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Future directions for forest restoration in Hawai'i

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Abstract Hawai'i has served as a model system for studies of nutrient cycling and conservation biology. The islands may also become a laboratory for exploring new approaches to forest restoration because of a common history of degradation and the growing number of restoration projects undertaken. Approximately half of the native ecosystems of Hawai'i have been converted to non-native conditions. Many restoration projects have focused on intensively managed out plantings of native plants with emphasis on threatened and endangered species. While these projects have been effective in stabilizing plant populations, this model is often prohibitively expensive for restoration at the scale needed to protect watersheds and provide habitat for rare bird species. Here we suggest ways of rethinking ecological restoration that are applicable across the tropics, particularly on islands and fire-prone grasslands. First, we suggest making use of non-native, non-invasive species to help reclaim degraded or invaded sites or as long-term components of planned restoration outcomes. Second, we suggest incorporating remote sensing techniques to refine where restoration is carried out. Finally, we suggest borrowing technologies in plant production, weed control, and site preparation from industrial forestry to lower restoration costs. These suggestions would result in ecosystems that differ from native reference systems in some cases but which could be applied to much larger areas

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than most current restoration efforts while providing important ecosystem services. We also stress that community involvement is key to successful restoration, as a major goal of almost all restoration projects is to re-connect the community with the forest.

Keywords Restoration ecology · Fire · Invasive species · Nurseries

Introduction

The Hawaiian Islands support a diverse array of native ecosystems, from warm, wet forests receiving more than 6000 mm of annual precipitation to dry scrub forests and sub-alpine shrublands at over 3000 m elevation (Mueller-Dombois and Fosberg 1998). These ecosystems feature a unique flora with a high rate of endemism (Wagner et al. 1999). Beginning with the arrival of Polynesian colonists about 1200 CE (Wilmhurst et al. 2011) and accelerating with Western contact, native ecosystems have been replaced by agriculture, ranching, and urbanization and degraded by wildfire, harvesting (particularly firewood and sandalwood extraction), and non-native plant and animal invasions (Trauernicht et al. in press; D'Antonio and Vitousek 1992; Cuddihy and Stone 1990; Scowcroft 1983). As a result, more than half of the original native ecosystems have been converted to systems dominated by non-native vegetation (Fig. 1) (Gon et al. 2006). This

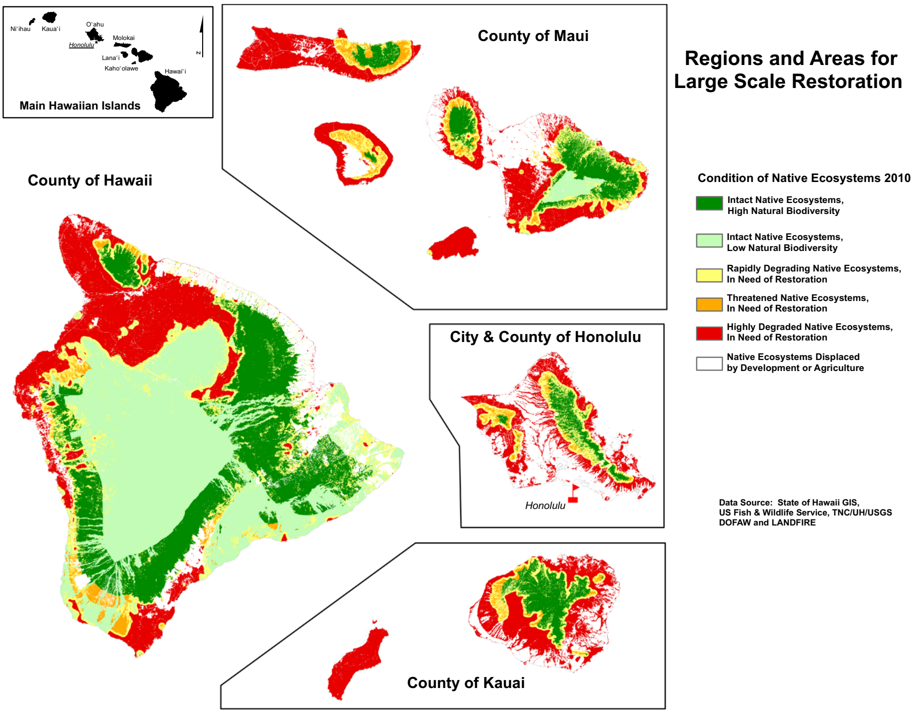

Fig. 1 Map of the Hawaiian Islands showing extent of degraded ecosystems in need of restoration. Data source State of Hawai'i GIS, US Fish and Wildlife Service, and LANDFIRE

loss of native ecosystems has resulted in Hawai'i becoming the "endangered species capital" of the United States, with 366 plant taxa and 30 bird taxa listed as threatened or endangered (Hawai'i Department of Land and Natural Resources). The loss of native ecosystems and endemic species is a critical problem worldwide, particularly on tropical islands (Denslow 2003) and in fire-prone grasslands (D'Antonio and Vitousek 1992). Innovations in restoring native ecosystems showing promise in Hawai'i have the potential to be applied across the tropics.

Forest restoration in Hawai'i encompasses both "rehabilitation" projects, where degraded native forests are protected and enriched, and "reconstruction" projects, where forests are re-established on land previously cleared and used for some time for agriculture or ranching (Stanturf et al. 2014). Forest rehabilitation projects in Hawai'i usually begin with fencing large areas to control feral ungulates (e.g. Cole et al. 2012), which is sometimes followed by control of invasive plants and outplanting of native species. Although native plant species regenerate passively (i.e., without outplanting or invasive species control) in some cases, particularly where the original forest has only recently been degraded (e.g. Scowcroft et al. 2008), forests in Hawai'i generally have been degraded so long and invasive species (particularly grasses) are so abundant that there is little passive native regeneration. Forestry for watershed conservation in Hawai'i dates back to the nineteenth century and reflected widespread concerns at that time about flooding and continued availability of irrigation for agriculture (Woodcock 2003). Most of the early forest restoration focused on establishment of single-species plantations of non-native tree species, after initial failures in establishing native species (Little and Skolmen 1989). Some of these introduced trees, such as *Fraxinus uhdei* (tropical ash) have since invaded native forests (Friday et al. 2008). Large scale restoration of native forest ecosystems began in the 1970s and 1980s with reforestation projects planting the fast-growing native legume tree *Acacia koa* to restore mesic upland pastures for habitat for endangered bird species (Horiuchi and Jeffrey 2002; Jeffery and Horicuhi 2003; Scowcroft and Jeffrey 1999; Scowcroft et al. 2008). Over half a million native plant seedlings have been planted over a landscape of 1100 ha at the Hakalau Forest National Wildlife Refuge since 1987. Unlike in other ecosystems where a mix of several pioneer and climax forest tree species is planted in a single step at the onset of the project (e.g. Elliott et al. 2003 in Thailand), restoration of these high-elevation pastures begins by planting a single-species stand of *Acacia koa* as a nurse crop to shelter subsequent underplantings of other species from seasonal freezing conditions (Scowcroft and Jeffrey 1999). Restoration of dry forests, which hold the greatest number of threatened and endangered plants in Hawai'i, has proven to be much more challenging (Cabin et al. 2002), as has restoration of lowland wet forests large populations of invasive woody plant species (Mascaro et al. 2008).

Restoration of many forests in Hawai'i has been driven by the need to increase populations of threatened and endangered plants (Cordell et al. 2008; Medeiros and von Allman 2006). Because wild populations of these plants are so small (many with fewer than 50 individuals), intensive out planting efforts, including hand-weeding and hand-watering, are justified and have been successful in significantly increasing population sizes (Cordell et al. 2008). These small projects have also been successful in providing field sites for community outreach and education in Hawaiian natural history and ethnobotany. Costs for these projects, however, can run into the tens of thousands of dollars per hectare. These small islands of restored native dryland forest exist in a matrix of vast fire-prone grasslands that threatens their existence (Trauernicht et al. in press). Although small-scale restoration projects meet some objectives such as stabilizing populations of rare plant species and providing outdoor classrooms, other objectives such as creating meaningful habitat for endangered bird species (Banko and Farmer 2014) or protecting entire watersheds require restoration of much larger areas of forest (Lamb et al. 2005). Here, we present some ideas on how to expand forest reconstruction in Hawai'i to a larger landscape scale.

Use of non-native species in restoration

We suggest that there are situations in Hawai'i where mixed native and non-native but non-invasive plants can provide significant ecosystem benefits and that restoration practitioners in Hawai'i consider moving away from the goal of restoring purely native ecosystems where project objectives can be attained more efficiently by restoring a mixed ecosystem. Ecologically-appropriate plant material used in restoration, as defined by Jones (2013), consists of plants fit for the planting site, compatible with the other members of the plant community, and not spreading to adjacent sites, although not necessarily native to the site. Restored plant communities may be made up of native and ecologically and culturally appropriate non-native plants if such plants help meet restoration goals. Such "novel ecosystems" (Hobbs et al. 2009) can be either temporary, where a non-native nurse crop can modify site conditions to allow establishment of native plants, or permanent, where a mixed forest of native and non-invasive, non-native species is the final goal (D'Antonio and Meyerson 2002; Schlaepfer et al. 2011). Non-native plants might be appropriately used in restoration projects in Hawai'i and in other island ecosystems where they can function as nurse crops for threatened and endangered plant species, provide habitat for rare fauna, protect soil and water quality, or restore watershed function. There is also potential for non-native plants, especially the early Polynesian introductions (i.e. those plants brought to Hawai'i by the first Polynesian settlers before Western contact), to provide important cultural ecosystem services such as food provisioning and ceremonial uses (Winter and McClatchey 2008; Pejchar and Mooney 2009). Non-native plants would not be appropriate where they are not compatible with the desired plant community, where they are invasive, or where they do not help meet overall project objectives.

Because many deliberately introduced plants have turned out to be invasive in Hawai'i (Daehler 2009) and other tropical islands (Denslow 2003), care must be taken in the selection of non-native plants for use in restoration projects. Daehler et al. (2004) developed a screening system for Hawai'i based on Australian and New Zealand models (Pheloung et al. 1999). The model analyzes risk of invasiveness using on a set of questions based on inherent plant characteristics and behavior in other geographic regions. The system as adapted for Hawai'i is available on line (www.plantpono.org) and currently widely used in planning forestry projects. The model was validated against expert opinion and correctly identified 95 % of potential invasive plants while clearing 85 % of non-invasive species. The system is transparent in that users may download the questions to examine conclusions for any particular plant.

Non-native plants would also not be appropriate if they change ecosystem properties in such a way to hinder restoration efforts, for example by facilitating more frequent and intense wildfires (e.g. Trauernicht et al., in press). Non-native grasses, particularly *Cenchrus setaceus* (fountain grass), *Schizachyrium condensatum* (bushy beardgrass), and *Melinis minutiflora* (molasses grass) in dry areas (D'Antonio and Vitousek 1992) and *Megathyrsus maximus* (Guinea grass) in wet areas (Ellsworth et al. 2013), can outcompete native plants and change disturbance regimes and are one of the major impediments to native plant restoration. Because the dominant tree of many of Hawaii's forests, *Metrosideros polymorpha* (ʻōhiʻa, Myrtaceae), is slow-growing (Mueller-Dombois and Fosberg 1998; Friday and Herbert 2006), establishing plantings of this species requires long-term weed management. Alternatively, fast-growing non-native tree species could be used to establish forest cover and create a matrix for planting native understory plants. However, the use of non-native and non-invasive trees as cover crops in Hawaiʻi has received very little attention, so specific recommendations for a given site rarely exist. Still, this is a potentially fruitful area for further exploration.

Plantations of non-native trees that can shade out grasses have been used in other areas of the world to recover grass-dominated sites (Parotta et al. 1997). However, extensive native plant colonization of exotic plantations has seldom been observed in Hawaiʻi, probably because most non-native plantation tree species can outcompete native vegetation and do little to create suitable habitat for native plants (Mascaro et al. 2008; Ostertag et al. 2008). Harrington and Ewel (1997) did find significant native vegetation growing under *Fraxinus uhdei* plantations in a wet forest on Hawaiʻi Island. Poor establishment of the *Fraxinus* plantation allowed the native *Metrosideros polymorpha* and other native plants to re-establish naturally, but it is unlikely that the *Fraxinus* functioned as a nurse crop. Mascaro (2011) documented two shade tolerant native understory tree species (*Psychotria hawaiiensis* and *Psydrax odorata*) growing under a mixed non-native plantation in greater abundance than they were growing in native forest, suggesting that there could be some role for non-native trees in restoration of a partly native forest. While most of the native wet forests in Hawaiʻi are dominated by a single species, *Metrosideros polymorpha*, there is more diversity in the understory and shrub layers (Mueller-Dombois and Fosberg 1998). In Costa Rica, Holl et al. (2010) found that the fast-growing, non-native, N-fixing trees *Erythrina poepiggiana* and *Inga edulis* facilitated growth of slower-growing native trees in the understory. In wet areas with dense growth of *Megathyrsus maximus, Hibiscus tiliaceus* (beach hibiscus), a native or possibly Polynesian-introduced shrub, could be planted to suppress the grass and then converted to other uses, as is done on other Pacific Islands (Ishmael Lebehn, pers. comm.). Plantations of non-native *Acacia auriculiformis*, *A. mangium*, and *A. confusa* have been used to reclaim degraded grasslands and allow establishment of native species in the understory on the Micronesian islands of Yap and Guam (Francis Ruegorong, Justin Santos, pers. comm.).

In Hawaiian lowland wet forest Ostertag et al. (2015) suggested using a functional trait approach to select species based on restoration objectives. Past restoration attempts have proven unsustainable, primarily because native species are highly conservative in regards to growth and resource acquisition and are quickly outcompeted by non-native species following disturbance. The main principles in the functional trait approach is that species' traits reflect their resource use and life history tradeoffs, and that species-level knowledge can in turn inform us about how ecosystems function (McGill et al. 2006). Plant functional traits can also be valuable predictors of restoration success as they are linked to ecosystem processes and services (Lavorel 2013). By selecting traits that may beneficially manipulate abiotic variables such as light and resource use and allocation, managers can construct species assemblages that favor both native biodiversity and provide resistance to problematic invaders.

Wildfires are common in dryland forests invaded by non-native grasses in Hawaiʻi and throughout the tropics, killing native trees and opening the forest canopy for increased invasion of grasses (Hughes et al. 1991; D'Antonio and Vitousek 1992; Trauernicht et al. in press). The dense canopies of some tree species can shade out grasses enough to break the grass-fire cycle (Trauernicht et al. 2012). In fire-prone sites on the island of Molokaʻi, plantations of non-native trees have been integrated into fuel breaks as a barrier to grass growth and thus fire spread (Lance DeSilva, pers. comm). In dry woodland areas in Hawai'i, non-native grasses will probably be impossible to exclude with native species except in small areas, as these open forests have sparse canopies that do not shade out invasive grasses (Denslow 2003). A promising management strategy is to enrich these sites with native plants that have been shown to resprout or regenerate profusely after fires, thus increasing the resiliency of the native forest to fire (Loh et al. 2004; D'Antonio et al. 2009). A mix of native and non-native trees and shrubs, even if sparse, would have some effect in breaking up fuel continuity and decreasing grass fuel loads. Rare plants could be grown in intensively-managed grass-free islands in a matrix of mixed native and non-native stands. Windbreaks and fire breaks composed of fast-growing, non-native, non-invasive species could also aid in restoration. Another mixed system that includes a significant native plant component is shade coffee under natural forest (Elevitch 2009). Some coffee farms also cultivate native understory plant species in a mix with the coffee trees.

Use of remote sensing to select favorable restoration sites

Remote sensing offers a way to prioritize field-based restoration efforts and to assess the progress, threats to, and success of a restoration project. Specifically, restoration areas can be prioritized by characteristics such as topography, soils, vegetation type, biomass, and fuel loads. This process of using a biophysical perspective afforded by high-resolution remote sensing can contribute to immediate and long term restoration planning. Questad et al. (2014) developed a technique to use remote sensing to identify areas of high suitability in a dry ecosystem where water is limiting to growth and desiccating winds reduce plant productivity and survival. In a subalpine dry site (<400 mm annual precipitation) on Hawai'i Island, they found that environmental growing conditions were superior on lee and toe slopes and that common and some at-risk native plants had higher biomass and were more abundant on these sheltered sites. From this information, they created a habitat suitability model by generating a high resolution digital elevation model generated by the Carnegie Airborne Observatory light detection and ranging (LiDAR) data (2.2 m scale) to map a 49,000 ha landscape. Preliminary data collected from Vandenberg Air Force base in California confirms the utility of the habitat suitability model in other dry regions. High resolution restoration potential maps such as the habitat suitability model in arid systems can help guide land managers to more efficiently and effectively use resources to prioritize lands that offer high potential for restoration success. This type of planning information often provides a quantitative approach that can be useful for making decisions on landscapes that are designated for multiple use activities where areas of high native plant suitability are set aside for restoration and conservation and low suitability areas can be released for other activities.

Production of improved planting material

Restoration projects could likely improve outplanting success and reforest more acreage at the same cost by contracting with large-scale professional nurseries. A major part of the high cost of restoration is the cost of planting material, yet many restoration projects in Hawai'i rely on very small dedicated nurseries run by general practitioners with little nursery experience. Larger, well-equipped nurseries will be able to produce better quality seedlings less expensively. They will be able to invest in proper forestry containers to produce healthy root plugs rather than polybags or flowerpots from local garden supply shops. While small, project-specific nurseries often produce only hundreds of seedlings per year, larger native plant nurseries produce hundreds of thousands of native plants annually, drastically reducing the overhead costs per seedling. Most restoration projects prefer to use local germplasm wherever possible, and larger nurseries have been able to collect seeds or planting material at the project site and propagate it at an off-site nursery. Transport of pests (Joe and Daehler 2008) and diseases (Uchida et al. 2006) on infected nursery stock is a major problem and larger nurseries will also be more likely to have professional staff with up-to-date knowledge of bio-security issues. We suggest that local restoration projects look beyond their immediate environs and solicit bids statewide for propagation of high quality planting materials. One drawback of outsourcing nursery production, however, is that local community groups would not be able to be involved in seedling production.

Restoration projects could also improve their success by paying more attention to planting stock and employing the "target tree" concept (Landis and Wilkinson 2014). The "target tree" is defined as the one that does best at the planting site. It requires that nurseries take into account (1) objectives of the project, (2) site conditions, (3) limiting factors at the site, (4) availability of mitigating measures for limiting factors, (5) species desired and genetics, (6) stock type, (7) outplanting tools and techniques available, and (8) timing of the outplanting window. For example, Jacobs et al. (unpublished) found that *Acacia koa* seedlings grown in larger containers (656 cm^3) and with higher rates of fertilization grew significantly taller and had larger root collar diameters than seedlings grown in smaller, conventional containers (49 cm^3) when planted at a mesic but weedy site with deep soils, and Pinto et al. (2015) found that even a modest increase in seedling container size (207 vs 111 cm^3) yielded increased growth. Larger size helped the trees overcome competition from grasses, although smaller seedlings would have been easier to transport and plant, important considerations for remote or rocky sites. Alternatively, restoring *Acacia koa* forests in Hawai'i can be accomplished using seedlings grown in small containers (65 cm^3) but with correspondingly higher investments in superior site preparation. Nutrient loading [i.e., applying fertilizers past the point of increased growth to where the seedlings take up excess nutrients (luxury consumption)] has also been shown to increase N concentration in *Acacia koa* seedlings (Dumroese et al. 2009), which should help them to establish well in N-poor grasslands. Cordell et al. (2008) found that vines had twice the survival rate of tree seedlings in an extremely dry site and recommended these for initial planting on disturbed sites. Coordination with field crews regarding timing of planting is critical, as trees can be vulnerable to frost at high elevation sites during winter months (Scowcroft and Jeffrey 1999), and managers need to be able to plant dry sites during the wettest part of the year.

Worldwide, information on propagation of rare and native plants has been lacking (Cabin et al. 2002), but nurseries are now able to benefit from decades of experimentation with plant propagation protocols published in books (Lilleeng-Rosenberger 2005) and on websites (e.g. the Hawaiian Native Plant Propagation Database http://www.ctahr.hawaii.edu/hawnprop/ and the Native Plant Network Propagation Protocol Database http://www.nativeplantnetwork.org/Network/). Information on seed biology for Hawaiian plants is also becoming more available (Yoshinaga 2001).

Even when the appropriate species are decided upon, mangers need to select plant material of appropriate genetics, both to ensure that plants do well in the site and to preserve the genetic diversity in the species overall. Because many Hawaiian plants are so rare and seeding is often irregular, growers are often forced to propagate from a limited pool of individuals. Collecting from small populations raises concerns about limiting genetic diversity of outplants and of impacting the ability of the wild population to reproduce (Smith et al. 2007; Schaal and Leverich 2005; Robichaux et al. 1997). There are agency specialists and programs, such as the Hawai'i Plant Extinction Prevention Program, dedicated to conserving the genetics of Hawaii's rarest species by collecting seeds from as many plants as possible. Concerns about local adaption in these situations are secondary to the need to prevent the species from going extinct (Schaal and Leverich 2005). For common plants, however, the general practice in Hawai'i is to collect from a number of local populations to retain diversity of planting material. Although outbreeding depression can be a concern when plants from widely spaced populations are planted together (Vander Mijnsbrugge et al. 2010), there is little local information available to evaluate possible negative effects. "Ecotypes" are not well defined for Hawai'i and there is limited information on whether local ecotypes are adapted for local conditions, although in one case Ares et al. (2000) found that *Acacia koa* trees from drier sites had a greater genetic water use efficiency than trees from wetter sites when grown in a common garden. In general, restoration practitioners seek to use propagation materials from the same island or region as the project, with each island becoming a "seed zone" (Vander Mijnsbrugge et al. 2010). This approach, however, may not be the best way to obtain locally adapted plants. Environments may vary greatly within a short distances and populations from a similar environments on a different island might be better adapted to the local planting site than populations from a different environment on the same island (Hufford and Mazer 2003). While most native Hawaiian species are mycorrhizal (Koske et al. 1992) and studies have shown that mycorrhizal inoculation improves early growth in some native species (Miyasaka et al. 1993; Habte et al. 2001; Gemma et al. 2002), native plant seedlings are seldom inoculated in the nursery. While inoculation may not improve growth on sites with healthy soils, many restoration sites also have low P soils where inoculation could improve P uptake and plant establishment (Gemma et al. 2002).

Lastly, an important aspect of the target tree concept is that the nursery manager works with the project manager to get feedback on how well the seedlings did at the site to make any necessary changes for the following year. Ideally, projects would have an initial phase of experimentation and adaptive management before undertaking restoration of large areas. However, often budget and grant concerns dictate planting of significant areas before any trials can be evaluated properly. Restoration after wildfires also can demand hundreds of thousands of seedlings at short notice without giving any time for experimentation as to what works best at the planting site.

Improved site preparation and weed control

Weed competition is one of the main reasons for reduced growth and mortality in restoration projects (Cabin et al. 2002; Denslow et al. 2006). Even though restoration projects usually do not have future commercial timber harvests in mind, more rapid growth of planted trees can achieve desired objectives, such as endangered bird habitat or carbon sequestration, in shorter time. Vegetation management is often the primary action in site preparation. While most seedlings are planted by hand or sometimes with gas-powered augers (Jeffery and Horicuhi 2003), one forestry company in Hawai'i has been having success using a spot tiller mounted on an excavator to prepare planting sites by scalping weeds from the soil surface and deep ripping the planting spot (http://www.wilco.co.nz/

cultivator.html). The excavator allows the cultivator to be moved to good planting locations and avoids disturbing the soil over an entire area. Another model for rare plant restoration abandons naturalistic random spacing of outplants and grows plants in rows as in an agricultural system (Burney and Pigott Burney 2009). This arrangement allows for weed control using standard agricultural implements. Both these techniques borrow from industrial forestry to improve efficiency and success in ecological restoration.

Most *Acacia koa* restoration activities take place in abandoned pastures dominated by *Cenchrus clandestinus* (Kikuyu grass). While glyphosate has generally been used for chemical weed control (Cabin et al. 2002; Brooks et al. 2009), combining glyphosate with imazapyr for longer term grass control has been shown to be more effective on weedy planting sites (Pinto et al. 2015; Leary and Gross 2013). Better grass control can lead to decreased moisture competition for seedlings (Pinto et al. 2015), an important consideration in mesic and dry sites. Timing of application is critical with herbicides, which are best applied while target plants are actively growing. Pre-plant applications are the most opportune to minimize collateral impact of broad-spectrum herbicides. Weed management options are greatly reduced once trees are planted. New information is also emerging on which herbicides are most effective against woody invasive plant species (Leary 2012). McDaniel et al. (2011) found that the best site preparation was achieved with a combination of turning over the soil surface to control grasses then applying herbicides to prevent re-establishment of grasses.

Direct seeding

Because of the high cost of seedling production and outplanting, there have been several attempts to restore native vegetation in Hawai‘i through direct seeding where natural regeneration is lacking (Denslow et al. 2006). In some instances where there has been good weed control and subsequent rainfall, native plants have been successfully restored by direct seedling (Brooks et al. 2009; Loh et al. 2004; McDaniel et al. 2011), but in many other trials direct seeding has not worked because of drought, poor seed germination, and weed competition (Ammondt et al. 2013). Direct seeding may be the only way to quickly restore large areas to native vegetation after wildfires, as most native seedlings take months to years to propagate. One constraint is the availability of the large quantities of native plant seed needed (Loh et al. 2004). To alleviate that bottleneck, native seed banks are now being developed in Hawai‘i (Hawai‘i Island Native Seed Bank Cooperative, http://www.hawaiiforestinstitute.org/our-projects/hawaii-island-native-seed-bank/). Loh et al. (2004) also suggested seed orchards for species with recalcitrant seeds that cannot be stored for long periods of time. A further need is the technology and information on questions of seed biology such as how to break dormancy, as many Hawaiian plant species have seeds that are slow to germinate unless scarified (Denslow et al. 2006; Allen 2002; Scowcroft 1978).

Community involvement

Whatever the technologies developed, forest restoration needs support from the community if it is to succeed. Worldwide, negative human impacts are usually the reason for ecosystem degradation, and community involvement in limiting anthropogenic impacts is often necessary before restoration can even begin (Walters 1997). The Society for Ecological Restoration (2004) noted that collective decisions by local communities are more likely to be adhered to by the communities than decisions made from above and that ecological restoration should encourage long term participation of local people. The need for community involvement is especially true in places such as Polynesia, Micronesia, and elsewhere where indigenous traditions of forest use are at odds with western-style management priorities (Friday and McArthur 2010). In Hawai'i, the first step in restoration is usually to fence and control feral ungulates (McDaniel et al. 2011). Public support for such actions will only be gained if the public values the rare and native plants and animals and services provided by restored ecosystems. As most wildfires are human caused (Trauernicht et al. in press), it is also critical to have community support for restored landscapes to avoid arson. Forest restoration on public lands can be more enthusiastically embraced by local communities if non-native but valuable and culturally important species can be incorporated into planting designs and made available for public use. Lamb et al. (2005) observe that tropical forest restoration has traditionally proceeded either by planting fast growing crops of non-native commercial tree species, which have high financial and livelihood, but low biodiversity benefits, or by planting mixes of native plants that maximize biodiversity but have little commercial value. They suggest that foresters can develop pathways between these two extremes that optimize total benefits, either by increasing a native species component in non-native plantations or by adding species of higher commercial or cultural benefits to native restoration projects. In Hawai'i, project managers have introduced breadfruit (*Artocarpus altilis*) as one tree component of the lowland wet forest restoration project described by Ostertag et al. (2015). Although not native to Hawai'i, breadfruit is one of the original Polynesian introductions to Hawai'i (Zerega et al. 2004) and valued both for local use and as a cultural icon. The tree's inclusion in the project generated local support. Private landowners could also be more attracted to planting native species in a mix of economically valuable trees where the two can be grown together.

Community members can also be sources of indigenous technical and ecological knowledge as well as labor for restoration projects, as was shown in forest restoration projects in the Philippines (Walters 1997). In Hawai'i, most groups doing restoration rely on volunteers to plant and weed and conduct public education and outreach efforts [e.g. Hakalau National Wildlife Refuge (www.friendsofhakalauforest.org), Hawai'i Volcanoes National Park (http://fhvnp.org/), Waikoloa Dry Forest Initiative (www.waikoloadryforest.org), Hui o Laka (www.kokee.org), Mauna Kea Restoration Project (www.restoremaunakea.org), Leeward Haleakalā Watershed Restoration Partnership (www.lhwrp.org) and (www.auwahi.org), and the Hawai'i Forest Institute Mahalo 'Āina program (http://www.hawaiiforestinstitute.org/our-projects/mahalo-aina/)]. Young people are especially attracted to volunteering in restoration efforts. As people of Hawai'i realize that their well-being is tied to preservation of ecosystems and that their culture is also inextricably woven into the native plants and animals of these islands, we anticipate increased success of native forest restoration despite the technical difficulties.

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