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Experimental restoration of mesic and wet forests in former pastureland, Kahuku Unit, Hawai`i Volcanoes National Park

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Abstract

The Kahuku unit of Hawaii Volcanoes National Park (HAVO) contains seven thousand acres of former forest that was converted to pasture for grazing cattle. There were several phases of forest clearing and pasture development (Parker Ranch 1912-1947, James Glover 1947-1958, and Damon Estate 1958-2000) creating an open pasture with scattered native trees and small remnant stands of native species. In 2005, methods to facilitate forest recovery were tested in four ungulate-proof exclosures (four hectares each). Within the exclosures, three temporary grass removal treatments (herbicide, soil turnover, and herbicide/soil turnover) were tested with the objective of finding a method that best promoted native forest recovery in conjunction with ungulate exclusion. In addition to monitoring plant recruitment from the natural seed bank in the soil, establishment by direct seeding and planting of native species in the different treatments was evaluated. By year one, rapid re-establishment of alien grasses occurred in all removal treatments, but was slowest in plots that received a combination of soil turnover and herbicide. Natural native plant recovery was evident in all grass removal treatments with a limited number of seedlings in the untreated grass control. Plant establishment from direct seeding for koa and pilo was highest in the combination soil turnover and herbicide treatment. No seedlings of *Pipturus albidus* (māmaki), *Cheirodendron trigynum* (‘ōlapa), *Coprosma pubens* (pilo), *Myoporum sandwicense* (naio) and very few *Acacia koa* (koa) and *Metrosideros polymorpha* (‘ōhi‘a) were observed outside of ungulate-proof exclosures. Planted seedling survival was moderate to high with no significant differences among sites and treatments (57-70%). Based on these results, temporary suppression of alien grasses in conjunction with ungulate exclusion can facilitate recovery of native species once abundant in the Kahuku region.

Introduction

The Kahuku Unit of Hawaii Volcanoes National Park (HAVO) provides important habitat to native species and links conservation lands in Kaʻū district to protected areas in the Kona district of the Island of Hawaii. There were several phases of forest clearing and pasture development (Parker Ranch 1912-1947, James Glover 1947-1958, and Damon Estate 1958-2000) creating an open pasture dominated by non-native grasses with scattered native trees and small remnant stands of native species. The restoration of approximately 3,000 hectares (7,400 acres) of former mesic to wet forest currently grazed by domestic cattle and impacted by wild moulfon sheep and feral pigs will perpetuate native biodiversity within the park and enhance regional conservation efforts. Today, only large old individuals of native ʻōhi'a and koa remain in the pastures along with small forest remnants that are slowing disappearing under continued pressure from ungulates that trample or eat seedlings, knockdown saplings and strip the bark of trees. Judging by relictual tree stands, and undisturbed forest vegetation in the bottom of pit craters, the forest was comprised of a rich variety of species (Benitez *et al.* 2008).

Throughout Hawaii, cattle ranching and logging converted large tracts of native forest to pasture land resulting in largely negative consequences for the native biota (Cuddihy and Stone 1990). Efforts to restore native forest by removing ungulates have had variable rates of recovery and levels of success due to intensity of past land use, changes in soil characteristics, availability of seed sources, and competition with non-native grasses. There is evidence that natural recovery of some native species, such as koa, may occur following ungulate exclusion if mature trees or a seed bank is present in the area (Tunison *et al*. 1995, Loh *et al.* 2005). However, other areas may remain in stasis due to the lack of native source propagules in the area and/or suppression of seedling recruitment by alien grasses. Past experiments by HAVO resource managers have focused on temporary grass removal using herbicide followed by planting native canopy species to shade out alien grasses and facilitate native seedling recruitment from planted individuals (McDaniel *et al.* unpub. data). Although successful, this approach is labor intensive and has only been applied on a small scale (< 20 ac).

In Kahuku, the large pasture size (3,000 ha), variable supply of native plant material and land use history (Parker vs. Damon period) provided an environment in which to test several forest restoration methods across a broader landscape.

In 2005, four ungulate proof exclosures (each 200 x 200 m) were constructed to evaluate mesic koa forest recovery in four sites which varied in moisture, elevation, proximity to native forest (Fig. 1). The exclosures excluded cattle from portions of the actively used pastures, as well as mouflon sheep and feral pigs which are also common to the area. Within the exclosures, three methods (herbicide, soil turnover, and combination herbicide and soil turnover) to temporarily remove the alien grasses were tested with the objective of finding a method that best promoted native forest recovery in conjunction with ungulate exclusion. In addition to monitoring plant recruitment from the natural seed bank in the soil, establishment by direct seeding and planting was evaluated within the different treatments. Native plant establishment was expected to vary across the four sites; with forest recovery potential to be highest at sites containing more remnant forest fragments and in closer proximity to the Kaʻū forest reserve, in contrast to sites that had endured a longer period of cattle grazing and had fewer native species remaining in the area (Fig 2).

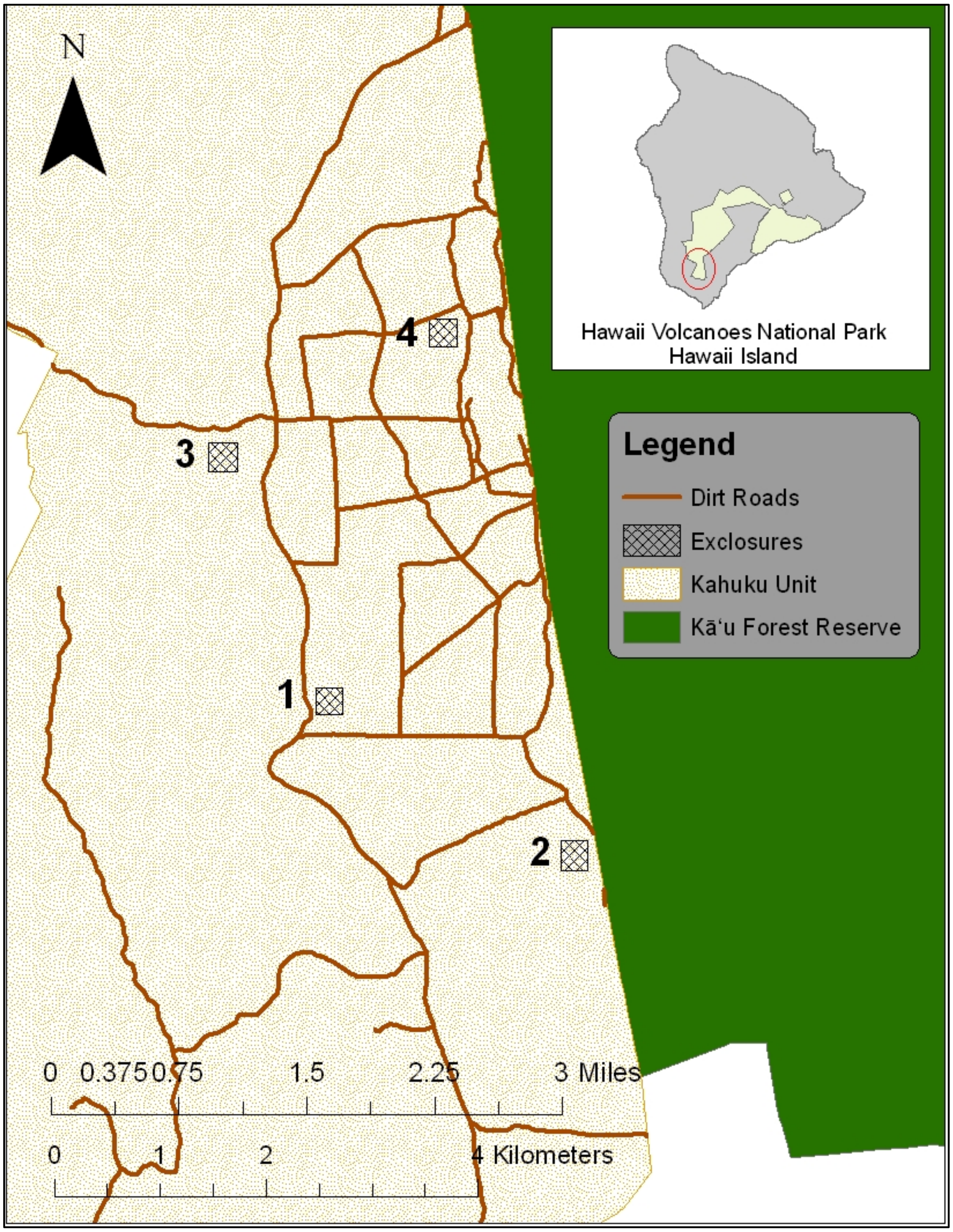

Figure 1. Distribution of the exclosures within the Kahuku Unit of Hawaii Volcanoes National Park. Sites 1 and 3 were under intense grazing for a longer period of time than sites 2 and 4.

Figure 2. Typical vegetation within the Kahuku pastures. The eastern sites (site 4 right upper and site 2 right lower) have a higher diversity of native species contained in forest fragments compared to the western sites (site 3 left upper and site 1 left lower).

Methods

SITE DESCRIPTION

The experiment was conducted within the central pasture region of Kahuku Ranch between 900 and 1350 m in elevation on the southwest slope of Mauna Loa. Annual rainfall in this region is between 1000-2000 mm (Giambelluca *et al.* 1986). The substrate is derived from Mauna Loa flows 1500-3000 years old. Although Kahuku was utilized as a cattle ranch beginning in 1861 and continued to be developed under Parker ranch (1912-1947), most of the intensive pasture development was conducted under the management of Glover (1947-1958) and Damon Estate (1958- early 1970s) who utilized bulldozers to clear vegetation (Avery 2009). The largest trees were left to provide shade and collect water from the fog. Some of the bulldozed trees were moved into piles around existing trees creating small islands of vegetation (Avery 2009). Judging from undisturbed vegetation in steep ravines and pit craters (Benitez et al. 2008) this area was once dominated by a closed ʻōhiʻa forest with a diverse array of native trees and ferns.

EXPERIMENTAL DESIGN

Fence Exclosure

In Fall 2005, four 200 x 200 m fenced exclosures were constructed (Fig. 1). Two exclosures (exclosures 2 and 4) were constructed on the wet eastern portion of the paddock system that had been developed under the management of Glover and Damon close to the Kā'u forest reserve; and two exclosures (exclosures 1 and 3) were constructed in sites located further away to the west which had been initially cleared under Parker management, where conditions appeared drier, and few relictual native species remained. Fence height was 6 ft tall to exclude all ungulates.

Grass Treatments

Within each exclosure, three grass removal treatments (herbicide, soil turnover, herbicide/soil turnover = combination) and an untreated control were replicated twice (50 x 100 m blocks; Fig. 3) in a randomized layout that was replicated at each exclosure. Grass removal treatments were conducted over the late summer/early fall of 2005 just prior to fence construction. For the herbicide treatment, one percent Round-up

(Glyphosate) was used for the initial grass removal in Summer 2005, followed by a touch-up application with two percent Round-up to treat re-growth in Fall 2005. Soil turnover was accomplished using a D4 catepillar with a ripping attachment that penetrated beneath the grass root layer approximately 50 cm (Fig. 4). In the combination treatment, herbicide application and subsequent touch-up was completed several months prior to the soil turnover.

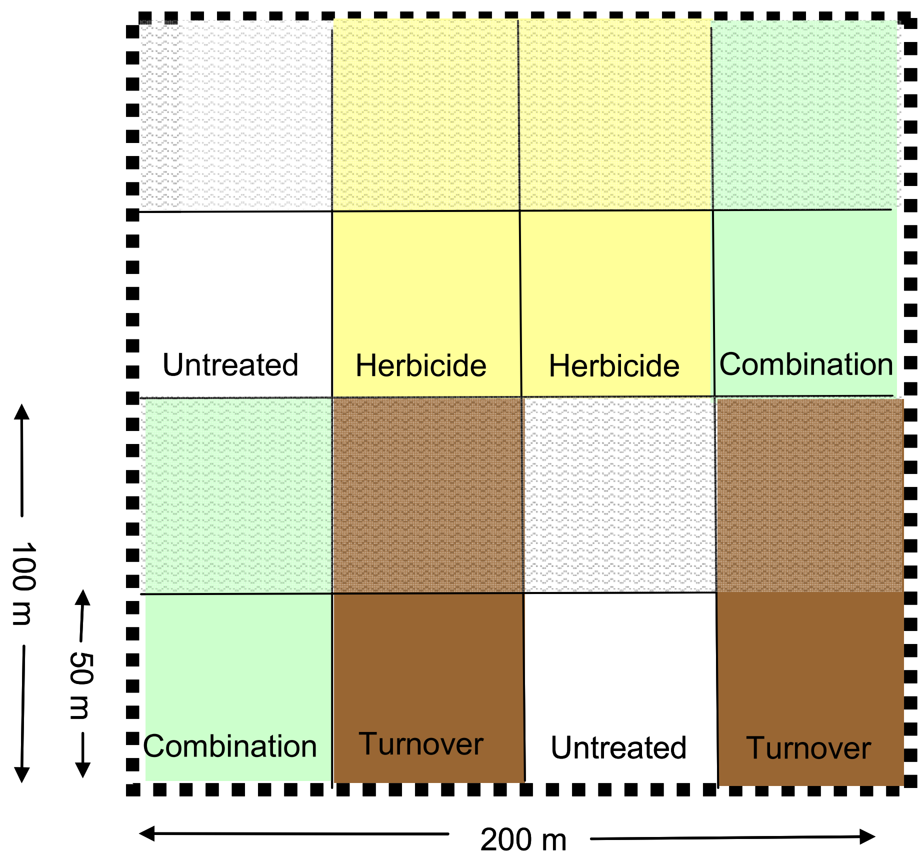

Figure 3. Layout of grass removal treatments within the fenced exclosure. Shading indicates seed and plant augmentation.

Figure 4. A D4 caterpillar with ripping attachment was used for the soil turnover treatment.

Each 50 x 100 m grass removal treatment and untreated control block was subdivided into two 50 x 50 m sections (Fig. 3). The southern 50 x 50 m section was used to monitor plant response to the different grass treatments in the absence of direct seeding or planting of native species (natural); while the northern section was used to monitor survival of planted individuals and seedling recruitment from direct seeding (augmented) into the different treatments.

Community Vegetation Response

A 30 x 30 m monitoring plot centered in each 50 x 50 m section was used to evaluate vegetation cover and species richness. Two 30 x 30 m monitoring plots were established outside each fence exclosure to serve as untreated controls. These control plots were at least 20 m away from the exclosure to isolate treatment and edge effects. Plots were read in summer 2005 just prior to treatment and December 2007 two years following treatment. Photos were taken at each monitoring interval to further document vegetation changes.

Vegetation cover (< 2 m) was estimated using the pole-intercept method along a 30 m transect that ran North-South through the plot. The pole (2 m in height) was placed vertically at 30 cm interval along the transect, and each plant species touching the pole was recorded once. If no vegetation intersected the pole, substrate type was recorded (soil, rock, litter, bryophyte, manure, woody litter (1-3”), log (3+”). Vegetation cover may be greater than 100% if multiple species are encountered at a single sample point. A photo-point was established at the start of each cover transect and a photo taken at the north end of the transect to the south. Species richness was determined by recording the presence of all species within the 30 m x 30 m monitoring area.

Changes in species richness were evaluated using a nested ANOVA (treatment nested within site) on the difference in the number of native and alien species documented between 2005 and 2007. Subsequently, an additional nested ANOVA (managed plots nested within treatment) was performed for each site to evaluate the effect of planting and seeding (managed plots). Vegetation cover was analyzed separately for 2005 and 2007 with a nested ANOVA (treatment nested within site). Vegetation cover between managed and unmanaged plots was analyzed separately by site with an additional nested ANOVA (managed nested within treatment).

Aboveground grass biomass was sampled in six 0.25 m^2 plots per treatment to evaluate the effect of grass removal treatment on biomass accumulation after one year (Oct. 2006). Six biomass plots (0.25 m^2) were also established in the unfenced area. All grass biomass was clipped to ground level and stored in paper bags until oven dried at 70 ˚C and weighed. Differences in grass biomass were evaluated using a nested ANOVA (treatment nested within site).

Establishment of Woody Native Plants by Seed Broadcast and Planting

Within the augmented section of each block (north), re-establishment through planting and/or seeding was conducted with 16 native species (Table 1). Seeds were collected from the pasture area and 11,975 seedlings were propagated at the Hawaii Volcanoes Native Plant Nursery and planted between Spring 2006 and Summer 2009 (1,727- FY06, 4,686- FY07, 3,929 FY08, 1,622 FY09). Seedlings were planted within the entire 50 x 50 m area. A subset of individuals that were planted in 2006 was flagged, and survivorship and vigor was recorded one year after planting. Differences in survivorship among grass treatments were analyzed using a nested ANOVA (treatment nested within site).

Seed broadcast was tested for four species in March 2006. Seeds were collected within the Kahuku pasture area unit. The number of founders was maximized, however, founder number was limited for some species (*e.g.*, *Tetraplasandra* *hawaiiensis*) (Table 1). The seed was removed from the fleshy part of the fruit and soaked in a solution of 5% bleach. Seeds were dried and stored in the refrigerator up to six months prior to field application. Twelve paired seeded and unseeded plots (1 m radius) were established at stratified random locations throughout the 30 x 30 m area within the augmented section to ensure adequate representation of the general area. Plots not reflective of the treatment type were rejected (*e.g.*, soil-turnover in the plot was incomplete). Six of the plots were seeded with a mix of ʻōlapa (*Cheirodendron trigynum*), kāwa'u (*Ilex anomala*), and pilo (*Coprosma pubens*) (Table 1). The other six plots were seeded with koa (*Acacia koa*).

Within each seeded and unseeded plot, seedlings were tallied by size class (0.1-10 cm, 10.1-50 cm, 50.1-100 cm, 100.1-150 cm, 150.1-200 cm, and 200+cm) at 6 months and two years post seeding. At 6 months, five seedlings from each size class greater than 10 cm were tagged and exact height measured to follow subsequent survival and growth rates. Differences in survivorship among grass treatments were analyzed using a nested ANOVA (treatment nested within site). Differences between seeded and unseeded plots across treatments at each site were analyzed with a Kruskal-Wallis rank sum test.

Table 1. Species planted or seeded within the fenced exclosures.

Scientific Name	**Common Name**	**No. of Founders**	**No. of Plants**	**No. of Seeds**
Acacia koa	koa	>100	1,600	64,000 *
Charpentiera obovata	pāpala	<10	43	
Cheirodendron trigynum	ʻōlapa	>100	2,179	532,416
Clermontia clermontioides	ʻōhā wai	<10	95	
Coprosma pubens	pilo	>100	3,106	28,800
Ilex anomala	kāwaʻu	>100	964	138,624
Melicope clusifolia	alani	25	122	
Myoporum sandwicensis	naio	25	253	
Myrsine lesertiana	kōlea	>100	3,016	
Perrottetia sandwicensis	olomea	25	16	
Phytolacca sandwicensis	popolo ku mai	2	111	
Pipturus albidus	māmaki	25	77	
Pittosporum hawaiiensis	hōawa	50-100	321	
Psychotria hawaiiensis	kōpiko	25	7	
Rubus hawaiiensis	ʻākala	25	56	
Tetraplasandra hawaiensis	ʻōhe mauka	6	9	
TOTAL			**11,975**	**763,840**

*Includes the additional 1400 koa seeds that were broadcast within the monitoring plots outside of the seed plots.

Results

COMMUNITY VEGETATION CHANGES

Grass Biomass

One year following treatment, grass biomass was lowest in the combination treatment and highest in the fenced and unfenced control treatments across sites ($p < 0.0018$, Fig. 5). Biomass of other herbaceous and woody species was negligible and was not formally analyzed.

Species Richness

Two years following treatment, there were small increases in the number of native and alien plant species across treatments and sites, but these were not statistically significant (Appendix A). Seventeen new alien species (7 grasses, 8 herbaceous and 2 woody) and 2 new native fern species were documented (appendix A). The new alien grasses were likely present prior to treatment, but were not identified due a lack of conspicuous inflorescences owing to active grazing. One of the new alien species, *Rubus niveus,* may be of management concern (Motooka et al. 2003). Several of the native and alien fern and herbaceous species documented prior to treatment were not observed two years following treatment (appendix A).

Augmentation of native species through a combination of planting and seeding increased species richness in the western sites (site 1, $p \leq 0.0023$, and site 3, $p \leq 0.007$). The eastern sites contained a higher number of native species in forest fragments prior to treatments, and the addition of species did not increase overall species richness in site 2 and 4. There were no significant differences for alien species between the planted/seeded treatments and those which did not receive additional seeds or plants.

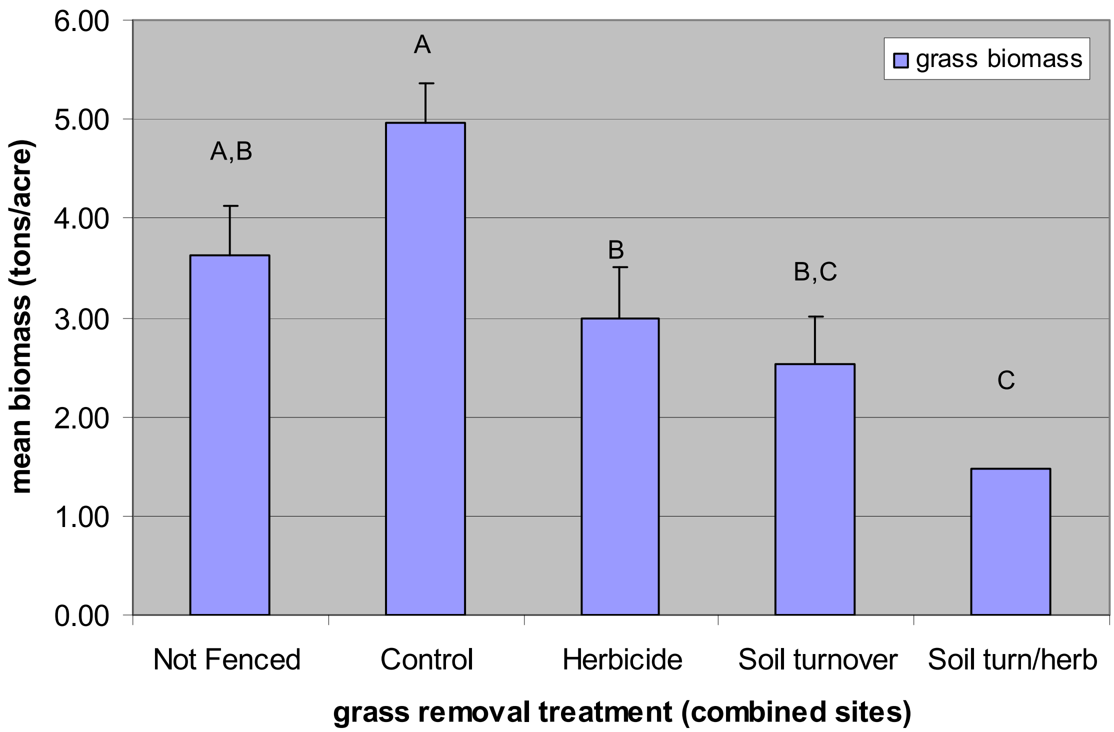

Figure 5. Grass biomass one-year following fencing and grass removal treatments across the four sites (means + SE). No grass removal was conducted outside of the fenced area. Different letters represent significant differences among treatments at $p \leq 0.05$. There were no significant site differences.

Vegetation Cover

Despite a temporary treatment reduction, alien grasses (> 100%), primarily kikuyu and pangola grass, continued to dominate the vegetation in the different grass treatments both inside and outside the fenced exclosures. Across sites and treatments, alien woody vegetation cover remained low (< 2%), and there were no differences among treatments (including unfenced) for native vegetation cover within each of the sites. Vegetation among sites differed in that the lower elevation sites 1 and 2 had significantly higher native vegetation cover in 2005 and 2007, dominated by woody species koa and ʻōhiʻa ($p \leq 0.0041$, $p \leq 0.008$, Fig. 6) compared with the upper elevation sites 3 and 4. Planting and direct seeding of native species did not significantly alter the native or alien vegetation cover at two years.

Establishment of Woody Native Plants

Natural Seedling Recruitment (no seed augmentation)

Koa, pilo, ʻōhiʼa, māmaki, and naio seedlings were the most common native species documented six months following fencing and temporary grass removal. Outside of the exclosure only two koa seedlings and fewer then 15 ʻōhiʻa seedlings (single site) were documented.

Recruitment of koa was evident within weeks of fencing and grass removal, but was highly variable among sites and treatments due to an uneven distribution of mature trees that served as propagules sources (site 1: 41 trees; site 2: 0 trees; site 3: 8 trees; and site 4: 2 trees). Although the difference between treatments was not statistically

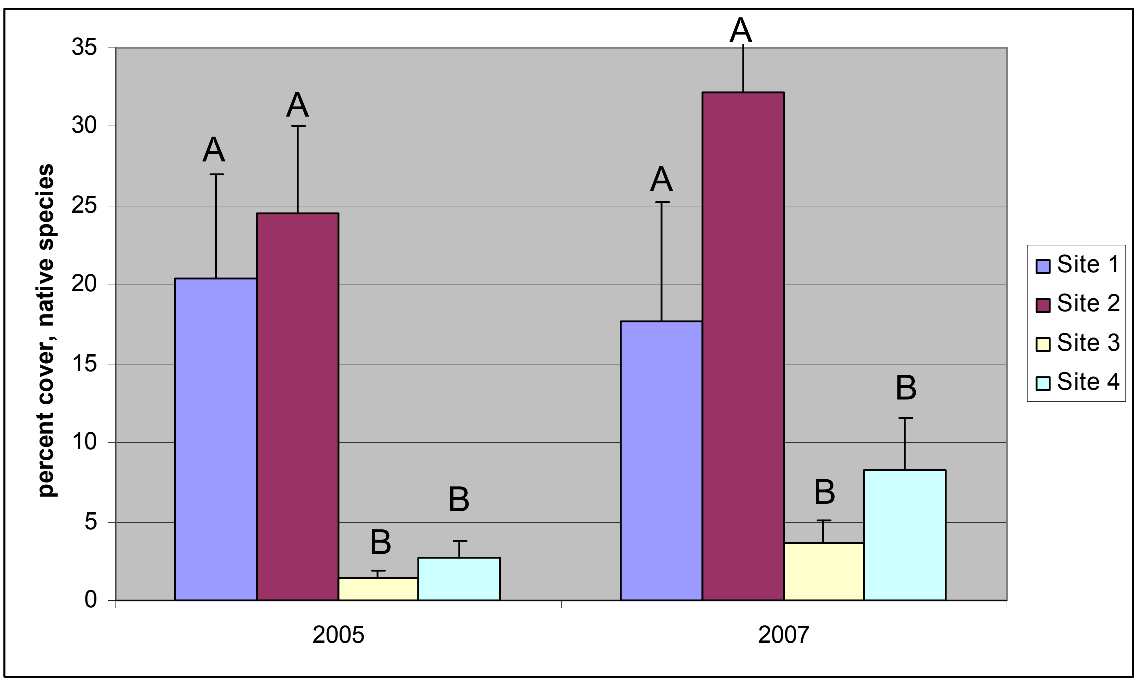

Figure 6. Percent cover of native species by site and year, different letters represent significant differences at p $\leq$ 0.05 (means + SE). There were no significant differences between treatments at each site.

significant from this data set, the visual difference along treatment boundaries is striking where there were existing mature koa (Fig. 7). The thickets grow quickly and appear to be starting to suppress the grasses (Fig. 8 and 9).

Only in site 1, the area with the highest number of mature koa, were any koa seedlings found in the control treatment. With the exception of koa, natural recruitment was too low to determine the effect of grass removal on other native woody species.

Seedling Recruitment in Augmented Plots (seeded)

Koa and pilo germinated in augmented plots across all sites and treatments within six months. ‘Ōlapa was only found in site 4 in the soil turnover/herbicide treatment. No seedlings of kawa’u were found. There was significantly higher koa recruitment in augmented plots than in non-augmented plots in sites 1, 3, and 4 ($p \leq 0.05$) by year two. In site 2, a temporary surge of koa recruitment observed in augmented plots observed at six months was no longer significantly different from non-augmented plots by year two. Also, seed augmentation significantly increased pilo seedling recruitment in sites 1,2, and 4 at both six months and two years. Among grass removal treatments, recruitment of koa ($p \leq 0.051$) and pilo ($p \leq 0.073$) was highest in the combination treatment by year two (Fig. 10 and 11).

Survival of tagged individual koa was high across treatments (74 - 89%), but no significant differences among grass removal treatments was detected, due to an insufficient number of tagged individuals over 10 cm available for analysis of grass removal effect at six months.

Establishment from Planted Individuals

Survival was moderate to high across species and treatments one year (1.5 years for koa) following planting (Table 2). Among sites, overall survival (combined species) was higher in sites 1,3 and 4 (~70%) than in site 2 (57%, $p \leq 0.01$). The lower survival at site 2 is largely due to lower survival of koa and pilo. Five of the species (hōawa, māmaki, kōlea, naio and ‘ōha) were not affected by site. Survival of ‘ōlapa was lowest in site 3, and for kāwa‘u lowest in site 4. There were no apparent differences among grass removal treatments. Across sites 1,3, and 4, over 70% of the naio, pilo and koa survived. Survival of māmaki, ‘ōha, and kōlea was between 60-63%. The

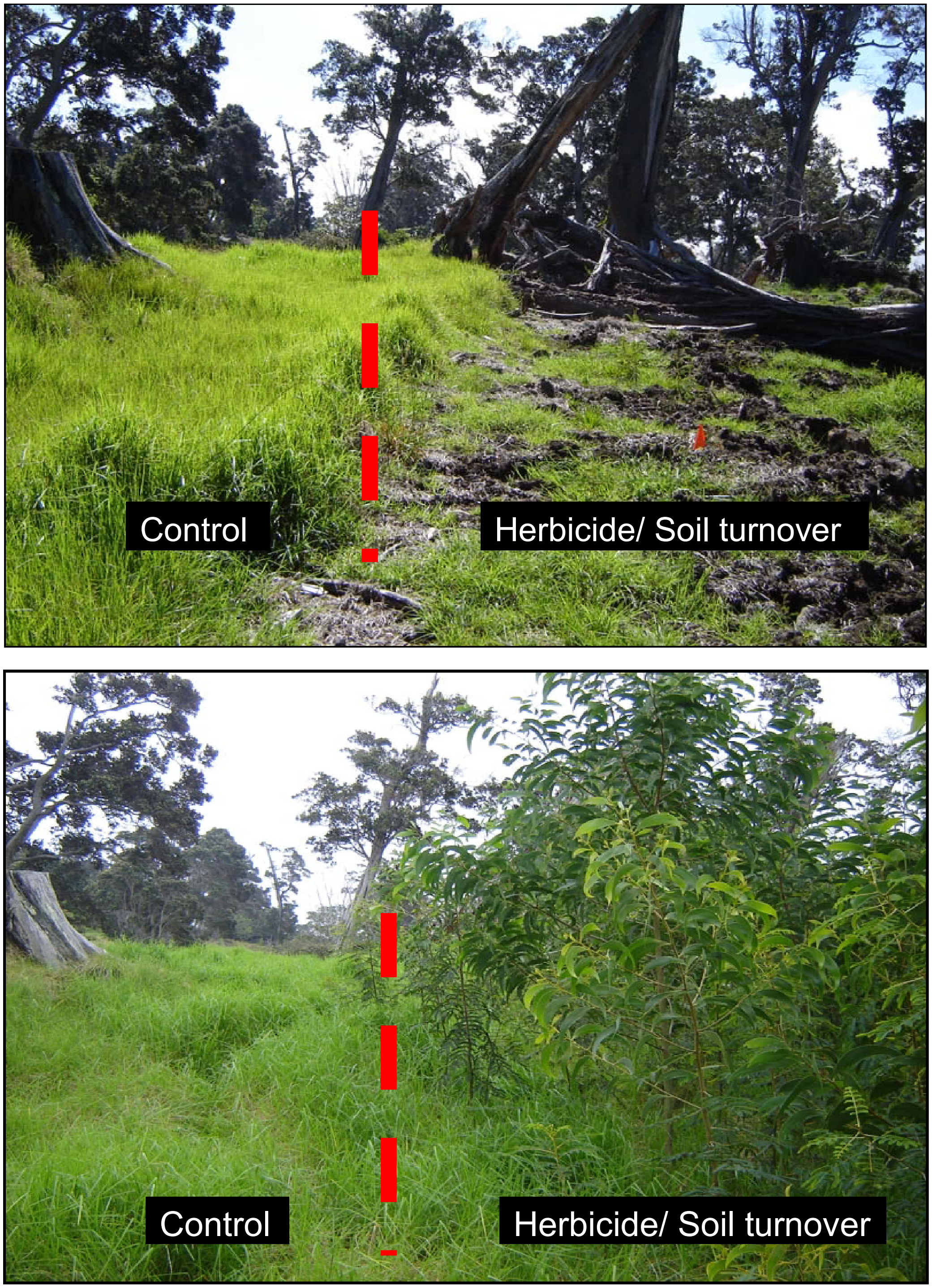

FIgure 7. Natural recruitment of koa in a combination plot compared with a control plot six months (top) and two years following treatment (bottom).

Figure 8. Natural recruitment of koa in a combination herbicide/ soil turnover plot, October 2007.

Figure 9. Typical grass growth under the re-generated koa thickets in combination herbicide/soil turnover plot, Oct. 2007.

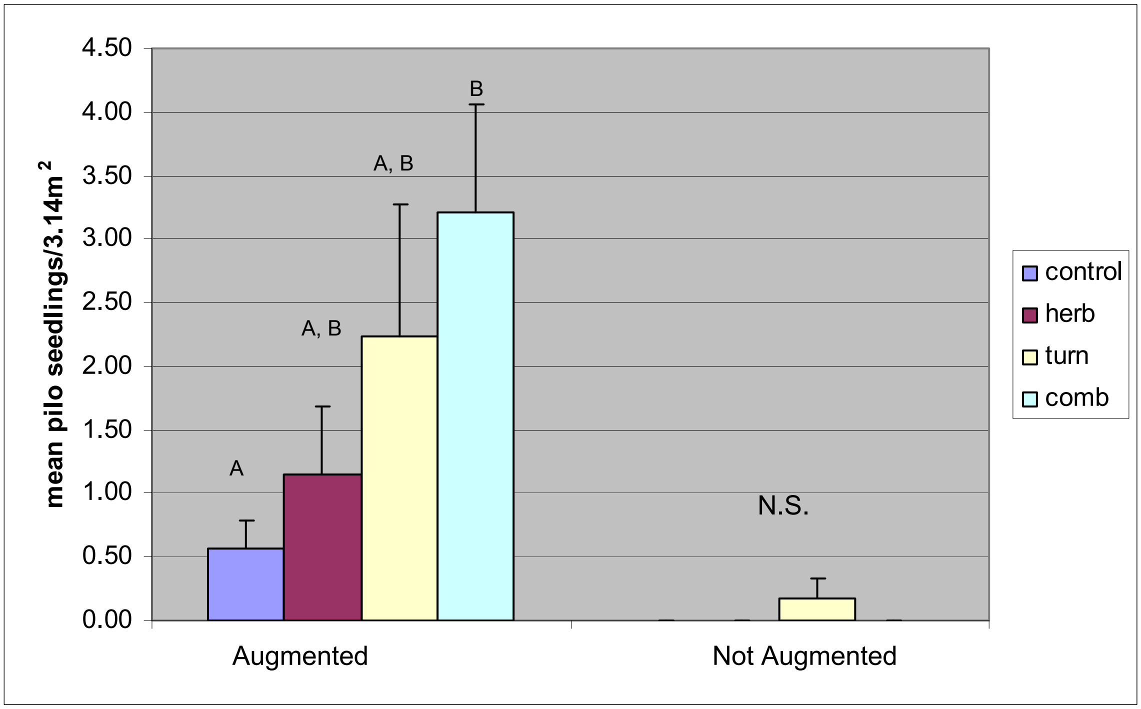

Figure 10. Mean number of pilo seedlings per plot established in the four grass removal treatments two years after application (means + SE).

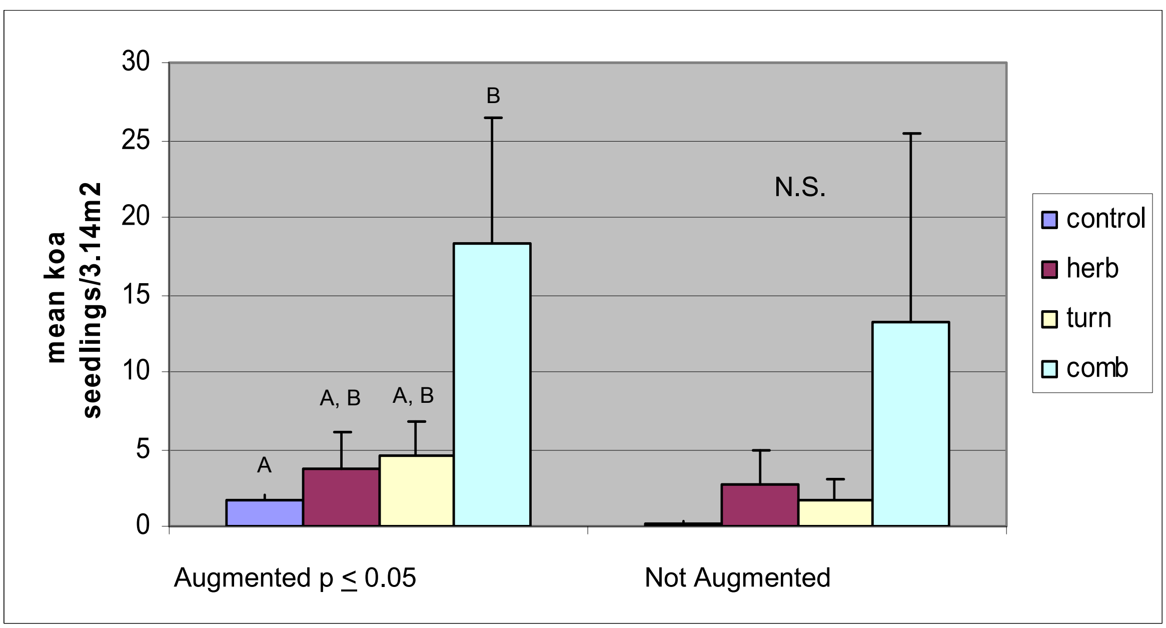

Figure 11. Mean number of koa seedlings per plot established in the four grass removal treatments two years after application (means + SE). Different letters represent significant differences at $p \leq 0.05$.

Table 2. Percent survival for nine species planted in 2006 within exclosures at four sites. Different letters represent a significant difference at $p \leq 0.05$. There were no significant differences between treatments. n = number of individuals monitored at each site.

	Common		**Site**					**All**
Species	**Name**	**n =**	**1**	**2**	**3**	**4**		**Sites**
Acacia koa	Koa	240	$72^{a,b}$	65^{b}	87^{a}	82^{a}		77
Cheirodendron trigynum	Olapa	80	63^{a}	$53^{a,b}$	25^{b}	$51^{a,b}$		48
Clermontia clermontioides	Ohe	24	80	61	45	16	n.s.	64
Coprosma pubens	Pilo	240	75^{a}	54^{b}	74^{a}	80^{a}		71
Ilex anomala	Kawau	56	57^{a}	57^{a}	$44^{a,b}$	16^{b}		50
Myoporum sandwicensis	Naio	32	78	47	81	91	n.s.	74
Myrsine lesertiana	Kolea	240	63	53	70	70	n.s.	64
Pipturus albidus	Mamaki	16	81	63	50	50	n.s.	61
Pittosporum hawaiiense	Hoawa	64	52	53	58	45	n.s.	52

lowest survival was found for olapa, kāwaʻu, and hōawa (48-52%). Rapid growth of koa plantings were evident one year after planting (Fig. 12). Although survival was high for the other species, growth was moderate to slow with some individuals remaining below the grass layer after two years (Fig. 12). This slow growth pattern may be due to competition with alien grasses.

Figure 12. Planted koa (left, December 2007) and planted kōlea (right, October 2007).

Discussion

Forest recovery following cessation of cattle ranching will depend on the implementation of effective techniques to stimulate native species recovery.   In this experiment, seedling recruitment of several native tree species was observed following ungulate removal, but without additional management seedlings remained scarce. As found in this and other studies in Hawaii (Cabin et al. 2002, Denslow et al. 2006) native recruitment is limited by both thick mats of non-native grass and lack of local seed sources.

Temporary removal of the grass maximized recruitment from the existing and augmented seed bank in the soil. By exposing the soil and providing a longer suppression of grasses, the combination treatment (soil turnover + herbicide) provided the least impediment to seedlings. This is in contrast to the separate herbicide or turnover treatments where grass biomass recovered more quickly.   Visual differences in recruitment of koa among grass treatments, while striking, were not statistically significant due to the (probably) patchy distribution of the seed bank in the soil. These differences between grass removal techniques were only statistically apparent in areas where augmentation (seed addition) took place.

Although the herbicide treatment was moderately successful in stimulating re-generation of native species, non-native grasses recovered quickly hindering recovery. New studies (Leary et al. unpublished) have documented that other chemical compositions (i.e. imazapyr) will suppress grasses for longer period than glyphosate (round-up). Experiments using different formations and rates may determine a more effective technique for long-term grass suppression and increased regeneration.

Augmentation using seeds or plantings combined with grass removal is a promising technique to increase species richness and rebuild forest structure. Due to the natural variability of the existing seed bank, temporary grass removal is not sufficient to ensure recruitment of native trees. Both direct seeding and planting were effective methods for restoring biodiversity to the area. Among the species tested, all nine were able to establish from plantings and two of the four species used for seed augmentation established into fenced enclosures. Tests of seed augmentation were limited by the scarcity of available seed sources in the area, so this method could not be tested for all species (e.g. ʻōha, naio, kōlea, māmaki, hōawa). For two species (ʻōlapa and kāwaʻu) where sufficient seed was obtained, long germination times combined with quick recovery of grasses may have prevented establishment. Additional experimentation with different species and substrates would be useful for refining this technique. Seeds of species with known long germination times may need to be seeded months before the grass is treated. Overall survival rates of planted seedlings were moderate to high one year post planting (57-70%). As these planted individuals mature and produce seed, thereby increasing the available propagule supply, forest recovery will be accelerated.

Previous work has demonstrated that the establishment of a closed native canopy in abandoned pasture will reduce the dominance of alien grasses and facilitate the recruitment of native species (Scowcroft et al. 1999, Medeiros and vonAllmen 2006, Scowcroft et al. 2008, McDaniel and Ostertag 2010). In this experiment, koa saplings quickly regenerated following grass removal (herbicide+ soil turnover) and will likely create canopy conditions within a few years. Long term monitoring will determine the efficacy of these thickets to suppress grasses. Although planting was successful in terms of survival, canopy conditions were not created as quickly due to slower growth rates and density of planted seedlings. Although not tested in this experiment, densely planted ʻaʻaliʻi (*Dodonaea viscosa*) has been highly successful at long term suppression of grasses following herbicide treatment of kikuyu grass in abandoned pasture (Mederios and vonAllmen 2006). This species grows faster than other species used in this experiment and should be evaluated as part of the matrix of species used to create canopy at Kahuku.

The creation of canopy at Kahuku will require different approaches based on past land use and the remaining vegetation in different areas. On the western end of the paddock system, biodiversity is severely depleted and can be increased through augmentation of native species. Strategically placed dense plantings to create forest islands will increase the amount and diversity of available seed in this section. In addition, within the islands, grasses will be suppressed and natural recruitment should be facilitated. Future management action (grass control) will facilitate expansion of these islands which will coalesce over time creating contiguous forest habitat. In areas where there are mature koa trees, the seed bank can be stimulated through grass removal. The resulting koa thickets will create shaded conditions facilitating the addition of understory species to increase biodiversity. This approach could also be applied in areas without koa if a sufficient amount of seeds could be collected to create a seed bank following grass removal. In the eastern section, forest fragments containing a high diversity of native species remain in the area. These fragments can serve as focal points to restore the surrounding grass dominated area. Removing grasses immediately adjacent to the forest fragments, where seed availability is higher, may serve to gradually expand the fragments. In addition, grass removal combined with dense planting or seeding between adjacent fragments will increase forest cover.

Disruptive invasive plant species have the capacity to outcompete native species and impede recovery efforts. Fortunately, the recruitment of alien woody species was low with less then 2% cover of non-native woody species two years following treatment in the exclosures. However, highly disruptive invasive species such as *Rubus argutus, Hedychium gardnerianum*, and *Solanum linnaeanum* are known from the general area. Consequently, periodic monitoring to detect and treat disruptive invasive species will be required to prevent their establishment in recovering forest.

Conclusions

1. **Ungulate removal** is essential to facilitate the recovery of forest ecosystems. Native forest seedlings such as pilo, māmaki, naio and ʻōlapa were only able to establish within the fenced exclosure.

2. **Grass manipulation** to temporarily suppress grass competition will greatly enhance the recovery of native species in areas where there is an existing seed source.

3. Seed and plant additions will increase species richness and provide additional seed sources where they are lacking. Many species are dispersal limited and augmentation will be necessary to restore the forest; particularly on the western end of the paddock system where fewer native plants remain in the area.

4. Preventing establishment of target weed species is essential to ensure continued forest recovery. Although establishment of disruptive weeds was low in the first two years, invasion prevention is vital to promoting forest development.

5. Continued monitoring and experimentation are needed to evaluate and improve restoration strategies and techniques. Mangers should explore additional methods to achieve grass suppression for a longer period including alternative herbicide formulations. Initial changes in community vegetation composition can be captured in two years; however, these sites should be re-monitored in the future to understand long term recovery patterns. In addition, future monitoring will allow mangers to evaluate long term establishment of planted individuals, time to reproduction and subsequent seed bank creation.

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Appendix A. Species documented before (2005) and after (2007) fencing and grass removal treatments in the Kahuku pastures.

alien grasses and sedges	**2005**	**2007**
Agrostis avenacea	x	x
Andropogon glomeratus		x
Andropogon virginicus	x	x
Anthoxanthum odoratum	x	x
Axonopus fissifolius	X	x
Bulbostylis capillaris	x	x
Cyperus sanquinolentus	x	x
Dactylis glomerata		x
Digitaria eriantha	x	x
Eragrostis brownei	x	
Erharta stipoides	x	x
Festuca arundinacea		x
Holcus lanatus		x
Kyllinga brevifolia	x	x
Melinis minutiflora		x
Paspalum conjugatum		x
Paspalum dilatatum	x	x
Pennisetum clandestinum	x	x
Poa pratensis	x	x
Rhynchospora rugosa	x	
Sacciolepis indica	x	x
Schizachyrium condensatum	x	x
Setaria gracilis	x	x
Sporobolus africanus	x	x
	18	22

alien herbaceous and ferns	**2005**	**2007**
Ageratina riparia	x	x
Anagallis arvensis	x	x
Arundina graminifolia	x	x
Asclepias physocarpa	x	x
Blechnum appendiculatum	x	x
Cardamine flexuose	x	x
Christella dentata	x	x
Christella parasitica	x	
Cirsium vulgare	x	x
Commelina diffusa	x	x

Conyza spp.	x	x
Cuphea carthagenesis	x	x
Cyrtomium falcatum	x	x
Deparia petersenii	x	x
Desmodium triflorum	x	
Drymaria cordata	x	x
Epilobium billardierianum	x	x
Erechtites valerianifolia	x	x
Geranium homeanum	x	x
Glycine wightii		x
Gnaphalium japonicum	x	x
Gnaphalium purpureum	x	
Hedychium gardnerianum	x	x
Hydrocotyle bowlesioides	x	x
Hypericum parvulum	x	
Hypochoeris radicata	x	x
Linaria canadensis	x	
Lotus subbiflorus	x	
Lotus uliginosus		x
Nephrolepis multiflora	x	x
Oxalis corniculata	x	x
Phaius tankarvilleae	x	x
Pityrogramma austroamericana	x	x
Plantago lanceolata	x	
Plantago major	x	
Prunella vulgaris	x	x
Rumex acetosella	x	x
Senecio madagascariensis	x	x
Stachytarpheta spp.	x	x
Tibouchina herbacea	x	x
Trifolium dubium	x	x
Trifolium repens	x	x
Veronica plebia	x	x
Veronica serpyllifolia	x	x
Youngia japonica	x	x
	43	37

alien woody	**2005**	**2007**
Buddleia asiatica	x	x
Physalis peruviana	x	x

Pluchea carolinensis	x	x
Rubus argutus	x	x
Rubus niveus		x
Rubus rosifolius	x	x
Schinus terebinthifolius	x	x
Solanum americanum		x
Solanum linnaeanum	x	x
	7	9

native grasses and sedges	**2005**	**2007**
Cyperus polystachos	x	x
Fimbristylis dichotoma	x	x
	2	2

native herbaceous and ferns	**2005**	**2007**
Amauropelta globulifera	x	
Asplenium adiantum-nigrum	x	
Asplenium polyodon		x
Asplenium spp.	x	x
Athyrium microphyllum	x	x
Cibodium glaucum	x	x
Cibodium menziesii	x	x
Dicranopteris linearis	x	x
Diplazium sandwichianum	x	x
Dryopteris fusco-atra	x	x
Dryopteris glabra	x	x
Dryopteris hawaiiensis	x	x
Dryopteris wallichiana	x	x
Gnaphalium sandwicensium	x	x
Hypolepis hawaiiensis		x
Marattia douglasii	x	x
Microlepia speluncae	x	
Microlepia strigosa	x	x
Nephrolepis cordifolia	x	x
Nephrolepis exaltata	x	x
Pseudophegopteris keraudreniana	x	x
Psilotum complanatum	x	
Psilotum nudum	x	x
Pteridium aquilinum	x	x
Rumex giganteus	x	x

Sadleria cyatheoides	x	x
Sadleria pallida	x	x
Smilax melastomifolia	x	x
Sphenomeris chinensis	x	x
Vandenboschia davallioides	x	
	28	23

native woody	**2005**	**2007**
Acacia koa	x	x
Cheirodendron trigynum	x	x
Clermontia clemontioides	x	x
Coprosma pubens	x	x
Dodonaea viscosa	x	x
Freycinetia arborea	x	x
Ilex anomala	x	x
Leptecophylla tameiameiae	x	x
Melicope clusiifolia	x	x
Metrosideros polymorpha	x	x
Myoporum sandwicensis	x	x
Myrsine lessertiana	x	x
Perrottetia sandwicensis	x	x
Pipturus albidus	x	x
Pittosporum hawaiiense	x	x
Psychotria hawaiiensis	x	x
Rubus hawaiensis	x	x
Vaccinium calycinum	x	x
Vaccinium reticulatum	x	x
	19	19