Forest Ecology and Management 400 (2017) 300–320

Contents lists available at ScienceDirect

Forest Ecology and Management

journal homepage: www.elsevier.com/locate/foreco

A watershed decision support tool for managing invasive species on Hawai'i Island, USA

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ARTICLE INFO

Article history:
Received 3 February 2017
Received in revised form 21 May 2017
Accepted 25 May 2017

Keywords:
Decision support
Hydrology modeling
Invasive species
Climate change
Ecosystem management
Climate change, Hawai'i

ABSTRACT

Non-native species invasions, growing human populations, and climate change are central ecological concerns in tropical island communities. The combination of these threats have led to losses of native biota, altered hydrological and ecosystem processes, and reduced ecosystem services. These threats pose complex problems to often underfunded management entities. We developed a watershed decision support tool (WDST) for the windward coast of Hawai'i Island aimed at prioritizing catchments for invasive species removal and native forest protection from non-native species invasions. Using the Ecosystem Management Decision Support (EMDS) system, we integrated spatial data from four sources: (*i*) native and invasive species coverage; (*ii*) modeled water yield; (*iii*) treatment cost and efficacy; and (*iv*) native species conservation value. We used a distributed hydrology model (DHSVM) to estimate catchment-level (~90 ha) water yield under six climate and non-native species invasion scenarios to identify where (1) invasive species removal and (2) protection from invasion would have the greatest benefit to increasing or maintaining native biodiversity and hydrologic functioning. The hydrology model predicted a 30% decline (386 Gl yr^{-1}) in total water yield under a drier future climate (20% reduction in rainfall), with an additional 2% reduction when catchments were fully invaded by non-native species. Increased temperatures had a small compensatory effect on water yield. The WDST identified 6.3% of the study area as high priority for invasive species removal, based on characteristics of large hydrological response to the removal treatment (concentrated in high rainfall areas), high quality road or trail access, and high conservation value. High protection priority from invasive species (5.9% by area) occurred in higher elevation catchments, near the upper range of strawberry guava (the main invasive species), where water yield was most sensitive to invasion. Climate change scenarios had little influence on the spatial distribution of priority scores despite large changes in overall water yield. In contrast, priority scores were sensitive to very high variation in treatment costs, which were influenced largely by travel times to catchments via road and trail networks. This last finding suggests that future management feasibility will hinge on improvements to road and trail networks, or development of alternative management strategies that reduce travel costs and time.

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1. Introduction

Ecological service-based decision support systems have been developed to facilitate ecosystem management across a wide variety of natural resource applications (e.g., Rauscher, 1999; Reynolds and Hessburg, 2005; Daily et al., 2009; Staus et al., 2010; Reynolds et al., 2012, 2014; Bremer et al., 2015). Such tools permit fully integrated analysis of multiple spatial datasets, and facilitate analytical processes that enhance one or more socio-ecological benefits, while simultaneously incorporating logistical and operational considerations (Reynolds et al., 2014). Decision support tools can provide: (1) a formalized process for engaging, negotiating, and mediating multi-objective decision-making; (2) a quantitative framework for selecting among management alternatives in a

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http://dx.doi.org/10.1016/j.foreco.2017.05.046
0378-1127/Published by Elsevier B.V.

spatial planning environment; and (3) an integrated strategy for justifying implementing specific management decisions (Reynolds et al., 2014).

While common in temperate regions, few documented applications of landscape-level decision support systems exist in tropical ecosystems (Westmacott, 2001; Bremer et al., 2015). However, the value of decision support in the tropics may be high because these tools assist managers in allocating limited resources to maximize treatment benefits. The advantages of such tools will increase in value as global changes progress and population-driven demands on watersheds increase.

A convergence of anthropogenic, climatic, and ecological stressors has created much uncertainty for tropical island forests. Particularly, their ability to sustain adequate freshwater to support local human populations, an expanding agricultural footprint, and other essential services (e.g., biodiversity protection, flood abatement, erosion control, carbon sequestration, recreation and tourism opportunities) into the future is unclear (Burns, 2002; Thaman, 2002; Parry, 2007; Duffy, 2011; White and Falkland, 2012). Successful mitigation of these impacts requires a strategic landscape-level approach in which managers and stakeholders effectively communicate and collaborate across ownerships, develop a shared understanding of cultural values and needs, and concentrate efforts to maximize management benefits. While access to decision support technology is currently limited, such approaches to managing landscapes are particularly relevant, and perhaps critical to watershed-based collaborations that seek to: (*i*) characterize land management needs; (*ii*) increase local resilience to climate and land use/land cover change; and (*iii*) efficiently use limited state and federal funding for management (White and Falkland, 2012). In the context of tropical landscapes, effective decision making must also emphasize cost-effectiveness, be relatively insensitive to limited data, and incorporate the diverse needs of complex ownerships that may include only a small proportion of shared public lands.

We piloted our tropical decision support tool effort in watersheds of windward Hawai'i Island. The objectives of the tool were to: *(i)* provide local land management cooperatives with a quantitative method to identify key areas for invasive species management; and *(ii)* demonstrate an application of state-of-the science ecological processing modeling within a decision support framework for a tropical island ecosystem. Together, our goal was to highlight how decision support applications can be tailored to specific management concerns and locally relevant ecosystem services.

We conducted our work across a highly constrained, hydrologic study system that encompasses a steep ridge-to-reef elevational gradient and varies widely in mean annual rainfall, temperature, and ownership patterns (Strauch et al., 2014, 2016a, 2016b, 2016c). This study area has undergone both significant agricultural land-use changes at low elevations as well as invasion by non-native plants in mid-elevation forests (~600–1200 m), with anticipated impacts to watershed hydrology. Hawai'i's climate and impacts to forests are also anticipated to change in the coming decades (Giardina, 2012; Timm et al., 2015). In Hawai'i, mid-elevation forests have already experienced a 0.163 °C decade^{-1} increase in surface temperature from 1975 to 2006, exceeding the global average (Giambelluca et al., 2008). Observed reductions in wet-season precipitation (October – March) and increased surface temperature have led to downward trends in stream base flow levels (Oki and County, 2004). Furthermore, climate projections point to continued warming, and for leeward areas of Hawai'i, a pattern of reduced rainfall, and altered precipitation distribution (Timm and Diaz, 2009; Timm et al., 2015). Yet, demand on water supply will likely increase as human population growth increases the number of water-users in the region. These factors, along with limited and aging water production systems, increased levels of freshwater contamination by saltwater, and high natural variability in the water supply due to ENSO events (Chu and Chen, 2005) leave many island nations facing a critical need for improved freshwater security.

As with much of the tropics, forests across Hawai'i have undergone significant change due to the combined influences of urban encroachment, intensive agriculture and forestry practices (Cuddihy and Stone, 1990), invasion by non-native plants and animals (Smith, 1985; Giardina, 2012), and climate change (Keener et al., 2012). Since European settlement in the mid-19th century, native forest area declined by nearly half (DLNR, 2011), and the introduction of non-native ungulates (e.g., feral pigs *Sus scrofa*), and invasive plant species (e.g., *Psidium cattleianum* or strawberry guava – the focus of the current study), have altered the structure, functioning, and connectivity of remnant forested patches (Nogueira-Filho et al., 2009; Strauch et al., 2016a, 2016b) . Forested watersheds in Hawai'i are particularly vulnerable to non-native species invasions due to the influence of the islands' remoteness of native flora and fauna, volcanic geology, complex topography, and steep climatic gradients (Loope and Mueller-Dombois, 1989; Loope, 1992; Gagne and Cuddihy, 1999; Leigh et al., 2007). Non-native invasive plants typically exhibit higher evapotranspiration rates compared to the native species they replace (Funk and Vitousek, 2007; Cavaleri and Sack, 2010), as well as lower canopy water storage capacity and cloud water capture (Takahashi et al., 2011), which leads to reduced freshwater retention in invaded ecosystems. For example, Giambelluca et al. (2009) found 27–53% higher evapotranspiration rates in a strawberry guava infested stand compared to a stand dominated by native 'ōhi'a (*Metrosideros polymorpha*) forest, with larger differences occurring during especially warm and dry periods (Giambelluca, 2009). As such, major foci of restoration activities in Hawai'i are the maintenance of native species assemblages, and the reduction in extent and level of invasion by non-native species (Conry and Cannarella, 2010).

In the current assessment, we first used a distributed hydrology model to quantify catchment-level water yield under various non-native species invasion and climate change scenarios, using standard modeling methods (Wigmosta et al., 1994). We then used the Ecosystem Management Decision Support (EMDS) software to build a watershed decision support tool (WDST) that prioritized treatments across the North Hilo-Hamakua districts of Hawai'i Island (Fig. 1). We chose this area because it: *(i)* includes a highly constrained gradient in mean annual rainfall (MAR), spanning 2000–6000 mm, where the elevational distribution of overstory species, soils, land use and non-native species invasion patterns is relatively constant across MAR; *(ii)* encompasses diverse ownerships aligned in the goal of managing for fresh water and controlling invasive species; *(iii)* is managed by a watershed management partnership (Mauna Kea Watershed Alliance) through which the WDST could be developed and implemented; and *(iv)* was identified by the *Rain Follows the Forest Initiative,* now the *Healthy Forests Initiative* (HFI), as a Priority I management area due to high rainfall, native wet forest extent, and concern about non-native species invasions (DLNR, 2011).

We used the following workflow to develop the WDST (Fig. 2):

Stakeholder input and data gathering: We collected stakeholder input on potential decision criteria to be used in the Decision Model, and gathered representative GIS layers. The intended users of the WDST were the diverse collection of land ownership partners in the area including federal, state, county, and private entities. Management priorities for each partner varied widely, which necessitated a flexible decision tool to incorporate the level of complexity in the overlapping and independent goals of the local land managers. The data for our WDST included information on

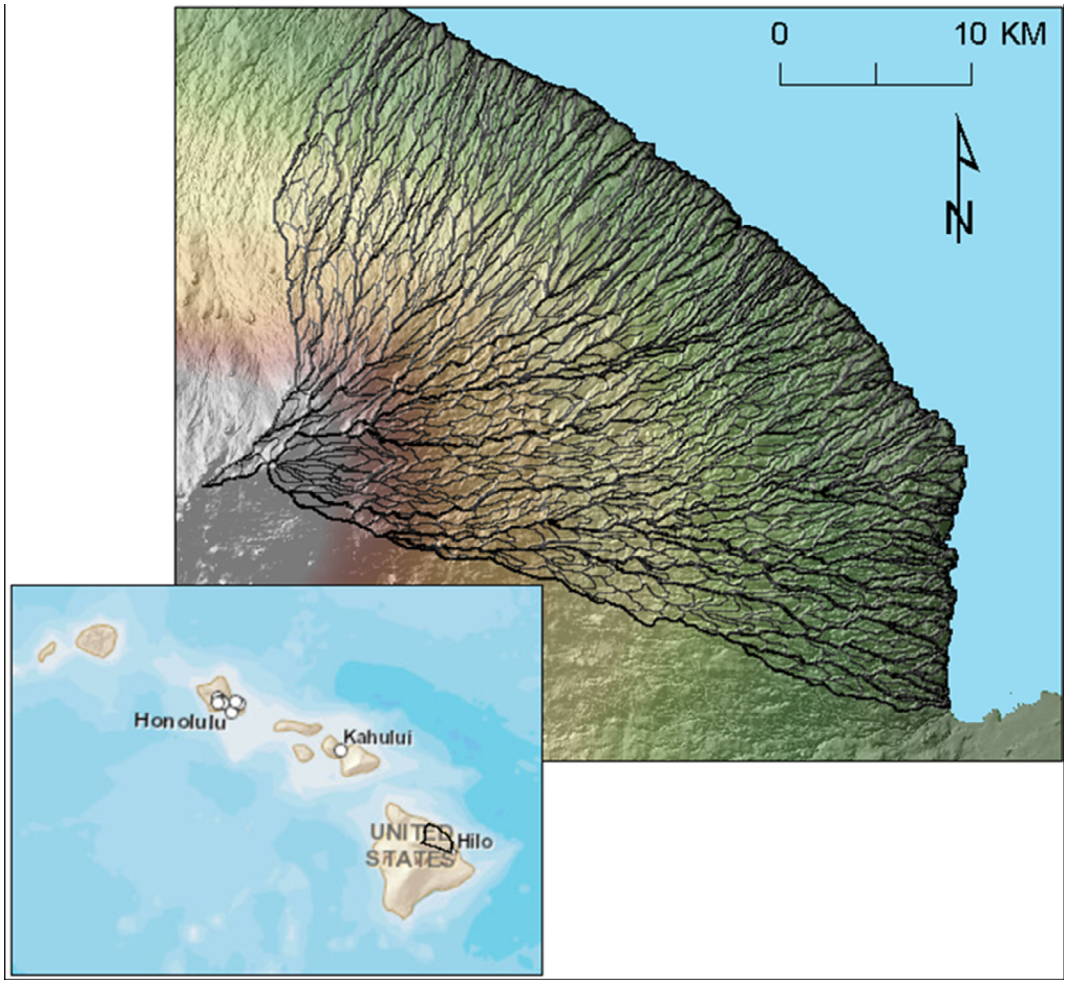

Fig. 1. North Hilo-Hamakua study area located on the windward side of the Island of Hawai'i. Dark lines represent watershed boundaries, and lighter lines represent boundaries of 904 catchments – the minimum management-relevant mapping unit used here.

treatment effort, travel time to treatment catchments, stream water quality, and land ownership/management objectives.

Hydrology modeling: Building on whole watershed modeling analyses for the study area (Strauch et al., 2016b), we used the Distributed Hydrology-Soil-Vegetation Model (DHSVM) (Wigmosta et al., 1994) to model sub-watershed, catchment-level water, energy, and mass balance across the North Hilo-Hamakua study area and to directly link the effects of climate change scenarios and non-native species invasion on freshwater supply across the study landscape.

Vegetation mapping: We developed a spatially explicit vegetation map for the study area under three non-native species invasion scenarios: *(i)* current invasion level, *(ii)* full invasion, and *(iii)* no invasion. We chose strawberry guava (SG) as our main non-native species of interest due to its low water-use efficiency (Giambelluca, 2009), high degree of invasiveness (Huenneke and Vitousek, 1990; Vorsino et al., 2014), wide distribution across the islands, and its relation with feral pigs as a vector of spread (Nogueira-Filho et al., 2009).

Cost analysis: We estimated SG treatment costs for: (1) SG removal; and (2) ungulate fencing treatments based on several factors, including travel time to catchments, size of catchment, and level of SG invasion;

Logic model development: We developed a logic model to assess the state of the system under each climate scenario, and map priority scores for fencing and SG treatments. Our logic model consisted of two main components: *(i)* percent SG aerial coverage (representative of the negative biological influence on forest structure and function), and *(ii)* relative hydrologic response to SG treatments

Decision outputs based on variable weighting schemes: Finally, we calculated priority scores under different decision model weighting schemes to determine how prioritizations change when different decision criteria were emphasized.

Through the application of this modeling framework in Hawai'i, we sought to achieve the following outcomes: *(i)* use a well-established distributed hydrology model that has been parameterized, including calibration and validation, for our study area (Strauch et al., 2016b) to quantify regional-level water yield patterns at catchment (30×30 m) resolution under multiple climate change and SG invasion scenarios; *(ii)* aggregate results into approximately 100 ha management scale catchments; *(iii)* prioritize catchments within the study region for management based on the level of SG invasion and relative hydrologic response to SG management; *(iv)* develop an interactive WSDT for managers to prioritize catchments for SG management activities within the study area; and *(v)* demonstrate the application of EMDS to tropical forested watersheds that in turn can be applied to other tropical ecosystems.

2. Methods

2.1. Study area description

We selected an 817 km^2 area of the North Hilo and Hamakua districts located on windward Hawai'i Island (Fig. 1). This study

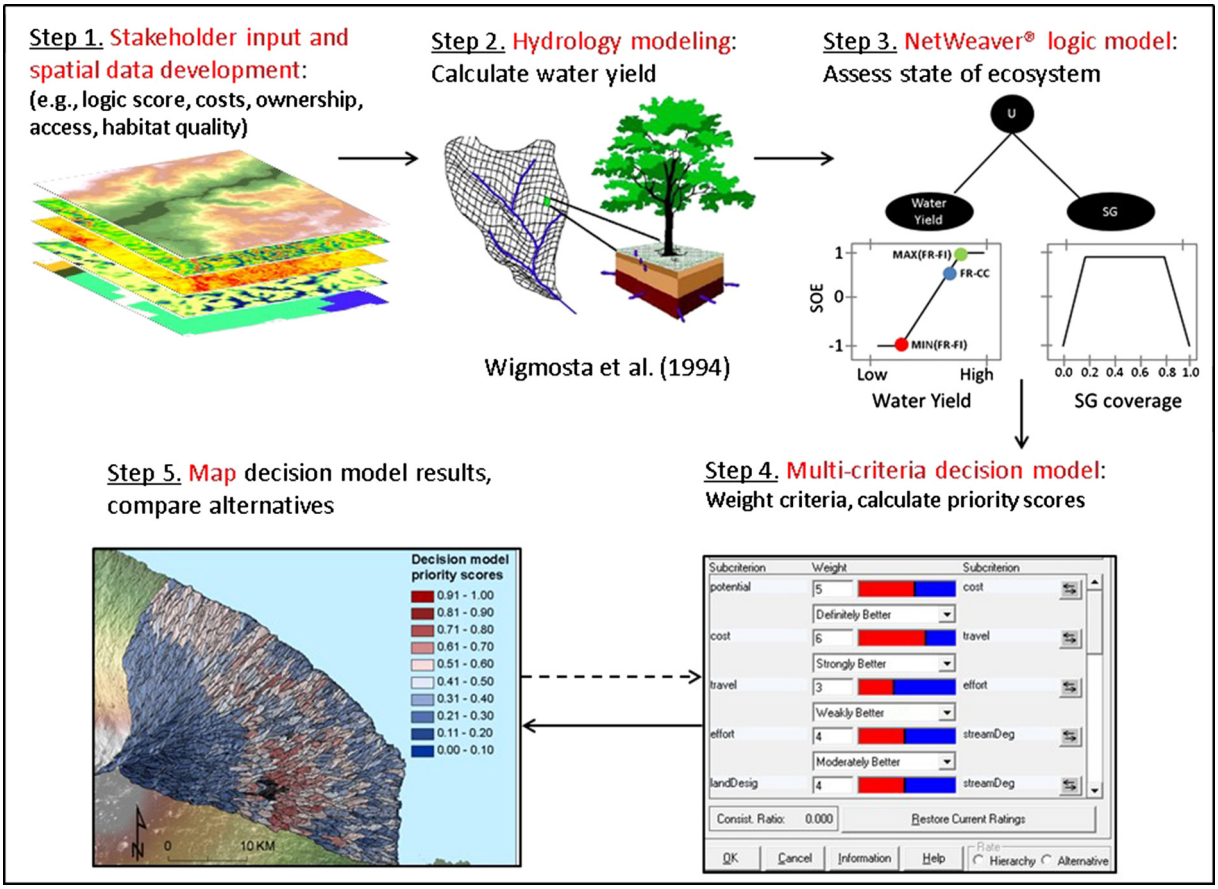

Fig. 2. Workflow of analyses used to quantify catchment-level water yield for the North Hilo-Hamakua, HI study area, which, along with a variety of other spatial datasets, was used to feed into the EMDS decision support software for landscape prioritization. SOE is "strength-of-evidence" score from the logic model, SG is strawberry guava.

area is dominated by spatially compact, narrow watersheds of volcanic origin (<100,000 years old). Elevations range from sea level to ~4200 m at the peak of Mauna Kea in the western portion of the study area. A steep east-west gradient in mean annual rainfall (MAR) of nearly 4000 mm exists with minimal variability in plant diversity, soil type, and geology (Strauch et al., 2015). Annual precipitation is influenced by prevailing southwest trending trade winds and steep elevation gradients and is highest in the southeast (max: 7629-mm) and lowest at high elevations (min: 204-mm). Average annual temperatures range from 3.6 °C at upper elevations to 25 °C at sea level.

Land-use patterns are characterized by *(i)* developed residential and agricultural activities in warm, wet low elevation settings below ~500 m ASL, *(ii)* forest reserves and forested watersheds in the cooler, typically wetter mid--elevations where watershed protection was initiated early in Hawai'i's history (Cuddihy and Stone, 1990), *(iii)* and working or abandoned, previously forested rangelands in cooler and much drier high elevation areas, which were suitable for domestic livestock production.

Large extents of native forest still remain in the mid-elevation areas of the project area, forming a belt between ~500 and 1800 m ASL, but these forests vary in the degree of non-native plant invasion. Native forests were historically dominated in the canopy by 'ōhi'a lehua (*Metrosideros polymorpha*) and koa (*Acacia koa*), with uluhe fern (*Dicranopteris linearis*), Hāpu'u 'i'i (*Cibotium menziesii*), and other native mid and understory species (Giardina et al., 2014; Ostertag et al., 2014). Non-native species dominate at lower elevations, and include African tulip (*Spathodea campanulata*), albizia (*Falcataria moluccana*), and SG. Occurrence of SG is pronounced between ~500 and ~1000 m ASL, and progressively declines as elevation increases. This elevation sequence of agriculture, invaded forest, native dominated forest, and previously forested rangeland is common across much of the MAR gradient (Strauch et al., 2014).

Access to forested areas is limited to informal trails near the middle of the study area, whereas drier forested areas, agricultural lands and upper elevation rangelands are more easily accessed via unimproved roads and established trails.

This study area provides a unique opportunity to apply the EMDS to a model "ridge-to-reef" hydrological study system that incorporates: *(i)* steep, complete temperature, rainfall, and land-use gradients; *(ii)* diverse assemblages of communities, cultures and industries in lower watershed reaches that depend on fresh water for agricultural, ranching, residential and industrial uses; and *(iii)* evolving policies and management that seek to balance the needs of water users and native ecosystems.

2.2. Data collection for distributed hydrology modeling

2.2.1. Topography data

Watershed and catchment boundaries, stream networks, and elevation data used in DHSVM were generated from a USGS 10-m resolution DEM obtained for the study area (http://planning.Hawai'i.gov/gis/). The DEM was post-processed to enforce downhill drainage throughout each study area watershed, and all hydrologic sinks were filled.

Channel initiation was assumed when drainage area equaled 100 ha, and additional downstream catchments were created when drainage area exceeded 50 ha. The resultant stream layer was compared to stream networks from the National Hydrograph

Dataset (NHD) and was modified by varying the number of upstream pixels to develop channel initiation points to reasonably match the NHD. This resulted in 904 catchments with a median size of 90 ha.

Stream layers were created by defining flow routes downstream of each initiation point. Parameters estimated for each stream segment included active channel width and depth, and channel gradient and roughness. Channel characteristics across the region were estimated based on published equations (Parker et al., 2007).

2.2.2. Vegetation mapping

A current 30 m vegetation land-cover map was obtained from the US Geological Survey (USGS) GAP analysis program (USGS, 2005), from which 39 vegetation and land cover types were identified. Data from established field plots within the Hawai'i Experimental Tropical Forest were used to validate SG cover estimates from the GAP analysis layer (Strauch et al., 2016b). Fifty, 30-m radius circular vegetation plots located within between 540 and 1640-m elevation in the center of our study area were used to refine SG cover estimates. SG cover was found to be generally underrepresented in the land cover map and was corrected using modeling based on established relationships of SG occurrence with elevation (30 m) and precipitation (250 m; Giambelluca et al. (2012)) data (Jacobi and Warshauer, 1992). A linear regression [Eq. (1)] ($F = 28.3$, $P < 0.005$, $R^2 = 0.74$, $n = 20$) was developed to correct GAP estimates of SG relative abundance (% stand basal area; SG_{RA}) via the regression function:

$$SG_{RA} = -194.1 + (0.0645 * Elev) + (0.0342 * Precip) \quad (1)$$

Regression model estimates were then used to reclassify each vegetation pixel into one of four SG invasion classes: *(i)* Fully invaded ($SG_{RA} > 10$), *(ii)* Moderately invaded ($5 < SG_{RA} \leq 10$), *(iii)* Lightly invaded ($0.1 \leq SG_{RA} \leq 5$), and *(iv)* Native/No invasion ($SG_{RA} < 0.1$).

Vegetation physical and ecophysiological parameter values required by DHSVM were established for each land-cover type from literature values specific or applicable to Hawai'i forests (Ares and Fownes, 1999; Asner et al., 2003; Giambelluca et al., 2009; Kagawa et al., 2009). Parameters included stand-level percent canopy closure, stomatal resistance, photosynthetically active radiation, rooting depth, leaf area index (LAI), among others. These parameters were used in the calculation of evapotranspiration from the soil, wet canopy, and incident solar insolation on the soil surface. Where we were unable to locate data from local studies, parameters were applied from comparable locations outside of Hawai'i. See Appendix A for all relevant parameters used in this study and Strauch et al. (2016b) for additional detail.

2.2.3. Meteorological data

Meteorological data, including precipitation, air temperature, wind speed, relative humidity, and short- and long-wave radiation, were collected from six climate stations from within or immediately adjacent to the study area. Climate stations were part of the University of Hawai'i and U.S. Department of Agriculture (USDA) Forest Service network of Climate Vegetation Observatory, the Remote Automated Weather Station (RAWS) network, the NRCS Soil Climate Analysis Network (SCAN), and the National Oceanic and Atmospheric Administration (NOAA) weather station network. Data were collected at 3 h time increments and interpolated for each pixel within the study area using inverse distance weighting from each weather station. Periods of missing data were filled in using regression analysis with other station data. Long-wave radiation was not measured at any of the stations, but was estimated from shortwave radiation, precipitation and air temperature (Bowling and Lettenmaier, 1997).

Fog drip, or cloud water interception, is the process by which fog is captured by tree crowns and thereby contributes to the total water input into the system. Fog drip can represent a significant proportion of total water inputs to humid tropical systems, but the proportion of water accounted for by fog drip is spatially and temporally variable (Juvik and Ekern, 1978), and poorly quantified. We did not include fog drip in the hydrology model because: *(i)* as of yet there is no parameter in DHSVM associated with this input; and *(ii)* no measurements of fog drip exist within our study area to validate its estimate contribution to the total water budget. However, the lack of fog drip did not appear to negatively influence DHSVM model calibration on the whole as predicted annual water yields matched well with observed stream flow data (see below). A likely explanation for this lack of influence may be because much of the stream flow in our study area is concentrated in areas with very high mean annual rainfall where contributions from fog drip may be insignificant compared to annual rainfall amounts. In terms of the potential influence of fog drip on our modeled climate scenarios, it is unclear how fog drip will vary over time as temperatures warm, Efforts to accurately quantify fog drip across large areas and identifying the likely response of fog drip to future climate change represent important areas of future research.

2.2.4. Soils and geology data

Soils and geology data were used in the DHSVM to determine the rate and volume at which water moves through the soil profile in both saturated and unsaturated conditions. Soil depth and textural classes were mapped from the USDA, Natural Resources Conservation Service (NRCS) 2008 dataset. Forty-two NRCS soil series were collapsed into 16 taxonomically similar soil types based on textural class. Soil texture type was estimated using published values (Coen and Wang, 1989; Abu-Hamdeh and Reeder, 2000; VanShaar et al., 2002; Hantush and Kalin, 2003; Saxton and Rawls, 2006; NRCS Soil Survey Staff, 2010). A single soil depth of 1.5 m was defined for the project area, which was based on the maximum depth available in soil surveys. The Acrudoxic Hydrudands were by far the most common soil type across the project area.

2.3. Distributed hydrology modeling

We used parameterization files for the DHSVM that had been developed for a coarse scale modeling analysis of ~90 watersheds that make up the study system (Strauch et al., 2016b). The calibration and final parameterization of the DHSVM was validated for the current study area. Our simulations are distinct from this earlier analysis as we used refined SG invasion scenarios and conducted analyses at the scale of 100 ha catchments versus much larger entire ridge to reef watersheds, which were scaled appropriately for prioritizing future vegetation treatments. Details of the DHSVM model are provided here, but see Strauch et al. (2016b) for further details.

The DHSVM model is a complete energy, mass, and water-balance model used to predict the influence of topography, vegetation, local climate, and soils on water flux through a landscape (Wigmosta et al., 1994). DHSVM provides a dynamic representation of the spatial distribution of soil moisture, evapotranspiration, and runoff produced at the resolution of the digital elevation model (DEM; 10-m in this study). The DHSVM model was chosen over other hydrology models because it is recognized as being responsive and robust for capturing meso-scale watershed response to climatic and vegetative conditions (Beckers et al., 2009).

Water yield was calculated in 3 h time steps across eight years (2005–2012) for each of the ~1 million pixels (30 m resolution) (Strauch et al., 2016b), and modeled water yields were summa-

rized to the catchment-level. For the purposes of the WDST, we were interested in comparing the sensitivity of each delineated catchment to climate change and SG invasion. For this reason, mean annual water yield was calculated as the independent contribution of a catchment from the SVM model outputs by subtracting the cumulative upslope water contributions to the catchment.

The model treated groundwater inputs and outputs by assuming a unidirectional loss to deep groundwater. Little local information existed regarding groundwater dynamics to better parameterize this aspect of the model.

Spatial data requirements included a DEM, a land cover map for each SG invasion scenario, soil type and depth, mean monthly precipitation for each climate scenario, and stream hydrography. Temporal data inputs included climate data summarized to 3-h time steps for air temperature, precipitation, relative humidity, solar radiation, and wind speed--from each of six climate-stations located within or adjacent to the project area--and stream gaging data to calibrate DHSVM flow projections.

Model calibration and accuracy - The DHSVM model was parameterized using current vegetation (see Section 2.2.2) and climatic conditions for the 2005–2012 water years. The model optimized modeled stream discharge with respect to observed water yields at the Honoli'i stream gauge (USGS gauge #16717000). This gauge was chosen as it represents average rainfall conditions across the study area and an extended period of record across diverse water years. Calibration consisted of varying the lateral and vertical hydraulic conductivity values to arrive at the best fit with the observed data until the gauge values were in agreement. Following calibration, high agreement was achieved between observed stream discharge and modeled mean monthly flow ($R^2 = 92\%$) and mean annual flow ($R^2 = 99\%$), while daily flow rates tended to underestimate peak flows and overestimate flow in the recession limb following peak flows ($R^2 = 0.62$). For this reason, the EMDS model was built using mean annual flow rates.

2.4. Climate and vegetation scenarios

To simulate effects of future interactions among predicted climatic and vegetation changes, vegetation and climate data were altered from the current observed conditions (Table 1). Six climate scenarios were simulated to depict changes in potential water yield due to predicted future climate change (Chu et al., 2010; Lauer et al., 2013).

The scenarios included: *(i)* current climate; *(ii)* a constant 2 °C increase applied to temperature record (hereafter, warmer); *(iii)* a constant 20% reduction in current annual rainfall (drier); (iv) 20% increase in annual rainfall (wetter); *(v)* temperature increase + rainfall decrease (warmer + drier); and *(vi)* temperature increase + rainfall increase (warmer + wetter). Future temperature and precipitation regimes were consistent with climate-model predictions and have been used in other similar studies (Benning et al., 2002; Timm and Diaz, 2009; Lauer et al., 2013).

Factorial combinations of the climate scenarios were paired with three vegetation scenarios: *(i)* current condition of the vegetation (CC); *(ii)* native forests fully invaded (FI) by SG ($SG_{RA} > 10$, see Eq. (1); full invasion of SG); and *(iii)* all forests fully restored (FR) to a hydrological native only/no invasion condition ($SG_{RA} < 0.1$, see Eq. (1); full removal of SG and replacement with native vegetation). The latter two vegetation conditions represented worst- and best-case scenarios, respectively, for the study area, in which no and complete vegetation management occurred to combat SG invasion. These scenarios allowed us to assess the total potential influence of vegetation on hydrologic output from the study area and compare them to current levels (Strauch et al., 2016b).

2.5. The EMDS decision support system

The EMDS is a two-phased spatial analysis framework that integrates logic and decision models to prioritize landscape units based on: *(i)* ecological assessments at spatial scales relevant to specific management questions; and *(ii)* user-defined and user-weighted decision criteria related to logistical and operational considerations of active management, respectively (Reynolds and Ford, 1999; Reynolds et al., 2014). The logic and decision models are complementary in the sense that the logic model is used to quantify the ecological "state of the system," while the decision model incorporates criteria such as treatment costs, land ownership, resource value, or special area designations to gauge the feasibility, efficacy, or other benefits of treating specific landscape units. The decision criteria influence the final prioritization through a series of weightings that can be altered based on stakeholder input. This, in turn, gives each stakeholder direct access to the decision-making process and creates a customizable model to suit the specific needs of the users. It does so by allowing alternative weights to be assigned to various decision criteria, which directly affects priority scoring of each catchment.

EMDS was used to prioritize landscape treatment units across the study area based on predicted ecological effects of treatments on water yield and other specified ecosystem values. In EMDS, logic models were designed with NetWeaver® software (Miller and Saunders, 2002) to evaluate the ecological status of a particular catchment. The logic model evaluated the strength of evidence that exists to satisfy stated propositions described below. Logic model output scores range between -1, representing no support for a proposition, and 1, representing full support for a proposition.

Decision models were designed using the Criterium DecisionPlus® software. They were used to evaluate results from the NetWeaver logic model (NWLM – see below) together with other data related to technical and economic feasibility of treatment, treatment efficacy, presence of designated critical habitat, and other logistical considerations that were relevant to management. Outputs from these models were combined to produce prioritization maps of decision scores related to the modeled criteria. Decision scores ranged between 0 (low) and 1 (high priority for treatment).

2.5.1. NetWeaver® logic model

The NWLM was used to evaluate the response of each modeled catchment-level water yield to SG invasion by calculating two sep-

Table 1
Mean annual water yield (Gl yr^{-1}) estimated using a distributed hydrology model for the North Hilo-Hamakua study area. Climate scenarios are (1) Current climate, (2) Wetter: +20% current precipitation, (3) Drier: −20% current precipitation, (4) Warmer: +2 °C current temperature, (5) Warmer +wetter: current temperature + 2 °C + 20% current precipitation, (6) Warmer + drier: current temperature + 2 °C − 20% current precipitation. Treatment scenarios are (1) Current Invasion, (2) Full restoration: Complete removal of SG, (3) Full invasion: Complete invasion of SG.

	Current climate	Wetter	Drier	Warmer	Warmer + wetter	Warmer + drier
Full restoration	1309.2	1721.7	923.0	1319.7	1733.2	932.2
Current invasion	1294.0	1706.1	908.3	1309.1	1722.5	922.0
Full invasion	1272.4	1682.7	889.5	1292.1	1703.9	907.1

arate strength-of-evidence (SOE) scores for: (i) restoration--evaluating the proposition that there was high strength of evidence to support SG removal to enhance water yield (i.e., chemo-mechanical removal); and (ii) protection--evaluating the proposition that there was high strength of evidence to support fencing for the prevention of SG spread.

Estimated water yields under current vegetation conditions (CC) were compared to water yields under full invasion (FI) and fully restored (FR) vegetation cover conditions. For each proposition, a ramp function was constructed to evaluate CC water yield against FR and FI water yields. SOE scores near + 1 indicate very high support for the proposed treatment (e.g., removal or protection), scores near -1 indicate very low support for the treatment, while scores near 0 indicate neutral support for either treatment. Simple logic networks were constructed for evaluating the main goals of restoration and protection separately (Fig. 3). Final logic model scores for each catchment were a function of the level of SG invasion and change in catchment-level water yield from current levels to those under complete SG removal (Restoration) or complete SG invasion (Protection). SG invasion and water yield scores were combined using a *union* (e.g., fuzzy set average) logic operator, which treats co-factors as additive and compensating. By incorporating SG invasion levels directly into the logic model structure we eliminated instances where modeled water yield changes were near zero across all vegetation and climate scenarios, which led to artificially high SOE scores despite having little to no SG invasion currently. This occurred specifically in high elevation catchments.

The water yield branch of the logic model was a piecewise linear ramp function with break points at the 90th and 10th percentile differences in water yield between the FI and FR vegetation conditions of all catchments (Fig. 3). High SOE support for restoration (i.e., SG removal) occurred when modeled water yields under CC and FI scenarios were similar (indicating that eradicating SG would lead to increased water yield). The second branch of the logic model emphasized moderately low to moderately high SG coverages. We considered areas fully invaded by SG as a lower priority (though still high priority) compared to those with 20–80% SG cover as these catchments would be less effective at controlling an area of active SG invasion, which was consistent with the HFI Initiative (DLNR, 2011) (Fig. 3).

Alternatively, high SOE support for watershed protection (i.e., fencing and prevention of feral pig invasion), occurred when CC was near the yield predicted for FR (indicating water yield was near its maximum potential), and where SG invasion was minimal. The 90th and 10th percentiles were used in lieu of maximum and minimum values to avoid the influence of outliers on SOE scores.

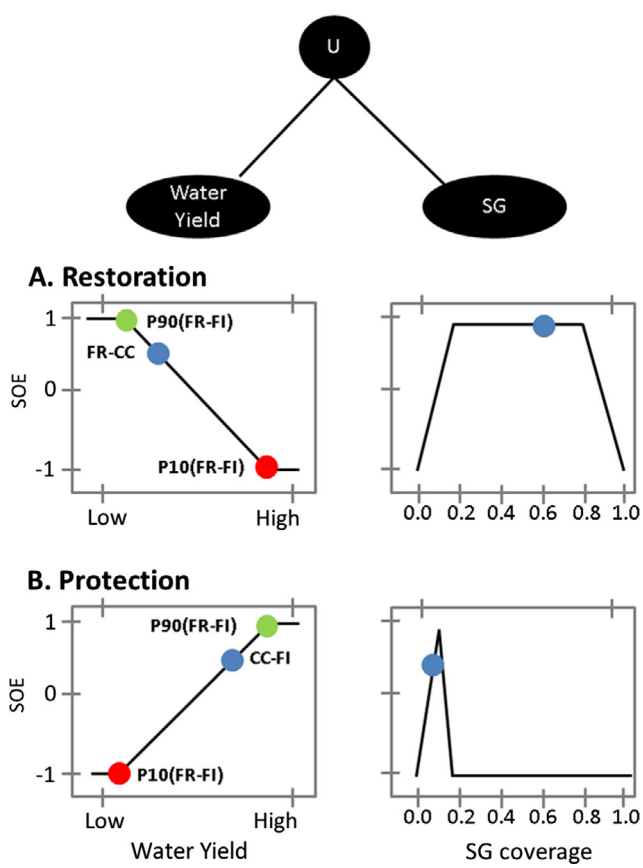

Fig. 3. Logic model depiction. Water yield was estimated for full restoration (FR), full invasion (FI), and current conditions (CC). High strength of evidence (SOE) for restoration (A) occurred where the difference between current conditions and full restoration was close to the 90th percentile (P90) difference between full restoration and full invasion scenario water yields when considering all catchments, and where strawberry guava levels were moderately high. High SOE for protection (B) occurred where the differences between full invasion and current condition water yields were near the 90th percentile difference between full restoration and full invasion scenario water yields, and where strawberry guava levels were low. A union (mean) operator was used to combine both ramps for each proposition. Each example shows high support for the respective proposition.

2.5.2. Multi-criteria decision model (MCDM)

Workshops and one-on-one meetings were held with a diverse set of land managers, which included state, federal and private managers representing >90% of the large ownership parcels within the project area, and thus, the largest primary stream managers within the study area. From these meetings, final decision criteria most influential to watershed management were decided upon. These included: *(i)* transportation costs; *(ii)* SG removal/fencing costs; *(iii)* 5 yr maintenance treatment costs; *(iv)* aquatic habitat quality; *(v)* conservation score determined by the primacy of ecological conservation goals by each landowner both within and adjacent to each catchment (see Appendix B and below); *(vi)* a critical habitat score representing the proportion of a watershed designated as US Fish and Wildlife Service critical habitat for rare or endangered species; and *(vii)* the catchment proportionate contribution to total water yield within its watershed (Fig. 1). We then developed spatial data layers to represent each factor, and summarized them to each catchment within ESRI ArcGIS 10.0 software. MCDM data are briefly described below, but see Appendix B for a complete description on how each data layer was developed.

SG removal treatment costs – Standard weeding practices for SG involve the mechanical cutting or wounding of individual stems, with subsequent silvicide application to prevent resprouting. Costs were divided into initial treatment and follow-up maintenance treatments, which included labor, materials (Fig. A3), and number of trips required to complete restoration (Fig. A4), maintenance (Fig. A5), and transportation costs (Figs. A5 and A6).

Protection fencing costs – Fences are used to limit the movement of feral ungulates into targeted conservation or protection areas. Fencing costs are highly variable depending on terrain, remoteness, soils, and vegetation. We worked with local managers to define average fencing costs for this region, and we assumed that fencing the widest part of each subunit would provide a reasonable scalar for determining catchment fencing cost.

Land conservation status – We used property ownership maps to classify land ownership into seven conservation categories (Fig. A7) that were classified on an ordinal scale with a value of seven (7) indicating the highest conservation value (for our study area, federal U.S. Fish and Wildlife Service lands), where natural resource conservation is the prime management objective. A low value of one (1) was given to privately owned lands where natural

resource conservation was not a priority. A conservation index was calculated for each catchment as the area-weighted mean score for each catchment, which in turn was averaged against the mean score for all adjacent catchments. Therefore, catchments that fell within and were surrounded by federal, and to a lesser extent state and county lands received higher scores.

Critical habitat: Critical habitat was identified from land surveys that identified core habitat areas for species of conservation concern (Fig. A8). The critical habitat score corresponded with the number of unique critical habitats identified within a catchment, and these scores ranged from zero (0) to five (5). See Appendix B for a complete list of species of concern.

Aquatic habitat quality – Stream habitat quality was estimated for each stream reach in the study area using the 2010 National Fish Habitat Partnership mapped indices of stream degradation risk (http://fishhabitat.org/content/nfhp-data-system), which estimates the cumulative impact of 15 different anthropogenic disturbance features on in-stream habitats (Esselman et al., 2011) (Fig. A9). The native scale for these values ranged from zero (0) to one (1), with one indicating the poorest predicted habitat condition (high degradation), and zero indicating high quality habitat (no or low degradation).

Decision model - Within the Criterium DecisionPlus® software, pairwise comparisons were conducted against each criterion to create a complete matrix of weights whose values ranged from 9 (Criteria A is critically more influential than Criteria B) to 1/9 (Criteria B is critically more important than Criteria A). The size of the weights matrix varied by management objective (9 × 9 for restoration, and 7 × 7 for protection) due to the different number of subcriteria for each. From these matrices, priority scores were calculated using the analytic hierarchy process (Saaty, 1988), which used manager-determined pairwise weights to calculate a weighted average among all criteria. The weighted average calculated for each catchment represented the final priority score for that subunit. Final priority scores output from the MCDM model ranged from 0 (no support for a management activity) to 1 (full support).

3. Results

3.1. Strawberry guava coverage

Estimated cover of SG represented approximately 290 km^2 or 35% of the study area, and was found at some level in nearly three-quarters of all catchments (Fig. 4A), with more than one third of all catchments showing >50% coverage of SG. Modeled major SG infestations were absent above 1370-m, and largely occurred below 1000-m, but above the forest line at 450-m.

3.2. Water yield

Under current vegetation and climate conditions, mean annual catchment-level water yield ranged from 0.0 gigaliters (Gl) yr^{-1} (highest-elevation catchments making up the alpine summit area of Mauna Kea), to 9.0 Gl·yr^{-1} (the southern and central portion of the study area), which corresponded with an area of extremely high annual precipitation (635+ cm yr^{-1}; Table 1, Fig. 4B). Mean annual water yield across all catchments was 1.43 Gl·yr^{-1}, and in total, the study area produced 1294 Gl·yr^{-1}.

Full SG invasion reduced annual water yield by 1.7% (22 Gl·yr^{-1}) across the study area. When full restoration and full invasion scenarios were compared, full restoration increased mean annual

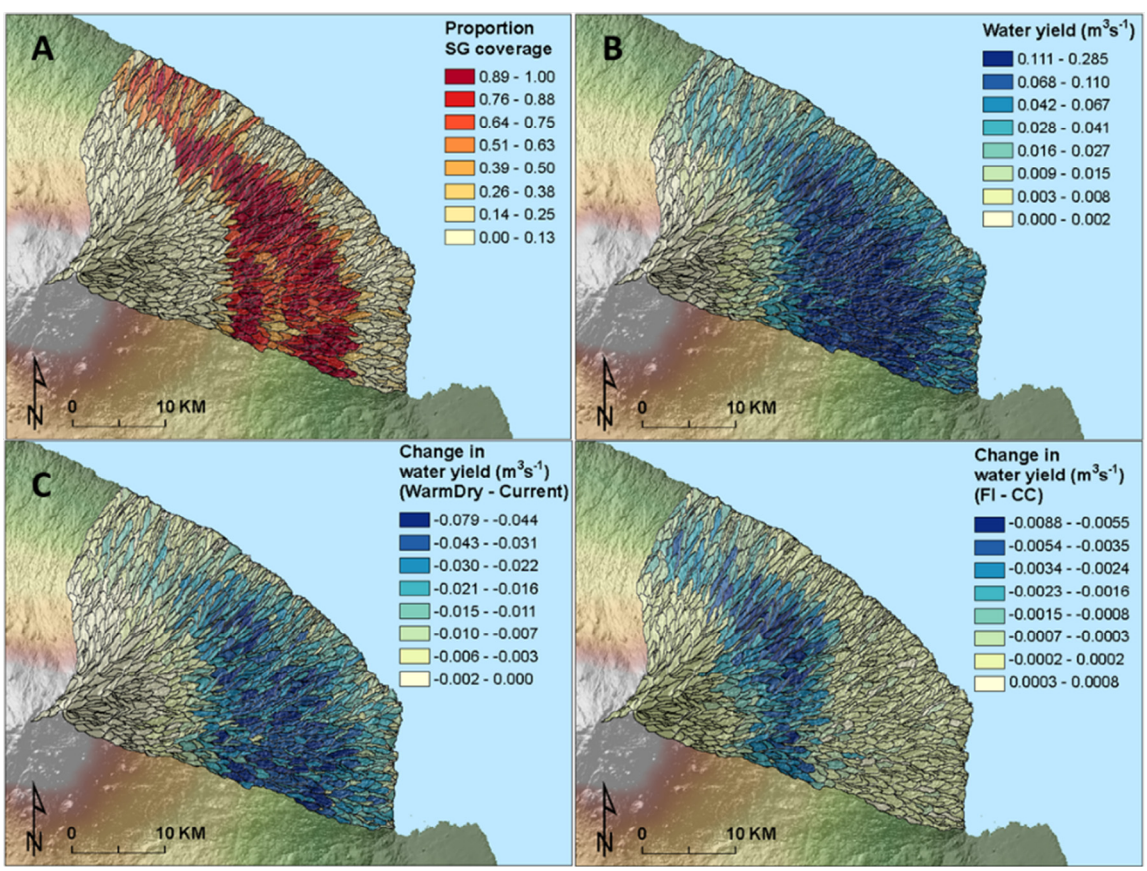

Fig. 4. (A) Modeled proportion of strawberry guava catchment area invaded by SG; (B) DHSVM-estimated water yield for each catchment; (C) difference in water yield between warm + dry scenario and current climate; and (D) difference in water yield between fully invaded and current invasion scenarios within the North Hilo-Hamakua study area. Abbreviations are: SG, strawberry guava; FI, Fully invaded; CC, Current condition (i.e., current level of SG invasion).

water yield by 2.8% (37 Gl·yr^{-1}) over the fully invaded scenario (Table 1, Fig. 5). In contrast, the warm-dry climate scenario led to a 29% reduction in water yield across all catchments compared to the current climate scenario (1294 vs 922 Gl·yr^{-1}, Table 1, Fig. 5). As expected, the greatest declines in water yield were located where current precipitation levels were highest. Across all climate scenarios, the highest elevations experienced minimal changes in water yield due to low overall water production in these settings. The FI *plus* warm-dry climate scenario reduced mean annual water yield by a 30% compared to current conditions. A similar increase was found (30.7%) when the FI *plus* warm + dry scenario was compared to the fully restored and current climate scenario. The largest differences in water yield were found between FI *plus* warm + dry compared to the FR *plus* warm + wet scenario (47.7% reduction, 826 Gl·yr^{-1}, Table 1).

3.3. Logic model

3.3.1. SG removal treatments

Across the study area, 4.6% of the catchments (6.3% by area) displayed high priority scores for SG removal (SOE > 0.5) in the logic model (Fig. 6). Nearly all these catchments were located in the south-central portion of the project area where annual precipitation and water yield rates were highest (Fig. 4). Approximately half of all catchments had SOE scores < −0.5, suggesting that most catchments had low or no support for restoration.

We compared logic model outputs for the current *plus* warm + dry climate scenarios. Despite large differences in total water yield across treatments (see above), differences in SOE scores were negligible (Fig. 6). Overall, the rank order of catchment SOE scores for restoration was relatively constant across all six climate scenarios (r > 0.98).

3.3.2. Protection treatments

Approximately 4.5% of catchments received high protection SOE scores. These units were located immediately adjacent to and uphill from catchments identified with high restoration SOE scores (Fig. 6). Some smaller catchments closer to the ocean also received high protection scores; these occurred where SG coverage was minimal and generally fell within the 0 to 1300-m elevation belt.

3.4. Multi-criteria decision model (MCDM)

Initial restoration treatment costs varied greatly among catchments (from <$10 to >$100,000,000 USD per catchment), which at an average size of 100 ha, corresponds to $1–$10,000,000 per

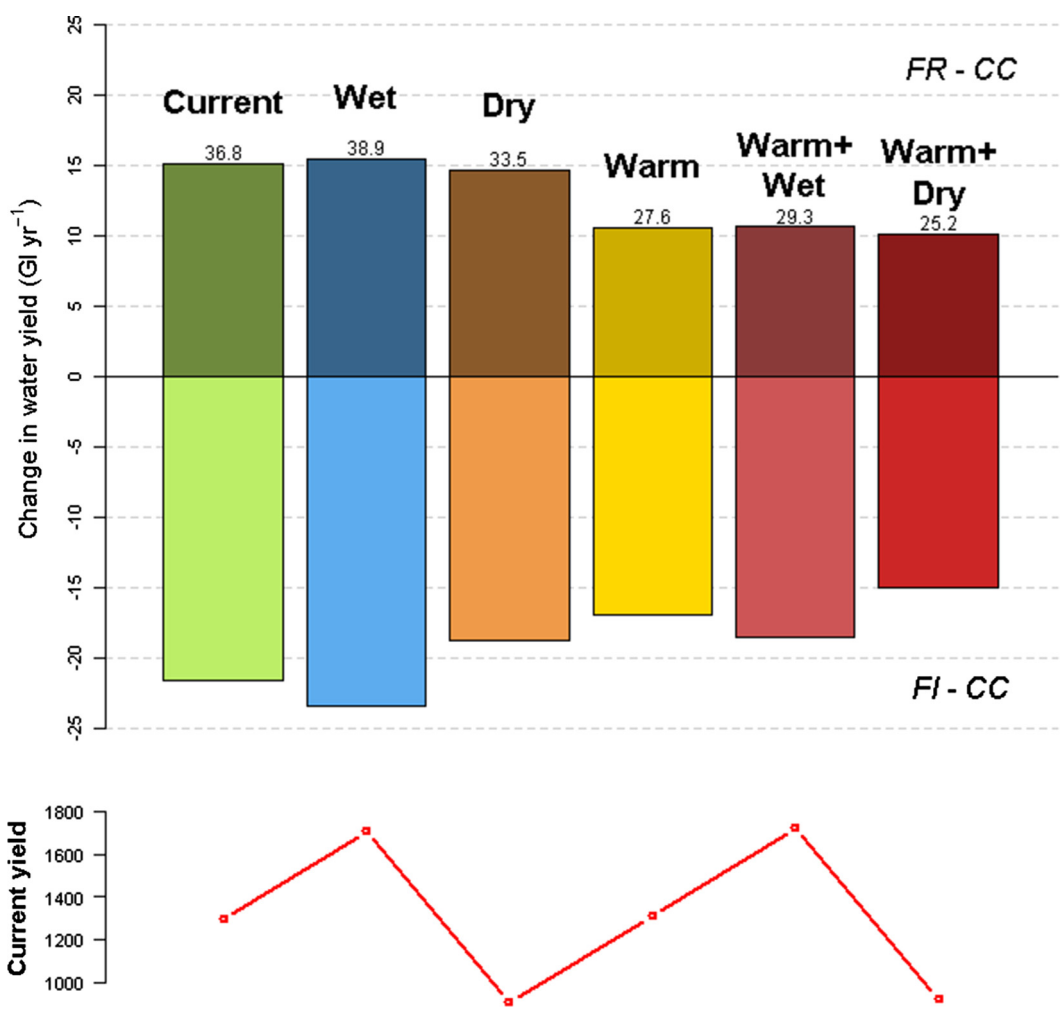

Fig. 5. Bar plot of the difference in total water yield (Gl yr^{-1}) associated with the fully restored and fully invaded strawberry guava (SG) conditions when compared with the current invasion condition for the North Hilo-Hamakua study area. Colored bars represent comparisons of total study area water yield by climate scenario. The portion of bars above the zero line (darker shading) represents the water yield gained by fully restoring the current condition, obtained by subtracting the current condition (CC) from the full SG restoration (FR) condition. Portions of the bar below the zero line (lighter shade) represent water yield lost due to full SG invasion (FI) when compared to the current condition (CC), obtained by subtracting the current vegetation condition (CC) from the fully invaded (FI) condition. Values immediately above the colored bars represent the total water savings (Gl yr^{-1}) from the FR scenario compared to the FI scenario. The red trace in the lower line graph represents the current water yield under each climate scenario.

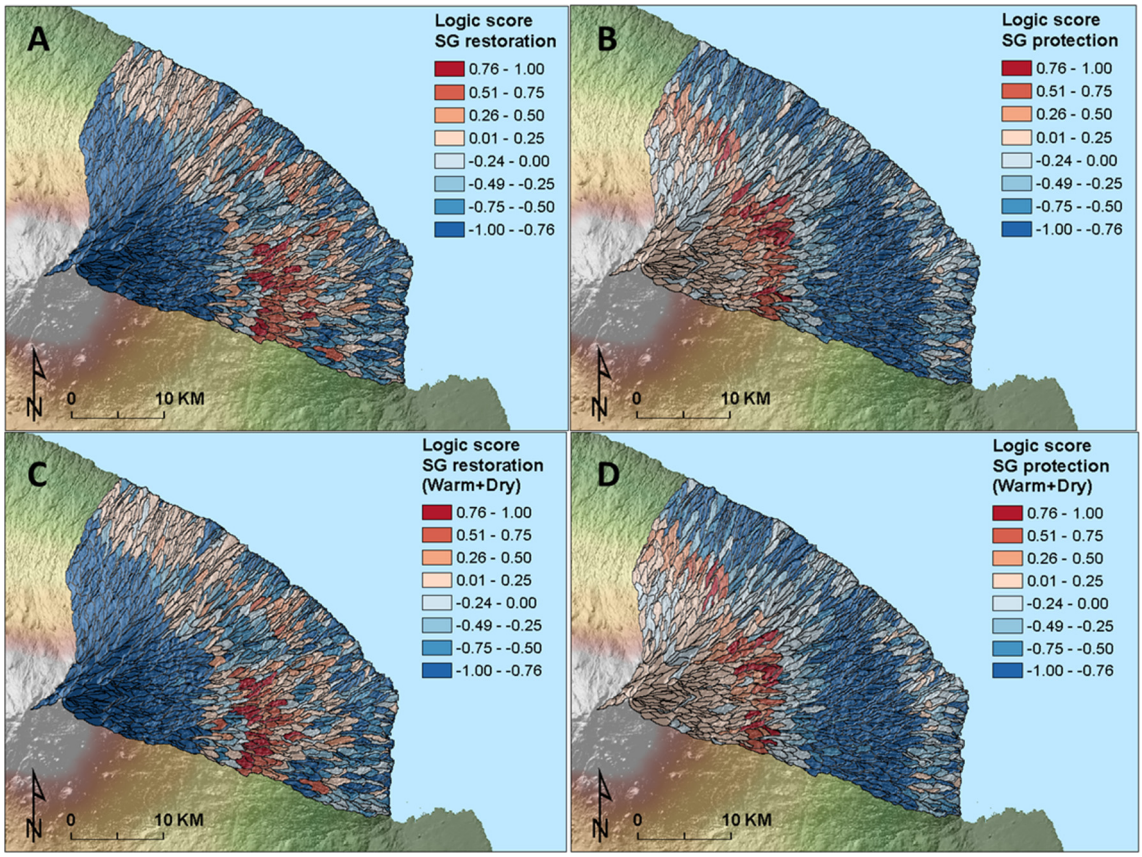

Fig. 6. Strength-of-evidence scores from the NetWeaver® Logic Model for strawberry guava (SG) removal (A) and protection from SG (B) in the North Hilo – Hamakua study area. Logic scores near 1 (dark red catchments) indicate a high level of support for either SG removal or protection, while values near -1 (dark blue) indicate no support for the propositions.

ha. High subunit costs were associated with poor access, long travel times, and heavy SG infestations that required many visits for initial treatment and maintenance. Two-thirds of all catchments required <90 min of one-way travel access time, while about 20% required ≥2 h of one-way travel time due to limited road or trail access (Fig. 7). Remote areas most often coincided with high SG invasion level, steep and highly dissected topography, high annual precipitation, and high potential water yield.

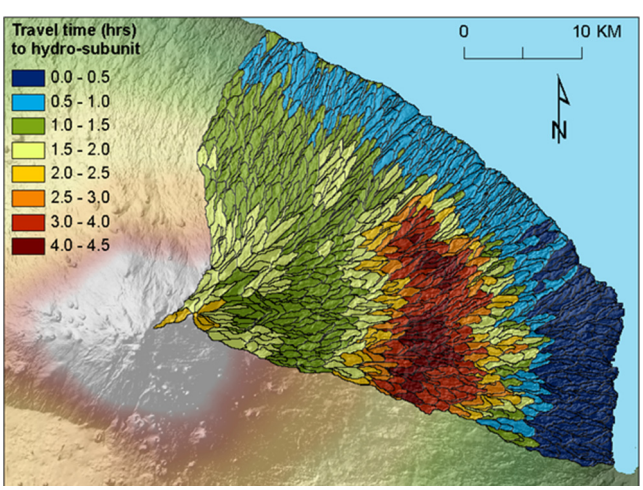

Fig. 7. One-way travel time (hours) to each catchment within the North Hilo-Hamakua study area.

In terms of landownership patterns, mid-elevation forests in the central and south-central portions of the study area contained large areas in state forest reserve, state natural area reserve, and federal wildlife refuge status. Management in these areas emphasizes conservation of biodiversity and natural resources, which in turn led to high landowner Conservation Scores in the MCDM. Conversely, low elevation areas were made up primarily of private urban, exurban, and agricultural lands, and received low conservation scores.

Within the MCDM, we chose an example weighting scheme in Fig. 8**,** in which primary topics from the logic model (Restoration Potential), which emphasizes managing for water yield, were most influential (potential weight = 0.583), and all other primary criteria were weighted equally (other criteria weights = 0.083; Fig. 8). Based on this weighting scheme, final priority scores for restoration from the MCDM ranged from 0.29 to 0.73. Possible values range between 0 indicating no support for management and 1, indicating full support for management (Fig. 9A and B). Approximately 10% of the study area (10,500 ha), received logic model scores >0.6, indicating a high potential benefit from restoration (Fig. 9**A**). Many of these units were located in the central and south-central portion of the study area where rainfall, potential freshwater yield, and SG coverage are high, but access is very limited.

Weighting factors can be independently adjusted, accommodating an extremely large range of manager preferences; here we present two alternative MCDM weightings to highlight the independent influences of treatment cost (Fig. 9C and D) and conservation value (Fig. 9E and F). When higher weights for Treatment Cost were used in the MCDM model, there was only weak support

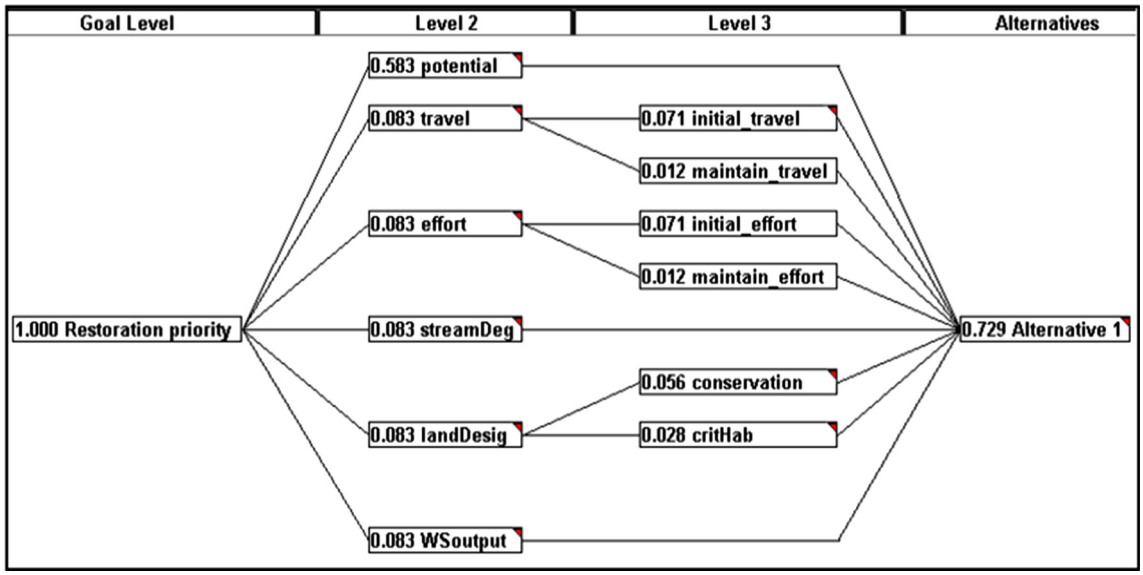

Fig. 8. Decision model architecture for non-native species restoration in North Hilo – Hamakua districts. Variable names are: (i) potential, restoration potential SOE scores from the NetWeaver® logic model; (ii) travel, cost of traveling to and from each catchment for both the initial and maintenance restoration; (iii) effort, materials cost related to treating each catchment; (iv) LandDesig, combination of conservation score and amount of critical species habitat; and (v) WSOutput, proportion of total water yield from each watershed contributed by each catchment. Numbers represent the relative contribution of each criterion to the restoration priority.

for restoration or protection across the study area. Conversely, when Conservation Value was emphasized in the MCDM, approximately 22% of the landscape received high priority scores for SG restoration and 25% for protection, which occurred primarily near the central and south-central portions of the study area, and coincided with federal and state conservation areas, including Hakalau Forest National Wildlife Refuge, Hilo Forest Reserve, and Laupahoehoe Natural Area Reserve. These conservation areas were set aside early in Hawai'i's history of resource extraction to ensure contiguous blocks of forest coverage to conserve water resources for future uses, and in turn were ranked highest in conservation value in our rating system.

Some catchments in the south-central region received high scores for both water yield potential (Fig. 9A and B) and conservation value (Fig. 9E and F) indicating opportunities to achieve multiple objectives.

4. Discussion

The *Rain Follows the Forest Initiative,* now the *Healthy Forests Initiative* (HFI), seeks to inspire and guide efforts to enhance the long-term sustainability of forest and freshwater resources. These visionary and long-term management plans emphasize landscape-level planning to protect forest resources and sustain freshwater quality and quantity. These goals are addressed while integrating the needs of diverse mixed-use land ownership patterns that intersect steep and compact biophysical gradients from coastal near-shore environments to high-elevation montane ecosystems. Successful application of landscape-level management will require implementing tools to facilitate multi-use and multi-resource planning that best serves diverse public interests.

Decision support models are one such tool that are designed to incorporate the complexity of socio-ecological systems into small- to broad-scale management planning across a variety of ecosystems (Reynolds et al., 2014). To this end, our WDST was developed to identify management strategies that enhance and preserve hydrologic function while considering land ownership patterns, accessibility, feasibility and cost of treatments. We showed through scenario building how landscape prioritizations can change as specific decision criteria are emphasized over others, and suggest that such modeling can provide stakeholders with an understanding of how each criterion influences decision making. Taken together, this tool is a vehicle for a transparent decision making process among representative stakeholders while supporting ecological processes for the future resilience of Hawai'i's native ecosystems.

4.1. Logic and decision support modeling

Climate variability, climate change and species invasions exert important influences on freshwater resources (Milly et al., 2005), which can result in water security concerns. Disentangling influences of global change factors on future freshwater yields and then designing efficient management strategies in a landscape context are becoming increasingly important to shaping effective land and water resource management. An important step towards this understanding is quantifying regional-level hydrological responses to global change and assessing management influences on potential future yields under various climate and vegetation scenarios (Safeeq and Fares, 2012).

As climate changes, reliance on water production from natural areas will require effective management strategies to help maximize ecosystem services through strategic landscape-level prescriptions (DLNR, 2011). Across our study landscape we found that the FI scenario led to an average 2–3% decline in total water yield under current climate conditions, but responses were highly variable with the largest proportional declines (30–80%) occurring in wetter mid-elevation regions, and the smallest changes (no change) occurring in the highest elevations where precipitation and water outputs were low. Under climate change alone, freshwater supply can only be maintained through possible physiological adjustments by plants that lead to reduced evapotranspiration, for example by plants reducing uptake or increasing water-use efficiency. Because such adjustments are unlikely (Strauch et al., 2016b), especially given concurrent increases in the cover of water demanding invasive plant species, climate mitigation would appear to be limited to management interventions that reduce the influence of water-demanding non-native tree species, such

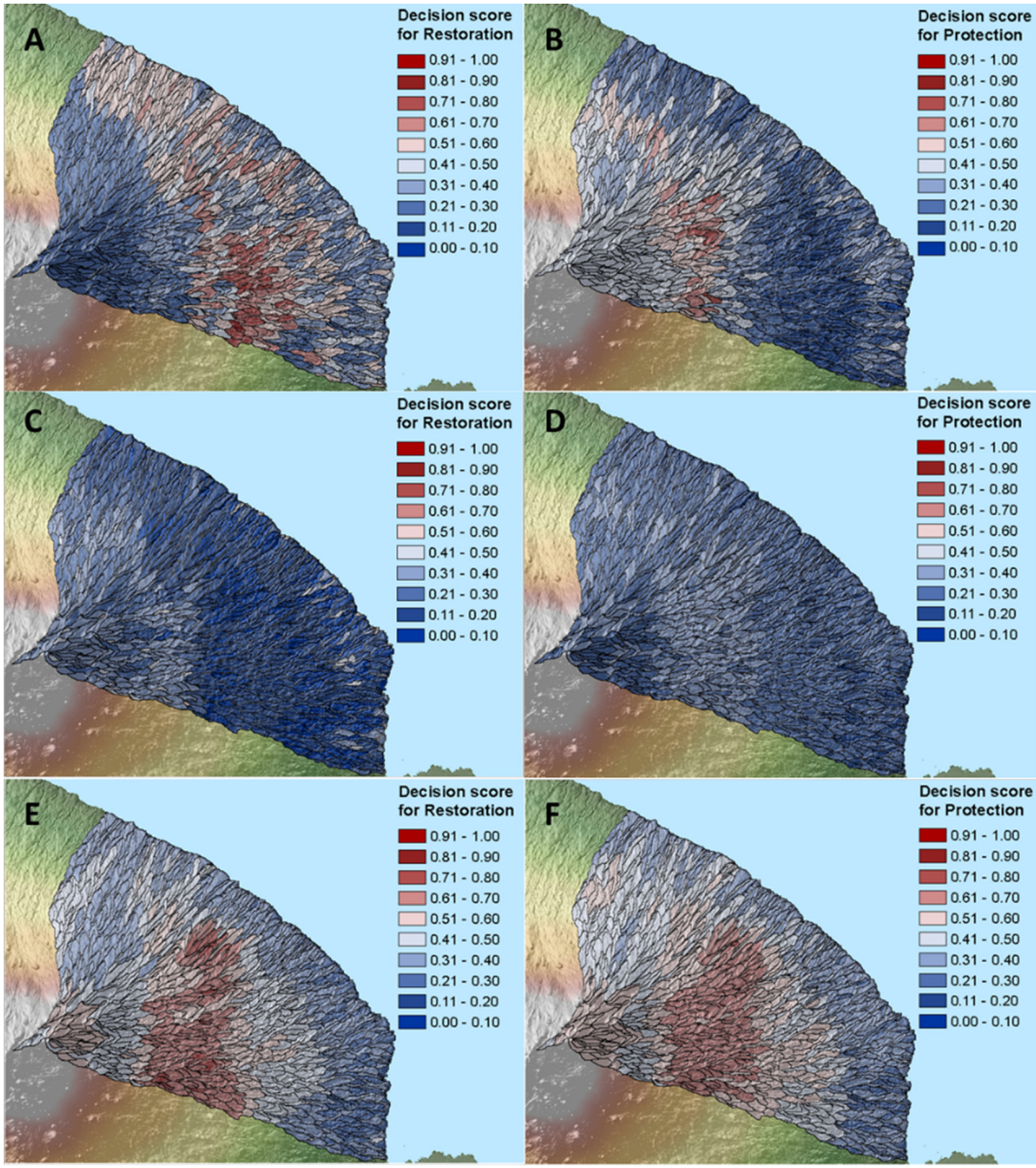

Fig. 9. Priority scores from the Decision Model for (A, C, E) SG restoration and (B, D, F) protection from SG. (A, B) using weightings from Fig. 7, (C, D) using weights to emphasize treatment costs, and (E, F) to emphasize land conservation value (critical habitat, and land ownership).

as SG, on hydrologic yields. Accordingly, management can be implemented that directly mitigates the effects of invasive species density and extent, and using tools such as the WDST developed here, these treatments can be targeted to areas where hydrological benefits are maximized and treatments are most feasible.

Our logic model and decision model scores showed little variation among climate scenarios. The proportional increases or decreases in water inputs across climate scenarios logically led to proportional increases or decreases in water yield (Figs. 3 and 4), and similarly, few differences in logic and decisions scores were apparent across scenarios (Fig. 5). This suggests that despite large differences in hydrologic output, climate change in our modeling did not appreciably affect landscape prioritizations and associated invasive species management strategies. That is, high priority areas under today's climate regime will remain high priority management areas in the future. This may be particularly true for protection prioritizations in which a distinct climate-driven elevation belt was identified immediately uphill from major SG infestations. While feral ungulates are not the only vector of SG spread, fence installation across this area may reduce the rate of SG spread into upper watershed areas currently free of SG, especially given continued warming (Giambelluca et al., 2008).

Our climate change scenarios did not include dynamically downscaled data from Global Climate Models, nor did we consider potential climate-driven range shifts of native and non-native species across the study area. The former would have allowed an assessment of spatial and temporal non-stationarity in the weather patterns under the climate change scenarios and concomitant changes in the modeled hydrologic response. Ecophysiological responses to climate change may cause differential expansion or contraction of niche space across species assemblages. However, the scenarios used here were within the late-21st century climate

under the business as usual emissions pathway (IPCC, 2013). Furthermore, the accuracy and utility of downscaled climate projections and assessments of potential range shifts from these model predictions is unclear due to the "compounding uncertainties" of these methods (Anderson and Ferree, 2010; Beier and Brost, 2010). The current assessment bridges an important gap between identifying impacts of future climates on natural resources and using these results to enable a transparent decision making process between stakeholders and managers to direct management across ownership boundaries (Fowler et al., 2007).

The EMDS model results corroborate the assumptions of the HFI (DLNR, 2011) that high precipitation areas are those at highest risk of non-native species invasions and will incur the greatest absolute losses of freshwater in response to climate change and forest degradation. Across climate scenarios, the wettest areas (MAR $\geq$ 6711 mm·yr^{-1}) received the highest logic model priority scores for protection or restoration (SOE > 0.5). While the HFI uses a coarse-scaled approach to mapping priority management zones, our WDST scaled the analysis to the small catchment-level, which provided a higher resolution view of potential changes in water yield, level of SG invasion, and relevant costs associated with restorative management. By delineating our project area into reasonably sized management units, and linking logic and CDP decision models into our WDST, we were able to identify treatment units that: *(i)* are tailored to the geography, hydrology, and ecology of the area; *(ii)* are predicted to have the largest net benefit to the ecology and hydrology of the region; and *(iii)* satisfy the needs of specific land owners while addressing both limitations and opportunities identified by the WDST.

For example, because road and trail access was a major constraint on effective restoration activities (Figs. 6 and 8C and D), some of the highest priority catchments identified by the logic model were subsequently removed from management consideration by the Decision Model, which downgraded their priority due to exorbitantly high treatment costs. In all, approximately 15% of the total study area (~128 catchments) required >2 h of one-way travel time to access, with the import effect of increasing restoration costs due to higher number of trips required to complete SG chemical and-physical control treatments. To overcome access limitations, alternative transportation methods may need to be considered, including helicopter transport for crews, extended backcountry stays, or the construction of a new access trails to expedite travel. Remote areas generally are located within state conservation lands, and improved access to these areas may serve to enhance multiple use management objectives such as gathering of non-timber forest products, camping and hunting. Finally, where improved access is not feasible because of rugged terrain or disease spread issues (e.g., see Mortenson et al., 2016), management may limited to aerial spread of the SG biological control agent, the Brazilian scale insect *Tectococcus ovatus* Hempel, which can limit SG growth, and presumably related water use and SG spread (Wikler et al., 2000; USFS PSW, 2013).

Because feral pigs consume SG fruits and then defecate viable seeds along with nutrient rich feces, they are vectors for SG spread. Thus, fencing is a management practice that can be used to limit pig-related spread of SG into low SG or SG-free areas. There are important limitations, however, to a fencing strategy to reduce SG spread. There are other vectors for SG spread (Cole et al., 2012), for example frugivorous birds, which typically are not deterred by fences. Fencing installations in remote areas also are very costly, and fencing projects can be the source of social conflicts. On this last point, while cost and ownership affinity for conservation can be viewed as social factors, our WDST does not consider the range of cultural or social factors that in certain natural resources management settings can strongly influence decision making (Kealiikanakaoleohaililani and Giardina, 2016). In addition to social factors not considered by the WDST, there are cultural factors that were not considered here. For example, a community-based strategy might identify different data layers or even approaches for prioritizing actions or protections compared with the approach conceptualized and data used in our agency driven approach. Such a broadening of this tool would require new communications with community based groups, for example hunters, native Hawaiian practitioner groups, or other non-agency users of the forests. Subsequent translation of the important human, cultural even spiritual priorities for landscape stewardship into spatial data layers would require both front-end relationship building and then creative and thoughtful methodologies not yet widely available (Hiiaka Working Group, 2011). We do envision that such methodologies could be incorporated via CDP into future modifications of our WDST, representing a future area of research with the potential to shape a new dialogue among agencies and communities.

5. Conclusions

Climate change and non-native species invasions will impact tropical watershed functioning with important implications for freshwater management. Our hydrology models predicted a ~30% reduction in water yield (~380 Gl yr^{-1}) across the study area under a warmer and drier climate with an additional ~15 Gl yr^{-1} being lost under full plant invasion (Fig. 4). Freshwater losses from SG invasion can be directly mitigated through the strategic allocation of restoration treatments aimed at both reducing the impact of SG in areas where it is already established, and by preventing the spread of SG into uninvaded areas,

Relying on input data layers describing hydrological response to invasive plant control, travel time and treatment costs, conservation focus of ownerships, critical habitat designations, and land and stream ecological condition, we found that our WDST was able to: *(i)* transparently translate drivers of decision making into spatial data layers that allow EMDS and CDP to distinguish low from high priority areas for SG management or prevention of SG spread; *(ii)* provide managers with easily understood approach to manipulate the weightings and so the importance of various data layers determining priority scores; *(iii)* provide managers with data to construct defensible positions on how management decisions can anticipate potential future changes in climate; *(iv)* precisely describe costs associated with SG control or prevention management for each of the 904 catchments examined in this analysis. Our analysis also provides a quantitative approach for retrospectively examining past management decisions and matching these decisions to CDP weighting scenarios, with the goal of helping managers to assess past decision making. A future benefit of this tool and tools like it are to improve the efficiency with which limited financial and human resources are allocated to managing areas such that priorities for management are clearly identified, defensible to the public and funders, and result in the highest potential for protecting or even enhancing future freshwater yields while restoring native species habitats. Furthermore, in a user community driven application of this WDST, future efforts can transparently balance costs and benefits of articulated management strategies. Using the WDST, we identified high priority watersheds and catchments in the study area for protection from further SG invasion, and these consistently aligned with attributes of high elevation, easy access plus low to no SG invasion (and so low cost), high modeled hydrological impact of invasion, and high conservation value. We also identified high priority areas for SG restoration, which aligned with high rainfall, high water yield, easy access plus moderate SG invasion (and so low cost), and high conservation value. While many catchments received high restoration scores,

the additional inference provided by CDP decision modeling suggested that in some areas high costs of restoration/protection and access limitations precluded the feasibility of treatment. On this aspect of implementing a WDST, coordinating restoration efforts among ownerships and implementing cost-effective prevention of invasive species spread are all actions critical to the success of future efforts.

Research needs going forward include: finer scale vegetation especially SG distribution mapping; additional plot scale monitoring to decipher ecophysiological characteristic of native and non-native invasive plan that lead to improved characterizations of non-native plant invasion on distributed hydrology; empirical data describing the time-series responses of climate, growth rates and SG spatial changes; the ecological, and hydrological benefits and costs of ungulate removal; refined estimates of climate change for the region (e.g., dynamic downscaled forcasts); forest structural information (as obtained by LiDAR, for example); and the sensitivity of key DHSVM vegetation parameters to climate change. Future WDST development needs include: continued dialogue and validation of assumptions associated with various decision criteria; tailoring of map products to manager needs; and continued collaborative efforts to implement the WDST into planning and decision making at the watershed partnership scale. Future efforts could also benefit from extending the current effort to new geographic areas with different climatic, species, soils and management objectives to gain a sense for the flexibility of the current platform in meeting diverse needs across Hawai'i and ideally the Pacific.

Acknowledgments

This research was supported by the Pacific Islands Climate Change Cooperative (PICC) grant # F11RG00295 and Region 5 of State and Private Forestry, USDA Forest Service. This project was supported in part by an appointment to the Internship/Research Participation Program at the Pacific Northwest Research Station of the USDA Forest Service, administered by the Oak Ridge Institute for Science and Education through an interagency agreement between the US Department of Energy and US Forest Service. We thank Heather Kimball for reviewing previous versions of this manuscript and assisting with cost analyses. We extend a special thank you to Cheyenne Perry (Mauna Kea Watershed Alliance), Kaeo Duarte (Kamehameha Schools), Robert Nishimoto (Hawai'i Division of Aquatic Resources), Colleen Cole (Three Mountain Alliance) and Steve Kendall (US Fish &Wildlife Service) for initial consultations on the development of the WDST presented here, including but not limited to: information on the factors that drive decision making; real costs for restoration and prevention management of SG; spatial data layers for ownerships, trails, and conservation status; ranking of ownerships by conservation status.

Appendix A. Vegetation parameters and descriptions applied to modified GAP land cover dataset to inform DHSVM simulations

Parameter	Description	Units	Comments
Vegetation description	Name of vegetation type	Text	
Overstory present	Whether an overstory is present	Binary	
Understory present	Whether an understory is present	Binary	
Fractional coverage	Canopy cover	Percentage	Assumes understory is 100%
Trunk space	The base height to the first branches	Fraction of total height	
Aerodynamic attenuation	Canopy attenuation coefficient for wind profile	Extinction coefficient	Not included
Radiation attenuation	Shortwave radiation extending to the ground surface (canopy attentuation)	Attenuation coefficient	Fixed value
Impervious fraction	Percent of ground cover that is impervious	Percentage	Estimated for rocky and built environments
Height	Height of each vegetation layer (canopy and understory)	Meters	Used plot data where available
Maximum resistance	Maximum stomatal resistance for each vegetation layer (canopy and understory)	$s \cdot m^{-1}$	Modeled from Giambelluca
Minimum resistance	Minimum stomatal resistance for each vegetation layer (canopy and understory)	$s \cdot m^{-1}$	Modeled from Giambelluca
Moisture threshold	Soil moisture threshold above which soil moisture does not restrict evapotranspiration for each vegetation layer	0–1	Constant from Giambelluca
Vapor pressure deficit	Vapor pressure deficit above which stomatal closure occurs for each vegetation layer	Pascuals	Modeled from Giambelluca
RPC	Fraction of shortwave radiation that is photosynthetically active for each layer	Fraction	Constant from Giambelluca
Number of root zones	Number of rooting zones	Integer	Estimated
Root zone depths	Depth of soil layers	Meters	Estimated

(*continued on next page*)

Vegetation parameters and descriptions applied to modified GAP land cover dataset to inform DHSVM simulations (*continued*)

Parameter	Description	Units	Comments
Overstory root fraction	Fraction of the roots of the overstory in each root zone	Fraction	Estimated
Understory root fraction	Fraction of the roots of the understory in each root zone	Fraction	Estimated
Overstory monthly LAI	Overstory leaf area index (one-sided) for each month	leaf area (m^2)/ground area (m^2)	Modeled from MODIS
Understory monthly LAI	Understory leaf area index (one-sided) for each month	leaf area (m^2)/ground area (m^2)	Modled from MODIS
Overstory monthly albedo	Overstory albedo for each month	Fraction	Modeled from MODIS
Understory monthly albedo	Understory albedo for each month	Fraction	Modled from MODIS

Appendix B. Multi-criteria decision model data development

Strawberry Guava (SG) Removal Treatment Costs – Standard weed removal practices for SG employ chemical and mechanical treatments, where a machete is used to stump (small diameter) or wound individual SG trees followed by an application of selective silvicide (tree herbicide; costs based on a Garlon® prescription) to the open wound. Cost for chemo-mechanical treatments was determined as a function of the following variables:

- Transportation time to a given pixel on the landscape (Figs. A1 and A2),
- Time for a 3-person crew to conduct treatment work in a pixel, based on a 10 h workday,
- Cost of materials to treat a pixel (Fig. A3),
- Number of trips for a 3-person crew to complete a hydro-subunit (Fig. A4),
- Number of trips, time, and cost of materials to complete follow-up maintenance treatments (Fig. A5).

Using a combination of TIGER road data (http://dx.doi.org/10.1016/j.foreco.2017.05.046) and field surveys by Mauna Kea Watershed Alliance partner organizations, a first approximation of a trail and road network map was constructed for the study area. Each road segment was given a surface classification and an average speed of travel to compute an approximate time to destination when originating from the Institute of Pacific Island Forestry (IPIF) in Hilo, Hawai'i. Transportation times were calculated as the least-cost distance (fastest) route from IPIF to every 30-m pixel along the mapped road and trail network. The point along a given road or trail closest to any given pixel in the study area (not road or trail) was identified and served as the stopping point for crew driving and foot travel to initiate the start of treatment. Transportation cost was calculated as the Internal Revenue Service standard mileage rate for travel ($0.35 km^{-1}), which was confirmed as the actual cost rate incurred by Mauna Kea Watershed Partnership personnel, who were practiced in SG treatment application in the field.

Travel time to a given pixel in the landscape from the nearest road or trail node was calculated using a path distance function

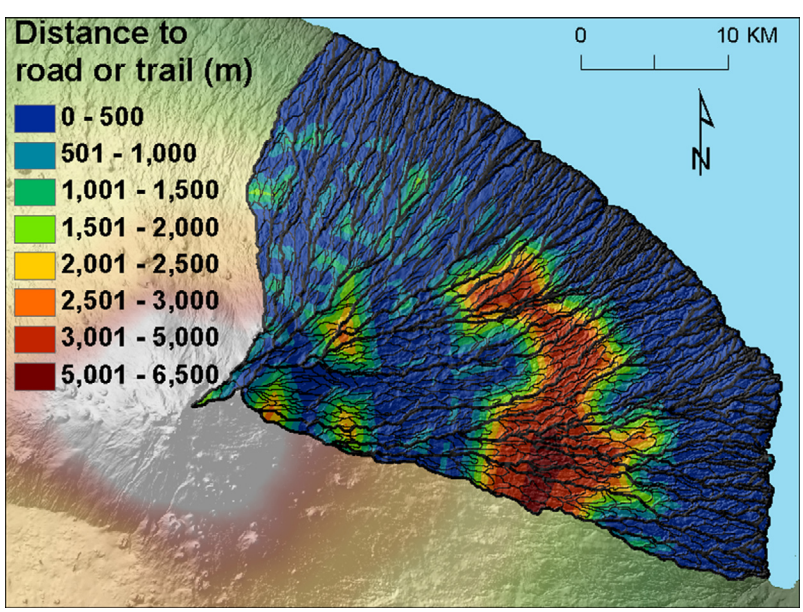

Fig. A1. Distance (m) from each pixel to road or trail.

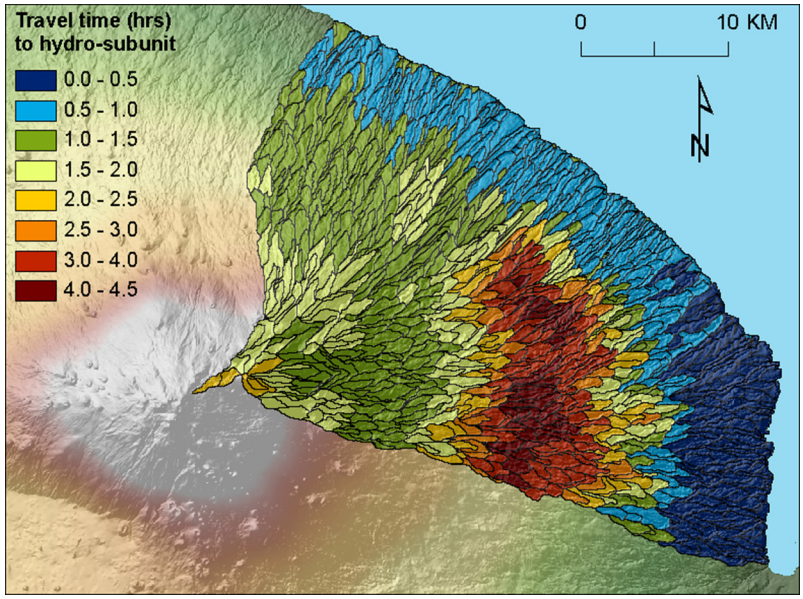

Fig. A2. Transportation time to each pixel.

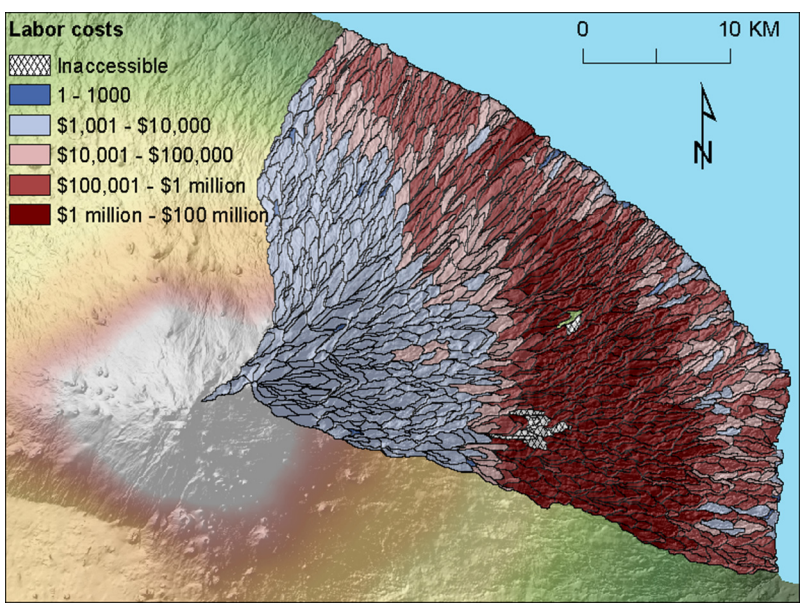

Fig. A3. Materials cost of treating each hydro-subunit.

(incorporating slope distance) and average walking speeds through each vegetation type. Walking speeds were based on the understory fraction of each vegetation type, where open grass would be at approximately 4 miles per hour (mph) on flat ground, with diminishing values for increased shrub and SG concentrations in forest environments (incorporating slope distance). Travel time to return to IPIF at the end of the day was also calculated for each pixel as twice the arrival time. Actual labor time (excluding travel time) spent on mechanical treatments was calculated using estimated single acre treatment times estimated from prior field campaigns by Mauna Kea Watershed Partnership field personnel:

- Fully Invaded: 350, 3-person crew hours/acre
- Moderately Invaded: 56, 3-person crew hours/acre
- Lightly Invaded: 5.6, 3-person crew hours/acre
- All others (sweep treatment): 0.4, 3-person crew hours/acre

These data were used in combination to estimate the: (i) number of return trips required for a three-person crew to complete the initial restoration of a hydro-subunit (assuming 10 h work days and time to actually conduct treatments); (ii) the cost of initial treatment (transport, labor, materials); and (iii) five-year maintenance treatment costs (transport, labor, materials). Item (iii) was calculated as:

$$\begin{aligned} \textit{Treatment costs} = {} & (\#\textit{trips} * \textit{vehicle costs}) + (\#\textit{trips} \\ & * \textit{labor costs}) + (\sum_{1}^{m}(\#\textit{acres SG classm} \\ & * \textit{treatment costs SG classm})) \end{aligned} \tag{2}$$

where the class value m was either fully-, moderately-, lightly-, or non-invaded by SG. Labor cost was assigned at $20.00 h^{-1} person,

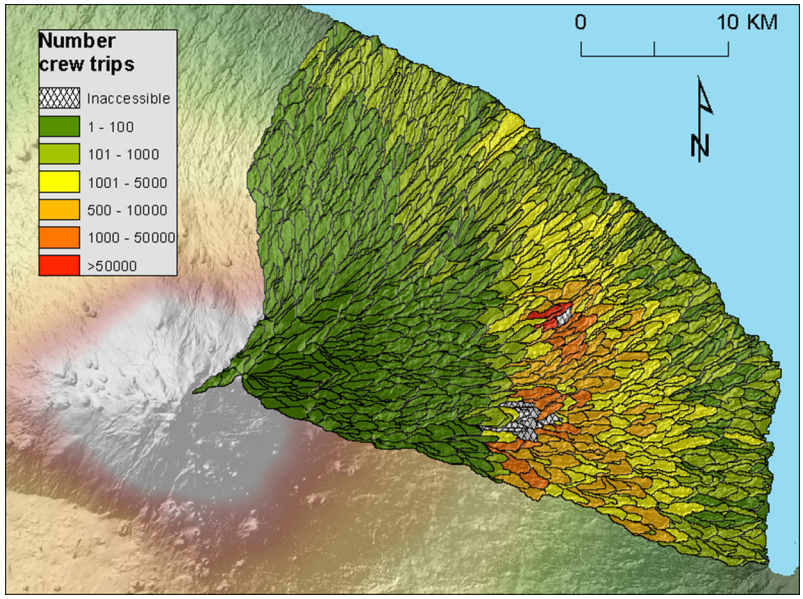

Fig. A4. Number of three-person crew trips required to treat each hydro-subunit.

and a 4 day week and 10 h work day was assumed to reduce treatment trip numbers to each unit and total cost.

Following initial treatments, maintenance was assumed to be the labor, transportation and materials costs for progressively declining SG invasion severity:

1. Stands with no/little SG incurred the cost of a sweep treatment once every five years, and as such were included only once for all pixels without SG invasion present.
2. Lightly invaded stands incurred the costs of a sweep treatment in year two.
3. Moderately invaded stands incurred the costs of a light invasion treatment in year two and a sweep treatment in year three.
4. Fully invaded stands incurred the cost of a moderate invasion treatment in year two, light invasion treatment in year three, and those of a sweep treatment in year four.

A five-year cost time series was generated to quantify the total cost of mechanical SG removal for all hydro-subunits.

Protection (fencing) costs – Fences are used limit the mobility of feral pigs into areas of special concern. Costs to implement fencing are highly variable and depend on the terrain, material, and transportation costs for crew access to shallow vs. steep terrain (ground vs. air transport; Fig. A6). We worked with managers of the three watershed partnerships on Hawai'i Island to evaluate a full range of fencing costs and the attributes that defined cost, and we assumed that fencing the widest part of each subunit would provide an approximate scalar for determining hydro-subunit fencing cost. We calculated the slope distance for each subunit at its widest point, and estimated the length of fencing required to span that distance. Total fencing cost was estimated as the average of high (contractor installed at \$93,206 km^{-1}. fenced) and low (watershed partnership installed at \$39,768 km^{-1}. fenced) cost installation. An

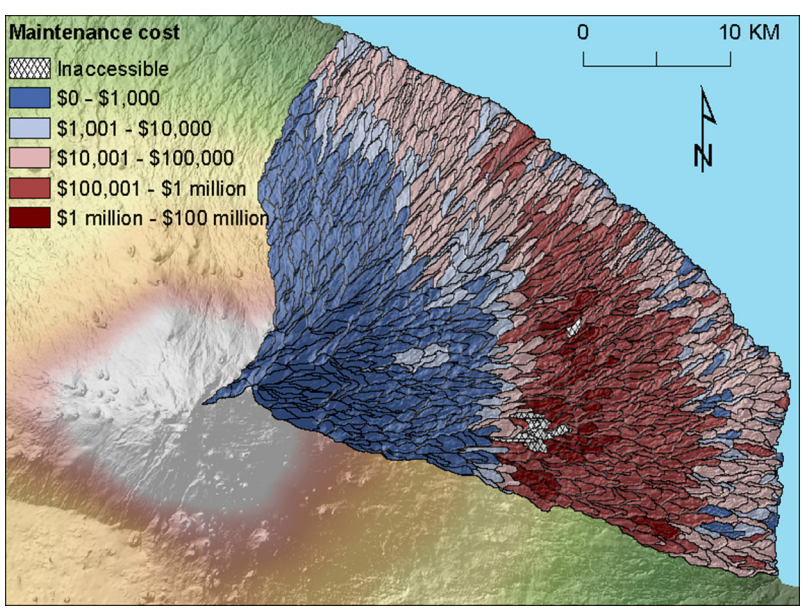

Fig. A5. Maintenance restoration costs for each hydro-subunit.

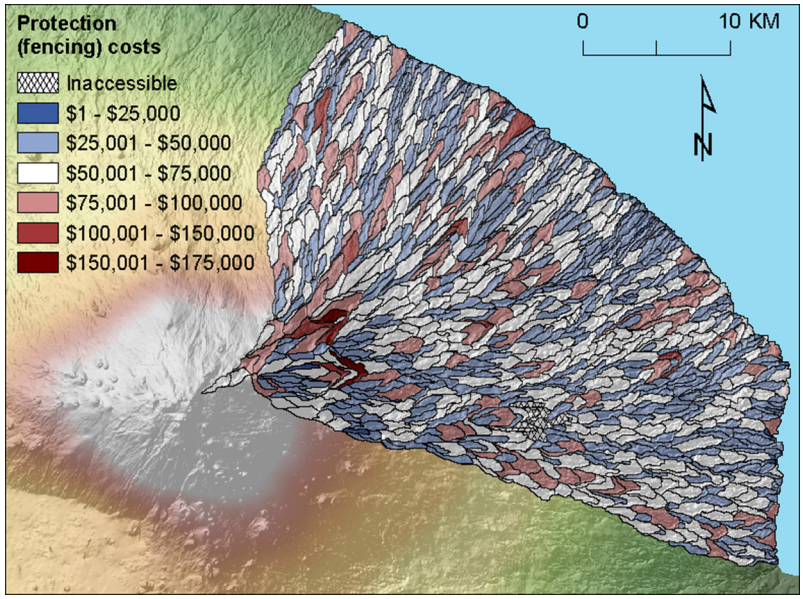

Fig. A6. Fencing (protection) costs Fig. A7. Conservation score. Scores were calculated as an area-weighted mean value assigned to each pixel within a catchment and a mean score from all adjacent catchment. Possible scores for each pixel were: 1. Private and other owned, 2. County of Hawaii, 3. State, other (not State Forest land), 4. Private Conservation land, 5. State Forest land, 6. State Conservation land, and 7. Federal land. Higher scores indicate a higher conservation value for a catchment.

average value of \$66.50 m^{-1} (\$41.32 km^{-1}.) was applied to the estimated width value to estimate the relative fencing costs of a subunit.

Land conservation status – We used property ownership maps (Tax Map Keys, TMK) to classify land ownership into seven conservation categories (Fig. A7). Categories were classified on an ordinal scale as follows, with a value of 7 indicating the highest conservation value:

1. Private and other owned	5. State Forest land
2. County of Hawaii	6. State Conservation land
3. State, other (not State Forest land)	7. Federal land
4. Private Conservation land	

The TMK maps were converted to raster format where each raster cell received a score of 1–7 based on the above classification. A zonal mean land status was computed in a GIS for each hydro-subunit, and scores were then altered based on an aggregation statistic where the mean land status score for each hydro-unit was averaged across all adjacent hydro-subunits. These two values were then averaged together to create a conservation status score.

Critical habitat – Critical habitat was identified from land surveys that identified core habitat areas for species of conservation concern (Fig. A8). Within our study area there were 1000 s of ha of USFWS designated critical habitat for the finch-like bird Palila (*Loxioides bailleui*), and several plant species including: *Clermontia pyrularia, Phyllostegia racemosa, Cyanea shipmanii, Clermontia lindseyana, Clermontia peleana, Phyllostegia warshaueri, Cyrtandra giffardii, Cyrtandra tintinnabula, Cyanea platyphylla*. Each critical

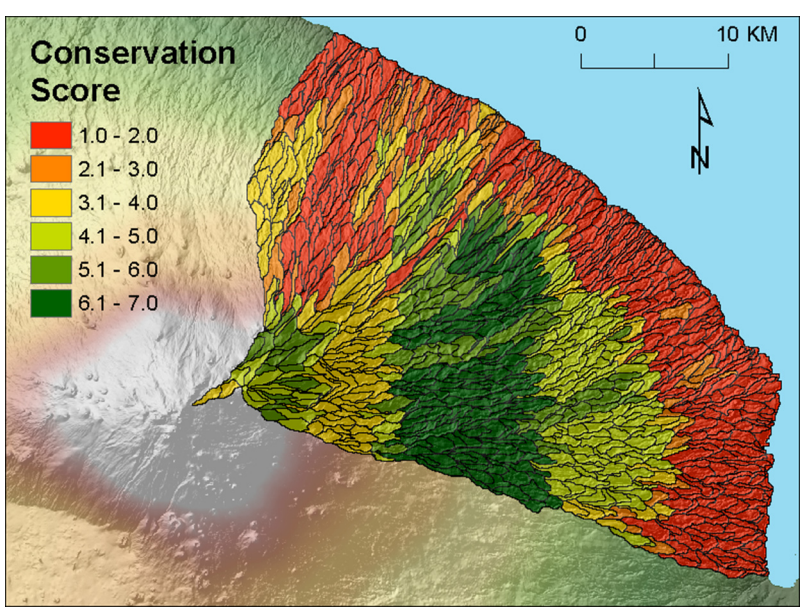

Fig. A7. Conservation score. Scores were calculated as an area-weighted mean value assigned to each pixel within a catchment and a mean score from all adjacent catchment. Possible scores for each pixel were: 1. Private and other owned, 2. County of Hawaii, 3. State, other (not State Forest land), 4. Private Conservation land, 5. State Forest land, 6. State Conservation land, and 7. Federal land. Higher scores indicate a higher conservation value for a catchment.

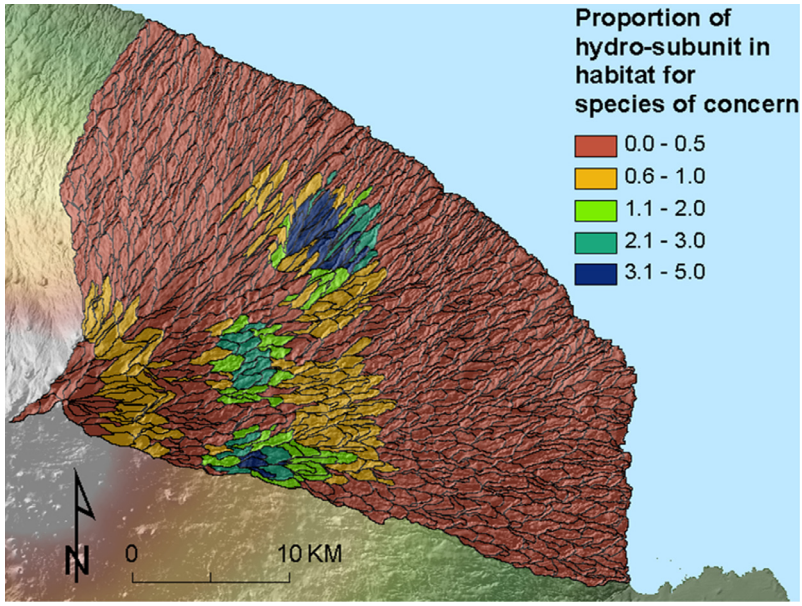

Fig. A8. Critical habitat.

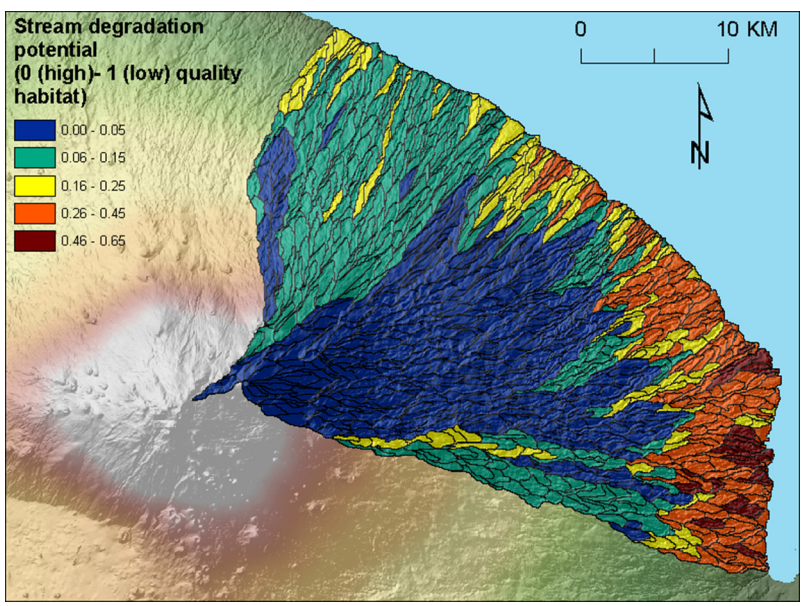

Fig. A9. Aquatic habitat quality.

habitat layer was converted to raster format, with the value of each raster corresponding to the number of species of concern found in that location. Final critical habitat scores ranged from 0 to 5, with 5 representing the maximum observed number of coincident species. Maps and associated GIS shapefiles were provided by the Mauna Kea Watershed Alliance.

Aquatic habitat quality – Stream habitat quality was estimated for each stream reach in the study area using the National Fish Habitat Partnership 2010 mapped indices of stream degradation risk (http://dx.doi.org/10.1016/j.foreco.2017.05.046), which estimate the cumulative impact of 15 different anthropogenic disturbance features that are known to degrade in-stream habitats [1] (Fig. A9). Disturbance features included: the amount of adjacent urban land use, row crop agriculture, pasture land, impervious land surfaces and densities of human populations, dams, roads and crossings, and permitted point sources of pollution and other toxic substances. We generated local catchment (node to node reaches between stream confluences) and upstream catchment (inclusive of that local reach and the entire upstream catchment of the watershed) scores for all reaches. Degradation risk was summarized for each reach, and to each hydro-subunit by means of area weighted averaging. Scores ranged from 0 to 1, with 1 indicating the poorest predicted habitat condition (high degradation potential) and 0 indicating high quality habitat (the best predicted habitat condition = no or low degradation).

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