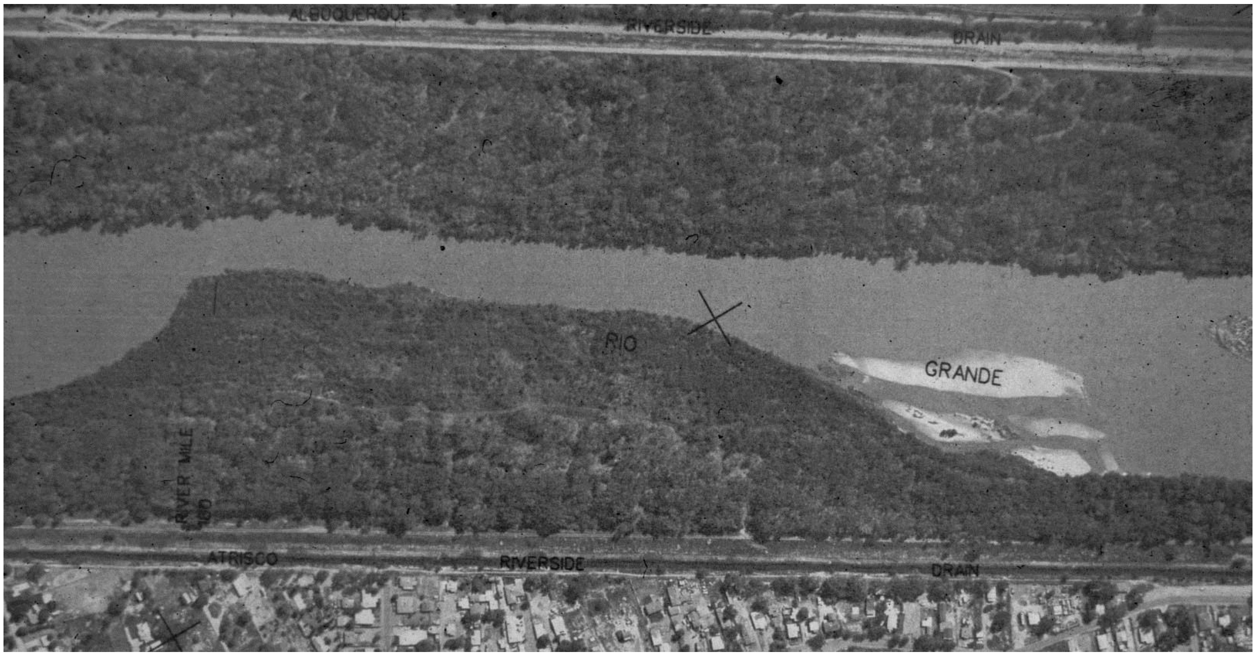


Figure 3. a) Average monthly spring-snowmelt period discharges in the months of May and June from 1970 to present; b) Average monthly discharges for the summer months of July, August and September from 1970 to present where flows are augmented by local monsoon thunderstorms (Albuquerque gauge station (8330000). Cochiti Dam was closed in 1973.

(A)

Figure 5.



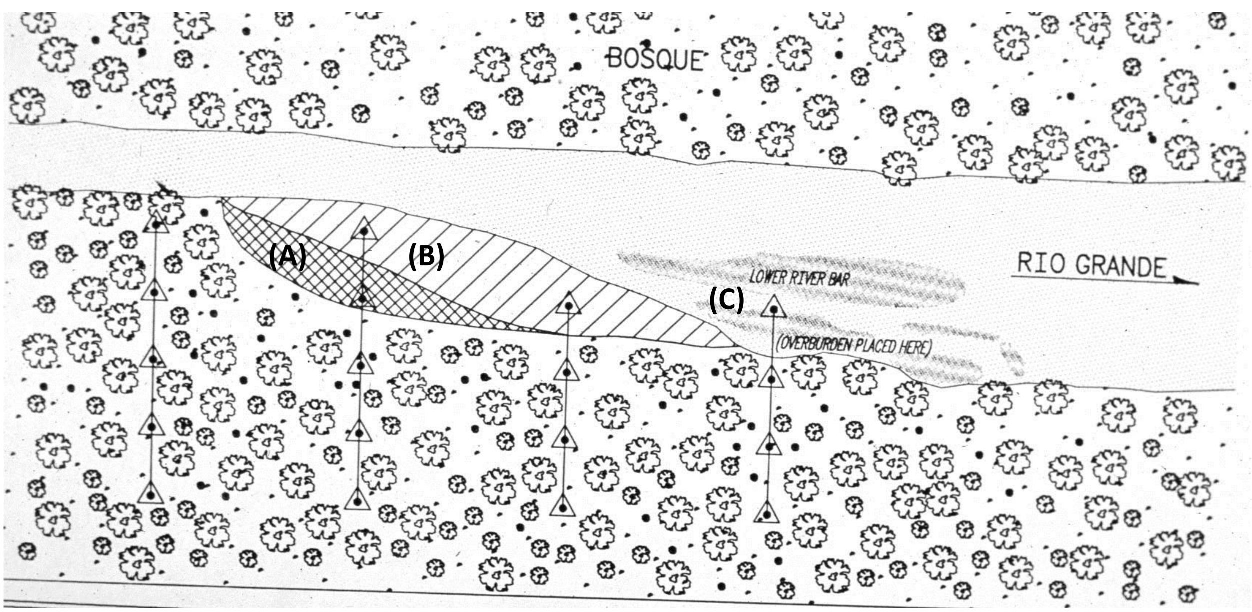


Figure 4. The design schematic for the AOP manipulation where (A) was the cleared but not lowered High Bar, (B) the cleared Lowered Bar, and (C) the Fill Bar that received overburden from the bar lowering of (B). Eighteen piezometer wells were also installed after the manipulation (noted as triangles in the drawing).

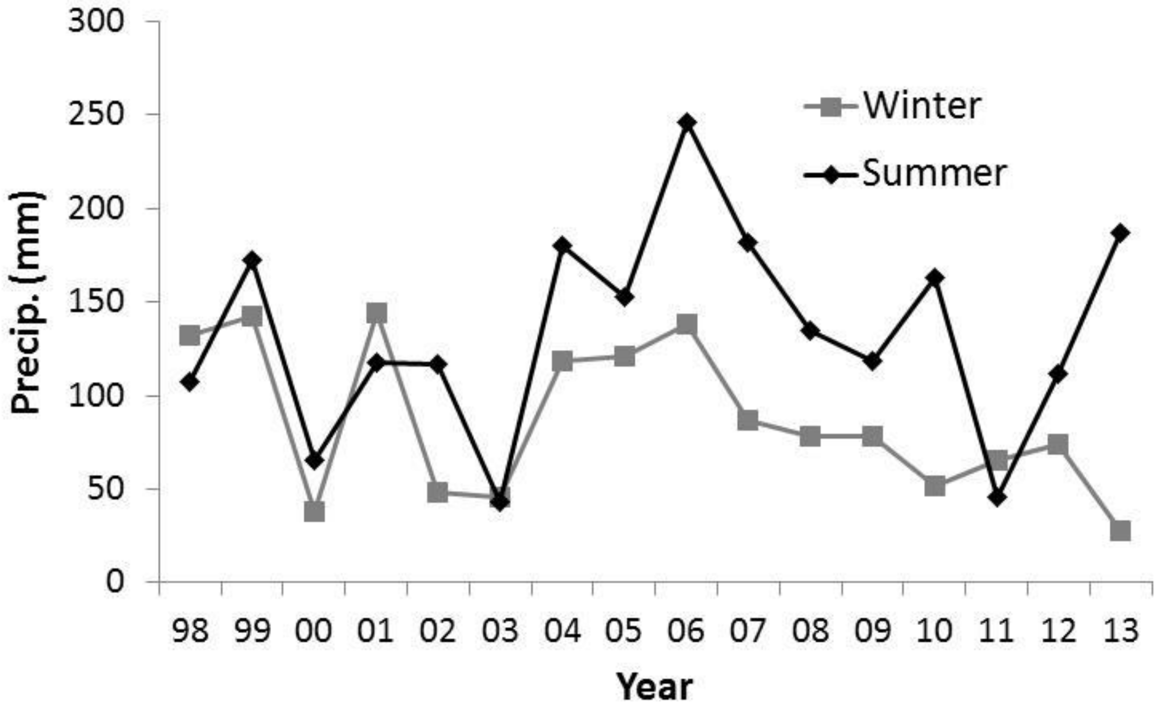


Figure 6. Mean summer (April-September) and winter (October-March) water-year precipitation as measured at the Albuquerque International Airport during the project period.

Figure 7. Mechanical manipulation on the AOP site prior to flooding: a) all introduced Russian olive was root plowed; b) the bar area adjacent to the river was lowered 0.6 m and channels and bars excavated; and c) the sediments from the lowering were deposited at the distal end of the bar.

Figure 5. The spring flooding events in 1998 (a) and 1999 (b). The flood flows moved a large snag downstream to the south end of the restoration site where it became an anchor for sediment accumulation and new vegetation development (c). This area is referred to as the New Bar zone.

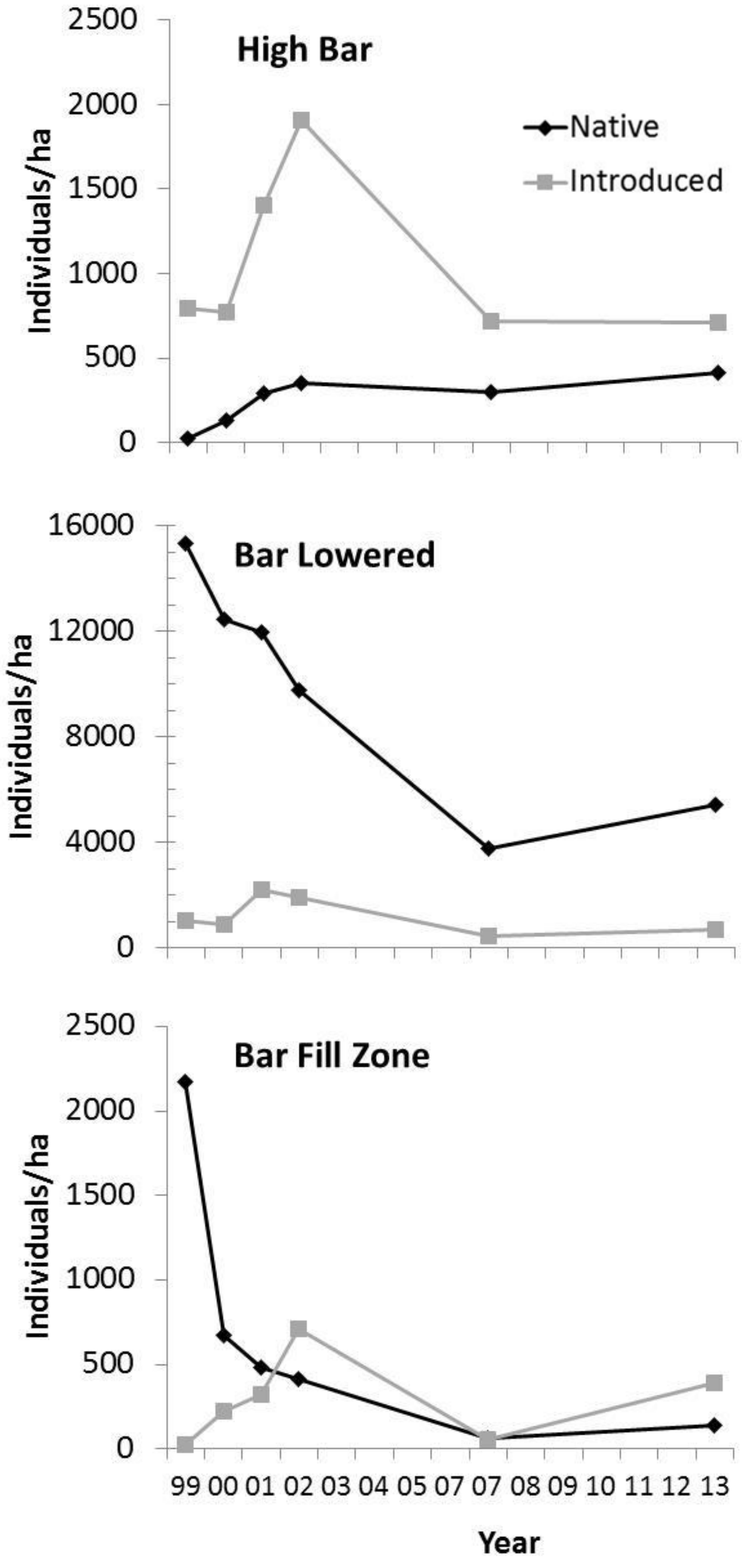


Figure 6. Mean yearly densities of native (mostly Rio Grande cottonwood) and introduced trees (mostly Russian olive) since the second growing season (1999) following manipulation (note: Y-axis scale differs between A, B and C). NEED TO LABEL GRAPHS A, B & C.

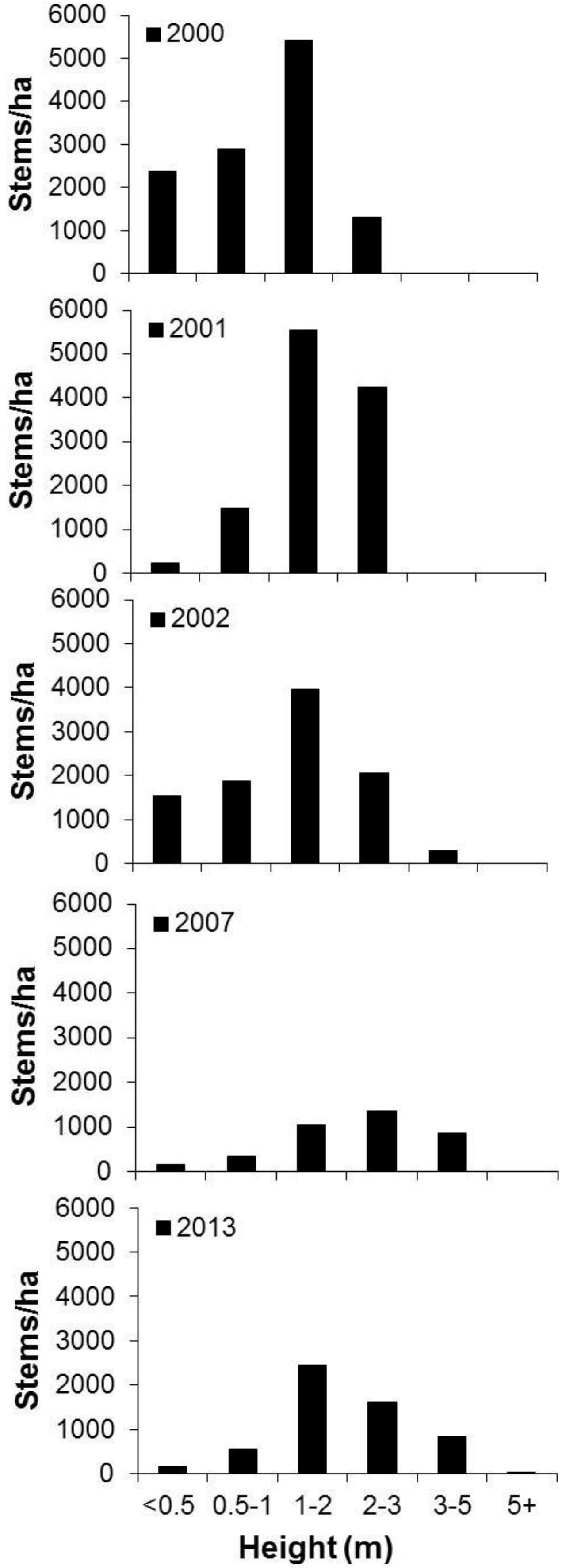


Figure 7. Rio Grande cottonwood stand structure based on a tree-height census that began in 2000, the fourth year of the AOP project.

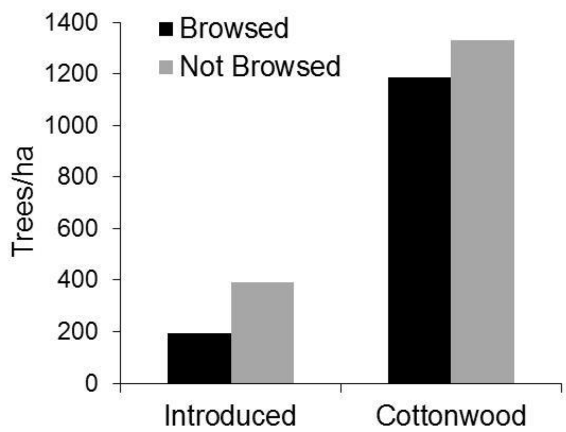


Figure 8. Beaver herbivory for Rio Grande cottonwood and combined introduced species (Russian olive, saltcedar, and Siberian elm) in 2013.

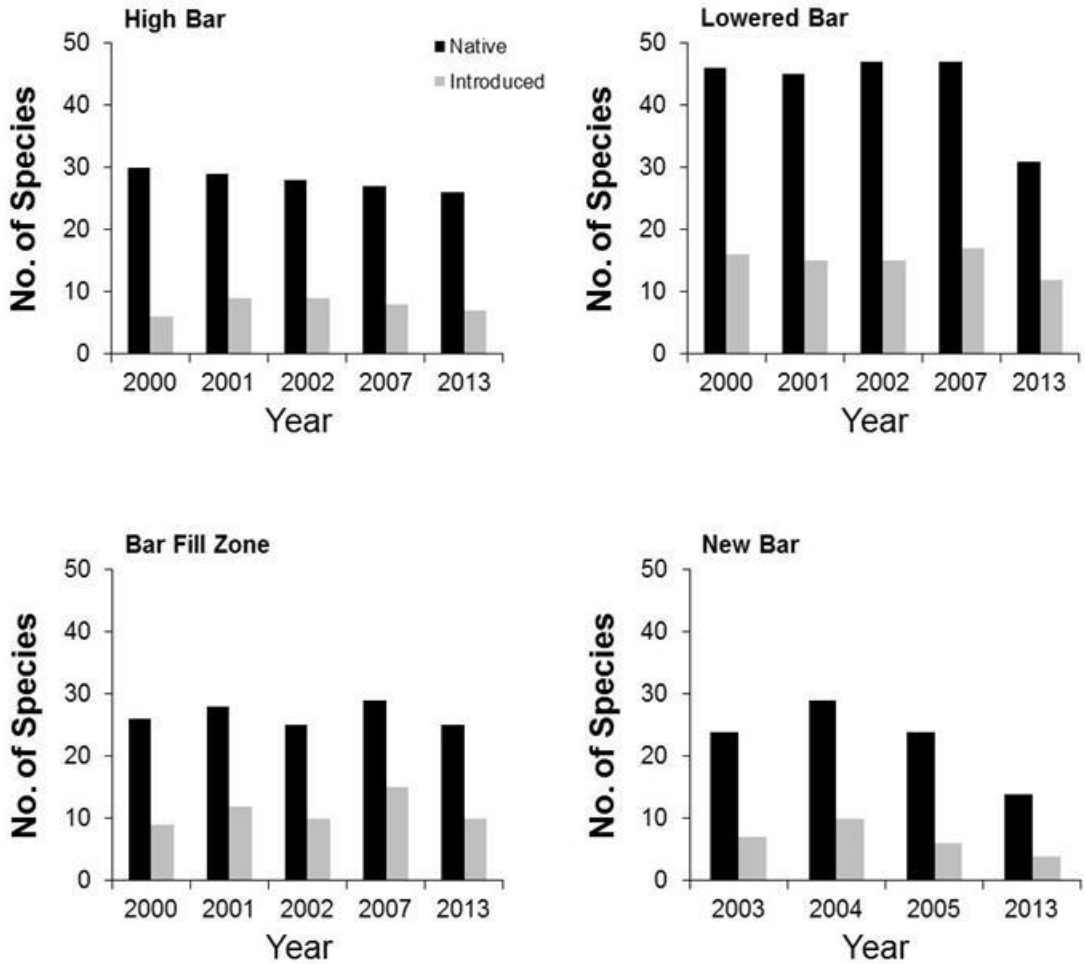


Figure 9. Native versus introduced plant species richness by sampling zone.

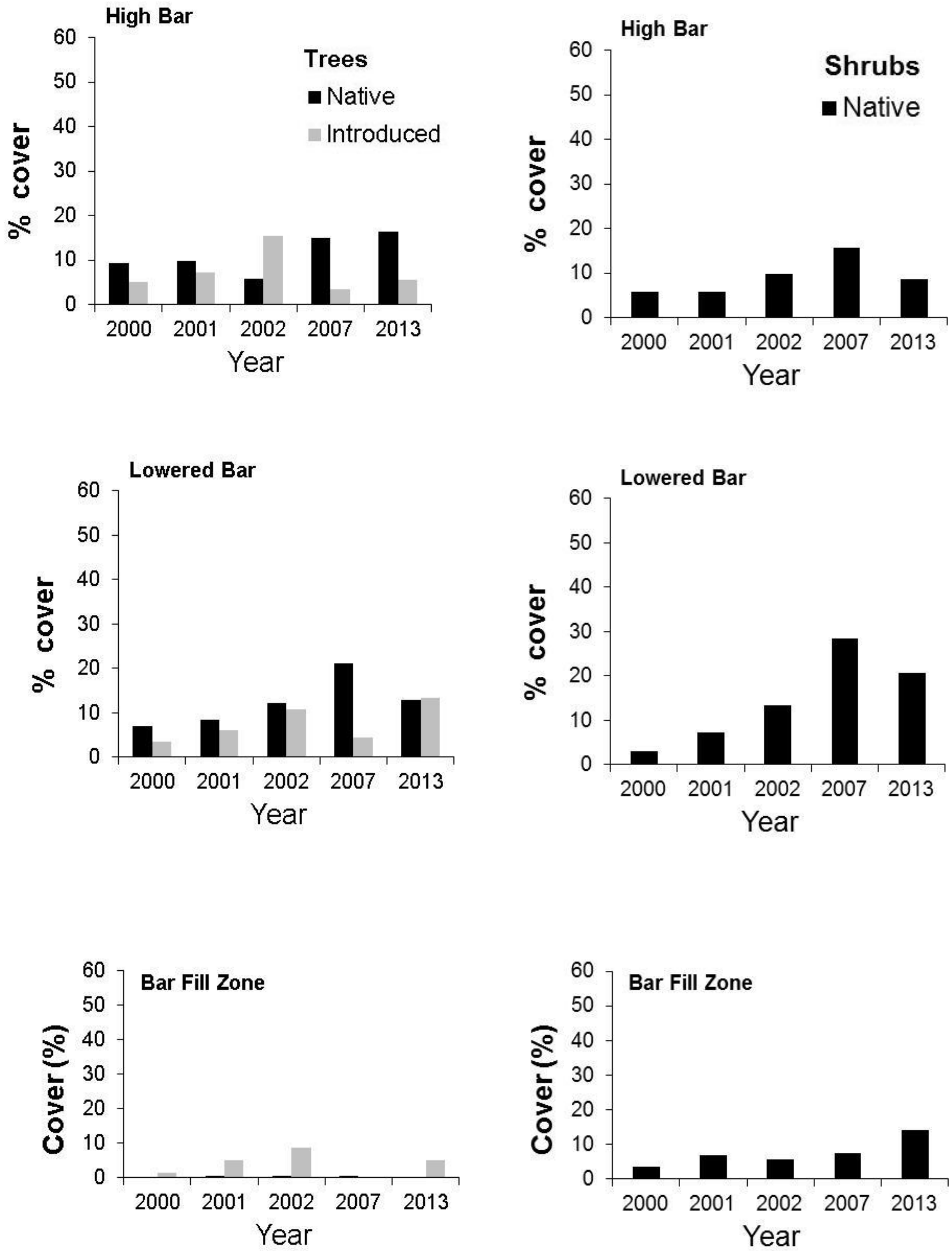


Figure 13. Percent canopy cover of native trees and shrubs by sampling zone (excluding the New Bar zone).

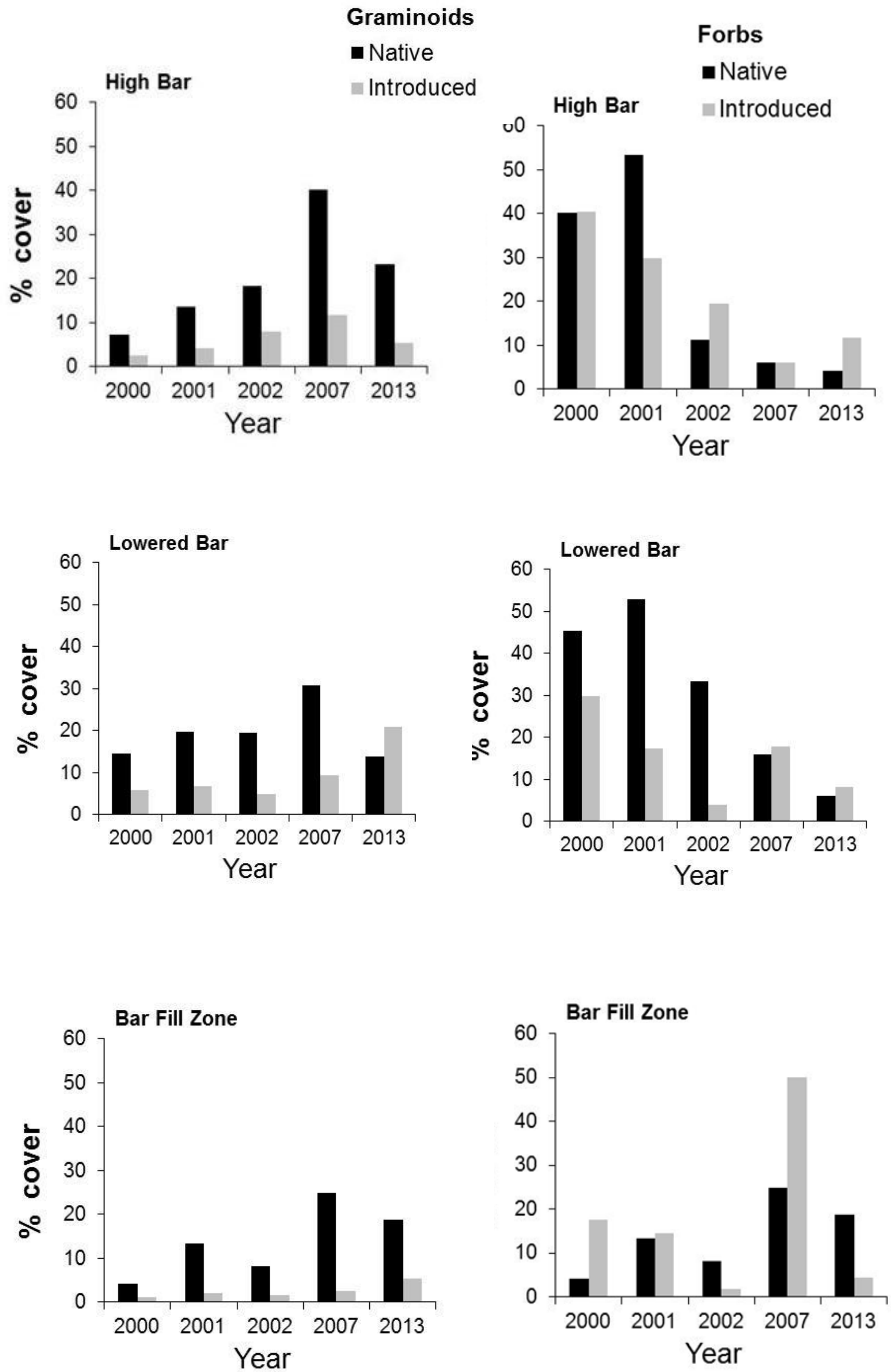


Figure 14. Graminoid and forb percent canopy cover of natives versus introduced species by sampling zone (excluding the New Bar zone).

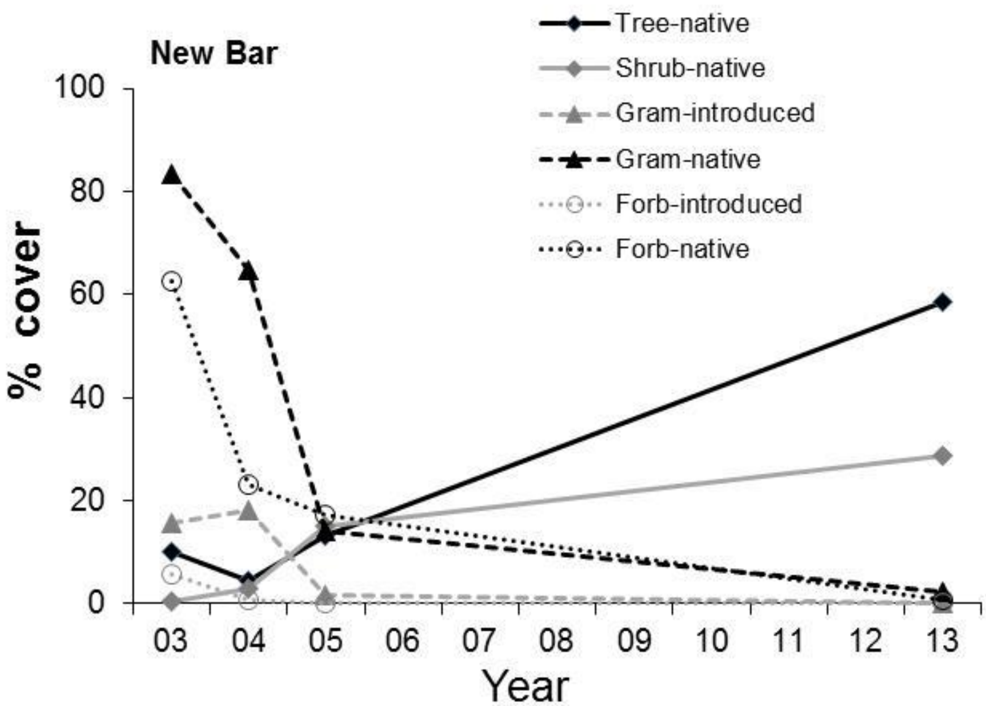


Figure 15. Canopy cover for trees, shrubs, graminoids (Gram), and forbs among natives and introduced species in the New Bar zone since 2003. 2003 is when the area began to develop perennial vegetation cover.