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Passive restoration augments active restoration in deforested landscapes: The role of root suckering adjacent to planted stands of *Acacia koa*

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ABSTRACT

Active forest restoration in Hawaii's Hakalau Forest National Wildlife Refuge has produced a network of *Acacia koa* tree corridors and islands in deforested grasslands. Passive restoration by root suckering has potential to expand tree cover and close gaps between planted stands. This study documents rates of encroachment into grassland, clonal stand structure, and tree architecture. Data were collected from random replicate strip transects that started inside 23-year-old koa plantations and ended either in open grassland or in adjacent planted stands. For the former, sucker densities increased from near zero inside planted stands to a maximum of 5–38 stems m^{-2} 5–14 m away from the edge of the plantation canopy, and then decreased to zero–a typical pattern for trees invading grassland. No suckers occurred more than 28 m from the canopy edge on east-facing slopes, or more than 18 m on south-facing slopes. Rates of expansion into grassland ranged from 0.8 to 1.5 m yr^{-1}; suckers had already filled gaps between closely spaced plantation stands located on north-facing slopes. Continued suckering should result in the eventual re-establishment of tree cover on deforested areas between planted tree islands and corridors, and without additional active restoration.

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1. Introduction

Extensive areas of tropical forests have been and continue to be converted to agricultural uses (FAO, 2010). Between 2000 and 2005 an estimated 27 million hectares of humid tropical forest were cleared (Hansen et al., 2008; Asner et al., 2009), a phenomenon driven mainly by rapid urbanization and increasing international demand for agricultural products (Butler and Laurance, 2008; DeFries et al., 2010). The net decline in natural forest area slowed during 2000–2010 (Meyfroidt and Lambin, 2011), an indication that losses were being partially offset by reforestation of abandoned agricultural lands.

Abandonment of degraded and deforested lands is driven by a number of factors, not the least of which is whether the agricultural use is economically viable and sustainable (Cramer et al., 2007; Rey Benayas et al., 2007). Once abandoned, land managers must decide whether restoration is desired to recover lost biodiversity and the suite of other ecosystem service provided by the original forest ecosystem, and if so, how should it be done (Holl and Aide, 2011). Answers vary with the degree of degradation, which will depend on the nature, duration, and intensity of the agricultural practices, the desired rate of recovery, and the desired state of the restored ecosystem (Aide et al., 2000; Holl, 2007). At one extreme are cases of small-scale, short-term subsistence agriculture where forests recover quickly after abandonment without human intervention and with the same species assemblages as formerly (passive restoration or secondary succession). At the other extreme are cases of large-scale, long-term deforestation that occurred when forest landscapes were converted to grasslands either intentionally for cattle pasture or unintentionally as a consequence of the invasive grass-fire cycle (e.g., Mack and D'Antonio, 1998; MacDonald, 2004). The biotic and abiotic legacy of such conversions can be stable alternative ecosystem states dominated by invasive species that resist colonization by native species following abandonment (Kulmatiski, 2006; Suding et al., 2004; Cramer et al., 2007). To speed ecological restoration managers must resort to assisted regeneration (active restoration) (Lamb and Gilmour, 2003), which generally has a negative benefit-cost ratio, except perhaps where the potential environmental cost of doing nothing outweighs the cost of intervention (Lamb et al., 2005; Birch et al., 2010).

Reestablishing of forest cover to deforested landscapes can involve both passive (natural regeneration by seeding, coppice, or root suckers) and active efforts (artificial regeneration by direct

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seeding or planting). Species that can regenerate by root suckering have an advantage over species that regenerate from seed in that they bypass or better cope with many impediments to natural recolonization of deforested landscapes. Those barriers include poor seed dispersal, seed predation, harsh environmental conditions, and competition from introduced grasses (Holl, 2002).

Establishment of tree islands (aka. applied nucleation) within deforested areas has been proposed as a way to overcome those barriers with only modest investment (Rey Benayas et al., 2008; Corbin and Holl, 2012). However, its success depends not only on increased seed dispersal, seedling survival, and species richness within tree islands, but also on the expansion of islands beyond their initial boundaries (Corbin and Holl, 2012). No matter how successful establishment of tree islands might be, if they do not expand into adjacent areas the landscape remains fragmented and only partially restored.

Here we present a case study describing how root suckering (passive restoration) is augmenting planting (active applied nucleation) to reestablish koa forest (*Acacia koa*) in abandoned pastureland. The questions of interest include the following—Might clonal expansion into unplanted grassland eventually close the gaps between planted stands? How long might it take? Are clonal trees and stands different from those of the same age but regenerated from seeds? Does slope aspect influence root suckering?

2. Methods

2.1. Site description

The study site is located inside the Hakalau Forest National Wildlife Refuge at 1980 m elevation (19°14′51″N/155°19′59″W) where the climate is cool and mesic (Scowcroft and Jeffrey, 1999). The Holdridge life zone for the site is Subtropical Montane moist forest (Tosi et al., 2002). Mean daily air temperature ranges from 9 to 14 °C, and freezing temperatures occur near ground level during clear winter nights. Annual rainfall is approximately 2000 mm. Soils are strongly acidic and classified as Medial over hydrous, ferrihydritic, isomesic Acrudoxic Hydrdands.

The 13,247-ha Hakalau Forest National Wildlife Refuge was established in 1985 to protect and manage endangered Hawaiian forest birds and their rain forest habitat. More than 100 years before establishment, approximately 2020 ha of mesic to wet native forest above 1700 m elevation were converted to pasture for cattle grazing. The vegetation in the deforested area was dominated by alien grasses, including *Pennisetum clandestinum*, which is a C_4 grass adapted to cool, fertile areas (Mears, 1970; Wilen and Holt, 1996), *Ehrharta stipoides*, *Anthozanthum odoratum*, *Holcus lanatus* and *Axonopus fissifolius*. The native grass, *Deschampsia australis*, and the short-stature native shrub, *Vaccinium calycinum*, were sparsely present. Remnant old-growth koa trees were largely confined to steep-sided gulches, although a few isolated individuals occurred elsewhere within the grassland matrix.

The first active reforestation effort was established in 1987 inside a 2.6-ha exclosure that was built that year within the deforested and actively grazed portion of the Refuge. It excluded pigs, goats and sheep as well as cattle. Grazing ended in 1990. Koa was chosen as the native tree species to plant. It was an important structural component of the former forest, and managers believed that it would establish despite grass competition and harsh environmental extremes, and subsequently physically change the abiotic environment, and in so doing directly or indirectly regulate the availability of resource to other species (Wright and Jones, 2006). No naturally occurring koa were present inside the exclosure or located within 0.5 km of it. Thirty blocks of koa were planted inside the Magnetic Hill Exclosure (Scowcroft and Jeffrey, 1999). Within each block, seedlings were planted on a 7 × 7 square grid, using one of two spacing: 2 and 2.5 m (Conrad et al., 1988). All blocks established successfully (Fig. 1a); some were located in drainage bottoms, while others were located on north-, east-, or south-facing slopes (Scowcroft and Jeffrey, 1999). Root suckers were first observed in 1990.

The success of the initial reforestation effort led managers to adopt planting as the method of choice. Active restoration was confined to 10-m-wide corridors, which consisted of three parallel rows of trees that were laid out to link remnant koa trees together and tie them to forest fragments and more densely wooded pastures located at lower elevation (Scowcroft and Jeffrey, 1999; US Fish and Wildlife Service, 2010). The corridor concept was similar in design to that implemented more recently in Columbia (Perfecto and Vandermeer, 2010; Murgueitio et al., 2011), but the intent differed. At Hakalau the intent was to create habitat that was suitable for establishment of native understory plant species, and to provide avenues for species movement into the heart of the deforested landscape. More than 312,000 koa seedlings were planted by 2007 (Unpublished report, 2007 Accomplishments and status – Hakalau Forest NWR, 2 p).

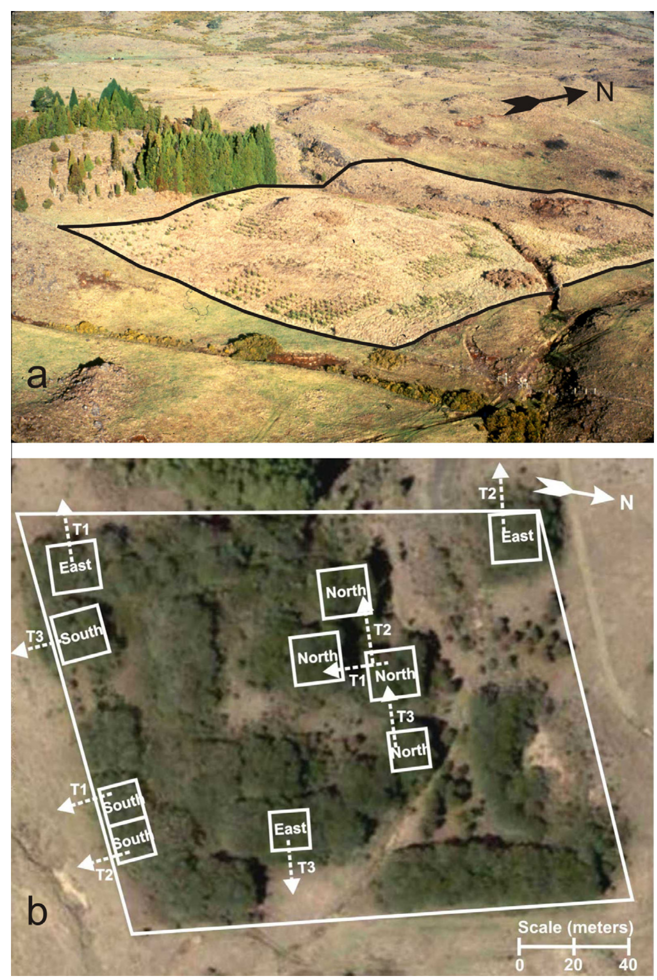

Fig. 1. The 2.6-ha Magnetic Hill Exclosure was the site of the first trial plantings (1987) of *Acacia koa* in deforested portions of Hakalau Forest National Wildlife Refuge. (A) The 7-by7-tree planting blocks were visible from the air in 1990. (B) The original boundaries of individual plantation blocks sampled in this study are shown superimposed on a 2006 Google Earth satellite image: two block sizes are shown depending on spacing (2- by 2-m or 2.5- by 2.5-m). Block labels denote general slope aspect. Dashed arrows denote the direction and approximate location and length of sample transects, which for a given slope aspect were labeled T1 through T3.