# Estimating Cost-Effectiveness of Hawaiian Dry Forest Restoration Using Spatial Changes in Water Yield and Landscape Flammability Under Climate Change 1

Authors: Wada, Christopher A., Bremer, Leah L., Burnett, Kimberly, Trauernicht, Clay, Giambelluca, Thomas, et al.

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# Estimating Cost-Effectiveness of Hawaiian Dry Forest Restoration Using Spatial Changes in Water Yield and Landscape Flammability under Climate Change[1]

*Christopher A. Wada,*[2,9] *Leah L. Bremer,*[2,3,4] *Kimberly Burnett,*[2] *Clay Trauernicht,*[5] *Thomas Giambelluca,*[6] *Lisa Mandle,*[4] *Elliott Parsons,*[7] *Charlotte Weil,*[4] *Natalie Kurashima,*[3,8] *and Tamara Ticktin*[3]

**Abstract:** Resource managers increasingly seek to implement cost-effective watershed restoration plans for multiple ecosystem service benefits. Using locally adapted ecosystem service tools and historical management costs, we quantified spatially explicit management costs and benefits (in terms of groundwater recharge and landscape flammability) to assist a state agency in evaluating cobenefits for a predefined restoration scenario (focused on biodiversity benefits) and to prioritize an expanded restoration scenario in the state-managed Pu‘u Wa‘awa‘a watershed (Hawai‘i) now and under the Representative Concentration Pathway (RCP) 8.5 midcentury climate scenario. Restoring all available areas increases recharge by ~1.74 million m$^3$/yr (5% of recharge over the entire watershed) under the current climate and does not meaningfully change recharge under RCP 8.5 midcentury, whereas climate change decreases recharge by ~50%. For landscape flammability, climate change increases the median and maximum probability of fire occurrence across all land use scenarios, and full restoration results in the greatest reduction in landscape flammability under both current and RCP 8.5 midcentury climate scenarios. We demonstrate that location and type of forest restoration influence overall cost-effectiveness of restoration, providing insights for landscape planning for ecosystem services under a limited budget. Across all scenarios, capturing potential benefits at low elevations requires greater expenditures ($13,161/ha) than at high elevations ($5,501/ha) due mainly to the substantial costs of removing *Pennisetum setaceum* (fountain grass), the dominant land cover below 1,000 m. If management focuses on groundwater recharge only, the most cost-effective areas occur at high elevations (>1,000 m), with ample fog interception, although recharge benefits decline across the landscape under RCP 8.5 midcentury. Focusing instead on cost-effective landscape flammability reduction as the primary management objective shifts emphasis toward dry low-elevation areas under the current climate. However, under the RCP 8.5 midcentury scenario, the most cost-effective areas for flammability management shift toward higher elevations with greater potential overlap with recharge benefits.

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[2] University of Hawai‘i Economic Research Organization, University of Hawai‘i at Mānoa, Honolulu, Hawai‘i 96822.

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[3] Department of Botany, University of Hawai‘i at Mānoa, Honolulu, Hawai‘i 96822.
[4] Water Resources Research Center, University of Hawai‘i at Mānoa, Honolulu, Hawai‘i 96822.
[5] Department of Natural Resources and Environmental Management, University of Hawai‘i at Mānoa, Honolulu, Hawai‘i 96822.
[6] Department of Geography, University of Hawai‘i at Mānoa, Honolulu, Hawai‘i 96822.
[7] Hawai‘i State Department of Land and Natural Resources, Division of Forestry and Wildlife, Honolulu, Hawai‘i 96813.
[8] Kamehameha Schools, Kailua-Kona, Hawai‘i 96740.
[9] Corresponding author (e-mail: cawada@hawaii.edu).

The majority of research on the benefits of native forest restoration focuses on changes in biodiversity, including measures such as abundance, species richness, and species diversity (Benayas et al. 2009). More recently, elevated attention to the potential for watershed restoration to provide additional benefits, such as groundwater recharge, soil retention, flammability reduction, and carbon storage, has led to increased interest in developing tools to measure and incentivize the provision of whole suites of ecosystem services (Bagstad et al. 2013, Sharp et al. 2016). Nonetheless, missing or incomplete assessments of restoration benefits continue to constrain funding availability in many regions across the globe (de Groot et al. 2013). Watershed-scale analyses that link specific management activities and their costs to ecosystem services of interest simultaneously provide valuable information for land managers planning future restoration projects, while also strengthening the general argument for increased investment in forest restoration.

Forest restoration is a common strategy for biodiversity conservation in Hawai‘i, where habitat destruction and invasive species have severely degraded native ecosystems (Friday et al. 2015). About 90% of Hawai‘i's vascular flora is endemic (Wagner et al. 1999), and over 40% of endemic species are listed as critically endangered or threatened (USFWS 2012). In addition to the benefits of biodiversity such as species richness and abundance, there is increasing interest and effort to restore and protect native forests for their potential contribution to groundwater recharge and reduced sediment delivery to coral reefs (DLNR 2011, Bremer et al. 2015). Although global research has demonstrated cobenefits of forest restoration beyond biodiversity (Benayas et al. 2009), this remains poorly explored in Hawai‘i, with the exception of limited research on evapotranspiration in native forest versus those dominated by nonnative species (Kagawa et al. 2009, Engott 2011, Burnett et al. 2017, Strauch et al. 2017). Although biodiversity will always be a critical objective of forest restoration in Hawai‘i, leveraging additional resources for forest restoration requires being able to prioritize and quantify potential outcomes based on cobenefits such as increased water yield and fire risk reduction. The ecosystem service assessment described in this article integrated economics and ecosystem service outcomes to widen the net of conservation benefits and inform decisions faced by land and resource managers about future investments in forest restoration at our study site.

We drew on a collaboration with resource managers at Pu‘u Wa‘awa‘a, a Hawai‘i state forest reserve, to undertake this assessment. Pu‘u Wa‘awa‘a is located on the leeward coast of Hawai‘i Island and, as in other dry regions across the state, restoration of dry forest species, fire prevention, and groundwater recharge are critical management objectives (DLNR 2003). Hawai‘i's dry forests have suffered major deforestation and degradation: more than 90% of their habitat has been lost and they contain the highest proportion of Hawai‘i's at-risk taxa (Bruegmann 1996, Sakai et al. 2002). Concerns about groundwater recharge are largely motivated by Hawai‘i's unique freshwater situation: groundwater accounts for 63% of the state's total freshwater withdrawals (Mandler 2017), and that number jumps to over 95% when only domestic uses are considered. Fire prevention has also recently become one of the top land management priorities in the region, given the increasing prevalence of wildfires in the drier, leeward watersheds of Hawai‘i and the threats they pose to both human communities and native biodiversity (Trauernicht et al. 2015). In these already dry areas, rainfall is declining (Frazier and Giambelluca 2016) and is expected to decrease further with climate change (Elison Timm et al. 2014).

Pu‘u Wa‘awa‘a watershed is an ahupua‘a (a traditional Native Hawaiian political-ecological land boundary) that spans 16,280 ha of the North Kona region of Kekaha, between Honōkohau and Pu‘u Anahulu. It stretches from sea level to within 2 km of Hualālai volcano's summit. Mean annual rainfall ranges from 240 mm at the coast to 1,500 mm at the upper reaches of the watershed (Giambelluca et al. 2013). The watershed contains some of the state's largest tracts of remnant native

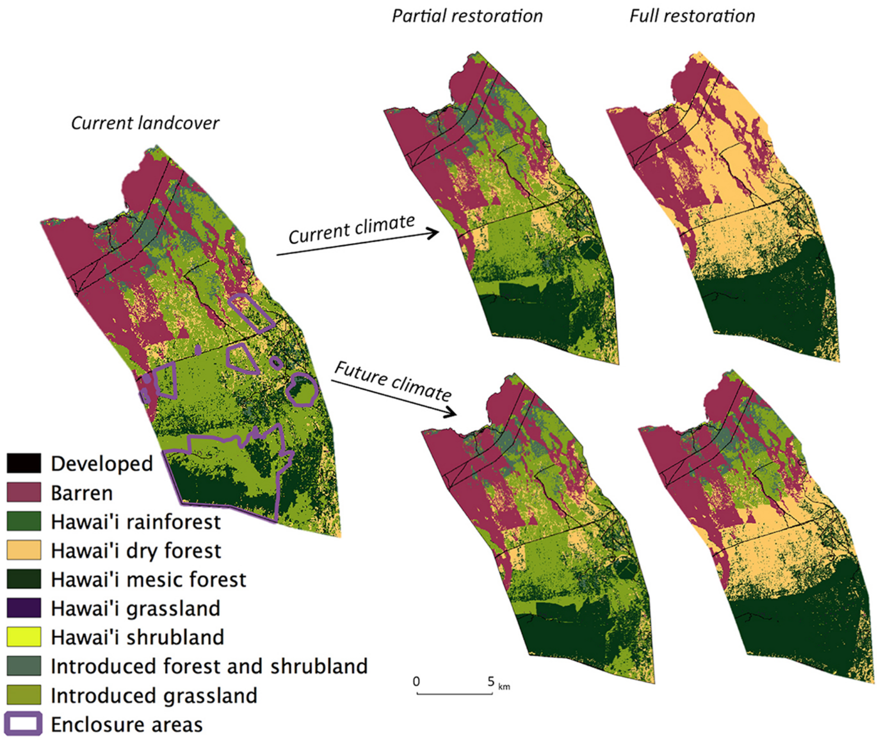


Figure 1. Land cover in Puʻu Waʻawaʻa under different restoration and climate scenarios. The purple contours on the current land cover map outline the enclosure areas corresponding to the partial restoration scenario. The upper maps on the right correspond to the evolution of land cover, under the current climate for the partial and full restoration scenarios. The maps below correspond to the evolution of the land cover, under the RCP 8.5 midcentury future climate projection for the partial and full restoration scenarios.

dry forest, but wildfires, decades of livestock grazing, and plant invasions have degraded much of the native vegetation over the past century, and the landscape is now primarily dominated by nonnative grasses (Blackmore and Vitousek 2000). Currently, the area contains a mix of land uses including managed grazing in the midelevation grasslands, with conservation and restoration efforts focused mostly within and adjacent to the forested areas at higher elevations (Figure 1). It is managed by the Hawaiʻi Department of Land and Natural Resources (DLNR) Division of Forestry and Wildlife.

Puʻu Waʻawaʻa managers have designed a restoration strategy focused on a series of high-feasibility and low-cost exclosures they have established. These exclosures were prioritized for restoration efforts because of their biodiversity conservation benefits: they (1) contain a relatively high proportion of native canopy trees and/or (2) contain a high number of endangered plants and/or (3) are good areas for replanting (good soil, close to a forest and therefore to dispersal of native seeds, etc.) (Garcia and Associates 2016). However, Puʻu Waʻawaʻa managers are very interested in factoring in other major concerns to their restoration and management strategies that they have not before, including considerations about their effectiveness now and under a future climate. The persistently dry climate and large, continuous expanses of nonnative grasses, especially fountain grass (*Pennisetum setaceum*), throughout the lower and midelevation areas of the watershed pose a major fire threat to the watershed's remaining forest tracts (DLNR 2003). As a result, managers have to plan for and integrate fire risk mitigation efforts at spatial scales much larger than the remnant forests and restoration areas. In addition, managers are interested in quantifying potential benefits in terms of groundwater recharge in a context of elevated interest in incentivizing forest restoration for water outcomes, particularly in the context of climate change (DLNR 2011).

In collaboration with Puʻu Waʻawaʻa managers, we therefore identified the potential cobenefits of currently planned and potential future restoration efforts. Specifically, we quantified spatially explicit management costs and benefits (groundwater recharge and landscape flammability) to identify the cobenefits of the current restoration plan (focused on biodiversity benefits from restoring the exclosures) and to prioritize an expanded restoration scenario based on groundwater recharge and reduced landscape flammability now and under the Representative Concentration Pathway (RCP) 8.5 midcentury climate scenario.

## MATERIALS AND METHODS

### *Scenarios*

We considered three land management scenarios: current land cover, partial restoration based on restoration plans by the state land managers (i.e., restoration of the series of exclosures), and full restoration of the ahupuaʻa. We evaluated these land management scenarios under the current climate and the RCP 8.5 midcentury climate scenario (IPCC 2014). RCP 8.5 was developed as a high emission scenario, though still within the range of business-as-usual projections, although emissions are currently on a trajectory to exceed it (Sanford et al. 2014).

CURRENT LAND COVER. Baseline land-cover maps were generated using Landfire (2012) data and corrected through consultation with the state manager, who knows the area well. All areas classified as Hawaiʻi lowland dry shrubland, Hawaiʻi lowland mesic shrubland, and Hawaiʻi montane-subalpine dry shrubland were reclassified as Hawaiʻi introduced perennial grassland, because these areas were deemed misclassified by the land manager.

PARTIAL RESTORATION (EXISTING EXCLOSURES RESTORED). Introduced perennial grassland, nonnative forest, and nonnative shrubland within planned state exclosures are restored to native forest. Restored forest types were determined by elevation gradients, in accordance with the current distribution of native forest in the region: Hawaiʻi lowland dry forest below 1,000 m and Hawaiʻi montane subalpine mesic forest above 1,000 m. Given the historical and current distribution of forest in the study region, we also assume that restoration is not possible in areas with less than 350 mm of annual rainfall. Under the current climate scenario, 1,437 ha (8.2% of the watershed), 99% of which falls into the perennial grassland category, is restored to 24% native dry forest and 76% native subalpine mesic forest. Under RCP 8.5 midcentury, total restoration in the exclosures is reduced by 1 ha to 1,436 ha because a small area falls below 350 mm of rainfall and is deemed too dry to restore.

FULL RESTORATION. Introduced perennial grassland, nonnative forest, and nonnative shrubland within the entire watershed are restored to native forest. As in the partial restoration scenario, we assume that restoration is not feasible in areas with annual precipitation below 350 mm. The total restorable area (6,520 ha, or 40% of watershed) is composed of 96% introduced perennial grassland, 2.5% introduced deciduous shrubland, 1.3% introduced wet-mesic forest, and <1% introduced dry forest. Under the current climate scenario, the postrestoration distribution of native forest type (68% native dry forest and 32% native subalpine mesic forest) was determined by elevation gradients and a precipitation constraint (dry forest below 1,000 m; no restoration below 350 mm of precipitation). Under the RCP 8.5 midcentury scenario, restoration is reduced to 5,294 ha given that a greater proportion of the watershed is too dry to restore. The postrestoration distribution of land cover shifts to 56% dry forest and 46% subalpine mesic forest. Table 1 presents the total restoration area for each of the scenarios, and Figure 1 illustrates land cover

TABLE 1

Land-Use and Climate Change Scenarios Evaluated in Pu'u Wa'awa'a

| Scenario | Description | Restoration Area |
|---|---|---|
| Current-current | Current land cover; current climate | — |
| Partial-current | Restoration of exclosures only; current climate | 1,437 ha |
| Full-current | Restoration of all nonnative vegetation; current climate | 6,520 ha |
| Current-RCP 8.5 | Current land cover; RCP 8.5 midcentury climate scenario | — |
| Partial-RCP 8.5 | Restoration of exclosures only; RCP midcentury climate scenario | 1,436 ha |
| Full-RCP 8.5 | Restoration of all nonnative vegetation; RCP midcentury climate scenario | 5,924 ha |

across the watershed for each management-climate pair.

### *Groundwater Recharge*

Given the lack of surface water flow in the Kona region (Brauman et al. 2014), we used a water balance approach where:

$$\text{Groundwater Recharge} = \text{Rainfall} + \text{Fog Interception} - \text{Evapotranspiration (ET)}.$$

We obtained current climate rainfall from the Rainfall Atlas of Hawai'i (Giambelluca et al. 2013) and used statistically downscaled rainfall data for RCP 8.5 midcentury (Elison Timm et al. 2014). [Note: The rainfall projection used here should be considered one of a wide range of possible futures, given uncertainties in (1) global models used, (2) downscaling method, and (3) parameter choices and other assumptions used in downscaling. For example, although results from a dynamical downscaling study for Hawai'i (Zhang et al. 2016) show some similarities in change patterns with results from the statistical downscaling approach, rainfall projections based on dynamical downscaling are generally higher.]

Following Engott (2011), we estimated fog as a function of elevation, vegetation, and precipitation:

$$F = P \times FIR \times FCE$$

where F = fog interception, P = precipitation as rainfall, FIR = fog interception ratio, and FCE = fog-catch efficiency. FIR and FCE values were based on Juvik and Ekern (1978 *in* Engott 2011) and Engott (2011); FIR increased with elevation, and FCE varied by land-cover type (FCE = 0 for grassland, 0.5 for shrubland, and 1.0 for forest). P varied under the midcentury RCP 8.5 climate scenario (Elison Timm et al. 2014), but FIR and FCE were held constant due to lack of information regarding their potential responses to changes in climate. FIR values were based on empirical observations of the relationships between fog captured in a mechanical fog gauge and a rain gauge for the study area (Juvik and Ekern 1978 *in* Engott 2011). FCE values were relative approximations of the effectiveness of different vegetation types in capturing fog based on vegetation height. Although FIR and FCE may change with climate change, we did not have sufficient information to adjust these parameters and recognize this as an important area for future research.

To estimate ET, we created regression equations with annual latent heat equivalent (LE) of ET (W $m^{-2}$) as a function of annual forcing variables (air temperature, net radiation, relative humidity, wind speed, available soil moisture, leaf area index, canopy cover, and vegetation height) (Giambelluca et al. 2014). One regression equation was estimated for each land cover that currently exists in Pu'u Wa'awa'a watershed using 288,007 annual ET data points across the Hawaiian Islands. We first tested for collinearity and excluded variables with a variance inflation factor (VIF) > 5 and then tested for spatial autocorrelation, selecting the models with the lowest Akaike Information Criterion (AIC) value (Zuur et al. 2009). For Hawai'i lowland dry forest, we were not able to satisfactorily fit a statewide model for this vegetation type. We therefore based our model on the subset of points receiving <1,200 mm rainfall, given that rainfall was <1,200 mm in this study site. Annual LE values predicted by the regression equations for our study site were converted into ET in water units (mm) (Giambelluca et al. 2014).

To test the model fit [Note: To truly validate the model, we would have split the ET data, created the regression models on one subset, and tested those models on the other subset. The test for model fit here should be interpreted as calibration skill or the ability of the simplified regression model to fit the ET predicted by the full process-based model.], we compared ET estimates from our regression model to the full model estimates produced by Giambelluca et al. (2014) for each land-cover type. Adjusted $R^2$ values were as follows: 0.91 for introduced perennial grassland (pasture), 0.92 for Hawai'i subalpine mesic forest, and 0.66 for Hawai'i lowland dry forest. $R^2$ is an imperfect, and likely conservative, measure of model fit given that it does not account for the effect of spatial autocorrelation captured in the regression model (Zuur et al. 2009). Within the watershed as a whole, the difference in estimated ET between the full model (Giambelluca et al. 2014) and the regression model was <8%. We used the mean percentage error (along with standard deviations) between the full model and the regression model for each land-cover class to estimate the uncertainty of model predictions (see the Appendix for a complete description of uncertainty methods).

The ET for each land management scenario was calculated by assigning the appropriate land use–based ET regression equation to each pixel in the scenario. Thus, areas undergoing restoration in the partial and full restoration scenarios were assigned the appropriate forest cover regression equation. Because we found no relationship between precipitation and leaf area index, canopy cover, or vegetation height, we used median values from each forest type from within existing forested area. We did not vary available soil moisture by land management, given that the available soil moisture calculations for the full ET model were based solely on precipitation for nonirrigated land covers (Giambelluca et al. 2014). Annual net radiation values were adjusted by land-cover type (see description following).

We calculated ET under the midcentury RCP 8.5 climate change scenario by adjusting temperature, net radiation, and available soil moisture values. We follow IPCC RCP 8.5 midcentury average global temperature projections and increased temperature by 1.4°C across the study region (IPCC 2014). [Note: Shortly after our analysis was completed, Elison Timm (2017) released statistically downscaled temperature projections for Hawai'i. Our assumption of a 1.4°C increase does not largely diverge from the statistically downscaled results, and much of the change occurs in the second half of the century, beyond our planning horizon.] To capture the likely impacts of land cover and climate on mean annual net radiation, we calculated annual net radiation as a function of precipitation and land-cover class (divided into forest, shrub, grass, barren, and developed).

To adjust available soil moisture under climate change (RCP 8.5 midcentury scenario), we used a relationship developed by Giambelluca et al. (2014) using empirical monthly rainfall and soil moisture data from seven HaleNet stations ($R^2 = .8$):

$$y = 0.182 * \ln(x) + 0.2632$$

where y = available soil moisture (percentage) and x = average of current and previous month mean rainfall (mm/day). Although available soil moisture varies based on a number of factors, including soil type and rooting depth, this information was not available at a sufficient spatial scale necessary to incorporate these values into the statistical relationship (Giambelluca et al. 2014). We used statistically downscaled changes in wet- and dry-season rainfall (6-month wet season and 6-month dry season) (Elison Timm et al. 2014) [Note: Corrected statistical downscaling estimates from the study authors were used given errors found in the originally published article.] to estimate future available soil moisture amounts. All scenario calculations were done in ArcMap raster calculator.

### *Landscape Flammability*

To assess how land use and climate influence fire occurrence at Puʻu Waʻawaʻa, we used a 20-yr, annual fire history (1992–2011) comprising 91 fire perimeters (ranging from 1 ha to >10,000 ha) for the NW quadrant of Hawaiʻi Island (ca. 3,000 $km^2$) mapped by the Hawaiʻi Wildfire Management Organization (Castillo et al. 2006). We used a GIS to randomly sample 30 $m^2$ pixels annually across the landscape ($N = 150{,}000$ total; mean of $N = 7{,}500$ per yr) and classified each pixel as burned or unburned according to whether or not it occurred within a fire perimeter. We used this binomial response (burned/unburned) to model fire occurrence as a function of the following predictors, also sampled at each pixel, using a logistic Generalized Additive Model (GAM): (1) mean annual rainfall; (2) mean annual temperature; (3) land-cover type [grassland, shrubland, forest, agricultural, developed, other (Landfire 2012)]; (4) wildfire ignition density [using a point-based data set (see Trauernicht et al. 2015)]; (5) aspect; and (6) annual rainfall anomaly [the difference between annual and mean annual rainfall (Giambelluca et al. 2013, Frazier et al. 2015)].

The predictors (1–6 above) used in the model were selected to capture the fundamental landscape drivers of ecosystem fire occurrence: climate, vegetation, ignition source, and topography (e.g., Pausas and Keeley 2009). The first three predictors were also selected and tested to integrate and assess the effects of changes in vegetation cover due to management decisions within the Puʻu Waʻawaʻa watershed and in climatic factors under the RCP 8.5 midcentury scenario. Sampling and classifying random points as burned or unburned annually across the NW region of Hawaiʻi Island enabled us to fit a logistic (binomial) GAM that predicts the annual probability of fire occurrence at each pixel in the watershed. However, because the probability is derived from the spatial extent of annual fire occurrence over 20 yr across the larger study region, the predicted, per-pixel fire probabilities, which we refer to as "landscape flammability," are best interpreted as the type of fire regime (e.g., high or low frequency) supported by the landscape and climatic features at each pixel. This approach differs from the predicted fire risk typically quantified with fire spread models that rely on more complex inputs such as weather and fuel-specific parameterizations of fire behavior, many of which are unavailable and/or poorly calibrated for Hawaiʻi (e.g., Benoit et al. 2009).

We fit GAMs of the probability of fire occurrence as a function of all possible combinations of predictor variables (1–6 above), including an interaction between rainfall and land cover, an interaction between rainfall anomaly and land cover, and sample year as a random effect (Preisler et al. 2004) and ranked these against the null model using corrected Akaike's Information Criterion (AICc) (Burnham and Anderson 2013). The top-ranked model [the global model (Akaike Weight > 0.99; Explained Deviance = 25.9%)] was used to predict the per-pixel annual probability of fire occurrence across the Puʻu Waʻawaʻa watershed under current and projected climatic and land-cover conditions with the Raster package in R (Hijmans and Van Etten 2013). The explanatory power of the top-ranked model (25.9% explained deviance) was constrained by the underlying uncertainty and high temporal variation in weather conditions and human-caused ignitions that also drive fire occurrence in Hawaiʻi (Trauernicht et al. 2015). In addition, we explicitly did not include a spatial term in the model that would have improved model fit but detracted from the objective of characterizing the influence of landscape factors within the watershed, as per-pixel probabilities would have been weighted toward areas in the study region where historical fires were most common.

### *Management Costs*

FOREST RESTORATION. For the purpose of calculating costs, we separated forest restoration into three main categories: 1) ungulate removal and maintenance via fence installation and monitoring, 2) removal of nonnative plant species (mainly grasses) from enclosed areas, and 3) planting of native species in cleared areas.

*Ungulate removal and maintenance.* Fence installation costs were estimated using information from planned/past work in or near Pu'u Wa'awa'a (E. Parsons, unpubl. data). The average per-meter cost for posts and wire from 16 sites was adjusted to include gate installation costs, following the gate-perimeter ratio of the Henahena management unit fence. Annual fence maintenance costs were assumed to be similar to the State Division of Forestry and Wildlife's average annual expenditures on fence checks and repairs in recent years (N. Agorastos, pers. comm.).

Because the water and fire models generate output for a single period of time, a lumpy trajectory of fence installation and maintenance costs will not be directly comparable. To remedy this incompatibility, we calculate the present value (PV) cost of management for each hectare, assuming that fixed costs (e.g., fence installation) are spread equally across all pixels being considered for restoration.

Ungulate removal costs can vary widely by the type of ungulate, the (typically unknown) number of ungulates in the area, the size of the exclosure, the eradication method(s) chosen, and accessibility of terrain, among other factors. With those caveats in mind, the per-hectare ungulate removal cost used for our study was calculated based on past methods employed within the 81 ha mixed-mesic Waihou unit in Pu'u Wa'awa'a. The eradication effort included a 20-staff drive, a five-staff dog hunt, and 72 hr of helicopter time (E. Parsons, unpubl. data). There are currently no funds dedicated to fence maintenance and prevention of ingress in Pu'u Wa'awa'a, but managers expect that the watershed can be monitored with roughly 10 hr of helicopter flight time per year. In calculating the per-pixel cost of restoration, we assume that ungulate removal occurs over the first 3 yr of management, and maintenance costs are incurred annually thereafter.

NONNATIVE PLANT REMOVAL AND MAINTENANCE. Removal of nonnative plant species is a multistep process, requiring high effort during the initial year for clearing, followed by lower-effort maintenance thereafter. Nonnative land cover in the study area is composed almost entirely of fountain grass (*Pennisetum setaceum*) and kikuyu grass (*Cenchrus clandestinus*). Fountain grass, a hardier species with a high rate of regrowth after initial removal, is found primarily below elevations of 1,000 m. Kikuyu grass, which dominates areas above 1,000 m, has a lower rate of recurrence after initial treatment and is thus less costly to eradicate. In addition, some fast-growing native species can be planted directly into existing kikuyu grass beds, which further reduces the effort required for site preparation (E. Parsons, pers. obs.).

Removal costs for both types of grasses were estimated based on past restoration work completed in Pu'u Wa'awa'a (E. Parsons, unpubl. data). We assume that some fixed materials are purchased once every 5 yr (2,600 hr of use at an average rate of 10 hr per week), but other variable materials are consumed with each hectare cleared. The estimates do not include costs related to base-yard operations, but the omission does not affect the relative ordering of costs by elevation-based categories. Although all fountain grass–dominated areas required intensive weed clearing before outplanting, preparation costs for upland areas were adjusted downward to account for the assumption that fast-growing natives would be planted directly into 90% of areas currently dominated by kikuyu grass.

In addition to initial site preparation costs, we allowed for 7.4 hr of weed maintenance per hectare annually thereafter. Weed maintenance costs were assumed to be identical for low- and high-elevation areas.

NATIVE SPECIES OUTPLANTING. Once a parcel is cleared of nonnative grasses and prepared for restoration, additional outplanting costs (acquiring native plant seedlings and paying wages) are realized. Low-elevation (≤1,000 m) dry-forest restoration requires a larger number of rarer native species that grow relatively slowly (5–12 months grow-out time). In contrast, the mix of native

TABLE 2

Plant Species Used for Outplanting by Elevation Category in Puʻu Waʻawaʻa

| Low Elevation (≤1,000 m) | | High Elevation (>1,000 m) | |
|---|---|---|---|
| Common Name | Scientific Name | Common Name | Scientific Name |
| ʻaʻaliʻi | *Dodonaea viscosa* | ʻaʻaliʻi | *Dodonaea viscosa* |
| alaheʻe | *Psydrax odorata* | hau kuahiwi | *Hibiscadelphus hualalaiensis* |
| hala pepe | *Chrysodracon hawaiiensis* | hōʻawa | *Pittosporum hosmeri* |
| kauila | *Colubrina oppositifolia* | ʻiliahi | *Santalum paniculatum* |
| kōlea | *Myrsine lessertiana* | koa | *Acacia koa* |
| lama | *Diospyros sandwicensis* | kōlea | *Myrsine lessertiana* |
| ʻōhiʻa | *Metrosideros polymorpha* | mamane | *Sophora chrysophylla* |
| wiliwili | *Erythrina sandwicensis* | ʻōhiʻa | *Metrosideros polymorpha* |

plants used for mesic-forest high-elevation (>1,000 m) restoration includes faster-growing varieties (2–6 months grow-out time). A representative list of species for each elevation category is provided in Table 2.

The cost of each native plant seedling varies, depending on whether it is fast or slow growing and the total size of the batch ordered from the nursery. Based on a 2014 contract for goods and services between the State of Hawaiʻi Department of Land and Natural Resources and a local nursery, the average cost for low- and high-elevation restoration is \$2.86 and \$2.60 per tree, respectively, assuming that plants are ordered in sufficiently large batches (>10,000 trees).

Outplanting labor costs were estimated based on restoration projects previously completed in Puʻu Waʻawaʻa. Over the period May 2012 to March 2013, 6,418 trees were planted using 2,496 volunteer hours of labor, which translates to an average planting efficiency of 2.6 trees per hour. Historically, planting densities at three lowland management units (Kīpuka Oweowe, Uhiuhi, and Hauaina) averaged 941 plants per hectare. In contrast, planting density averaged 346 plants per hectare in the mixed-mesic upland Waihou management unit. The large difference in densities is driven by the spatial variation in existing land cover. In lowland areas, rare dry-forest canopy and understory species are planted in close proximity after fountain grass is completely eliminated from restoration plots. At higher elevations, plant spacing is wider because larger fast-growing native species are often incorporated directly into areas currently dominated by kikuyu grass.

In summary, the spatial variation in restoration costs in our model is driven primarily by two factors: (1) the type of nonnative grass being removed and (2) the types of native plants chosen for outplanting. Those factors, in turn, are determined largely by the elevation gradient; low-elevation dry-forest restoration is more expensive because fountain grass is more persistent than kikuyu grass and lower-elevation dry-forest native species tend to be costlier to propagate.

FIRE PRESUPPRESSION. Fire presuppression efforts consist primarily of fire and fuel break establishment and are focused along the perimeter of exclosures and the existing road network. Establishing fire and/or fuel breaks requires mechanical removal (i.e., weed whacking) of vegetation, primarily nonnative grasses, followed by regular herbicide application to prevent regrowth. Restoring areas dominated by nonnative grasses within the exclosures also reduces flammability, but those costs are included as part of forest restoration and therefore will not be discussed here. The costs of fire and fuel break establishment and maintenance are estimated based on past work in Puʻu Waʻawaʻa (E. Parsons, unpubl. data). Once a fire or fuel break is established, herbicide is applied from a spray-tank to prevent overgrowth and maintain a desired buffer. Each tank is composed of 738 liters of water, 15 liters of Roundup PowerMax [Note: The appearance of a product name or brand is not an endorsement of that product or brand.], and 4 liters of Turf Marker (blue dye). The costs of the spray-tank and truck are not included; they are assumed to be part of the base-yard operations. In practice, the frequency of maintenance varies from two herbicide applications during dry years to four applications during wet years. Because the temporal component of our model does not include annual climate-change projections, we did not explicitly account for wet and dry years in the cost calculations. Rather, we assumed that maintenance requires three applications on average.

TOTAL MANAGEMENT COSTS. For fence construction and maintenance, ungulate removal and maintenance, and firebreak maintenance, the present value cost (PVC) was calculated over a 50-yr planning horizon, given a positive discount rate of 2%. We then aggregated PVC for each management category i and divided the result by the total restored area (A):

$$PVC_0 = \sum_i\{\sum_{t=0}^{50} [(1.02)^{-t}(C_{it})]/A\},$$

where $C_{it}$ is the cost incurred for management category i in year t. $PVC_0$ represents the per-hectare cost attributed to management activities that are shared equally across pixels.

In contrast, the costs of weed removal and maintenance and outplanting vary with elevation, which means that their PVCs cannot simply be aggregated and then distributed evenly over the entire study area. Instead, we calculated the PVC for each management category j in elevation group n and divided by the restorable area *within* that elevation group:

$$PVC_n = \sum_j\{\sum_{t=0}^{50} [(1.02)^{-t}(C_{jnt})]/A_n\},$$
$$n = \text{low, high}$$

The total PV per-hectare management costs are $TPVC_{low} = PVC_0 + PVC_{low}$ and $TPVC_{high} = PVC_0 + PVC_{high}$ for low- and high-elevation areas, respectively. This approach condenses costs that vary over elevation gradient and time into PV costs that are directly comparable.

*Cost-Effectiveness*

We define the cost-effectiveness of a pixel as the change in quantity of biophysical outputs (ecosystem services) resulting from restoration in that pixel per year over the 50-yr time horizon per dollar invested in restoration: the larger the benefit gained per dollar, the more cost-effective your investment in that pixel. Using this definition, we generated cost-effectiveness maps using ArcMap raster calculator by dividing biophysical output change rasters by restoration cost rasters, the latter of which were generated by assigning the appropriate TPVC to each pixel based on elevation.

## RESULTS

*Groundwater Recharge*

CLIMATE CHANGE. At the watershed scale, groundwater recharge ranged from 37.5 million cubic meters per year (MCMY) in the unrestored scenario to 38.9 MCMY in the fully restored scenario under the current climate and from 18.8 MCMY in the unrestored scenario to 20.0 MCMY in the fully restored scenario under the RCP 8.5 midcentury scenario. In other words, this climate scenario resulted in an 18.7–18.9 MCMY (49%–50%) decrease in annual groundwater recharge across land-use scenarios at the watershed scale (Figure 2).

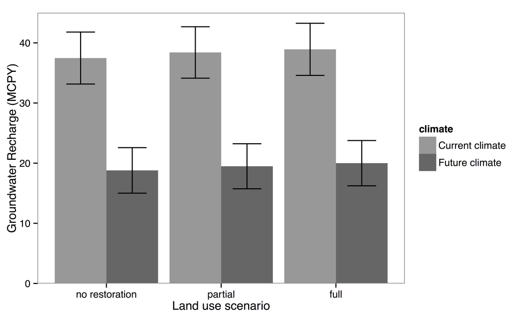


FIGURE 2. Groundwater recharge in Pu'u Wa'awa'a under no restoration, partial restoration, and full restoration in the current midcentury and under RCP 8.5 (future climate) scenarios (full ahupua'a). Error bars = 1 standard deviation of error associated with ET estimates.

LAND USE CHANGE. *Restoration of enclosed areas* (*current land cover to partial restoration*). Under the current climate, restoration of the enclosed areas increased groundwater recharge by 0.92 MCMY (increase of 22% in the exclosure areas; 2% over the watershed) relative to the current land cover. Restoration of enclosed areas under the RCP 8.5 midcentury climate scenario followed a similar trend, with groundwater recharge increasing by 0.23 MCMY (11% in the exclosure areas; 1% over the watershed) (Figure 3, Table 3).

*Full restoration of watershed* (*current land cover to full restoration*). Under the current climate, full restoration significantly increased groundwater recharge by ~1.74 MCMY (12% in the restored area; 5% over the watershed). Groundwater recharge also increased under RCP 8.5 midcentury climate change (0.36 MCMY; 10% in the restored area; 2% over the watershed), but the increase was not meaningfully different (Figure 4, Table 4). The area available for restoration was also reduced under the RCP 8.5 midcentury climate scenario.

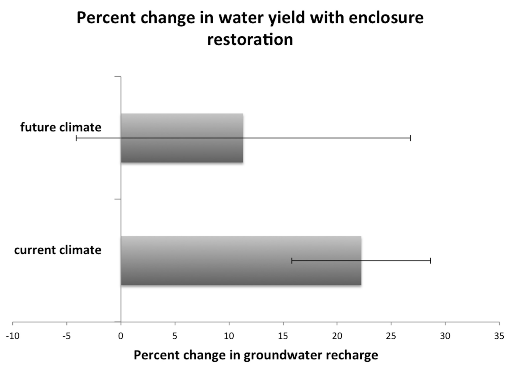


FIGURE 3. Percentage change in groundwater recharge in enclosed areas with restoration (current land cover to partial restoration). Partial restoration increased groundwater recharge [in millions of m$^3$ per year (MCMY)] within each climate scenario in the exclosure areas (though only significantly so in the current climate scenario). Error bars are error (1 SD) of difference between before and after restoration; where they do not overlap with zero, there is confidence in the direction of change.

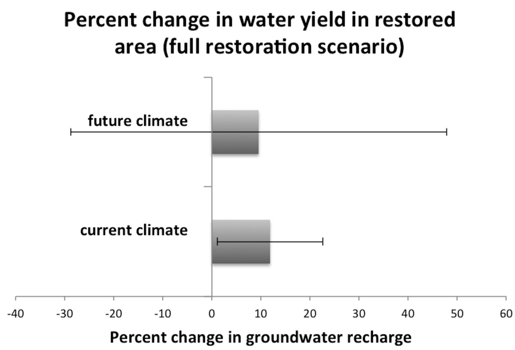


FIGURE 4. Percentage change in groundwater recharge in restored area with full restoration (current land cover to full restoration). Restoration increased groundwater recharge in the full restoration scenario, but only significantly so in the current climate. Error bars are error (1 SD) of difference between before and after restoration; where they do not overlap with zero, there is confidence in the direction of change.

TABLE 3

Evapotranspiration (ET), Rainfall, Fog Interception, and Groundwater Recharge (million cubic meters per year [MCMY]) in Exclosure Areas under Current and Future Climate (RCP 8.5 midcentury)

| Scenario | Evapotranspiration (SD) [MCMY] | Rainfall [MCMY] | Fog Interception [MCMY] | Groundwater Recharge (SD) [MCMY] |
|---|---|---|---|---|
| Current climate | | | | |
| Unrestored | 5.58 (± 0.29)b | 9.71 | 0.00 | 4.13 (± 0.29)b |
| Partial restoration | 6.01 (± 0.22)a | 9.71 | 1.35 | 5.05 (± 0.22)a |
| RCP 8.5 midcentury | | | | |
| Unrestored | 5.23 (± 0.20)b | 7.27 | 0.00 | 2.04 (± 0.20) |
| Partial restoration | 6.03 (± 0.22)a | 7.27 | 1.03 | 2.27 (± 0.22) |

*Note:* Different letters indicate where restored versus unrestored ET and groundwater recharge differed according to uncertainty estimates within each climate scenario (see Appendix). Area of restoration corresponds to partial restoration scenario (1,437 ha current climate; 1,436 ha future climate).

TABLE 4

Evapotranspiration, Rainfall, Fog Interception, and Groundwater Recharge [million cubic meters per year (MCMY)] in Full Restoration Scenario under the Current and Future Climate

| Scenario | Evapotranspiration (SD) [MCMY] | Rainfall [MCMY] | Fog Interception [MCMY] | Groundwater Recharge (SD) [MCMY] |
|---|---|---|---|---|
| Current climate | | | | |
| Unrestored | 24.56 (± 0.88) | 39.13 | 0.03 | 14.61 (± 0.88)b |
| Full restoration | 25.85 (± 1.29) | 39.13 | 3.06 | 16.35 (± 1.29)a |
| RCP 8.5 midcentury | | | | |
| Unrestored | 19.41 (± 1.02) | 23.14 | 0.02 | 3.75 (± 1.02) |
| Full restoration | 20.90 (± 0.94) | 23.14 | 1.87 | 4.11 (± 0.94) |

*Note:* Different letters indicate where restored versus unrestored ET and groundwater recharge differed significantly within each climate scenario. Area of recharge corresponds to full restoration scenario (6,250 ha current climate; 5,294 ha RCP 8.5).

*Landscape Flammability*

Fire occurrence varied greatly year-to-year across the 3,000 $km^2$ NW quadrant of Hawai‘i Island, ranging from 0 to >13,000 ha (4.3% of the study region) burned annually. Despite this annual variability, the top-ranked, global model of landscape flammability indicated distinct patterns in the probability of fire occurrence that were driven most strongly by vegetation type and mean annual rainfall (Akaike weight > 0.99; explained deviance = 25.9%). Among the three dominant vegetation types, probability of fire occurrence was greatest for grasslands, followed by shrublands and forest (Figure 5). The relationship between the probability of fire occurrence and mean annual rainfall (MAR) further illustrated a peak, or climatic “sweet-spot,” in which flammability was reduced at wetter and drier sites, which constrain fuel ignitability and availability, respectively (Murphy et al. 2011). Peak flammability, however, varied among vegetation types (Figure 5). Based on observed landscape-scale fire occurrence, fire probability for grassland peaked at drier conditions (450 mm MAR) when compared with shrubland and forest (650 mm MAR) (Figure 5).

Projecting per-pixel predictions of fire occurrence probability across the Pu‘u Wa‘awa‘a watershed allowed for comparisons of landscape flammability among scenarios in terms of both (1) maximum frequency, or the highest predicted probability of fire occurrence; and (2) relative extent, or the number of pixels falling within the upper ranges of the predicted probabilities, across the watershed.

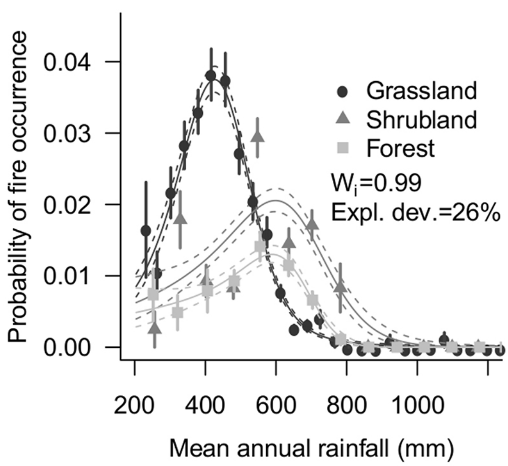


FIGURE 5. The probability of fire occurrence for grasslands, shrublands, and forest. Points indicate actual probabilities calculated as the proportion and standard deviation (error bars) of area burned binned across the rainfall gradient. Solid lines indicate model predictions, and dashed lines indicate the upper and lower 95% confidence intervals.

CLIMATE CHANGE. Climate change increased the median and maximum probability of fire occurrence across all land-use scenarios and full restoration resulted in the greatest reduction in landscape flammability under both current and RCP 8.5 midcentury climate scenarios (Figure 6).

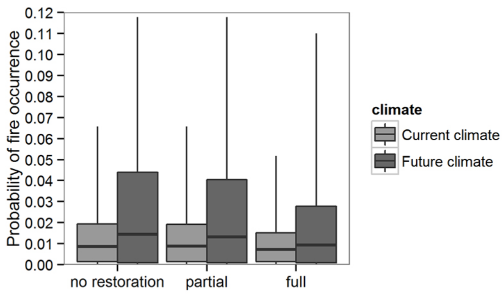


Figure 6. Landscape flammability plotted as the distribution of the per-pixel probability of annual fire occurrence across all pixels in the watershed under different land-use and climate scenarios. The center lines indicate median values, the box indicates lower (25th) and upper (75th) quartiles, and the whiskers indicate minimum and maximum values.

land use change. We projected landscape flammability for both partial and full restoration scenarios for priority ranking and cost assessments (see earlier in this section) but only present the spatial results for full restoration here to illustrate the effects and spatial patterns of reforestation and the RCP 8.5 midcentury climate change scenario on landscape flammability (Figure 7). Under current land cover and climate, flammability was greatest across the midelevation, grassland areas of the watershed (Figure 7*a*). Under full restoration and current climate, landscape flammability was reduced both in terms of extent and maximum frequency (Figure 7*b*). In contrast, current land cover under a warmer and drier future climate greatly increased landscape flammability in terms of frequency (ca. 0.10) and extent. In addition, the area of peak flammability shifted toward the upper-elevation portion of the watershed due to projected declines in annual rainfall that shifted conditions toward peak flammabilities for all vegetation types (i.e., 400–600 mm) (Figure 7*c*). Under future climatic conditions, full restoration constrained flammability in terms of both frequency and extent when compared to current land cover; however, the area of highest flammability still shifted toward upper elevations (Figure 7*d*).

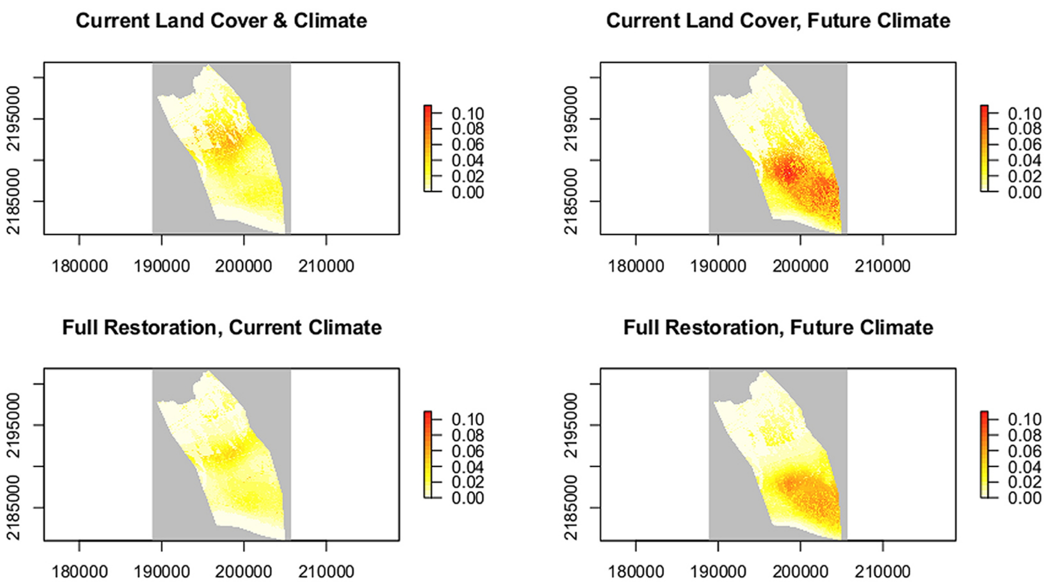


Figure 7. Landscape flammability illustrated as the predicted, per-pixel probability of fire occurrence projected across the Pu‘u Wa‘awa‘a watershed under (*a*) current land cover, current climate; (*b*) current land cover, future climate; (*c*), full restoration, current climate; and (*d*) full restoration, future climate. Future climates consisted of midcentury projections of mean annual rainfall and temperature under the RCP 8.5 scenario.

*Management Costs*

FOREST RESTORATION. *Ungulate removal and maintenance.* Costs to install a fence surrounding the entire restored area totaled \$3.5 million for the partial restoration scenario and \$4.9 million for the full restoration scenario. Annual fence maintenance costs differed by a similar factor, amounting to \$4,664 and \$6,594 for the entire fence perimeter in the partial and full restoration scenario, respectively. Staff effort, materials, and other relevant cost parameters are listed in Table 5.

Ungulate removal costs totaled \$163 per ha. In the present value calculations, we assume that this cost is spread over the first 3 yr of restoration. Thereafter, a maintenance cost of \$1.63 per ha (primarily attributed to monitoring effort) is incurred annually. Staff effort, helicopter time, and other relevant cost parameters are itemized in Table 5.

*Nonnative plant removal and maintenance.* Site preparation costs for lowland dry forest restoration (≤1,000 m) totaled \$6,031 per ha. More than 75% of the cost was attributed to labor, and fixed and variable material costs contributed \$623 and \$682, respectively. Detailed quantities and costs of herbicide, weed removal equipment, and labor are provided in Table 5.

Site preparation costs for upland mixed-mesic forest restoration (>1,000 m) totaled \$196 per ha, or less than 5% of the cost estimated for dry forest restoration. The difference is largely due to the land manager's assumption, based on past restoration work in the region, that only 10% of every high-elevation hectare requires extensive clearing before outplanting. Table 5 includes quantities and costs of herbicide, weed removal equipment, and labor. Site preparation is allocated evenly over the first 20 yr of management (i.e., the entire area is not cleared during the first year).

We estimated an annual maintenance cost per hectare of \$159 for both types of restoration. Labor and material costs are detailed in Table 5.

*Native species outplanting.* High-elevation, mixed-mesic forest restoration outplanting costs totaled \$3,908 per ha. Low-elevation, dry forest restoration outplanting costs were substantially larger, amounting to \$10,366 per ha (Table 5). The difference was driven by two factors: (1) higher planting densities at low elevations (and consequently higher total labor costs), and (2) higher costs associated with obtaining rarer native species for planting. Outplanting is assumed to be evenly spread over the first 20 yr of management (i.e., the entire area is not planted during the first year).

FIRE PRESUPPRESSION. We assumed that firebreaks were established along the fence perimeter in each of the land-use scenarios as part of the installation process. Fire presuppression costs, therefore, only included annual maintenance. Table 5 lists the cost parameters used to calculate the per-meter maintenance cost of \$0.10 per year. Total annual maintenance cost amounted to \$4,511 and \$6,377 for the partial and full restoration scenario, respectively.

PRESENT VALUE MANAGEMENT COSTS. The total present value cost of full restoration ranged from \$53.9 million (future climate) to \$68.0 million (current climate), assuming a 50-yr planning horizon and a discount rate of 2%. The PV cost per hectare was slightly higher under the future climate scenario for both upland and lowland restoration because the fixed costs were spread across fewer restorable pixels (Table 6).

*Cost-Effectiveness*

GROUNDWATER RECHARGE. Full restoration (from current land cover) increased groundwater recharge across both climate scenarios, but the direction of change varied spatially (Figure 8). Several factors explain this spatial distribution. First, the type of restoration matters, as well as the land cover it replaces. In general, forest restoration increased evapotranspiration relative to perennial grassland, but this increase was greater with restoration of Hawai'i subalpine mesic forest (above 1,000 m) than of Hawai'i lowland dry forest (below 1,000 m). However, with fog interception (above 760 m), the net water balance can still be positive, although

TABLE 5

Management Costs for Fence Installation, Ungulate Control, Site Preparation, Weed Control, Outplanting, and Firebreak Maintenance

| Fence Costs | Unit | Unit Cost | Partial Restoration Quantity | Full Restoration Quantity | Partial Restoration Cost | Full Restoration Cost |
|---|---|---|---|---|---|---|
| Fence installation | Meters | $74 | 44,422 | 62,796 | $3,270,340 | $4,623,045 |
| Vehicle gate installation | Gates | $1,865 | 11 | 40 | $74,600 | $20,515 |
| Pedestrian gate installation | Gates | $1,236 | 100 | 142 | $123,600 | $175,512 |
| ATV gate installation | Gates | $1,236 | 33 | 47 | $40,788 | $58,092 |
| Total fence installation | | | | | $3,510,392 | $4,877,448 |
| Fence check labor costs | Meters | $0.0188 | 44,422 | 62,796 | $833 | $1,177 |
| Fence material costs | Meters | $0.0075 | 44,422 | 62,796 | $333 | $471 |
| Total quarterly fence maintenance | | | | | $1,166 | $1,648 |
| Total annual fence maintenance | | | | | $4,664 | $6,594 |

| Ungulate Management Costs | Unit | Unit Cost | Units per Hectare (≤1,000 m) | Units per Hectare (>1,000 m) | Cost per Hectare (≤1,000 m) | Cost per Hectare (>1,000 m) |
|---|---|---|---|---|---|---|
| Initial hunting drive effort | Hours | $22 | 0.74 | 0.74 | $16 | $16 |
| Follow-up dog hunt effort | Hours | $22 | 0.99 | 0.99 | $22 | $22 |
| Aerial hunting effort | Hours | $22 | 1.19 | 1.19 | $26 | $26 |
| Helicopter time | Hours | $1,000 | 0.10 | 0.10 | $100 | $100 |
| Total ungulate removal | | | | | $163 | $163 |
| Staff annual maintenance effort | Hours | $22 | 0.0046 | 0.0046 | $0.10 | $0.10 |
| Helicopter time | Hours | $1,000 | 0.0015 | 0.0015 | $1.53 | $1.53 |
| Annual ungulate maintenance | | | | | $1.63 | $1.63 |

| Site Preparation Costs | Unit | Unit Cost | Units per Hectare (≤1,000 m) | Units per Hectare (>1,000 m) | Cost per Hectare (≤1,000 m) | Cost per Hectare (>1,000 m)* |
|---|---|---|---|---|---|---|
| Clearing (weed whack) effort | Hours | $18 | 99 | 5.2 | $1,782 | $94 |
| 1st herbicide application effort | Hours | $18 | 36 | 3.5 | $648 | $63 |
| Woody invasive shrub-removal effort | Hours | $18 | 74 | 0 | $1,332 | $0 |
| 2nd herbicide application effort | Hours | $18 | 36 | 0 | $648 | $0 |
| Follow-up spot-spray effort | Hours | $18 | 18 | 0 | $324 | $0 |
| Total labor | | | | | $4,726 | $156 |
| FS240 bicycle handle weed whacker[a] | | $690 | 5 | 2 | $3,450 | $1,380 |
| Weed whacker safety helmet[a] | | $75 | 5 | 2 | $373 | $149 |
| Lopper[a] | | $55 | 5 | 0 | $275 | $0 |
| Stihl 41-cm (16-inch) chain saw[a] | | $405 | 5 | 0 | $2,025 | $0 |
| Leather gloves[a] | | $9 | 5 | 2 | $45 | $18 |
| Total fixed material (5-yr material life span) | | | | | $623 | $5 |
| Roundup PowerMax[b] | Ounces | $0.26 | 1,001 | 60 | $260 | $3 |
| Big Foot Blue Dye[c] | Ounces | $0.21 | 373 | 18 | $78 | $1 |
| Element 4[c] | Ounces | $0.36 | 89 | 0 | $32 | $0 |
| 4.5 kg (10 lb) roll weed whacker line[d] | | $130 | 1 | 1 | $130 | $130 |
| Box of Shindaiwa 2-cycle oil[d] | | $82 | 1 | 1 | $82 | $82 |
| Champion carb cleaner[d] | | $15 | 1 | 1 | $15 | $15 |
| Box of disposable chemical gloves[e] | | $80 | 1 | 1 | $80 | $80 |
| Total variable material | | | | | $682 | $35 |
| Total site preparation | | | | | $6,031 | $196 |

(*continued on following page*)

TABLE 5 (continued)

| Weed Maintenance Costs | Unit | Unit Cost | Units per Hectare (≤1,000 m) | Units per Hectare (>1,000 m) | Cost per Hectare (≤1,000 m) | Cost per Hectare (>1,000 m) |
|---|---|---|---|---|---|---|
| Herbicide application (spot-spray) | Hours | $18 | 7.4 | 7.4 | $133 | $133 |
| Roundup PowerMax[b] | Ounces | $0.26 | 73 | 73 | $19 | $19 |
| Big Foot Blue Dye[c] | Hours | $0.21 | 27 | 27 | $6 | $6 |
| Total annual weed maintenance | | | | | $159 | $159 |

| Outplanting Costs | Unit | Unit Cost | Units per Hectare (≤1,000 m) | Units per Hectare (>1,000 m) | Cost per Hectare (≤1,000 m) | Cost per Hectare (>1,000 m) |
|---|---|---|---|---|---|---|
| Planting effort | Hours | $22 | 362 | 133 | $7,919 | $2,917 |
| Lowland plant material cost | Plants | $2.60 | 941 | 0 | $2,447 | $0 |
| Upland plant material cost | Plants | $2.86 | 0 | 346 | $0 | $991 |
| Total outplanting | | | | | $10,366 | $3,908 |

| Firebreak Maintenance Costs | Unit | Unit Cost | Partial Restoration Quantity | Full Restoration Quantity | Partial Restoration Cost | Full Restoration Cost |
|---|---|---|---|---|---|---|
| Herbicide application effort | Hours | $18 | 142 | 201 | $2,592 | $3,664 |
| Roundup PowerMax | Liters | $8.90 | 178 | 261 | $1,594 | $2,253 |
| Big Foot Blue Dye | Liters | $7.26 | 45 | 64 | $325 | $460 |
| Total annual firebreak maintenance | | | | | $4,511 | $6,377 |

[a] Source: Farm and Garden Kona.
[b] Source: Crop Production Services.
[c] Source: BEI Hawai‘i.
[d] Source: Farm and Garden Kona.
[e] Source: Airgas Kona.
* Assumes that only 10% of each upland hectare requires preparation.

TABLE 6

Present Value (PV) Costs for Full Restoration under Current and Future Climate (RCP 8.5 Midcentury)

| Climate | Total PV Cost | PV Cost/ha Lowland | PV Cost/ha Upland |
|---|---|---|---|
| Current climate | $68.0 million | 13,161 | 5,501 |
| Future climate (RCP 8.5 midcentury) | $53.9 million | 13,415 | 5,754 |

we acknowledge that we were not able to fully model changes in fog under RCP 8.5 midcentury. Accordingly, the greatest benefits were seen in the midelevation belt where increased fog interception resulting from restoration of native dry forest increased overall groundwater recharge (dark blue pixels in Figure 8). Under the current climate, high returns were also seen in high-elevation areas, whereas the response is variable under the RCP 8.5 midcentury climate scenario. Under RCP 8.5 midcentury, there are areas where restoration of native lowland dry forest actually increased groundwater recharge through a reduction of ET in comparison with perennial grassland.

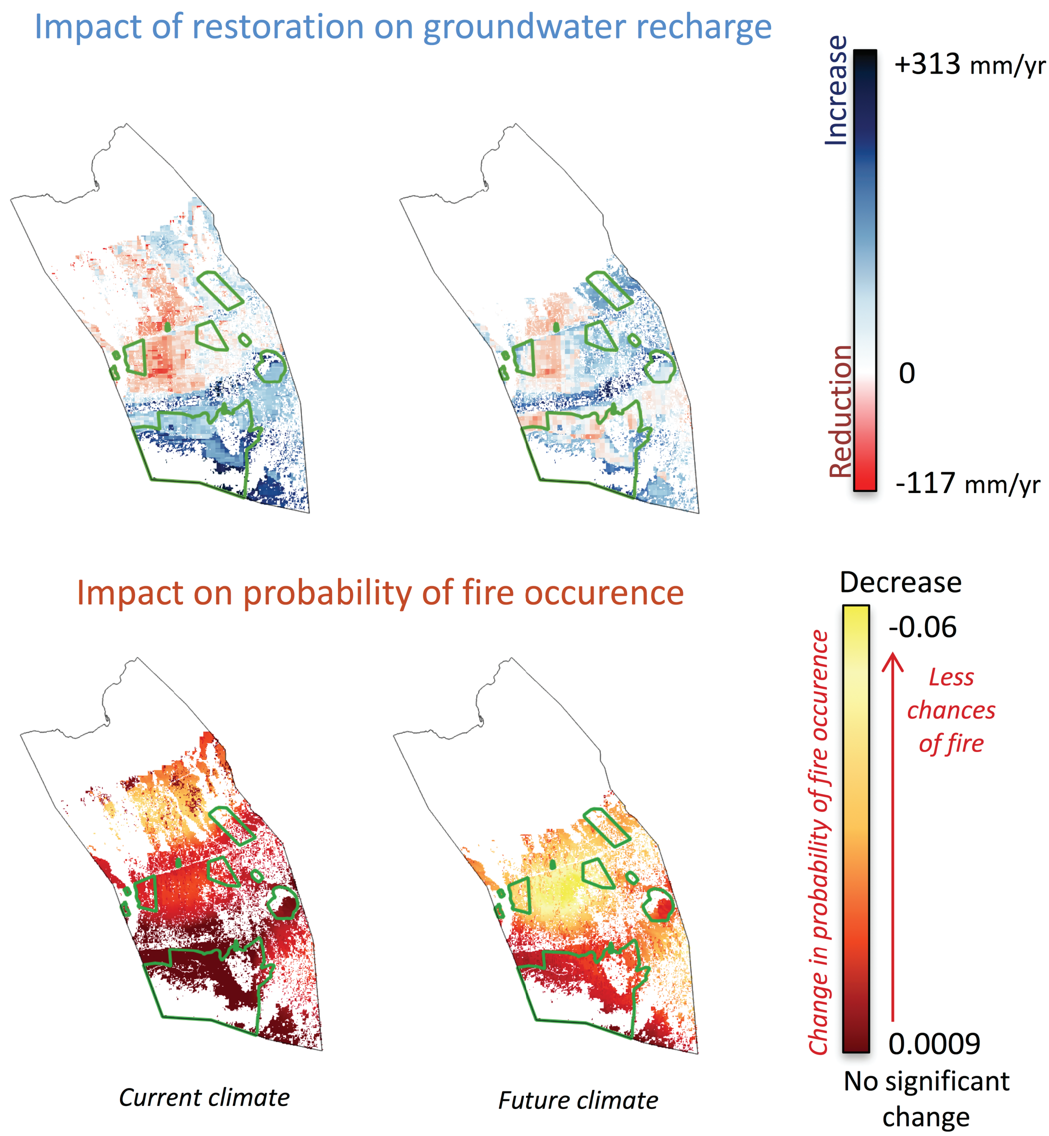


Figure 8. Impact of restoration on groundwater recharge and landscape flammability in Pu'u Wa'awa'a. The green contours outline the exclosure areas corresponding to the partial restoration scenario in which only these regions undergo some changes. In the top half of the figure, blue corresponds to an increase in groundwater recharge due to restoration, and red corresponds to a decrease. In the bottom half of the figure, yellow corresponds to a decrease in landscape flammability due to restoration, and red means no significant change.

Cost-effectiveness of restoration for recharge is largely driven by the magnitude of biophysical outputs because high-output areas largely overlap low-cost areas. In other words, there are few opportunities to improve cost-effectiveness of investment by switching to lower-cost/lower-output areas and/or higher-output/higher-cost areas. Under the current climate, recharge ranges from a low of –0.1 $m^3$/pixel/dollar/year to a high of 0.3 $m^3$/pixel/dollar/year (Figure 9), and the most cost-effective areas are found at higher

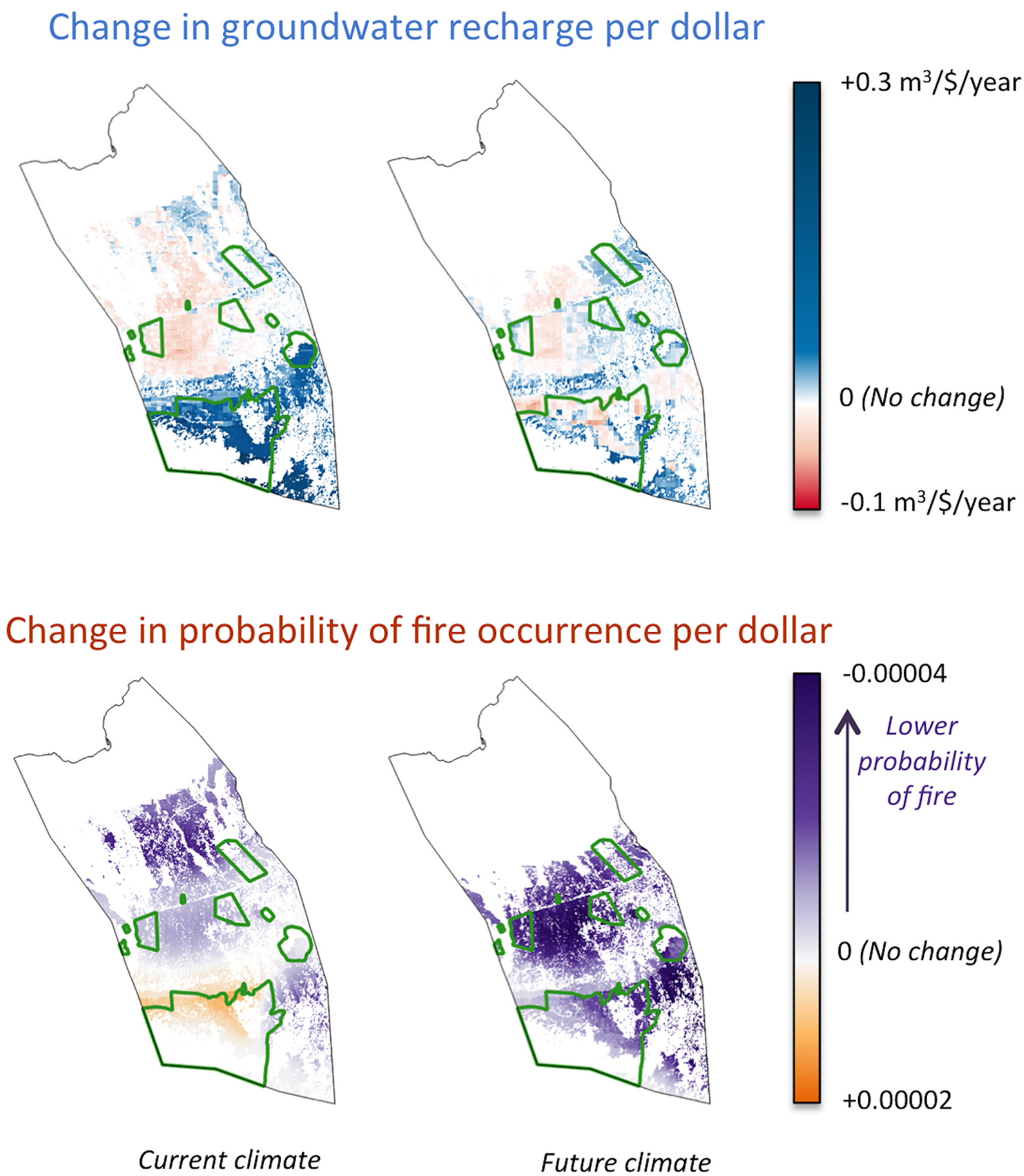


FIGURE 9. Cost-effectiveness of forest restoration in Puʻu Waʻawaʻa in terms of recharge and probability of fire occurrence. The enclosure areas are outlined in green.

elevations. Under RCP 8.5 midcentury, cost-effectiveness values decline across most of the landscape, and high-elevation areas remain relatively cost-effective.

LANDSCAPE FLAMMABILITY REDUCTION. Under current climatic conditions, full restoration primarily reduces flammability at midelevations with little to no change at the upper-elevation areas of the watershed (Figure 8). This indicates that increasing forest cover would substantially reduce flammability across an area that is currently dominated by grasslands (the most fire-prone vegetation type in the watershed) and falls within the peak range of flammability for this vegetation type in terms of rainfall (Figure 5). Under RCP 8.5 midcentury conditions, temperature increases, but more important is that decreased rainfall across the mid- to upper-elevation portions of the watershed increases peak flammability and shifts the most flammable areas to higher elevations (Figure 8). Therefore, the net decrease in flammability due to restoration is larger in terms of both reduced frequency and extent for future climates, and these benefits are realized at higher elevations in the watershed. Future climates will also constrain flammability in the lower extent of the watershed due to lack of rainfall, which will limit fuel production (e.g., Murphy et al. 2011).

The change in probability of fire occurrence under the current climate ranges from –0.00004/pixel/USD/year to 0.00002/pixel/USD/year. Although we acknowledge that incremental changes in probability resulting from marginal increases in expenditures are difficult to interpret (flammability shifts at the watershed scale are more meaningful), we generated cost-effectiveness maps for fire occurrence primarily to aid in illustrating trade-offs between multiple ecosystem services. The most cost-effective areas largely correlate with high-benefit areas at lower elevations (Figure 9). Although restoration costs are high in these areas, the benefits are large enough to offset those costs. There is also a smaller region in the upper watershed that is fairly cost-effective, wherein moderate benefits can be gained at relatively low cost. Under the RCP 8.5 midcentury climate scenario, investments in higher elevations become more cost-effective. This shift is driven mostly by the shift in flammable area (Figure 8).

UNCERTAINTY REGARDING COST ESTIMATES. Given that the effect of restoration on ecosystem service provision is relatively small in percentage terms, uncertainty regarding cost estimates may have a nontrivial effect on the relative desirability of restoration areas. Some lower-elevation areas may be relatively more cost-effective than our calculations suggest if actual costs are lower. This might be true, for example, if site preparation costs are strongly related to grass density and some pixels are substantially less dense than others. In that case, low-elevation areas would remain preferable for flammability reduction but may also become slightly more desirable for recharge. On the other hand, some higher-elevation areas may be relatively less cost-effective than our calculations suggest if actual costs are higher. The assumption that only 10% of higher-elevation areas need to be cleared before outplanting is based on past work in the region, but the percentage is likely to vary across pixels. Restoration of a given pixel is less cost-effective if more labor-intensive clearing and site preparation is required. In that case, high-elevation areas may become less desirable for recharge and would remain undesirable for flammability reduction. Generally, cost-effectiveness of a particular pixel depends on both the benefit in terms of recharge or flammability change and the restoration cost. The extent to which either is uncertain determines how cost-effectiveness varies across the landscape.

## DISCUSSION AND MANAGEMENT IMPLICATIONS

Effective forest restoration requires close consideration of management objectives as well as potential future environmental and fiscal conditions (Bullock et al. 2011). In this analysis, we work closely with a state reserve manager to consider these factors simultaneously by assessing the benefits in terms of management objectives (groundwater recharge and landscape flammability) and management costs of various types and locations of forest restoration. Several lessons from the Pu‘u Wa‘awa‘a management area inform restoration priorities in Hawai‘i and beyond. We show that the location and type of land-use change associated with restoration can influence the net benefits gained from restoration activities. The management choice for location and size of restoration area will vary depending on restoration objective, such as native species conservation, groundwater recharge, and/or fire reduction. Preferred restoration site and size will also vary given single (groundwater recharge only) versus multiobjective (water enhancement and fire reduction) restoration goals.

Our results show minimal overlap of high-benefit areas for groundwater recharge and landscape flammability objectives under the current climate, demonstrating that prioritizing based on single ecosystem services can result in important trade-offs in other objectives (Bullock et al. 2011). Focusing exclusively on groundwater recharge generates results that favor restoration of higher-elevation areas where fog interception increases precipitation enough to increase overall groundwater recharge despite some small increases in evapotranspiration. High-elevation areas also correspond to lower restoration costs, explaining their relatively high cost-effectiveness. On the other hand, even in areas low enough to require expensive removal of fountain grass and restoration to dry forest (<1,000 m) but high enough for fog interception (~>750 m), groundwater recharge benefits can be large and restoration moderately cost-effective. That restoration of forest can provide moderate increases in water availability in some locations is similar to the findings of a field hydrological study carried out in Kona, Hawaiʻi (Brauman et al. 2012), but contrasts with global trends of decreasing water yield (associated with greater evapotranspiration) with forestation (Bosch and Hewlett 1982, Farley et al. 2005). This suggests that tropical dry forest restoration results in very different hydrological outcomes than global trends based primarily on temperate ecosystems.

If focusing on cost-effectiveness of flammability reduction, however, the priority areas are almost completely flipped, apart from some overlap in the upper watershed. Under current climate conditions, lowland restoration yields the highest flammability benefits, because the grasslands that dominate this area constitute the most flammable portion of the watershed. Although low-elevation areas are relatively expensive to restore, they are most cost-effective for flammability reduction because the benefits are much larger than in the upper watershed.

Our results also point to the importance of considering the interaction of land use and climate change in prioritizing for ecosystem services (Runting et al. 2016). In the RCP 8.5 midcentury climate scenario, high-elevation areas continue to generate the highest recharge benefits and remain relatively cost-effective, although overall recharge benefits of native forest restoration decline across the landscape. In terms of flammability reduction, investments at mid- and high elevations in the watershed become more cost-effective as the climate becomes drier. Given this, opportunities for simultaneously reducing fire risk and increasing groundwater recharge are greater under RCP 8.5 midcentury than under the current climate.

If working with a limited budget under the current climate, managers may be most effective by focusing effort on either a low- or high-elevation area exclusively to capture the largest benefits for one of the biophysical outcomes. Relaxing the budget constraint beyond a certain point, however, would create the opportunity to capture additional benefits with a second targeted management unit, leaving the least cost-effective midelevation area unrestored. Under RCP 8.5 midcentury, high-elevation areas remain the most cost-effective for recharge but also become cost-effective for flammability reduction. Thus, management efforts, even with a limited budget, can capture cost-effective areas for both recharge and flammability by focusing on higher elevations. Given the expected shift in benefits over time under the assumption of climate change, a dynamic management strategy might start with exclosures in high-elevation, low-cost areas and expand downward over time as the cost-effectiveness of management for landscape flammability shifts upward with the drying climate.

Although we do not measure biodiversity benefits in this study, biodiversity (particularly endangered species) conservation is a critical management objective of the land manager. The current and proposed exclosures in the partial-restoration scenario were chosen to protect existing endangered species and/or as optimal sites for restoration of native plants based on this objective. It is interesting that these exclosures also encompass much of the most cost-effective areas for reducing landscape flammability (under current and future climate scenarios) but little of the high-priority areas for maximizing ground water recharge (Figure 8). That areas critical for biodiversity conservation provide certain cobenefits demonstrates that including ecosystem service objectives can potentially "widen the net" of conservation (Reyers et al. 2012) through increasing funding tied to outcomes such as reduced landscape flammability. However, our results also indicate that prioritizing for ecosystem services will not necessarily result in optimal protection of areas most important for biodiversity. As the number of biophysical objectives increases, so does the challenge of minimizing trade-offs and costs given a limited budget.

In Pu'u Wa'awa'a, landscape flammability, ground water recharge, and endangered species conservation represent critical and growing concerns. Our methodology can be applied to other areas where these or other ecosystem services are important and where resource managers are interested in cost-effective management for multiple benefits.

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## Appendix

### Uncertainty Associated with Groundwater Recharge Estimates

To estimate the error in modeled evapotranspiration (ET) between the full model (Giambelluca et al. 2014) and the simplified regression model, we calculated the mean absolute and mean percentage error (along with standard deviations) between the full model and the regression model for each land cover class (at a 250 m pixel scale) within the watershed of Pu‘u Wa‘awa‘a. We used the following equations:

1. Mean absolute error for given land class
   $$= \frac{\sum_{i \in K}(ET1 - ET2)}{n}$$
   where $i$ = a pixel in land class $k$; $n$ = number of pixels in land class $k$; $ET1$ = evaporation as calculated by regression model; and $ET2$ = evapotranspiration as calculated by full model.

2. Mean percentage error for given land class
   $$= \frac{\sum_{i \in K}(ET1 - ET2)}{n}$$
   where $i$ = a pixel in land class $k$; $n$ = number of pixels in land class $k$; $ET1$ = evaporation as calculated by regression model; and $ET2$ = evapotranspiration as calculated by full model.

To characterize the uncertainty of the modeled scenarios, we adjusted the ET estimates calculated using the regression equations as follows:

3. Adjusted ET
$= ET1/(1 - F1)$
where $ET1$ = evapotranspiration as calculated by regression model and $F1$ = fraction underestimate of regression model compared to full model (per equation 2).

We reported error estimates as 1 standard deviation around the adjusted ET in terms of percentage difference between the regression model and the full model.

To calculate the difference (or change) between scenarios we used the following equations:

4. Mean difference between scenarios
$= ET1adj - ET2adj$
where $ET1$adj = bias-adjusted ET scenario 1 and $ET2$ adj = bias-adjusted ET scenario 2.

5. SD of difference
$= \sqrt{(SDET1adj^2 + SDET2adj^2)}$
where $SDET1adj$ = standard deviation of bias-adjusted ET scenario 1 and $SDET2adj$ = standard deviation of bias-adjusted ET scenario 2.

To translate this into percentage change, we used the following equations:

6. Percentage change between scenarios
$$= \frac{(ET1adj - ET2adj)}{ET1} * 100$$

7. SD percentage change
$$= \frac{SD(ET1adj - ET2adj)}{ET1} * 100$$

We considered scenarios meaningfully or significantly different where the difference in ET between scenarios was greater than the SD of the change.