National Park Service
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Natural Resource Stewardship and Science

Natural Resource Condition Assessment

Haleakalā National Park

Natural Resource Report NPS/HALE/NRR—2019/1977

ON THIS PAGE
View of Kīpahulu Valley from West Camp.
(© ELIZABETH URBANSKI)

ON THE COVER
A family of endangered nēnē (Hawaiian geese) and pūkiawe shrubs with Haleakalā crater in background.
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Natural Resource Report NPS/HALE/NRR—2019/1977

Paul D. Krushelnycky,[1] Charles G. Chimera,[2] Eric A. VanderWerf[3]

[1]University of Hawaiʻi at Mānoa
Dept. of Plant and Environmental Protection Sciences
3050 Maile Way, Gilmore 310
Honolulu, HI 96822

[2]Charles Chimera Consulting
PO Box 532
Honokaʻa, HI 96727

[3]Pacific Rim Conservation
3038 Oahu Avenue
Honolulu, HI 96822

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Executive Summary

Natural Resource Condition Assessments (NRCAs) evaluate current conditions for a subset of natural resources and resource indicators in national park units. Haleakalā National Park (HALE), the unit that is the focus of this NRCA, is a 33,719 acre (13,645 ha) natural area on the island of Maui, Hawaiʻi. HALE includes lands from sea level to the summit of Haleakalā Volcano at 10,023 ft (3,056 m) elevation, and protects a remarkable diversity of biomes and ecosystem types. These include, but are not limited to: coastal strand and scrub; dry, mesic and wet lowland forests; montane mesic or wet forests and bogs; subalpine grasslands and mesic or dry shrublands, which grade into alpine desert; and freshwater ecosystems.

The NRCA for HALE covers 16 focal resources that were chosen for assessment, split among three overarching categories: Life-Supporting Environment, Biological Integrity, and Landscape Condition Context. Current conditions for each of these resources, and for constituent indicators that were considered to be most relevant in characterizing the resources, were evaluated according to all available information and assigned to one of three broad categories: good, warranting moderate concern, and warranting significant concern. When sufficient temporal information was available, an increasing, stable, or declining trend in the condition was also assigned. When insufficient information was available, current conditions and/or trends were categorized as unknown.

Of the three focal resources pertaining to the Life-Supporting Environment, Water Quality was judged to be in good current condition with an unknown trend, Air Quality was judged to be good with an improving trend, and Soil Quality was judged to warrant moderate concern. Water Quality is characterized by no or few known violations of state standards in levels of total phosphorus, total nitrogen, and turbidity, but somewhat more frequent exceedances for nitrate + nitrite. Air Quality is often very good, but is periodically impacted by volcanic emissions from Kīlauea Volcano on Hawaiʻi Island, which is a natural process. Visibility showed an improving trend as volcanic emissions decreased between 2010 and 2015, but it remains to be seen how the 2018 eruption will affect that trend. Soil Quality is generally good in most regions of the park with the exception of the newer Nuʻu Parcel, where severe erosion caused by large numbers of feral ungulates has long been a problem. This condition is currently stable, but is anticipated to improve in the future as large portions of the parcel are fenced to exclude the ungulates, and stabilizing vegetation recovers.

Ten focal resources pertain to Biological Integrity, and their assessment comprised the bulk of the NRCA. Only three of these resources (Cave Ecosystems, Sub-alpine Grassland Ecosystems, and the Crater ecoregion) were judged to be in good condition. All three were assigned a stable trend. Cave Ecosystems support a number of unique cave-adapted invertebrate species, and these communities appear relatively undisturbed. Sub-alpine Grassland Ecosystems are currently minimally invaded, being characterized by native plant diversity, and especially cover, that is higher than that of non-native plants. The Crater also supports native plant diversity and cover that is substantially higher than that of non-native plants, a condition due in large part to the fencing and removal of ungulates. However, recent evidence indicates that changing climate is impacting at least some plants, like ʻāhinahina, and may be an important new influence in Crater ecosystems. A moderate fraction of shrubland arthropod communities is composed of non-native species, including the highly damaging Argentine ant and western yellowjacket. The distribution of the Argentine ant in the Crater is currently limited but expanding, and is contributing to a worsening condition of native invertebrate communities.

Three resources pertaining to Biological Integrity (Coastal Ecosystems, the Nuʻu Parcel, and T&E Species) were judged to warrant significant concern. Within Coastal Ecosystems, most vegetation communities have been heavily modified, and are dominated by non-native plant species. This habitat degradation, along with pressures from abundant invasive invertebrates, contributes to the poor condition of native coastal invertebrate communities. Shorebirds are present but not abundant, and almost no seabirds nest in coastal portions of the park, likely owing to predation by invasive mammals. Similarly, native passerines are absent from coastal ecosystems. This condition was judged to be stable (unchanging). In the relatively new Nuʻu Parcel, most vegetation communities have been heavily modified by cattle grazing and feral ungulate browsing, and are now dominated by non-native plant species. Forests and shrublands are mostly converted to open or bare ground, resulting in severe soil erosion. Most or all native bird species likely exist at low population densities, and non-native predators are currently unmanaged. The highly degraded condition of habitats in Nuʻu makes it very likely that only small remnant populations and low diversities of native arthropods remain. There was insufficient information to assess a trend in this condition.

Currently, HALE harbors 56 T&E Species. Estimation of population trends is not possible for nearly all of the 43 listed plant taxa, but anecdotal evidence suggests that at least several are declining, while possibly only one may be increasing. Most taxa persist with very small and precarious populations, and natural regeneration has been observed for less than half of the extant taxa. Only three out of seven extant listed bird species currently appear to have either stable or increasing population trends, with information on the remaining four being limited. Future climate warming may cause the conditions of listed forest bird species to decline strongly. Information on the single listed invertebrate and bat is too limited to assess current condition for either species. Temporal information was also too limited to infer a trend in the overall condition of T&E Species.

The remaining four resources in the Biological Integrity category (Freshwater Ecosystems, Forest Ecosystems, Bog Ecosystems, and Shrubland Ecosystems) were judged to warrant moderate concern, the intermediate condition category. All were assigned a stable trend. Within Freshwater Ecosystems, all five species of ʻoʻopu (native gobioid fish) are present and apparently stable in the park, with populations in several streams. However, native freshwater molluscs and shrimps appear to have undergone dramatic population declines in recent decades, with this situation currently unchanging. Aquatic insects appear to be in relatively good condition, although the information supporting this inference is limited.

The condition of Forest Ecosystems differs sharply between low and high elevation zones, separated roughly by the 4,000 ft elevation level. Native plant diversity and cover has remained high, and non-native plant invasion has been comparatively low, in high-elevation forests. Similarly, the six extant forest bird species have exhibited stable population trends. Low elevation forests, in contrast, are much more heavily invaded by non-native plants, and support lower diversity and densities of forest birds. Similar patterns may apply to native invertebrate communities, but information is too limited to be sure. Like other regions of the park, Forest Ecosystems have benefitted from fencing and control of ungulates, but incursions of pigs into wet forests have increased recently. Bog Ecosystems also differ in condition between upper and lower elevation regions. Some upper elevation bogs have increased in richness and cover following pig exclusion, but others remain moderately invaded. Lower elevation bogs are heavily modified and are unlikely to increase in native species richness or diversity.

Within Shrubland Ecosystems, both on the West Slope and in Kaupō, native plant diversity and cover is substantially higher than that of non-native plants, especially in the canopy layer. This good condition appears to have improved since the fencing and removal of ungulates. However, shrubland arthropod communities are heavily invaded, and are especially threatened by several highly damaging species of invasive ants and wasps. The poor condition of arthropod communities appears to be worsening, as the number of established non-native arthropod species continues to grow, and the invasive Argentine ant continues to spread.

Three focal resources (Fuel and Fire Dynamics, Soundscape, and Viewscape) fell within the Landscape Condition Context category. Fuel and Fire Dynamics were judged to warrant moderate concern, with an unknown trend. General risk of wildfire is low, but regions with extensive areas of unnatural fine fuel loads, especially fire-prone non-native grasses, exist, especially in the Kaupō and Nuʻu regions. Fire frequency and size has likely increased substantially from the natural historic fire regime in most of HALE, but area burned has been very small in recent decades, suggesting that fire management strategies have been effective. The Soundscape was also judged to warrant moderate concern, with an unknown trend. Existing ambient sound levels in wilderness areas are unusually low, especially during quiet conditions. However, noise from helicopters, mainly commercial air tours, are a frequent disruption to natural sound levels. The HALE Viewscape, in particular the quality of dark night skies, was judged to currently be in good condition. Light pollution in the form of sky glow from Maui's urban environment is low at the summit and west rim of the Crater, and is presumably even lower in backcountry wilderness areas. Point sources of light pollution are also minimal in wilderness areas. The trend in this condition was unknown.

Overall, six of the 16 focal resources (38%) were judged to be in good condition, seven (44%) were judged to warrant moderate concern, and three (19%) were judged to warrant significant concern. Collectively, this assessment suggests that natural resources at HALE are under considerable strain. The geographic resources that are most degraded, namely Coastal Ecosystems and the Nuʻu Parcel, experienced wholesale conversion many decades to centuries ago. Some areas within these have transitioned into communities dominated by non-native species, while others are now recovering from former agricultural use. For substantial portions of both types of areas, it is unlikely that they will return to a condition in which native species dominate, at least given current management capabilities and levels of investment. Yet, certain native species can persist within them, giving them conservation value, and other portions of these ecosystems are somewhat less degraded and may passively recover after the removal of feral ungulates.

Most resource management efforts are currently targeted at the remaining focal resources related to biological integrity, to prevent those warranting moderate concern to degrade further, and to protect those still in good condition from incipient threats. Across HALE, the primary force impacting biological resources is the collective pressures exerted by non-native species. As a consequence, conserving these resources continues to be most effectively achieved by attacking new invasive species, such as incipient weeds, and maintaining adequate control of persistent invasive species, such as feral ungulates and small mammals. Addressing the impacts of non-native invertebrates remains one of the most difficult challenges. Other major resource management challenges include the potential arrival of devastating forest pathogens like Rapid ʻŌhiʻa Death (*Ceratocystis* spp.), and understanding how to most effectively deal with climate change.

Confidence for most of the resource assessments was not high, largely as a result of insufficient information. A substantial amount of the available information is qualitative in nature, and many information gaps exist, which is not surprising for such a large and complex natural area as HALE. Aside from an absence of information in certain areas, the lack of repeated monitoring data for all but a few resources made it difficult to assess trends in many cases. The relatively new NPS I&M program is an important step in rectifying this situation, and will undoubtedly make future resource assessments more robust.

Despite its resource management challenges, HALE remains one of the most valuable natural areas in Hawaiʻi, protecting a wide diversity of ecosystems and a large number of threatened and endangered species. This circumstance is in large part owed to its inclusion within the National Park system, and the concerted management efforts that were subsequently undertaken. Major past achievements, such as the fencing and exclusion of feral ungulates from the majority of the park, provide hope that current and future threats may yet be mitigated with sufficient determination, creativity, and collaborative effort.

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Acronyms

EWM: Endangered Wildlife Management unit of the HALE Resources Management division.

FAM: Feral Animal Management unit of the Resources Management division of HALE

FR: Forest Reserve

HALE: Haleakalā National Park

I&M: Inventory and Monitoring Program

NAR: Natural Area Reserve

NPS: National Park Service

NRCA: Natural Resource Condition Assessment

RM: Resources Management division of HALE

RMP: Resources Management Plan

SotP: State of the Park

T&E: Threatened and Endangered

TWI: Trade wind inversion

VM: Vegetation Management unit of the Resources Management division

Chapter 1. NRCA Background Information

Natural Resource Condition Assessments (NRCAs) evaluate current conditions for a subset of natural resources and resource indicators in national park units, hereafter "parks." NRCAs also report on trends in resource condition (when possible), identify critical data gaps, and characterize a general level of confidence for study findings. The resources and indicators emphasized in a given project depend on the park's resource setting, status of resource stewardship planning and science in identifying high-priority indicators, and availability of data and expertise to assess current conditions for a variety of potential study resources and indicators.

NRCAs Strive to Provide…

- *Credible condition reporting for a subset of important park natural resources and indicators*
- *Useful condition summaries by broader resource categories or topics, and by park areas*

NRCAs represent a relatively new approach to assessing and reporting on park resource conditions. They are meant to complement, not replace, traditional issue-and threat-based resource assessments. As distinguishing characteristics, all NRCAs

- Are multi-disciplinary in scope;[1]
- Employ hierarchical indicator frameworks;[2]
- Identify or develop reference conditions/values for comparison against current conditions;[3]
- Emphasize spatial evaluation of conditions and Geographic Information System (GIS) products;[4]
- Summarize key findings by park areas;[5] and
- Follow national NRCA guidelines and standards for study design and reporting products.

Although the primary objective of NRCAs is to report on current conditions relative to logical forms of reference conditions and values, NRCAs also report on trends, when appropriate (i.e., when the underlying data and methods support such reporting), as well as influences on resource conditions. These influences may include past activities or conditions that provide a helpful context for

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[1] The breadth of natural resources and number/type of indicators evaluated will vary by park.

[2] Frameworks help guide a multi-disciplinary selection of indicators and subsequent "roll up" and reporting of data for measures ⇨ conditions for indicators ⇨ condition summaries by broader topics and park areas

[3] NRCAs must consider ecologically-based reference conditions, must also consider applicable legal and regulatory standards, and can consider other management-specified condition objectives or targets; each study indicator can be evaluated against one or more types of logical reference conditions. Reference values can be expressed in qualitative to quantitative terms, as a single value or range of values; they represent desirable resource conditions or, alternatively, condition states that we wish to avoid or that require a follow-up response (e.g., ecological thresholds or management "triggers").

[4] As possible and appropriate, NRCAs describe condition gradients or differences across a park for important natural resources and study indicators through a set of GIS coverages and map products.

[5] In addition to reporting on indicator-level conditions, investigators are asked to take a bigger picture (more holistic) view and summarize overall findings and provide suggestions to managers on an area-by-area basis: 1) by park ecosystem/habitat types or watersheds, and 2) for other park areas as requested.

understanding current conditions, and/or present-day threats and stressors that are best interpreted at park, watershed, or landscape scales (though NRCAs do not report on condition status for land areas and natural resources beyond park boundaries). Intensive cause-and-effect analyses of threats and stressors, and development of detailed treatment options, are outside the scope of NRCAs.

Due to their modest funding, relatively quick timeframe for completion, and reliance on existing data and information, NRCAs are not intended to be exhaustive. Their methodology typically involves an informal synthesis of scientific data and information from multiple and diverse sources. Level of rigor and statistical repeatability will vary by resource or indicator, reflecting differences in existing data and knowledge bases across the varied study components.

The credibility of NRCA results is derived from the data, methods, and reference values used in the project work, which are designed to be appropriate for the stated purpose of the project, as well as adequately documented. For each study indicator for which current condition or trend is reported, we will identify critical data gaps and describe the level of confidence in at least qualitative terms. Involvement of park staff and National Park Service (NPS) subject-matter experts at critical points during the project timeline is also important. These staff will be asked to assist with the selection of study indicators; recommend data sets, methods, and reference conditions and values; and help provide a multi-disciplinary review of draft study findings and products.

NRCAs can yield new insights about current park resource conditions, but, in many cases, their greatest value may be the development of useful documentation regarding known or suspected resource conditions within parks. Reporting products can help park managers as they think about near-term workload priorities, frame data and study needs for important park resources, and communicate messages about current park resource conditions to various audiences. A successful NRCA delivers science-based information that is both credible and has practical uses for a variety of park decision making, planning, and partnership activities.

Important NRCA Success Factors

- *Obtaining good input from park staff and other NPS subject-matter experts at critical points in the project timeline*
- *Using study frameworks that accommodate meaningful condition reporting at multiple levels (measures ⇨ indicators ⇨ broader resource topics and park areas)*
- *Building credibility by clearly documenting the data and methods used, critical data gaps, and level of confidence for indicator-level condition findings*

However, it is important to note that NRCAs do not establish management targets for study indicators. That process must occur through park planning and management activities. What an NRCA can do is deliver science-based information that will assist park managers in their ongoing, long-term efforts to describe and quantify a park’s desired resource conditions and management targets. In the near term, NRCA findings assist strategic park resource planning[6] and help parks to report on government accountability measures.[7] In addition, although in-depth analysis of the effects of climate change on park natural resources is outside the scope of NRCAs, the condition analyses and data sets developed for NRCAs will be useful for park-level climate-change studies and planning efforts.

NRCAs also provide a useful complement to rigorous NPS science support programs, such as the NPS Natural Resources Inventory & Monitoring (I&M) Program.[8] For example, NRCAs can provide current condition estimates and help establish reference conditions, or baseline values, for some of a park's vital signs monitoring indicators. They can also draw upon non-NPS data to help evaluate current conditions for those same vital signs. In some cases, I&M data sets are incorporated into NRCA analyses and reporting products.

NRCA Reporting Products…

Provide a credible, snapshot-in-time evaluation for a subset of important park natural resources and indicators, to help park managers:

- *Direct limited staff and funding resources to park areas and natural resources that represent high need and/or high opportunity situations* ***(near-term operational planning and management)***
- *Improve understanding and quantification for desired conditions for the park's "fundamental" and "other important" natural resources and values* ***(longer-term strategic planning)***
- *Communicate succinct messages regarding current resource conditions to government program managers, to Congress, and to the general public* ***("resource condition status" reporting)***

Over the next several years, the NPS plans to fund an NRCA project for each of the approximately 270 parks served by the NPS I&M Program. For more information visit the NRCA Program website.

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[6] An NRCA can be useful during the development of a park's Resource Stewardship Strategy (RSS) and can also be tailored to act as a post-RSS project.

[7] While accountability reporting measures are subject to change, the spatial and reference-based condition data provided by NRCAs will be useful for most forms of "resource condition status" reporting as may be required by the NPS, the Department of the Interior, or the Office of Management and Budget.

[8] The I&M program consists of 32 networks nationwide that are implementing "vital signs" monitoring in order to assess the condition of park ecosystems and develop a stronger scientific basis for stewardship and management of natural resources across the National Park System. "Vital signs" are a subset of physical, chemical, and biological elements and processes of park ecosystems that are selected to represent the overall health or condition of park resources, known or hypothesized effects of stressors, or elements that have important human values.

Chapter 2. Introduction and Resource Setting

2.1. Introduction

2.1.1. Enabling Legislation and Historical Overview

Lands encompassing the crater and summit area of Haleakalā volcano were first placed under federal protection on August 1, 1916, with the establishment of Hawaiʻi National Park in the U.S. Territory of Hawaiʻi (39 Stat. 432). Hawaiʻi National Park originally included tracts on Kīlauea and Mauna Loa volcanoes on Hawaiʻi Island, in addition to approximately 21,150 ac (8,563 ha) on Haleakalā volcano on the island of Maui, which were "perpetually dedicated and set apart as a public park or pleasure ground for the benefit and enjoyment of the people of the United States" (16 U.S.C. section 391). Initial motivations for establishing a national park in Hawaii focused primarily on the well-known geological resources at Kīlauea, and the spectacular but less-accessible Haleakalā section was added only in the last version of the bill that created the park (Jackson 1972, Nakamura 2010).

The remote nature of the Haleakalā section of Hawaiʻi National Park, along with legislative restrictions that tied additional federal support to land accessibility, impeded the progress of park improvements in the early years following establishment. The designated lands were not formally deeded to the federal government until 1928, and NPS presence at the Haleakalā section was minimal during this period, being limited to occasional staff visits from the headquarters at Kīlauea (Jackson 1972). However, the construction of a road to the Haleakalā summit during 1933-35, resulting from advocacy by both Maui residents and Hawaiʻi National Park leadership, greatly increased access to the Haleakalā section and spurred the development of additional infrastructure and the advancement of the park's mission. Visitation rates to the summit jumped immediately by several orders of magnitude to over 16,000 in 1936, new structures at the 7000 ft (2134 m) elevation level and a visitor center near the summit were built from 1935-36, and the first district ranger was stationed at Haleakalā in 1935 (Jackson 1972, NPS 2016).

The Kīlauea and Haleakalā parcels on the two islands remained under combined jurisdiction until July 1, 1961, when Hawaiʻi National Park was split into two: Hawaiʻi Volcanoes National Park on Hawaiʻi Island (HAVO), and Haleakalā National Park (HALE) on Maui (Public Law 86-744). HALE continued to undergo subsequent changes, the first of which was the acquisition of portions of Kīpahulu Valley on the eastern border of the existing park boundary in 1969 (NPS 2016). The remainder of Kīpahulu Valley, portions of upper Hāna Rainforest, and Manawainui and Kaumakani Planezes were added in 1974 (NPS 1999). In 1976, 24,770 ac (10,028 ha) of HALE were recognized as designated and potential wilderness (Public Law No. 94-567), with all but 51 of the 5,500 ac (2,247 ha) of potential wilderness converted to designated wilderness in 2002 (Federal Register 02-3563). Additional land acquisitions include the Puhilele section in 1997, Kaʻāpahu section in 1999, Nuʻu section in 2008, and an un-named parcel adjoining Kaʻāpahu in 2016 (NPS 1999, NPS 2016). HALE, along with HAVO, was designated an International Biosphere Reserve in 1980 (UNESCO 2016).

The rationale for establishment of the park is set forth in HALE's purpose statement:

> *For the inspiration of current and future generations, Haleakalā National Park protects a wild volcanic landscape with a wide array of fragile and diverse native ecosystems, including plant and animal species found nowhere else on Earth. Our stewardship perpetuates the unique and continuing connections between Hawaiian culture and this sacred and evolving land.* (NPS 2015)

As further guidance, HALE has developed the following current mission statement:

Haleakalā National Park is an International Biosphere Reserve that supports native ecosystems in a maturing volcanic landscape. As stewards of this park, we will incorporate Native Hawaiian protocols and generational knowledge for the perpetuation of cultural resources, and implement traditional and modern methods for the preservation of natural resources. With the community and other partnerships, we will protect, manage, and interpret these unique resources for the education, experience and inspiration of all peoples and future generations. (NPS 2016)

The explicit recognition that Hawaiian culture and natural resource protection are inter-related is explained in the park's Foundation Document:

> *Haleakalā National Park is a sacred place to kānaka maoli (Native Hawaiians) and is fundamentally linked to their traditional and contemporary beliefs, practices, and way of life. The concept of kuleana (responsibility) is central to these beliefs, passed on from the kupuna (ancestors) to future generations to ensure stewardship and respect for all things spiritual and physical. Closely connected to kuleana is the concept of mālama ʻāina, caring for and nurturing the land so it continues to provide the essential means and resources necessary to sustain life for present and future generations.* (NPS 2015)2.1.2 Geographic Setting

HALE is located on East Maui, which is comprised of Haleakalā volcano, the larger of the two volcanic mountains that form Maui Island (Figure 2.1.1-1). Maui is the second youngest island in the Hawaiian archipelago, with East Maui being approximately 1.2 million years old (Clague 1996), older only than Hawaiʻi Island lying to the southeast (Figure 2.1.1-1). Haleakalā is the third highest volcano in the island chain. HALE occupies much of the higher elevation portions of Haleakalā, including the summit at 10,023 ft (3,056 m), as well as several valleys and tracts of land that stretch down to and include the coast at Kīpahulu, Kaʻāpahu, and Nuʻu. Total current acreage for the park is 33,719 ac (13,645 ha).

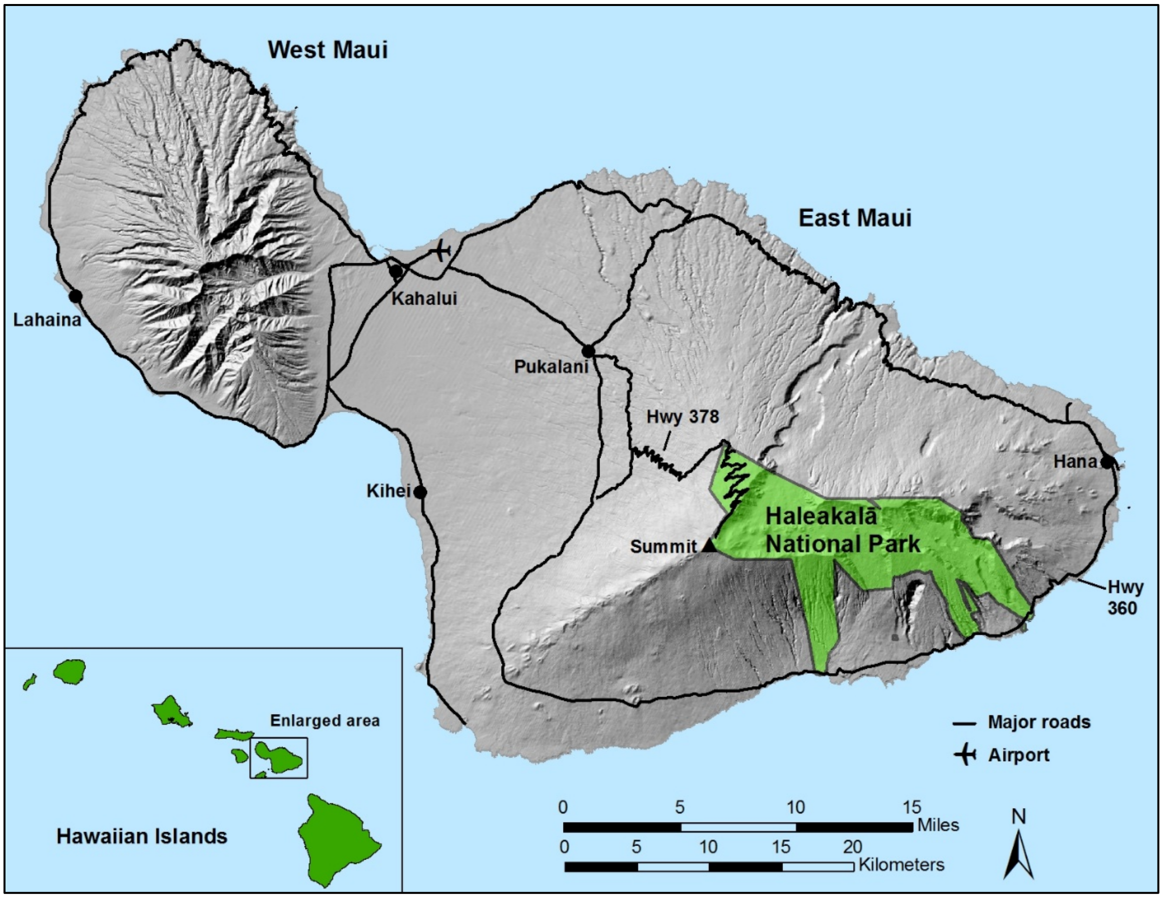

Figure 2.1.1-1. Location of HALE within Maui and the Hawaiian Island chain. Shown are major towns and cities on Maui, and the two main access roads to the park (Hwy 378 and Hwy 360). Source: NPS.

HALE has two administrative sections, the Summit District and Kīpahulu District (NPS 1999). The Summit District includes the large central "crater" at the top of Haleakalā volcano, as well as the upper western, northern and southern rims and outer slopes of the crater (Figure 2.1.1-2). Haleakalā's crater is not a true volcanic crater, but is instead a large depression formed from the joining of two massive erosional valleys that cut into the northern and southern flanks of the shield volcano, and whose amphitheater-headed upper sections eventually met in the central highlands (NPS 1999). Much of the two valleys were subsequently filled by renewed, secondary volcanic activity, with these younger lava flows, cinder fields and cinder cones today forming the floor of the depression (<50,000 years old, Sherrod et al. 2006). Despite the well-known geological inaccuracy of the term, "crater" is used pervasively as a shorthand description of this feature both in colloquial use and in park and scientific literature, and we therefore also use it throughout this assessment. The sides of the ancient valleys described above form the two gaps in the walls of the crater that are also part of the Summit District: Koʻolau Gap in the northwest and Kaupō Gap in the southeast. The administrative center of both the Summit District and the entire park is the Park Headquarters, located on the west slope of the mountain at 7,000 ft (2,134 m) elevation along the road to the summit (Figure 2.1.1-2). Three cabins, three campgrounds, and approximately 32 miles (51.5 km) of trails are available for visitor use in the Summit District.

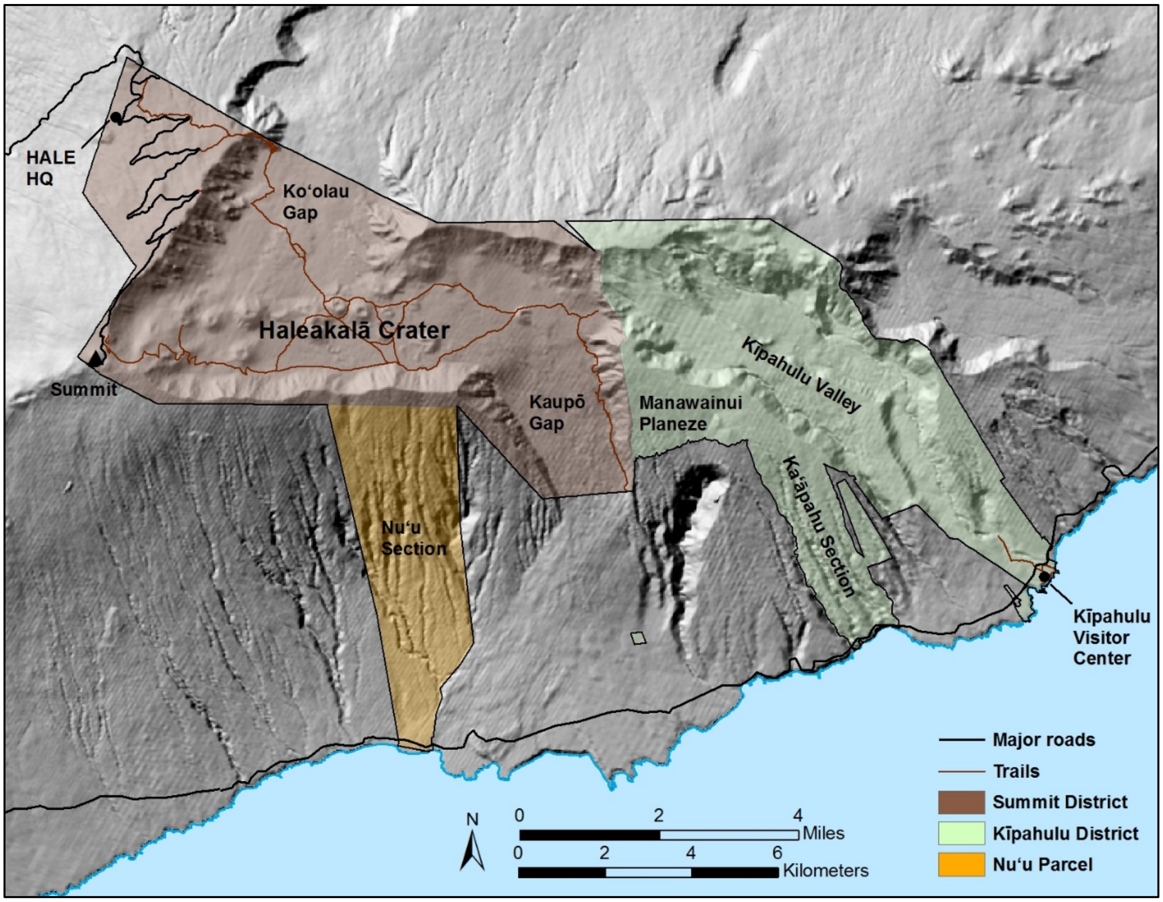

Figure 2.1.1-2. Major geographical areas of HALE. Also depicted are the two park districts (plus the newer Nu'u Parcel) and their administrative centers: HALE Headquarters (HQ) and offices at the Kīpahulu Visitor Center. Note that district boundaries are approximate, as these are not precisely delineated. Source: NPS.

The Kīpahulu District is comprised of Kīpahulu Valley, which extends from the eastern terminus of the crater down to the coast near the settlement of Kīpahulu, together with nearby park lands. The upper valley is a closed biological reserve that is off limits to the public, while the lower valley and coastal area contains a smaller administrative center (Figure 2.1.1-2), several trails (totaling ~4 miles (6.4 km)), and a campground. Adjoining Kīpahulu Valley to the south is the Manawainui Planeze; the Ka'āpahu section, which reaches the coast below Manawainui but has no park facilities or infrastructure; and tracts of lowland and montane forest and bogs north of Kīpahulu Valley. All of these latter sections are also closed to the public, as is the relatively recently acquired parcel, which has not yet been administratively attached to either district (Figure 2.1.1-2).

HALE has two primary points of access (Figure 2.1.1-1). The Summit District is accessible via Highway 378 (Crater Road), which splits off Highway 377, originating in the town of Pukalani. Beyond the park entrance near Headquarters, the road continues to the summit. Kīpahulu District is transected near the coast by Highway 360 (Hana Highway), with Hana being the closest town. Haleakalā Crater can also be accessed via Kaupō Trail, which enters the park in Kaupō Gap. Most lands surrounding HALE have conservation designation, and are managed by a variety of public and private landowners, and include State Forests, State Natural Area Reserves, Hawaiian Homelands, and The Nature Conservancy's Waikamoi Preserve (Figure 2.1.1-3). The remainder of lands adjacent to the park are agricultural, used primarily for ranching. These non-urban buffer zones contribute greatly to the ecological integrity of the park, and to its remote and natural aesthetic. One exception concerns the continued development of the cluster of astronomical facilities ("Science City") immediately adjacent to the park boundary near the summit, despite objections from NPS (NPS 1999). These observatories and related buildings are perceived by some to degrade cultural resources and diminish the summit viewshed, and their construction represents a potential pathway for invasive species introduction to sensitive high-elevation ecosystems.

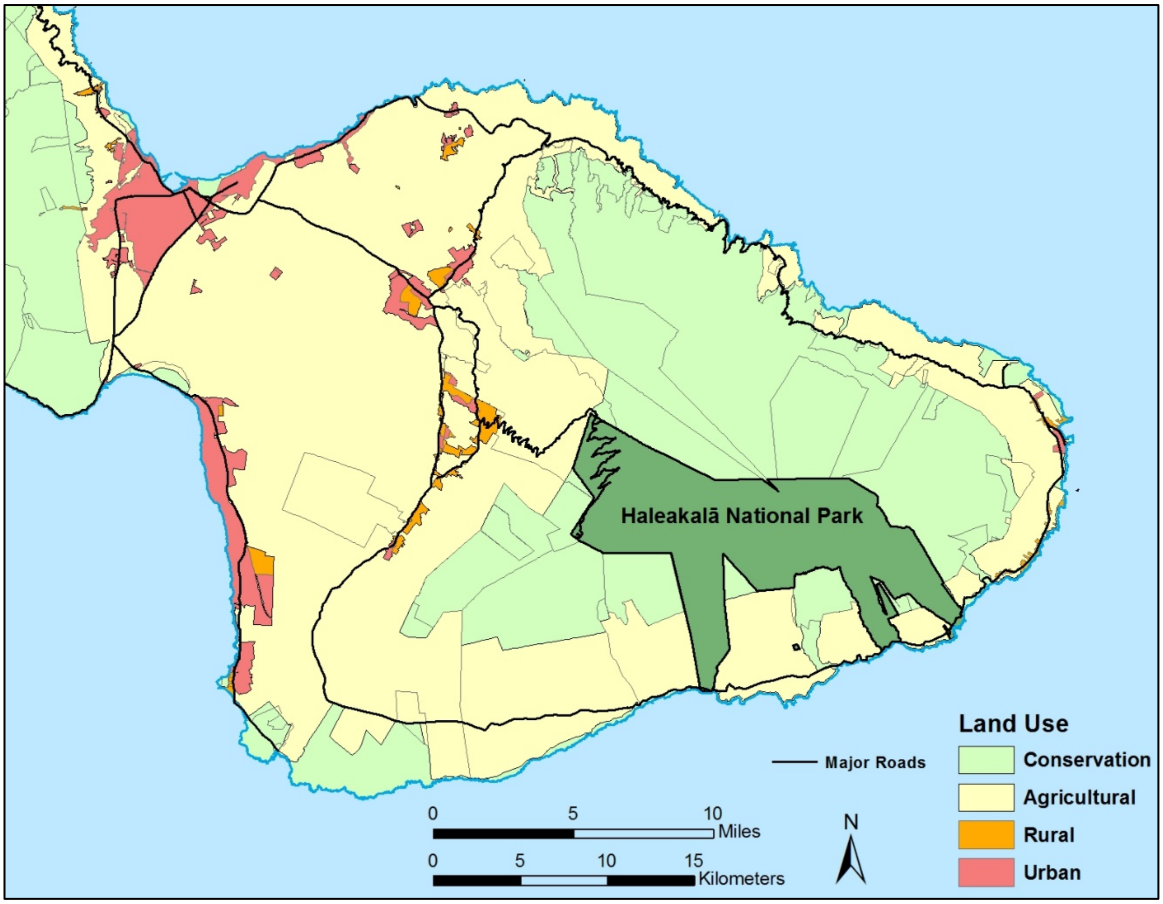

Figure 2.1.1-3. Land use designations of lands surrounding HALE. Divisions within land use categories indicate boundaries of land ownership and stewardship. Source: Hawai'i Office of State Planning.

2.1.2. Visitation Statistics

Visitation to HALE increased steadily from the 1960's to the 1990's, after which is has generally fluctuated between 1.0 and 1.6 million visitors per year (NPS 2017a; Figure 2.1.2-1).

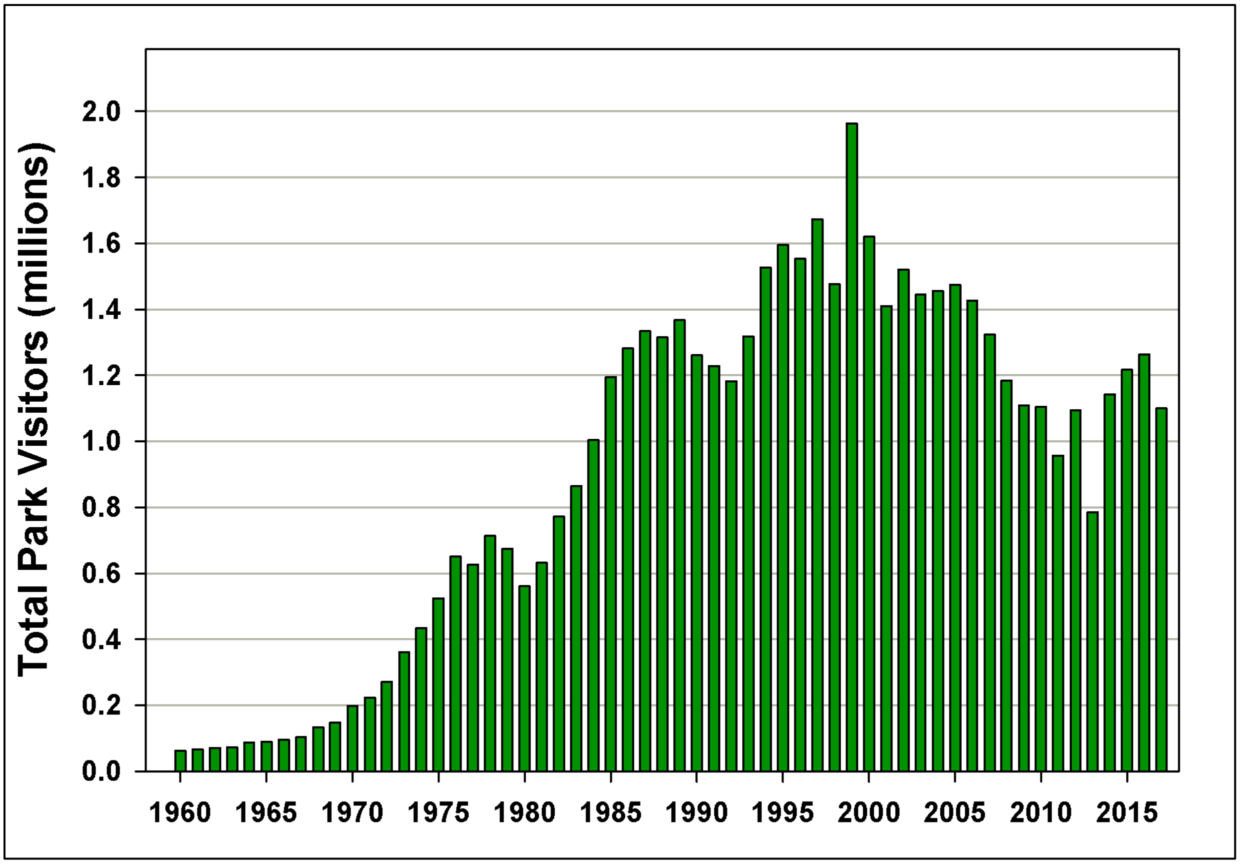

Figure 2.1.2-1. Total number of visitors at HALE between 1960 and 2016. Source: NPS (2017a).

Over the past few decades, visitation has typically been higher in the Summit District than the Kīpahulu District (Figure 2.1.2-2). Summit visitation has generally varied between 600,000 and one million people per year, while Kīpahulu visitation has typically ranged between 300,000 and 600,000 people per year.

Most of this visitation is restricted to a few small areas of the park (NPS 1999), including Headquarters, the visitor centers at the summit and Kīpahulu, and several overlooks on the western rim of the crater. Trail use is generally heaviest along the initial section of Keonehe‘ehe‘e (Sliding Sands Trail) originating near the summit, and along the trails around ‘Ohe‘o Gulch. Backcountry trail use is substantially lower; for example, there has consistently been fewer than 5,000 backcountry campers per year from 1979 through 2016 (NPS 2017a), although numbers of day-hikers are higher but are not tracked. HALE prohibits off-trail hiking throughout the park to protect numerous endangered species and sensitive ecosystems (NPS 1999).

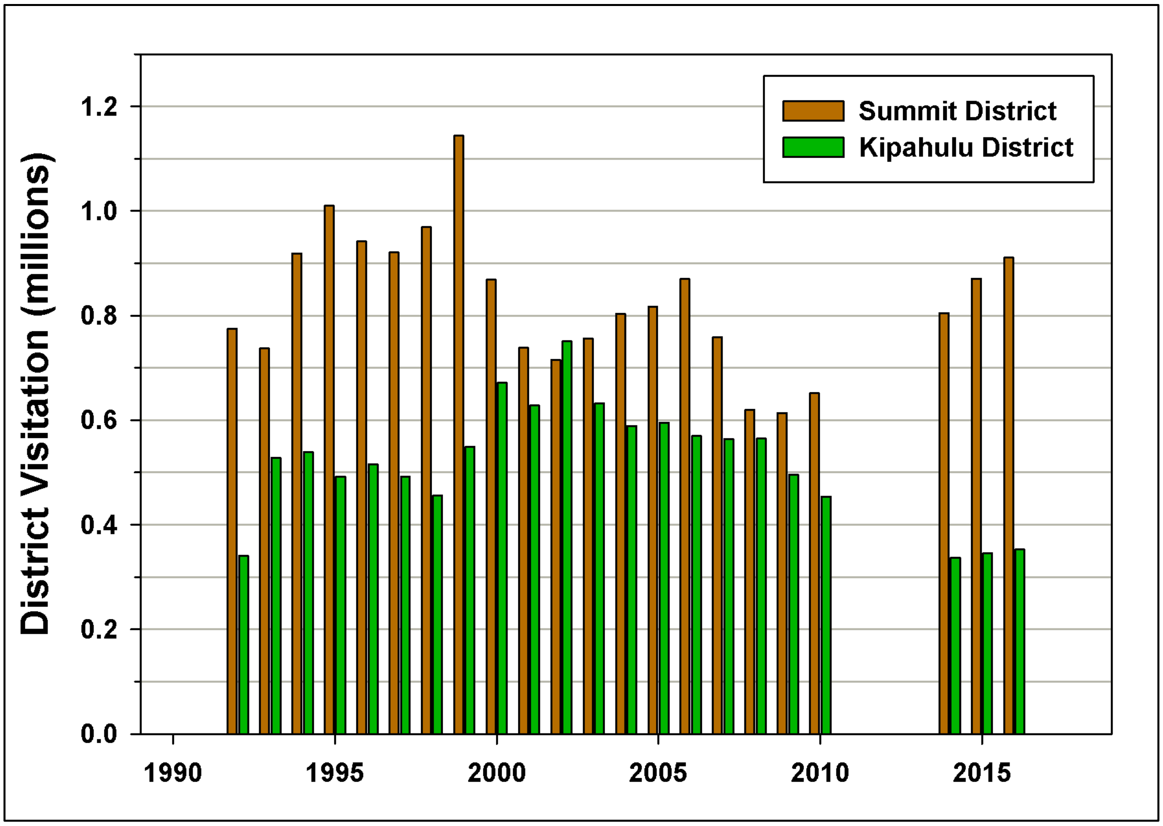

Figure 2.1.2-2. Visitation to the two districts in HALE between 1992 and 2016. Source: NPS (2017a).

2.2. Natural Resources

2.2.1. Ecological Units and Watersheds

The wide range of climatic conditions prevailing across Haleakalā volcano results in a remarkable diversity of biomes and ecosystem types within a relatively small geographic area. Climate in Hawai‘i is governed by two main forces: orographic rainfall generated from prevailing northeasterly trade winds, and the trade wind temperature inversion (TWI) that influences vertical development of weather (Giambelluca and Nullet 1991, Loope and Giambelluca 1998). Northeasterly trade winds forced up the slopes of Haleakalā produce persistent clouds on the windward faces, with this cloud layer bounded below by the lifting condensation level (~1,312-3,280 ft (400-1,000 m) elevation) and above by the TWI base height (~7,082 ft (2,159 m) elevation on average) (Longman et al. 2015b). This creates wet to very wet conditions on northeast, windward slopes of the mountain, and much drier conditions on the southwest, leeward side of the mountain (Giambelluca et al. 2013). Above the TWI, atmospheric conditions are generally clear and dry (Longman et al. 2015b). Although alternate weather patterns periodically bring precipitation to leeward slopes and high elevation areas normally situated above the TWI, the dynamics summarized above are typical. The resultant rainfall gradient across the mountain is extremely steep, with rainfall in the park ranging from less than 20 inches (508 mm) per year to over 400 inches (10,160 mm) per year (Figure 2.2.1-1). Owing to its low latitudinal, subtropical location, Hawai‘i lacks the strong seasonality of temperate areas, but experiences two main annual phases: a wet season, approximately from November through April, and a dry season, approximately from May through October (Longman et al. 2015b). These strongly varying precipitation patterns interact with the substantial temperature gradient spanning the 10,000 ft (3,050 m) elevation range in the park to produce a large variety of micro-climate zones.

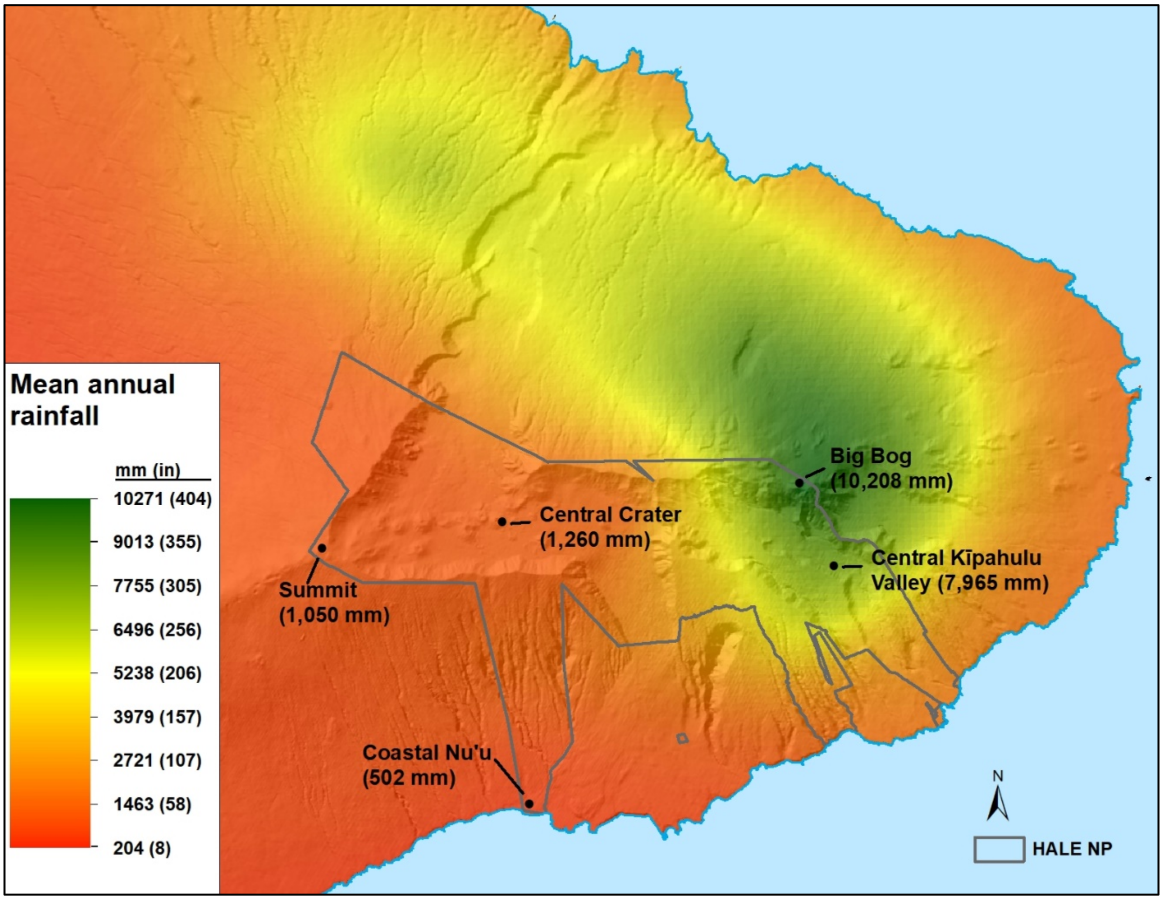

Figure 2.2.1-1. Rainfall patterns on Haleakalā volcano. Mean annual rainfall in HALE is estimated to range from just over 500 mm (19.7 in) at the Nu‘u coastline to over 10,200 mm (402 in) in the windward-facing montane bogs. Source: Giambelluca et al. 2013.

Major biomes supported across these micro-climate zones are characterized by all combinations of dry, mesic and wet moisture regimes with grassland, shrubland and forest vegetation formations (Jacobi et al. 2017; Figure 2.2.1-2). Ecosystems and habitat types within HALE can be further subdivided by coastal, lowland, montane and subalpine/alpine elevation zones, and by vegetation structure. Ecosystem types include: coastal strand and scrub; dry, mesic and wet lowland forests; montane mesic or wet forests, bogs, and grasslands; and subalpine mesic or dry shrublands, which grade into alpine desert areas supporting very sparse vegetation near the summit and on very young, porous soils above the TWI. Several classification systems of vegetation community types within each of these ecosystems have been developed, largely based on combinations of biome, physiognomy and dominant plant taxa. These include both archipelago-wide classification efforts (e.g. Gagné and Cuddihy 1999, Jacobi et al. 2017) and those specific to smaller regions in and around HALE (Whiteaker 1980). This assessment will primarily use the vegetation community classification system developed for HALE by the NPS I&M Program (Green et al. 2015; Figure 2.2.1-3), because it is recent, covers the entire park, and will be most relevant to park managers and future natural resource assessments at HALE.

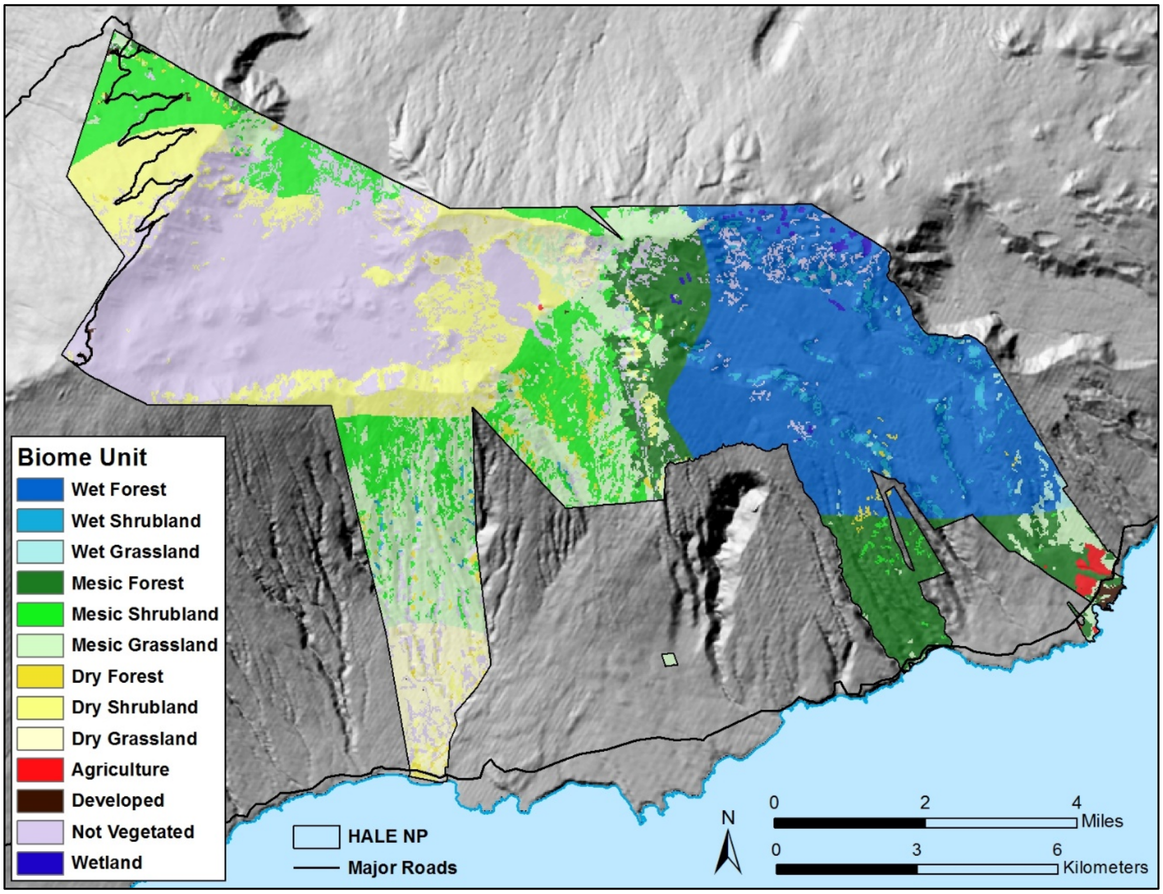

Figure 2.2.1-2. Major biome types in HALE. Areas categorized as Not Vegetated support less than 5% cover of vegetation. Source: Jacobi et al. (2017).

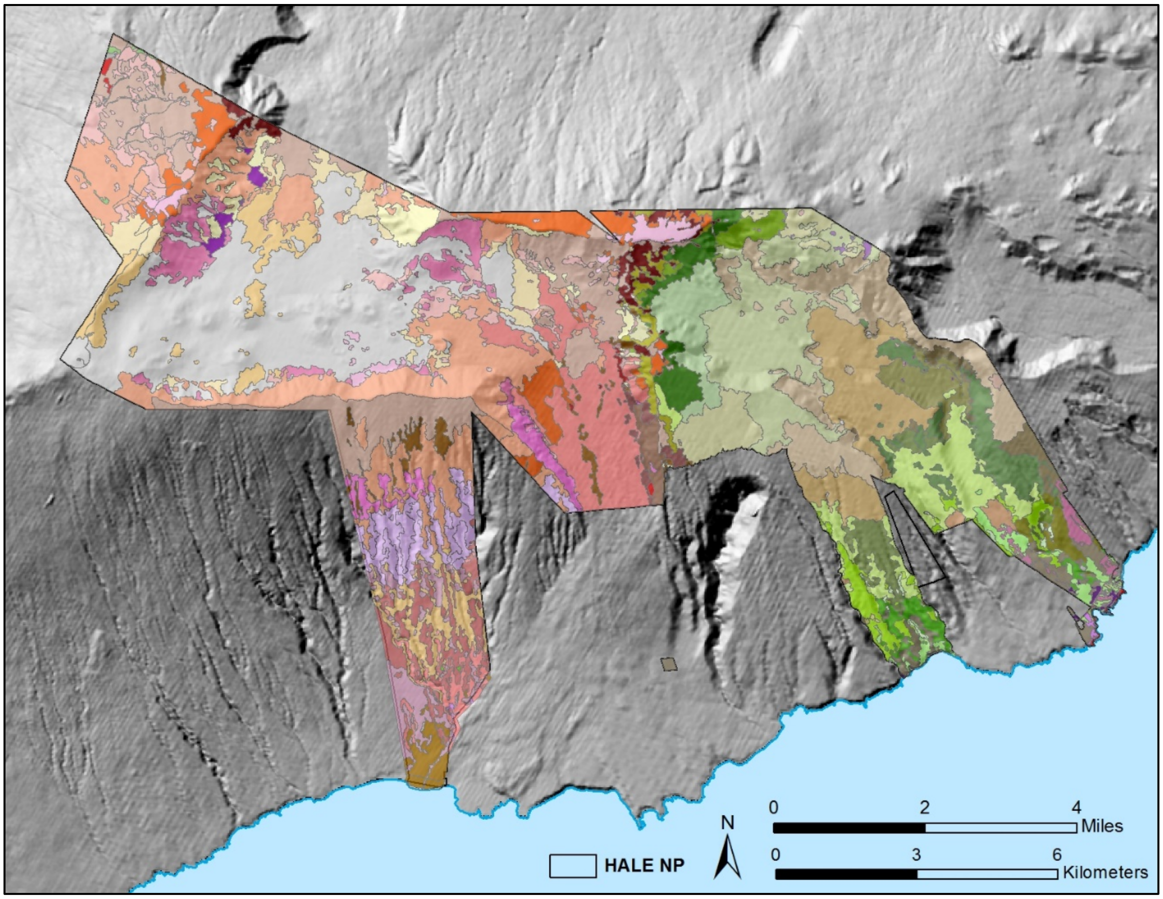

Figure 2.2.1-3. Distribution of major vegetation community types in HALE, as determined in baseline NPS I&M vegetation mapping. A legend for the 72 mapped community types is not shown in the interest of visual clarity; these are best viewed in the original report or geodatabase. Source: Green et al. (2015).

In addition to the biomes and ecosystems summarized above, HALE contains freshwater ecosystems, in the form of perennial and intermittent streams and several small lakes. These streams serve as drainages for several of the numerous watersheds on East Maui, depicted in Figure 2.2.1-4. The condition of freshwater resources within the park further affect the condition of nearshore marine environments, which fall outside park boundaries.

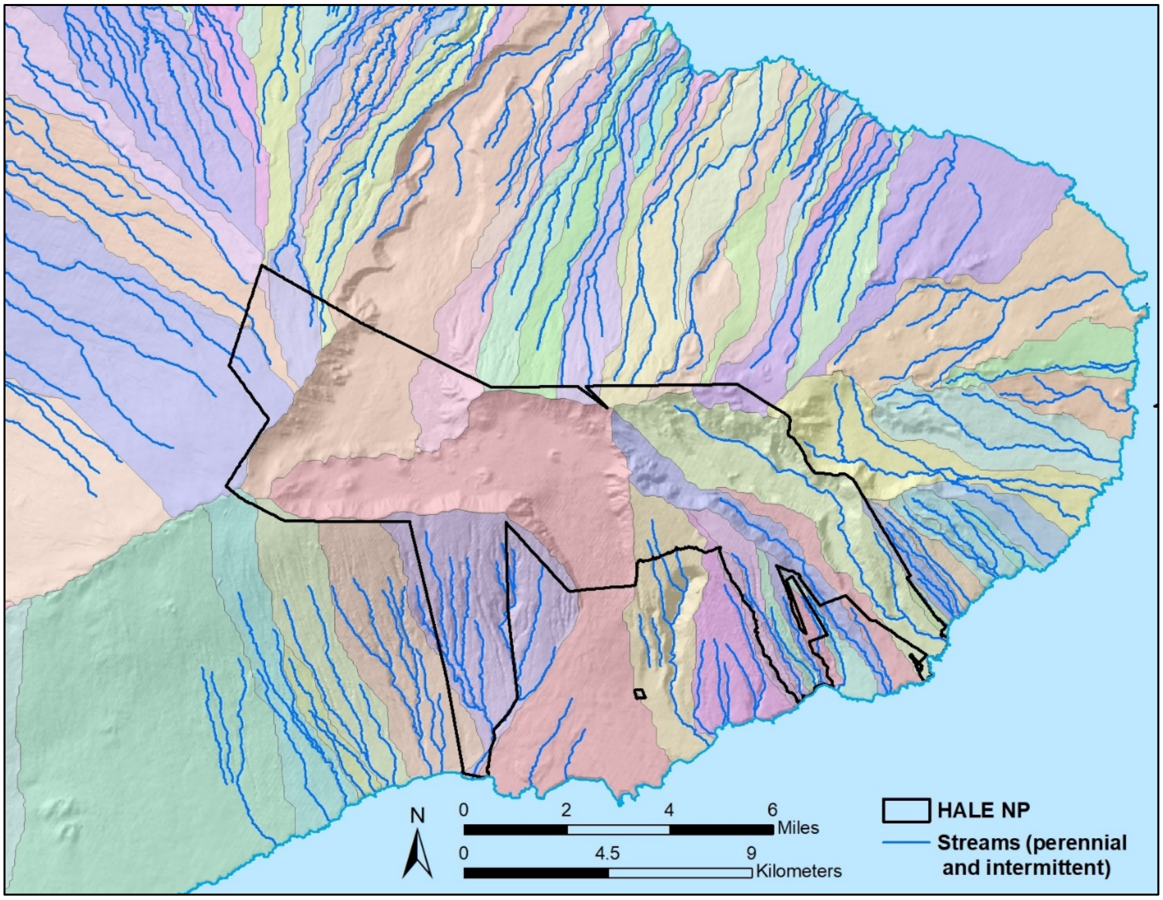

Figure 2.2.1-4. Watersheds and streams of East Maui. Source: Hawai‘i Office of State Planning.

2.2.2. Resource Descriptions

Many of the park’s major resources are delineated by ecosystem types and/or geographic areas. Other resources consist of important biological taxa, including rare and endangered species. Brief introductions to these resources are given below; more extensive overviews are provided in Chapter 4.

Coastal ecosystems

The boundaries of HALE reach the coast in three locations: a strand between Nu‘u and Huakini Bays, at Ka‘āpahu Bay, and between Awapaewa‘a and Pepeiaolepo Bays. Approximately 5.2 km of coastline occur in the park (Natividad Bailey 2009). Coastal ecosystems in HALE, as in much of the Hawaiian Islands, have been dramatically altered by both agricultural practices of early Polynesians and those that continued after European contact. Although non-native vegetation now dominates most locations, a narrow strip of strand vegetation lines the coast. Characteristic plant species include naupaka-kahakai (*Scaevola taccada*), hala (*Pandanus tectorius*), nanea (*Vigna marina*), and the dwarf mat-forming sedge *Fimbristylis cymosa*.

Protected and less degraded coastal ecosystems in Hawai‘i may support unique native animal communities, including coastal nesting seabirds; shorebirds that forage along coastlines and lowland areas; marine animals, including sea turtles that nest in sandy substrates and monk seals that haul out above the waterline; and crabs, insects, and related invertebrates that utilize coastal habitats. These coastal animals are poorly represented at HALE owing to a combination of limited suitable habitat and the presence or a variety of threats.

Lowland and montane forests and bogs

Forest ecosystems of the Hawaiian Islands occur across a broad range of elevation and rainfall gradients, from lowland dry forests which occur below about 4,900 ft (1,500 m) elevation and receive less than 50 inches (ca. 1,300 mm) of annual rainfall, up to montane wet forests which occur at elevations of about 3,280-6,230 ft (1,000–1,900 m) and receive rainfall amounts exceeding 98 inches (2,500 mm) and reaching as high as 400+ inches (>10,000mm) (Giambelluca et al. 2013, Gustafson et al. 2014). Within HALE, the majority of forest ecosystems occur in Kīpahulu Valley, the Kaʻāpahu section of the park, and the Manawainui Planeze. Scattered within wet forests in locations of poor drainage are a series of montane bogs, within Hāna Rainforest on the outer northeast rift at 5,380-6,150 ft (1,640- 1,875 m) elevation, and along the Palikea Stream on the lower shelf of Kīpahulu Valley between elevations of 2,400 and 2,950 ft (730-900 m).

The two most common canopy trees in HALE forests are *Metrosideros polymorpha* (ʻohiʻa lehua) and *Acacia koa* (koa). Subcanopy and lower strata support a high diversity of other tree, shrub and herb species, with composition of these communities changing with precipitation regime and substrate (Gagné and Cuddihy 1999). Bogs contain a distinct assemblage, with many species restricted to these ecosystems or similar areas. The relatively large tracts of continuous forests on East Maui, of which HALE is an integral part, provide the foundation for some of the best remaining habitat for associated invertebrates and birds in Hawaiʻi.

High elevation shrublands, grasslands and alpine desert

Subalpine shrublands occur on the upper slopes of Haleakalā mainly on the drier, western half of the volcano (Medeiros et al. 1998). Wet shrublands also exist in areas above the treeline on eastern flanks, but these are generally narrower zones that transition quickly into forest. Larger expanses of native shrubland dominate the western slopes and upper portions of Koʻolau and Kaupō Gaps within HALE. The most common shrub species in these ecosystems is pūkiawe (*Leptecophylla tameiameiae*); other dominant shrubs include māmane (*Sophora chrysophylla*), ʻōhelo (*Vaccinium reticulatum*), pilo (*Coprosma montana*), ʻaʻaliʻi (*Dodonaea viscosa*), and kūpaoa (*Dubautia menziesii*). A relatively uncommon small tree, ʻiliahi (*Santalum haleakalae*), is scattered throughout shrublands, while the tangled, sometimes prostrate shrub ʻūlei (*Osteomeles anthyllidifolia*) becomes more common in Kaupō Gap. Numerous additional species of shrubs, herbs and grasses support many unique arthropod species, and are used seasonally for foraging by honeycreepers that nest in nearby forests (Medeiros et al. 1998).

Shrublands grade into alpine desert ecosystems near the summit of Haleakalā, and on young cinder and lava-dominated substrates on the western crater floor (Medeiros et al. 1998). Although mean annual rainfall (ca. 1,000-1,300 mm) is considerably higher than most true deserts, the young, porous soils, generally dry atmospheric conditions above the TWI, and large daily temperature fluctuations create conditions of water limitation and climatic stress for plants. Vegetation is generally sparse.

Plant diversity is low in these ecosystems, and is dominated by pūkiawe (*L. tameiameiae*), kūpaoa (*D. menziesii*), and in places by the unusual ʻāhinahina (Haleakalā silversword, *Argyroxiphium sandwicense* subsp. *macrocephalum*), a close relative of kūpaoa. Other plants occurring in this ecosystem include *Tetramalopium humile*, *Silene struthioloides*, and several grasses, sedges and herbs. Several insect species are known only from alpine deserts and nearby shrublands, such as the flightless moth, *Thyrocopa apatela*, and several species of predatory ground beetles (family Carabidae).

A high-elevation grassland ecosystem, unique in Hawaiʻi (Gagné and Cuddihy 1999), exists on the northeast slope of Haleakalā, adjacent to the eastern crater and western Kīpahulu Valley. This mesic Kalapawili grassland is dominated by large tussocks of the endemic grass *Deschampsia nubigena* (Medeiros et al. 1998). Other associated native species include scattered ʻōhelo shrubs (*V. reticulatum*), the sedges *Uncinia brevicaulis* and *Carex macloviana*, and the rush *Luzula hawaiiensis* (Medeiros et al. 1998). Smaller high-elevation grasslands occur on Kuiki (the peak on the eastern side of Kaupō Gap) and within the crater. These grasslands are frequented by nēnē (Hawaiian goose, *Branta sandvicensis*), and are used for both nesting and foraging.

Streams

Freshwater ecosystems in Hawaiʻi are principally comprised of streams, with wetlands and occassional small lakes occurring on some islands. HALE's freshwater ecosystems are mainly represented by numerous intermittent and perennial streams, and three small montane lakes, in the wetter eastern portion of the park. Additional drainages on the western and southern faces of Haleakalā volcano flow only rarely, during extreme rainfall events, but may hold standing water in pools for longer periods of time.

One of the longest undiverted streams on Maui, Palikea, sits entirely within HALE, in Kīpahulu Valley. Palikea and nearby streams in HALE are important habitat for a range of aquatic animals. These include five species of ʻoʻopu, or gobioid fish: ʻoʻopu ʻakupa (*Eleotris sandwicensis*), ʻoʻopu naniha (*Stenogobius hawaiiensis*), ʻoʻopu nākea (*Awaous guamensis*), ʻoʻopu nōpili (*Sicyopterus stimpsoni*), and ʻoʻopu alamoʻo (*Lentipes concolor*). Two species of freshwater crustaceans (ʻōpae kalaʻole, mountain shrimp, *Atyoida bisulcata*) and (ʻōpae ʻoehaa, freshwater prawn, *Macrobrachium grandimanus*), and two species of mollusc (hīhīwai, *Neritina granosa*; and hapawai, *Theodoxus vespertinus*), inhabit HALE streams. A diverse aquatic insect fauna also lives in these freshwater ecosystems, including at least five species of damselflies in the genus *Megalagrion*.

Plants

As with all groups of organisms in Hawaiʻi, total richness of plants in the archipelago is relatively low for a subtropical region (ca. 1,100 species, Wagner et al. 1999), owing to the isolation of the islands and the resultant difficulty of natural colonization. Also like other taxonomic groups, however, Hawaiian plants have evolved into a highly unique and unusual flora, characterized by numerous large radiations (speciation events) stemming from the few colonizers that were successful, and complete absences of other groups that are conspicuous components of floras in other parts of the world (termed taxonomic disharmony, Funk and Wagner 1995). Some of the largest radiations of plants in Hawaiʻi include the Lobelioids (>110 species in six genera, family Campanulaceae), *Cyrtandra* (60 species, family Gesneriaceae), *Phyllostegia* (35 species, Lamiaceae), and the 'silversword alliance' (>30 species in three genera, family Asteraceae) (Wagner et al. 1999, Lindqvist and Albert 2002, Cronk et al. 2005, Givnish et al. 2009). The silversword alliance is one of world's best examples of adaptive radiation in plants, in which one herbaceous colonizer diversified across a wide range of climate zones into a remarkable variety of growth forms, including prostrate herbs, shrubs, trees, cushion plants, and alpine rosettes (the famous silverswords) (Robichaux et al. 1990). This flora serves as the foundation for all terrestrial ecosystems in Hawaiʻi, and was the template for diversification and co-evolution among many groups of invertebrates and birds (e.g., Roderick and Percy 2008, Bennett and O'Grady 2012).

Medeiros et al. (1998) recorded at least 246 native plant species, not including ferns and their allies, in HALE, of which 77% are endemic to Hawaiʻi and 17% are endemic to Maui. Currently, there are believed to be 412 native plants, including 129 ferns and allied plants, 58 monocots and 225 dicots (NPSpecies 2017). These include representatives of all of the major radiations mentioned above, plus many others (Medeiros et al. 1998). Chief among them are the ʻāhinahina, the most abundant remaining taxon of silversword in the islands, as well as the related greensword (*Argyroxiphium grayanum*) and several kūpaoa (*Dubautia*) species, and numerous species of Lobelioids in the genera *Cyanea*, *Clermontia*, *Lobelia* and *Trematolobelia*. The diversity of plant forms across the many biomes and ecosystem types in HALE is striking, and the protection of this diversity is a central mission of the park.

Birds

With the exception of one species of bat, birds are Hawaii's only native terrestrial vertebrates. Several avian groups radiated extensively after colonizing the islands, especially the Hawaiian honeycreepers (family Fringillidae), which diversified into at least 50 species across the archipelago (Scott et al. 2001, Pratt 2009). Many of these species evolved highly specialized bills for different modes of foraging, and some species with long, curved bills are thought to be important pollinators for rare plants with long, curved, tubular corollas, such as the Lobelioids mentioned above. Another avian group that colonized the Hawaiian Archipelago and then diversified were thrushes in the genus *Myadestes* (family Turdidae), which are primarily frugivorous and were important seed dispersers. Unfortunately, over half of the entire Hawaiian avifauna is now extinct, and many remaining endemic species are highly endangered (Scott et al. 2001, Banko and Banko 2009). Fossil evidence indicates that many flightless species and seabirds disappeared shortly after human arrival, and numerous other species went extinct within the last 150 years (Olson and James 1982, Scott et al. 1986). The full effects of the concomitant loss of ecological services such as plant pollination, seed dispersal, nutrient import, and others, are difficult to quantify but are likely substantial.

HALE serves as an important refuge for several groups of birds. ʻUaʻu (Hawaiian petrel, *Pterodroma sandwichensis*), nēnē (Hawaiian goose, *Branta sandivicensis*), and pueo (Hawaiian short-eared owl, *Asio flammeus sandwichensis*) nest in the high elevation shrublands, grasslands, and crater ecosystems. The latter two species also use these areas for foraging. Forests and mesic to wet shrublands inside the park are home to several species of honeycreepers (Lerner et al. 2011): Maui ʻamakihi (*Chlorodrepanis virens wilsoni*), ʻapapane (*Himatione sanguinea*), ʻiʻiwi (*Drepanis* *coccinea*), Maui ʻalauahio (Maui creeper, *Paroreomyza montana newtoni*), ʻākohekohe (crested honeycreeper, *Palmeria dolei*), and kiwikiu (Maui parrotbill, *Pseudonestor xanthophrys*). The latter three species are endemic to Maui, the latter two species are listed as endangered, and HALE supports the majority of their global populations. Several additional species of seabirds are known to nest in the park, including koaʻe kea (white-tailed tropicbird, *Phaethon lepturus dorotheae*) and noio (Hawaiian or black noddy, *Anous minutus melanogenys*). Several species of migratory shorebirds use park lands during the winter months when they are not nesting, primarily September-April, including the kōlea (Pacific golden plover, *Pluvialis fulva*), ʻakekeke (ruddy turnstone, *Arenaria interpres*), and ʻūlili (wandering tattler, *Tringa incanus*), and a few individuals stay over during the summer months. These species primarily use coastal habitats in the park, but they occasionally venture into higher altitude grassland and shrubland habitats.

Invertebrates

Taxonomic disharmony, large radiations stemming from few colonizers, and unique evolutionary products reached its pinnacle among Hawaiʻi's invertebrates. Within the arthropods alone (insects and their relatives), there is an estimated total of perhaps 10,000 native species, although only about half have been formally described so far (Howarth 1990, Medeiros et al. 2013). Nearly all of these species are thought to be endemic to Hawaiʻi (Medeiros et al. 2013). The largest known radiation is in the fly genus *Drosophila* (and related genus *Scaptomyza*, family Drosophilidae), which is estimated to encompass perhaps 1,000 species, all originating from a single colonizing species (Kaneshiro 2006, O'Grady et al. 2010). Another remarkable group is the *Hyposmocoma* moths (family Cosmopterigidae), with over 350 species recognized and many more undescribed (Rubinoff 2008). The caterpillars of these moths have also diversified in ecological roles, being herbivores, detritivores and even carnivores, and occupying nearly all ecosystems and habitat types, including underwater in streams (Rubinoff 2008). Other unusual adaptive ecological shifts among invertebrates include ambush predatory caterpillars in the genus *Eupithecia* (family Geometridae), and predatory/scavenging seed bugs (family Lygaeidae) (Montgomery 1982, Howarth 1987). The range of evolutionary and ecological novelties among Hawaiian invertebrates are too numerous to detail here, but collectively encompass a wide range of important ecological roles (Medeiros et al. 2013). As an example, it is estimated that 67% of Hawaiian plant species rely on insect pollinators (Sakai et al. 2002).

A consequence of the extreme taxonomic disharmony in the Hawaiian invertebrate fauna is the absence of groups that are ecologically significant elsewhere (Howarth 1990). For example, it is thought that few if any species of ants, termites, or any other social insect group naturally colonized Hawaiʻi (Wilson 1996), and that the subsequent arrival of these groups has led to outsized impacts on the native fauna (Zimmerman 1970, Howarth 1990). Invasive predatory snails, which were formerly absent, and rodents are thought to have decimated the native snail fauna, which is known to be highly diverse but is largely still undescribed (Cowie 2001).

Numerous unique and interesting invertebrate species are known from the park. In a survey of portions of the Summit District, Beardsley (1980) reported that 35% of the native insect species collected were endemic to Haleakalā volcano. The current number of native invertebrates (including insects, arthropod relatives, and molluscs) known to occur in the park is 575 (NPSpecies 2017), though there are likely many more. Most work on invertebrates has taken place in the shrublands on the west slope, and in other parts of the Summit District (e.g. Beardsley 1980, Brenner 2003, Krushelnycky et al. 2007). Only very preliminary surveys have been conducted within the mesic and wet forests dominating the eastern half of the park (e.g. Wilson 1967, Villegas 1976, Kaholoa‘a unpub. data), which likely support considerably higher diversity than the drier lands to the west. The PACN monitoring plan chose to postpone development of inventory and monitoring protocols for invertebrates until phase 2 (HaySmith et al. 2006), due to the difficulty of studying this group.

Threatened and Endangered Species

Hawai‘i currently has 543 federally listed threatened or endangered (T&E) species, more than any other state (USFWS 2017). The reasons for this situation are summarized in section 2.2.3, below. Because HALE encompasses some of the best remaining habitat for native plant and animals in the islands, it also contains many T&E species. Currently, there are 56 listed species known or thought to occur in the park, including species that may now be extinct (NPS 2012, NPSpecies 2017, P. Welton pers. comm. 2018). The majority of these are plants, but also include 10 birds, 2 mammals, and 1 insect (Table 2.2.2-1).

Table 2.2.2-1. Federally listed T&E taxa recorded in HALE. Source: NPSpecies (2017) and P. Welton (pers. comm. 2018).*

Taxon	Endangered	Threatened	Total
Plants	42	1	43
Birds	8	2	10
Mammals	2	0	2
Invertebrates	1	0	1
Total	**53**	**3**	**56**

2.2.3. Resource Issues

In common with most areas of the world today, oceanic islands are stressed by a variety of forces of global change, including habitat loss and fragmentation, invasive species, and climate change. Island ecosystems, however, are particularly vulnerable to these stressors owing to factors such as evolutionary isolation of their floras and faunas and small land areas. Small land area results in highly localized distributions of many species, which consequently have limited options for contending with changing conditions. These vulnerabilities are reflected in the very large number of endangered species in Hawai‘i and in HALE.

Mitigation of external stressors has been the central focus of resource management strategies at HALE since its inception, and these efforts have expanded substantially over time. The most recent Resource Management Plan (RMP) for HALE (NPS 1999) stated that “Active resources management efforts begun in the mid-1970's, accelerated in the mid-1980's, and continuing into the 1990's, have begun to reverse the chronic decline of the park's resources.” The RMP also provided an overview of the main perceived threats and issues to the park at the time, which included:

- Feral goats and pigs, and other browsing and predatory invasive mammals
- Invertebrate pests
- Alien plants
- Endangered plants and animals
- Native plant restoration
- Biological and air quality monitoring
- Climate change and climate monitoring
- Aircraft noise impacts
- Park visitor and outside development impacts
- Fire management

Non-native, invasive species figure prominently in this list. Human facilitated biological invasions are well-documented causes of global biodiversity loss and extinction and are especially damaging to the biota of oceanic islands (MacDonald et al. 1989, Denslow 2003). Hawaiian Island ecosystems are believed to be more vulnerable to invasion than are continental ecosystems because their biota have evolved in isolation from pressures such as foraging and trampling by herbivorous mammals, predation by ants, rodents and other mammals, other pests and pathogens, and frequent fire (Loope and Mueller-Dombois 1989). These factors have contributed to the dramatic alteration of low and middle elevation ecosystems of the islands, and are having similar if less pronounced effects on many of the higher elevation ecosystems of HALE (Loope and Medeiros 1994).

Feral and invasive animals

Among the most damaging of invasive species in Hawai‘i are feral ungulates, including pigs (*Sus scrofa*), goats (*Capra hircus*), sheep (*Ovis* spp.), cows (*Bos taurus*), and more recently, axis deer (*Axis axis*). In HALE, goats and pigs have had the greatest ecological impacts. Feral cows remain a problem in localized areas of lower Kīpahulu, and in the past cattle herds were transported through Haleakalā Crater and pastured in certain areas of the park before and even in the years after federal designation and protection (Loope and Crivellone 1986). In response to browsing and trampling damage from ungulates, mainly goats, the park initiated hunting programs to reduce their numbers in and around the crater area from the 1940’s through the 1970’s (NPS 1999). Beginning in the 1980’s, HALE initiated an ambitious effort to erect a network of fences to completely exclude feral ungulates from the park. This network, which fully surrounded some management units, and utilized a combination of fencing and natural barriers in others, was mainly completed in 1988 with some later additions (NPS 1999, Figure 2.2.3-1). Animals were subsequently removed from management units using a combination of ground-based and aerial shooting and snaring. Total numbers of animals removed from HALE from 1986 to present by the Feral Animal Removal and Management (FARM) unit of the Resources Management Division (RM) are shown in Table 2.2.3-1. The completed fence network is not totally impervious, as animals continue to find locations to circumvent natural barriers that were thought to be impassible, and tree falls and other damage create temporary entry points. Inspection and maintenance of all boundary fences in perpetuity, as well as cyclic replacement of the

Kīpahulu District fences due to corrosion and damage from water and wind, and continued removal of feral animals from all park ecosystems, remain the prime responsibility of FARM staff.

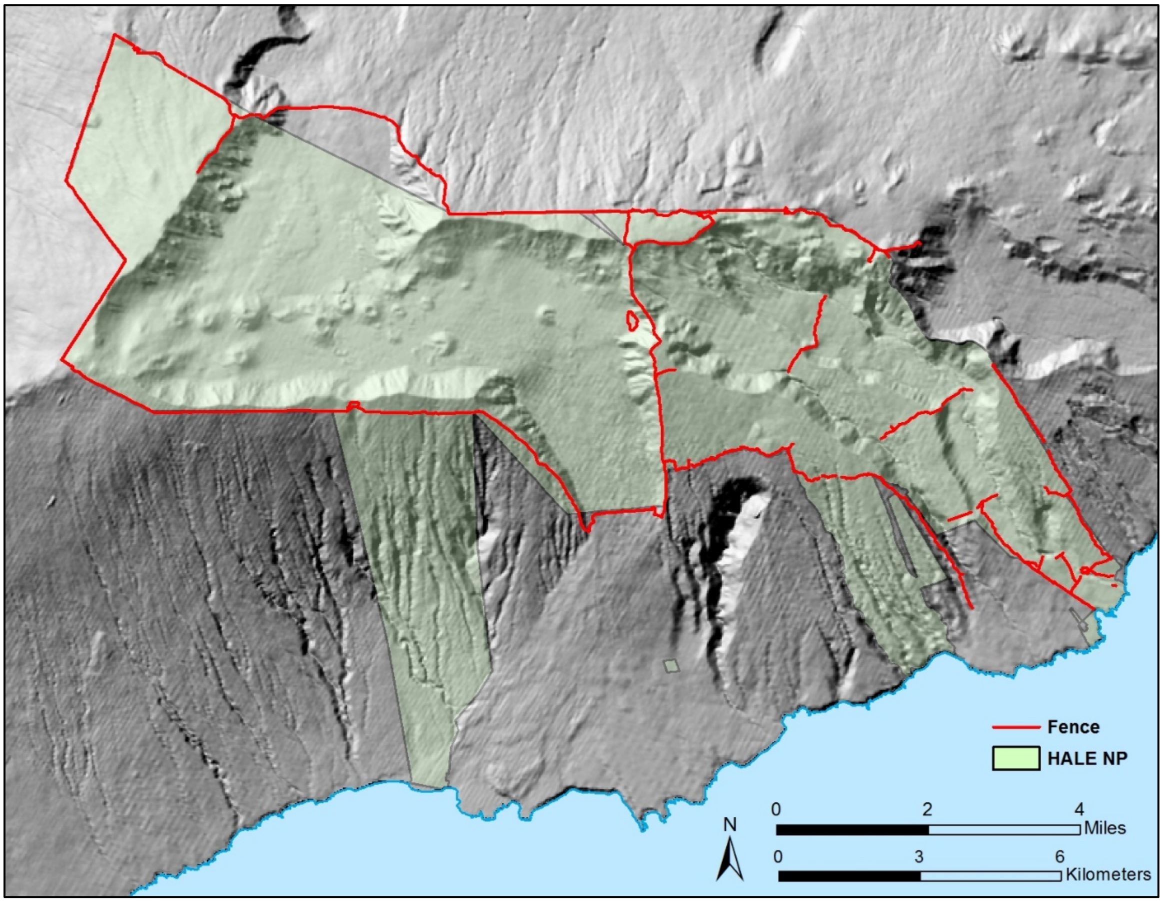

Figure 2.2.3-1. Locations of fence lines in and around HALE, designed to exclude feral ungulates from the park. Fencing for the Nuʻu section is currently under construction. Source: NPS HALE.

Table 2.2.3-1. Total numbers of feral animals removed by Feral Animal Removal and Management staff from 1986 through 2017. Source: NPS (2017b).

Year	Pigs	Goats	Deer	Cattle
1986	<50	>1000	0	N/A
1987	<50	>1000	0	N/A
1988	<50	>1000	0	N/A
1989	<50	>1000	0	N/A
1990	<50	>1000	0	N/A
1991	<50	<500	0	N/A
1992	<50	<100	1	N/A
1993	>50	<50	0	N/A
1994	>50	<50	0	<5

Table 2.2.3-1 (continued). Total numbers of feral animals removed by Feral Animal Removal and Management staff from 1986 through 2017. Source: NPS (2017b).

Year	Pigs	Goats	Deer	Cattle
1995	>50	<50	4	<5
1996	>50	<50	0	<5
1997	>50	<20	3	<5
1998	>25	0	0	<5
1999	23	0	0	<5
2000	21	0	0	<5
2001	17	0	0	<5
2002	13	0	0	<5
2003	20	0	0	<5
2004	20	0	0	<5
2005	6	0	0	<5
2006	11	30	0	<5
2007	0	22	0	<5
2008	4	19	0	<5
2009	0	7	0	<5
2010	5	6	1	<5
2011	9	8	0	<5
2012	4	6	0	<5
2013	30	24	0	<5
2014	16	8	0	<5
2015	3	14	0	<5
2016	1	12	0	<5
2017	36	17	0	2

Smaller invasive mammals, including rats (mainly *Rattus rattus*, but also *R. norvegicus* and *R. exulans*), mongoose (*Herpestes auropunctatus*), cats (*Felis catus*), and sometimes dogs (*Canis familiaris*), are significant predators of native birds (van Riper and Scott 2001). Rats (and mice, *Mus musculus*, also non-native) are also known to feed heavily on native plant seeds and fruits, and native invertebrates (Cole et al. 2000, Chimera and Drake 2011, Pender et al. 2013, Shiels et al. 2013). In 1991, a population of 97 European rabbits (*Oryctolagus cuniculus*), believed to have originated from an initial release of six rabbits by a pet owner in 1989, was detected and subsequently eradicated from high elevation native shrubland before causing significant damage to the park's natural resources (Loope et al. 1992). In order to protect populations of endangered ʻuaʻu (*Pterodroma sandwichensis*) and nēnē (*Branta sandivicensis*), the Endangered Wildlife Management (EWM) Program of RM established several traplines in HALE in the vicinity of major ʻuaʻu colonies and nēnē nesting areas (NPS 2012). These traplines were established in 1981, and currently employ approximately 425 live traps that target rats, mongoose and cats (NPS 2012). Foothold traps are also sometimes strategically deployed in specific locations to target heightened predator activity (NPS 2012).

Invasive plants

Invasive plants that crowd out native plant species are major threats to native Hawaiian ecosystems (Stone et al. 1992). Their impacts and spread are often promoted in the presence of invasive ungulates, which create disturbed space that facilitates weed germination and which may also disperse weed seeds. Following the removal of invasive mammals in HALE, alien plants are considered to be the most serious problem facing resources management (NPS 1999). Over 400 non-native plant species are now established in the park (Medeiros et al. 1998, NPSpecies 2017), and managing their spread is the main focus of the Vegetation Management (VM) Program. Some of the most damaging invasive weeds, and thus the primary targets of VM, include Koster's curse (*Clidemia hirta*), kahili ginger (*Hedychium gardnerianum*), strawberry guava (*Psidium cattleianum*), several species of pines (*Pinus* spp.), blackberry (*Rubus argutus*), gorse (*Ulex europaeus*), and pampas grass (*Cortaderia jubata*). Mapping and control of targeted invasive weeds is conducted within 40 management units that strategically segregate the park (Figure 2.2.3-2), facilitating the planning, coordination, and tracking of these efforts.

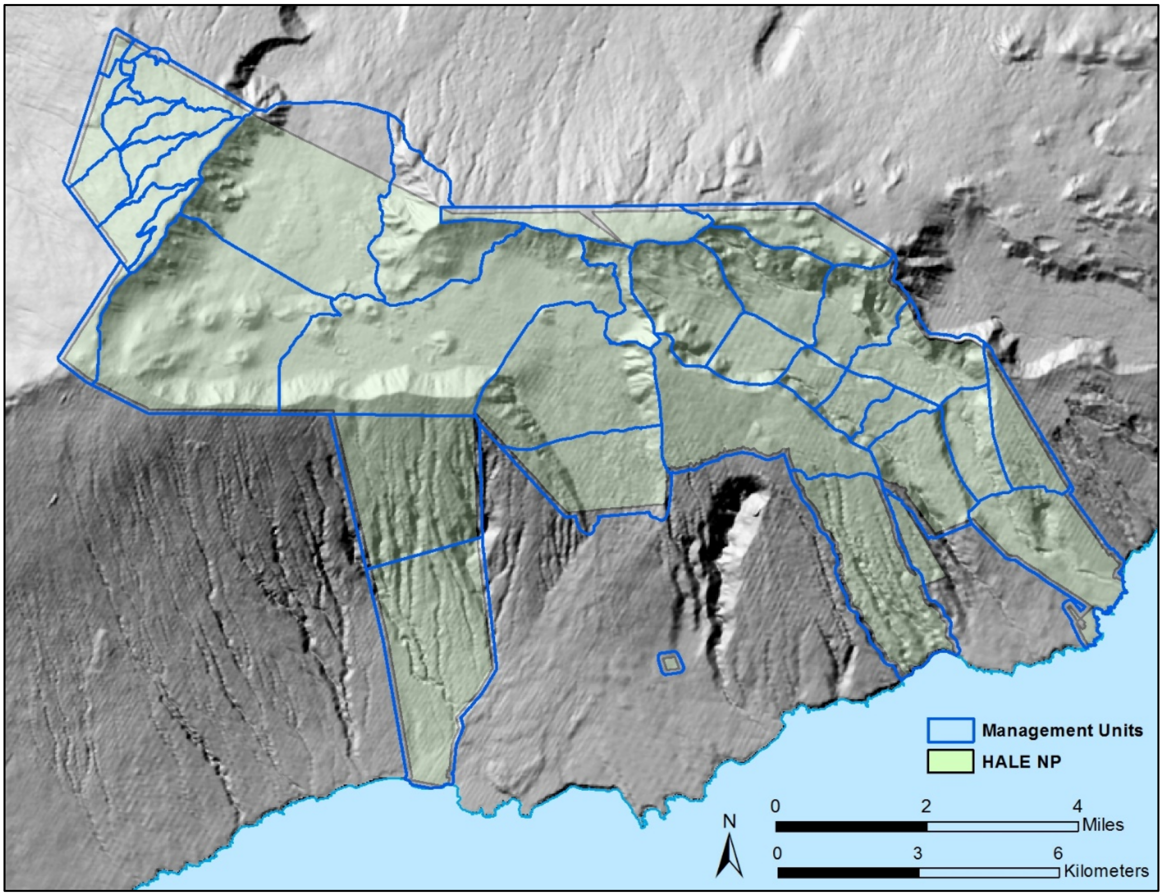

Figure 2.2.3-2. Management unit divisions used by Vegetation Management Division to plan and conduct invasive plant control efforts. Source: NPS HALE.

Disease and pathogens

More recently, several plant pathogens have emerged in Hawaiʻi that threaten to devastate native forests. In particular, fungal pathogens (*Ceratocystis* spp.) are responsible for widespread mortality of *M. polymorpha* on Hawaiʻi Island, but have not yet been detected on Maui (Keith et al. 2015, Mortenson et al. 2016). The resulting disease has been termed "Rapid ʻŌhiʻa Death" (ROD) because healthy trees die within weeks after first showing symptoms. If the pathogen becomes established on Maui, the loss of *Metrosideros* trees would completely transform high elevation forests and likely result in the subsequent decline of much of the remaining associated native flora and fauna. Another fungal pathogen (*Puccinia psidii*), commonly referred to as ʻōhiʻa rust in Hawaiʻi, has killed many trees of related species in the family Myrtaceae, but so far has shown limited pathogenicity on *Metrosideros* species (Uchida et al. 2006, Loope and La Rosa 2008). If a strain more pathogenic to *Metrosideros* were to arrive, however, results could be similar to those feared in conjunction with ROD.

Non-native invertebrates

A large number of invasive insects and other invertebrates have also established in Hawaiʻi, and many of these are thought to impact native insect communities through competition, predation and parasitism (Zimmerman 1970, Howarth and Gagné 2012). Some, like the koa psyllid (*Psylla uncatoides*) and naio thrips (*Klambothrips myopori*) may attack native plants, or vector plant diseases. Non-native mosquitos transmit diseases that have decimated native bird populations at lower elevations. Although accurate numbers of non-native invertebrate species established in the park are difficult to estimate, a survey of portions of the Summit District recorded over 150 non-native insect species in the 1970's (Beardsley 1980). The current total number of all non-native invertebrate species recorded in the park is 396 (NPSpecies 2017), but there are likely more. Several species are especially problematic at HALE, including the western yellowjacket (*Vespula pensylvanica*) and the Argentine ant (*Linepithema humile*). Both species of social Hymenoptera are voracious predators of native insects and appear to be decimating native insect communities (Cole et al. 1992, Krushelnycky and Gillespie 2008, Wilson et al. 2009). Argentine ants in HALE occupy over 1,606 ac (650 ha) and are spreading (Krushelnycky et al. 2011). Control efforts were attempted but eventually discontinued because they were insufficiently effective, and the spread of this species into shrubland and crater ecosystems remains one of the most serious problems facing the park. Colonies of western yellowjackets are individually detected and controlled by EWM as staffing levels permit.

Climate change

Finally, climate change has emerged as one of the most challenging issues threatening species, communities and ecosystems worldwide. Islands are typically considered to be especially susceptible to the potential impacts of climate change (Keener et al. 2012, Harter et al. 2015). Rising sea levels will impact coastal communities, and the combination of small land area and restricted ranges among many island species will often limit opportunities to shift distributions in response to changing climate conditions. This is particularly true for mountaintop communities, which are generally expected to move upslope to track rising temperatures (Diaz and Eischeid 2007, Dirnböck et al. 2011). Temperatures in Hawai'i have been increasing over the previous century, with rates of warming being highest in recent decades and at higher elevations (Giambelluca et al. 2008), and there is little uncertainty that temperatures will continue to rise over the coming century (IPCC 2013). Hawaiʻi has also been experiencing drier conditions over the past decades (Chu and Chen 2005, Banko et al. 2013, Longman et al. 2015a), with resultant ecological effects apparent (Banko et al. 2013, Krushelnycky et al. 2013, 2016). Long-term reductions in stream base flow have occurred across the state (Oki 2004), which may impact freshwater communities.

Future projections of precipitation patterns in Hawaiʻi have considerably more uncertainty than those concerning temperature. Two of the methodological approaches used to downscale future climate projections in Hawaiʻi, statistical downscaling (SD) and dynamical downscaling (DD), have yielded precipitation projections that are not in good agreement. On East Maui, SD modelling predicts that rainfall will decrease across both wetter windward and drier leeward areas, with some leeward areas becoming much drier, depending on the assumed greenhouse gas emission scenario (RCP4.5 or 8.5) and future timeframe (mid-century or end of century) (Elison Timm et al. 2015). In contrast, DD modelling predicts that by the end of the century, rainfall will increase in wetter windward areas, and will increase slightly or remain unchanged in leeward areas within park boundaries (Zhang et al. 2016). Future rainfall patterns will have strong effects on the distributions of species, communities and entire ecosystems at HALE (e.g., Crausbay and Hotchkiss 2010), so current uncertainty about future changes makes management planning a significant challenge.

Changes in the TWI (trade wind inversion) will also have ecological consequences. There is evidence that the frequency of occurrence of the TWI has increased in Hawaiʻi in recent decades (Longman et al. 2015a), and that this has led to drier, sunnier conditions above its base height on Haleakalā (Longman et al. 2015b). These changes may in turn underlie strong population declines of the iconic ʻāhinahina plant (Haleakalā silversword) in the park (Krushelnycky et al. 2016). The TWI is predicted to not only increase in prevalence in the future (Zhang et al. 2016), but also to possibly drop in elevation (Lauer et al. 2013), which would lead to downward contraction of native cloud forest on the mountain (Loope and Giambelluca 1998, Crausbay et al. 2014a, b). While climate change has potential implications for all ecosystems at HALE, and recent values of temperature and precipitation parameters in the park have been extreme relative to historical values (Monahan and Fisichelli 2014), the consideration of climate in this assessment will be restricted to cases in which empirical evidence suggests that changing conditions are already impacting the condition of resources.

2.3. Resource Stewardship

2.3.1. Management Directives and Planning Guidance

The HALE Foundation Document provides guidance for park planning and management (NPS 2015). Features and qualities of HALE that are deemed to be of special value and meritorious of federal protection are reflected in the park’s significance statements:

1) Rising 10,000 ft from the sea to the summit of Haleakalā, the park protects a striking variety of natural landscapes, ranging from tropical rainforest to subalpine desert. Within these lands, extreme gradients of rainfall and temperature shape the park’s remarkable biodiversity.

2) Haleakalā National Park protects unrivaled examples of native Hawaiian ecosystems, providing a home for diverse threatened and endangered species, including some that exist nowhere else in the world, and still others yet to be discovered.
3) From its windswept cinder fields to its lush rainforest, the Haleakalā Wilderness provides a panorama of exceptional grandeur where people may find solitude and inspiration within a vast and colorful landscape.
4) Haleakalā National Park preserves places, resources, stories, and intangible elements of profound sacred importance to Native Hawaiians. Collectively, these are linked by the piko, the life-line that honors the past and connects the living Hawaiian culture of today to future generations.
5) Haleakalā National Park is known for its exceptional scenery, including sunrises and sunsets above the clouds; coursing waterfalls, clear pools, and crashing waves; lush rainforests; and sparkling, star-filled skies. These and countless other sights and scenes provide transformational experiences for residents and visitors alike.
6) Visitors to the park can enjoy a broad spectrum of natural sounds, including a rare opportunity to experience intense quiet inside the Haleakalā Crater. Sound levels in the crater are among the lowest recorded in any national park.
7) The Haleakalā shield volcano, one of the highest peaks in the Pacific, is the result of countless volcanic eruptions during the past two million years, and unique erosion in action.
8) At Haleakalā, the volcano’s height, landscape, air quality, and location on Earth provide for excellent, clear night skies. From ancient Polynesian navigators to current day astronomers, people have and continue to use the summit of Haleakalā to study and view the night sky. Numerous light-sensitive species, whose lives are negatively impacted by artificial light, depend on Haleakalā’s natural lightscapes for survival.

The HALE Foundation Document also identifies the park’s Fundamental Resources and Values, which are those “attributes determined to warrant primary consideration during planning and management processes because they are essential to achieving the purpose of the park and maintaining its significance” (NPS 2015). These values for HALE include:

- **Natural Sounds, Viewsheds, and Dark Night Skies**—Natural sounds, panoramic views, and dark night skies greatly contribute to Haleakalā’s unique sense of place.
- **Kīpahulu Moku District (including ‘Ohe‘o Gulch and Palikea Stream)**—Handed down over the centuries through oral tradition and practice, the ‘Aha Moku system is the traditional Hawaiian system of natural resource division and management for ocean and land resources. The undiverted free-flowing Palikea stream and ‘Ohe‘o Gulch are part of an intact East Maui watershed that begins at the piko, or navel, of the island.
- **Wilderness**—Approximately 24,000 ac of Haleakalā National Park is federally designated wilderness. The wilderness area includes the majority of the Haleakalā Crater and the Kīpahulu Biological Reserve, which protects one of the most intact rainforest ecosystems in the Hawaiian Islands.

- **Ongoing Connections to Living Hawaiian Culture**—Haleakalā National Park has cultural and spiritual value for Native Hawaiians who have used particular places, sites, and resources in the park for a broad range of activities from ancient times to the present. Among these traditional cultural activities are ritual ceremonies, spiritual training, and practices related to birth and burial.
- **Outstanding Geological Resources, Including the Haleakalā Volcano and Crater**—Rising to 10,023 ft in elevation, the Haleakalā volcano—also known as the East Maui volcano—is the primary geological feature of the park, and preserves a record of Maui's volcanic history.
- **Archeological and Historic Resources Associated with Native Hawaiian Culture**—Haleakalā National Park preserves a high density and variety of precontact and historic archeological resources. These resources exist in many locations and include Native Hawaiian temples (heiau), trails, altars, fishing shrines, house platforms, and other features.
- **Native Hawaiian Biological Diversity**—Haleakalā, rising from sea to summit, and exposed to both the windward moist tradewinds and leeward drying air, features a striking variety of ecosystems that support a tremendous range of native biological diversity.
- **Kuleana**—The Native Hawaiian concept of kuleana is generally recognized as the responsibility passed down from the kūpuna (ancestors) to present and future generations for stewardship and respect for all things spiritual and physical.
- **Museum Archive and Collections**—The Haleakalā National Park museum collection documents the cultural and natural history of the park. The collection is divided into three main categories: natural history, cultural resources, and archives.

Finally, the foundation document provides summaries of current condition and trends, threats, opportunities, existing data and plans, data and/or GIS needs, and planning needs for each of these Fundamental Resource Values (NPS 2015, Appendix B). However, specific descriptions of desired conditions or management targets, or measurement indicators for such conditions, are generally not articulated.

A document of primary relevance to natural resource management is the Resources Management Plan for HALE (NPS 1999). Although now somewhat dated, this report briefly reviewed the status of major natural resources at the time, and identified the following management objectives for natural resources:

- Re-establish and perpetuate as nearly as possible the mosaic of ecosystems which would have evolved without the interference of human technology.
- Protect and restore native biota by controlling non-native plants and animals, particularly those aggressive species which out compete native forms.
- Maintain the human altered Kīpahulu coastal area in its present state, with latitude for restoration of native plant communities where appropriate.
- Isolate and carefully restrict use of the upper Kīpahulu Valley in order to ensure the perpetuation of nearly pristine native plant communities, native Hawaiian birds, and other native species.

- Encourage a comprehensive park research program for improvement of management and interpretation of Haleakalā's geologic, biotic, and cultural values.
- Include within the park boundaries those adjacent lands which are important to the protection of existing park resources and their interpretation, contain significant resource values, or are essential for visitor access.

The RMP also provided strategy narratives for dealing with the main resource threats and issues in the park. These strategies, together with the management objectives, are helpful for contextualizing the current status and trends of certain focal resources in the park.

2.3.2. Status of Supporting Science

The Inventory and Monitoring Program (I&M) of NPS is an important source of supporting science for park units, providing baseline inventories and conducting subsequent monitoring of selected resources or indicators of ecological conditions ("Vital Signs"). Specifically, Vital Signs are defined as "a subset of physical, chemical, and biological elements and processes of park ecosystems that are selected to represent the overall health or condition of park resources, known or hypothesized effects of stressors, or elements that have important human values" (HaySmith et al. 2006). HALE is part of the Pacific Island Network (PACN) of the I&M Program, which includes 12 park units spread across Hawaiʻi, Guam, the Northern Mariana Islands, and American Sāmoa (HaySmith et al. 2006).

PACN developed a total of 31 Vital Signs deemed important to units within the network (HaySmith et al. 2006). Of these, 18 will be monitored across the network by the I&M Program in either Phase 1 or Phase 2 efforts, and require the development of standardized monitoring protocols. The remaining 13 Vital Signs are monitored by individual park units or other cooperating agencies, although not all are relevant to all units, and some may not be possible to monitor at present. The Vital Signs relevant to HALE, and their monitoring designation, are listed in Table 2.3.2-1.

Table 2.3.2-1. NPS I&M Vital Signs for HALE. Phase 1 and 2 Vital Signs bolded. See notes for designation descriptions. This table includes recent updates by PACN and HALE.

Level 1	Level 2	Vital Sign	Designation*
Air & Climate	Air Quality	Visibility and particulate matter	HALE, other
	Weather & Climate	Climate	1
Geology & Soils	Geologic Processes	Seismic activity	HALE, other
	Soil Quality	Erosion and deposition	2
Water	Hydrology	Stream flow	HALE, other
	Hydrology	Sea level	HALE, other
	Hydrology	Groundwater dynamics	1
	Water Quality	Water Quality	1
	Water Quality	Toxics	Not now

* Designation Values: 1 = Protocol development and implementation in phase 1 of network-wide I&M program; 2 = Protocol development and implementation in phase 2 of network-wide I&M program; HALE, other = Vital sign monitored by HALE, other NPS program, or other agency; Not now = Vital sign which cannot currently be implemented, future monitoring possible.

Table 2.3.2-1 (continued). NPS I&M Vital Signs for HALE. Phase 1 and 2 Vital Signs bolded. See notes for designation descriptions. This table includes recent updates by PACN and HALE.

Level 1	**Level 2**	**Vital Sign**	**Designation***
Water (continued)	Water Quality	Microorganisms	Not now
Biological Integrity	Invasive Species	Status and trends of established invasive plant species	1
	Invasive Species	Early detection of invasive invertebrates	2
	Focal Species or Communities	Benthic marine community	1
	Focal Species or Communities	Marine fish	1
	Focal Species or Communities	Sea turtles	Not now
	Focal Species or Communities	Hawaiian monk seal	HALE, other
	Focal Species or Communities	Freshwater animal communities	1
	Focal Species or Communities	Cave community	2
	Focal Species or Communities	Focal terrestrial plant communities	1
Biological Integrity	Focal Species or Communities	Anchialine pools (animals)	1
	Focal Species or Communities	Terrestrial invertebrate communities	2
	Focal Species or Communities	Nēnē distribution/abundance	HALE, HAVO
	Focal Species or Communities	Waterbird distribution/abundance	Not now
	Focal Species or Communities	Landbirds	1
	Focal Species or Communities	Seabirds (Hawaiian Petrels)	1
	Focal Species or Communities	Bats	2
Human Use	Consumptive Use	Fish harvest	2
	Visitor & Recreation Use	Visitation	HALE, other
Landscapes	Landscape Dynamics	Landscape dynamics (now called Environmental Settings)	1

* Designation Values: 1 = Protocol development and implementation in phase 1 of network-wide I&M program; 2 = Protocol development and implementation in phase 2 of network-wide I&M program; HALE, other = Vital sign monitored by HALE, other NPS program, or other agency; Not now = Vital sign which cannot currently be implemented, future monitoring possible.

Monitoring protocols have now been developed for phase 1 Vital Signs that PACN will implement. Initial inventory and/or baseline efforts, which are to be repeated at future intervals (i.e., monitoring), have now been completed at HALE for Landbirds (Judge et al. 2013), Focal Terrestrial Plant Communities (PACN and Ainsworth 2016), Water Quality (Raikow and Farahi 2016), and Established Invasive Plants (Gross et al. 2017). These inventories served as important information sources for this NRCA effort. Development of the PACN Vital Signs Monitoring Plan also produced a useful overview of important HALE resources (Aruch 2006). For more reports, visit the Pacific Island Inventory Reports and the Pacific Island Monitoring Reports.

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Chapter 3. Study Scoping and Design

3.1. Preliminary Scoping

Preliminary scoping for this assessment project began with an initial meeting on October 13, 2015, held at the Pukalani Community Center, Maui, Hawaiʻi. The meeting included the assessment team (Krushelnycky, Chimera and VanderWerf) and eight members of the Resources Management (RM) staff of HALE, representing a wide range of divisions and expertise. NPS attendees included the NRCA Project Manager (Patti Welton) and one additional assessment committee member (Cathleen Bailey). A full list of attendees and meeting notes are provided in Appendix A. The group primarily discussed the list of potential resources and indicators that was prepared by the NPS assessment committee and given priority rankings of 1 to 3 prior to the project initiation (Appendix B), and which was provided in the project funding opportunity announcement (FOA). Discussions included which resource priority ranks to focus on, potential sources of information for these resources, framework structures and options, and project timeline. It was agreed that the framework used to structure potential resources and indicators in the FOA, which is essentially a combination of the Heinz Center framework and the NPS Ecological Monitoring framework, would serve as the framework for the NRCA, subject to modification if needed. Although many details remained unresolved in this initial meeting, it was decided that follow-on questions would be addressed in personal communications between the assessment team (mainly the lead, Krushelnycky) and the NPS assessment committee (mainly the project manager, Welton), either via phone, email, or in periodic visits to HALE by Krushelnycky in the course of ongoing research at the park.

Over the ensuing three months, discussions between the assessment team and the NPS committee resulted in the refinement of the project framework and scope of resources to be assessed. Essentially, most of the priority level 1 and 2 resources (see Appendix B) were retained, while priority level 3 resources were excluded. This also resulted in a slight reorganization of the original framework subheadings. Indicators of the retained focal resources were also modified in some cases. The revised framework was included in the project study plan, which was submitted on January 25, 2016. A subsequent phone conference was held between assessment team lead Krushelnycky, NPS project manager Welton, and two NRCA advisors (Marsha Davis, NPS Geologist in Pacific West Region, and Cathy Schwemm, Institute for Wildlife Studies and NRCA Project Ecologist), to clarify lingering questions regarding the most effective framework structure and resource designations, and methodological approaches. During this January 11, 2017 conference, it was decided that prior framework formulations that specified invasive species and other stressors as focal resources, should instead be recast such that stressors would be treated as factors influencing conditions and trends within each focal resource. It was also clarified that indicators of focal resources chosen early in the assessment scoping and development process could be modified during assessment production if more appropriate indicator designations and categories became apparent (which was often the case).

3.2. Study Design

3.2.1. Indicator Framework, Focal Study Resources and Indicators

The chosen indicator framework for the HALE NRCA places focal resources within three broad, overarching categories borrowed from the Heinz Center-modified NRCA framework: Life-

Supporting Environment, Biological Integrity, and Landscape Condition Context. The full framework, and final 16 focal study resources contained therein, is shown in Table 3.2.1-1.

Indicators for each focal study resource are also shown in Table 3.2.1-1. Although NRCAs typically attempt to define quantitative measures for each indicator and/or focal resource, this approach was often ineffective for the HALE NRCA for several reasons. First, the park study team defined focal resources as broad ecosystems or geographic regions, with the conditions of biological components of the ecosystems serving as indicators of the condition of the overlying ecosystem or region. This often cast indicator components as resources themselves, especially within the Biological Integrity category, each of which could typically be assessed with many potential measures. In many cases, however, robust quantitative data on such measures are lacking for HALE. In light of the current gaps in quantitative data, it was deemed important to utilize all available information, frequently qualitative or piecemeal in nature, in assessing current conditions of the indicators and focal resources. For these reasons, specific quantitative measures for indicator components are typically not identified or highlighted as the only criteria used to assess the conditions of those indicators, but rather are discussed and evaluated within the context of all available information in assessing an indicator's condition. In all cases, quantitative measures were used to maximum extent possible, but were simply embedded within the overall assessment of indicator condition. The conditions of all component indicators are then integrated when assessing the overall condition of the overlying focal resource. While this may differ somewhat from other NRCAs, it was determined to be the most effective approach for the specific resources and goals of the HALE NRCA.

Table 3.2.1-1. Final indicator framework, modified after Heinz, showing focal resources, their indicators or measures, and reporting area for each focal resource.

Resource Category	Focal Resource	Indicators of resource	Reporting area
I. Life-Supporting Environment	Air Quality	• Ground-level ozone • Wet deposition of N and S • Visibility	Park-wide
	Soil Quality	• Erosion • Soil contamination	Park-wide
	Water Quality	• Total Phosphorus • Total Nitrogen • Nitrate + Nitrite • Turbidity	Major perennial streams
II. Biological Integrity	Coastal Ecosystems	• Coastal vegetation • Coastal vertebrates • Coastal invertebrates	Park-wide
	Freshwater Ecosystems	• Fish • Aquatic invertebrates	Major perennial streams

Table 3.2.1-1 (continued). Final indicator framework, modified after Heinz, showing focal resources, their indicators or measures, and reporting area for each focal resource.

Resource Category	Focal Resource	Indicators of resource	Reporting area
II. Biological Integrity (cont'd)	Forest Ecosystems	• High elevation vegetation (>4,000 ft) • Low elevation vegetation (1,000-4,000 ft) • High elevation bird communities (>4,200 ft) • Low elevation bird communities (1,000-4,200 ft) • Invertebrate communities	Park-wide
	Bog Ecosystems	Bog vegetation	Park-wide
	Cave Ecosystems	• Cave-dwelling invertebrates and other fauna • Vegetation	Park-wide
	Shrubland Ecosystems	• West slope vegetation • Kaupō vegetation • Invertebrates	West slope and Kaupō geographic units
	Sub-alpine Grassland Ecosystems	Grassland vegetation	Park-wide
	Crater	• Vegetation • Invertebrates	Geographic unit
	Nu'u Parcel	• Vegetation • Birds • Invertebrates	Geographic unit
	T&E Species	• Plants • Birds • Invertebrates • Bat	Park-wide
III. Landscape Condition Context	Fuel and Fire Dynamics	• Fuel characteristics • Fire frequency and size	Park-wide
	Soundscape	Natural Quiet	Park-wide
	Viewscape	Dark Night Skies	Park-wide

3.2.2. Reporting Areas

Reporting areas for each focal study resource are shown in Table 3.2.1-1. For water resources, including water quality and freshwater ecosystems, reporting areas are the major perennial streams in the park. For most other resources, reporting areas cover particular ecosystems or geographical units; these are often complementary, because many ecosystems at HALE are spatially restricted to one or a few geographical units. For the remaining resources, reporting areas are park-wide.

3.2.3. General Approach and Methods

Information used to evaluate the current condition and trends in the focal resources in Table 3.2.1-1 included published literature in scientific journals, published and unpublished NPS and other agency reports, published and unpublished raw data, and expert opinion. Raw data were obtained directly from HALE RM and PACN I&M staff, or were extracted from published reports or digital datasets. In most cases, raw data were used to produce new data summaries in order to show patterns or trends not otherwise analyzed or reported in the original presentations, to make comparisons across multiple original information sources, or to provide support for or refute inferences made from other information sources. More complex analyses of raw data were generally not performed, as this is not a function of NRCAs.

Assessments of current resource condition attempted to evaluate the entirety of information available for each resource, placing the greatest weight on reports and analyses supported by robust, comprehensive data sets. In many cases, however, such data are lacking or are very limited in scope, in which case assessments sometimes considered expert observations and opinions if available, made inferences based on comparisons with similar resources at other locales in Hawaiʻi, or concluded that valid assessments are not currently possible. Throughout the assessments in Chapter 4, consistent effort was made to provide a clear narrative of both the sources of information used as well as the rationale for making particular interpretations of that information, and for arriving at overall conclusions. Transparent statements regarding sources and degree of uncertainty and gaps in knowledge are provided.

Trends in the condition of each resource were determined when there were either sufficient temporal data to allow a robust evaluation of trend, or when changes in condition over some time scale were of such magnitude or scope that unambiguous trends could be concluded even when empirical data were sparse or lacking. Otherwise, trends in condition were not addressed.

For each focal resource, we provide an introduction that gives background and context, describe the indicators used to evaluate the condition, and list the sources of information available. We use these information sources, as summarized above, to assess the current condition and, if possible, trend in that condition, for each indicator evaluated for each focal resource. We also discuss the level of confidence in each of these assessments. We use the NPS State of the Park (SotP) categories and symbol set to report and depict resource and indicator conditions, trends, and levels of confidence. The symbol set is shown in Table 3.2.3-1, and examples of how to interpret the symbols are provided in Table 3.2.3-2.

Table 3.2.3-1. Standard NPS categories and corresponding symbols for current condition status, trend in condition, and confidence in assessment.

Condition Status		**Trend in Condition**		**Confidence in Assessment**	
🔴	Warrants Significant Concern	⇧	Condition is Improving	◯	High
🟡	Warrants Moderate Concern	⬄	Condition is Unchanging	○	Medium
🟢	Resource is in Good Condition	⇩	Condition is Deteriorating	◌	Low
No Color	Current condition is Unknown or Indeterminate	No Trend	Trend in condition is Unknown or Not Applicable	–	–

Table 3.2.3-2. Example symbols for assessments and their interpretations.

Symbol	**Assessment**
🟢⇧	Resource is in good condition, its condition is improving, high confidence in the assessment.
🟡⬄	Condition of resource warrants moderate concern; condition is unchanging; medium confidence in the assessment.
🔴	Condition of resource warrants significant concern; trend in condition is unknown or not applicable; low confidence in the assessment.
◌	Current condition is unknown or indeterminate due to inadequate data, lack of reference value(s) for comparative purposes, and/or insufficient expert knowledge to reach a more specific condition determination; trend in condition is unknown or not applicable; low confidence in the assessment.

To arrive at an overall assessment of condition and trend for each focal resource, we used the SotP rules for aggregating the conditions and trends for the multiple indicators of each resource (Tables 3.2.3-1 and 3.2.3-2 and aggregation rules were borrowed directly from NPS NRCA Guidance Update dated January 20, 2014.). The rules for aggregating conditions of indicators into an overall condition for a resource are as follows:

> *To determine the combined condition, each red symbol is assigned zero points, each yellow symbol is assigned 50 points, and each green symbol 100 points. Open (uncolored) circles are omitted from the calculation. Once the average is calculated, apply the scale below to determine the resulting color.*

Table 3.2.3-3. Scores used to determine condition status of resources.

Score 0 to 33	Score 34 to 66	Score 67 to 100
Red[1]	Yellow[2]	Green[3]

[1] Indicates that the resource warrants significant concern.

[2] Indicates that the resource warrants moderate concern.

[3] Indicates that the resource is in good condition.

The rules for aggregating trends of indicators into an overall trend for a resource are as follows:

> *To determine the overall trend, subtract the total number of down arrows from the total number of up arrows. If the result is 3 or greater, the overall trend is up. If the result is -3 or lower, the overall trend is down. If the result is between 2 and -2, the overall trend is unchanged. Sideways trend arrows and cases where trend is unknown are omitted from this calculation.*

The outcomes of these aggregations, however, were subject to modification or adjustment, as per NPS NRCA guidance, based on extenuating information, unequal weighting of particular indicators relative to others, or other overarching judgments made by the assessment team. Any such qualifications are explained in the discussion of the overall condition of each resource. The resultant overall condition, trend, and level of confidence for each focal resource is represented using the SotP standard symbol set at the beginning of each section of Chapter 4. The same symbols are also used to represent the conditions, trends, and levels of confidence for each indicator of each resource in summary tables in Chapter 5.

Chapter 4. Natural Resource Conditions

This chapter presents the individual assessments of current condition and trends for each of the focal resources outlined in the assessment framework (Table 3.2.1-1). For each resource, the following components are included:

- Condition Summary – a synopsis of the findings for current condition and trend of the assessed resource. Includes a cross-reference link to the overall condition discussion near the end of the resource section.
- Relevance and context – describes why the resource is important to the park.
- Indicators of resource condition – characteristics of a resource which can be evaluated to determine the condition and trend in condition of the resource.
- Data and methods – presents indicator data and discusses its evaluation and analysis to determine current condition and trend. Includes sub-sections on sources of information and description of reference condition.
- Current condition and trend, including level of confidence and information gaps – discusses evaluation and analysis of existing data and historical record of indicators within the context of threats and stressors to assess their current condition and trend. Indicator assessments are then rolled up to provide an overall condition and trend of the focal resource with a determination of level of confidence. Includes a cross-reference link to take the reader back to the beginning of the resource section. Gaps in resource information are presented to identify areas of potential research that would benefit the park.
- Literature cited

If applicable, ouside sources of expertise (other than the authors) used to assess resource condition and trend are identified.

4.1. Life-Supporting Environment

4.1.1. Air Quality

Condition Summary

Overall resource condition is good with an improving trend; confidence in condition determination is high. Rationale is discussed in the "*Overall assessment*" section and following.

⇧

Relevance and Context

Air quality affects both the natural resources and the recreational values of National Parks, collectively referred to as Air Quality Related Values (AQRVs) (Sullivan 2016a). A variety of man-made and natural air pollutants impact air quality, and may thereby affect soils, aquatic life, terrestrial plants and animals, and visibility within National Parks (Sullivan 2016a). Under the Organic Act and the Clean Air Act (CAA), NPS is mandated to manage air quality:

> *Accordingly, the Service will seek to perpetuate the best possible air quality in parks to (1) preserve natural resources and systems; (2) preserve cultural resources; and (3) sustain visitor enjoyment, human health, and scenic vistas. … The Service will actively promote and pursue measures to protect these values from the adverse impacts of air pollution. In cases of doubt as to the impacts of existing or potential air pollution on park resources, the Service will err on the side of protecting air quality and related values for future generations. (NPS 2006)*

HALE is categorized under the CAA as a Class I air quality area, the highest designation, giving it the strongest level of protection from air pollutants (Sullivan 2016b). Hawaiʻi Volcanoes National Park (HAVO) is the only other Class I unit within PACN (Sullivan 2016b). Located on a remote island distant from large population centers, HALE generally experiences very good air quality. However, HALE is sometimes affected by volcanic emissions from Kīlauea Volcano, approximately 100 miles away on the adjacent Hawaiʻi Island. These emissions include sulfur dioxide (SO_2), sulfuric acid, and sulfate particles. Although HALE is far enough from Kīlauea to avoid dangerous levels of these gasses and particles, the volcanic smog, or "vog", that is produced can extend to Maui and increase haze at HALE. This occurs most often during southeasterly "kona" wind interruptions to the prevailing trade wind pattern. In addition, local sources of anthropogenic pollution, including power plants, vehicles, and sugar cane and other agricultural production can impact air quality at HALE (NPS ARD 2016a).

The NPS Air Resources Division (ARD) identifies five categories of indicators for assessing condition and trends in air quality: ground-level ozone (O_3) concentration, total nitrogen (N) and sulfur (S) wet deposition, visibility (Taylor 2017). All of these can impact AQRVs. This NRCA focuses on the three indicators (O_3, N and S wet deposition, visibility) that were previously recommended for air quality evaluation (NPS ARD 2013). Ozone is a respiratory irritant that can trigger a variety of human health problems including chest pain, coughing, throat irritation, and congestion. Ozone can also reduce plant growth and vigor and cause tissue damage, with some species being more sensitive to exposure than others. Nitrogen and S deposition can lead to acidification, and N deposition can also enrich soil and water nutrients, which may promote growth of invasive plants or cause other undesired changes to biological community composition and ecosystem function. Reductions in visibility from haze, primarily caused by sulfate (SO_4^{2-}) emissions from both natural and human sources, obscure scenic views in National Parks (Sullivan 2016b). Concentration of SO_2 in the air is an important additional indicator of air quality at HAVO owing to the heavy localized output of this gas from the active volcano, but SO_2 concentration is not measured at HALE because it does not reach levels of human health concern (NPS ARD 2016a).

Measures

- Ground-level ozone
- Wet deposition of N and S
- Visibility

Data and Methods

This assessment considers conditions and trends of the three key air quality indicators listed above.

Sources of information

Information on ground-level ozone was obtained from Kohut (2004), Sullivan (2016b), and NPS ARD (2016a, b). Information on N and S wet deposition was obtained from Sullivan (2016b) and NPS ARD (2016a, b). Information on visibility was obtained from Sullivan (2016b), NPS ARD (2016a, b) and the Interagency Monitoring of Protected Visual Environments (IMPROVE) Network website (IMPROVE 2016).

Reference condition

NPS ARD has defined condition categories for the three air quality indicators listed above, specifying ranges of values that can be considered good condition, warranting moderate concern, or warranting significant concern (Taylor 2017). These are shown in Table 4.1.1-1, with ground-level ozone split into two component metrics for human and plant health, respectively. As the NPS strives to attain the best possible air quality in parks, this assessment considers air quality values meeting Good Condition criteria as the reference condition. For human health effects from ground-level ozone, this corresponds to concentrations below 55 parts per billion, assessed as the five year average of the 4th-highest daily maximum concentration. For vegetation effects from ozone, this corresponds to a 3-month maximum 12-hr W126 of <7 ppm-hrs (the W126 statistic is a biologically relevant metric that preferentially weighs ozone concentrations most likely to affect plants, and is a better predictor of vegetation response than the human health standard (Taylor 2017)). For wet deposition of N and S, this corresponds to rates less than 1 kg per hectare per year. For visibility, this corresponds to a difference of less than 2 deciviews (dv) between average current visibility and average natural visibility, where average visibility is calculated as the mean of values between the 40th and 60th percentiles (Taylor 2017).

Table 4.1.1-1. Condition categories for air quality indicators defined by Taylor (2017).

Condition	Symbol	Ozone Human Health (ppb)[1]	Ozone Plant Health (ppm-hrs)[2]	Wet deposition (kg/ha/yr)	Visibility condition (dv)[3]
Warrants Significant Concern	🔴	≥71	>13	>3	>8
Warrants Moderate Concern	🟡	55-70	7-13	1-3	2-8
Resource is in Good Condition	🟢	≤54	<7	<1	<2

[1] The five year average of the 4th-highest daily maximum concentration

[2] Calculated as the 3-month maximum 12-hr W126

[3] Calculated as average visibility – estimated average natural visibility condition

Condition and Trend

Ground-level ozone

The ARD's condition assessment for human health risk from ozone is directly related to the U.S. Environmental Protection Agency's (EPA) human health-based primary National Ambient Air Quality Standard. Risk assessment for ozone damage to plants is based on three general parameters: ozone sensitivity of plants, level of ozone exposure, and the soil moisture status of the environment. The latter variable influences stomatal condition of plants, and therefore levels of gas exchange and ozone absorption (Kohut 2004).

An assessment of ozone sensitivity of plants at HALE identified only four species known to be sensitive, all in the family Pinaceae: *Pinus jeffreyi* (Jeffrey pine), *Pinus ponderosa* (ponderosa pine), and *Pinus radiata* (Monterey pine) and *Pinus strobus* (eastern white pine) (Kohut 2004, Sullivan 2016b, NPSpecies 2018). None of these species are native to HALE or to Hawai‘i (Wagner et al. 1999).

Ozone data were not available for HALE, so Kohut (2004) could not assess ozone exposure.

The US Department of Agriculture's Palmer Z Index is a standardized metric of soil moisture status that can be used to evaluate the ozone exposure environment. This index was not calculated for Pacific Islands, and therefore this parameter also could not be assessed for HALE (Kohut 2004). Generally, soil moisture levels in HALE vary strongly across ecosystem types, as mean annual rainfall in the park is estimated to range from 500 to over 10,250 mm/yr (Giambelluca et al. 2013). Plants occurring in areas with more favorable soil water status can be expected to absorb more ozone than those in drier areas.

Due to lack of data on two key parameters, especially background ozone concentrations, NPS ARD was unable to assess risk of ozone injury to plants and human health at HALE, and was unable to assess current condition or trend of this measure of air quality (NPS ARD 2016b). There is insufficient information to assess condition and trend of ground-level ozone at HALE at this time.

Wet deposition of N and S

Airborne pollutants are eventually deposited through either wet deposition (i.e., rain, snow, clouds, and fog) or dry deposition (i.e., particles and gases) onto vegetation, soils, streams, and lakes. The deposition of N and S compounds can damage ecosystems through both acidification and through nutrient enrichment of soils and eutrophication of aquatic environments (Sullivan et al. 2011, NPS ARD 2016a).

Ecosystems vary in their sensitivity to acidification, with higher sensitivity often occurring in higher elevations and on steeper slopes with base-poor soils. These characteristics reduce the amount of acid neutralization that can occur in the soil. In addition, some types of plants, such as mosses and lichens, are known to be more sensitive to acidification (Sullivan et al. 2011). There are six pine and cedar species known to be acid-sensitive at HALE (NPSpecies 2018), but none are native to Hawaiʻi (Wagner et al. 1999). Using a quintile relative ranking system among all I&M Parks, Sullivan (2016b) determined that ecosystem sensitivity to acidification at HALE was Very High, the highest category. This was based on the relatively steep terrain slopes in the park, averaging between 10° and 20°, which increases potential for acidification of aquatic systems, and on the park's high diversity of mosses, a plant group known to be higher in acid sensitivity. The >100 species of mosses documented in and around HALE (Waite 2007) may be especially vulnerable to N, S and heavy metal deposition (Sullivan 2016b).

Steep slopes at HALE may promote rapid run-off with little time to buffer acidic ions, contributing to the susceptibility to acidification (NPS ARD 2016a). HALE freshwater systems may be particularly sensitive to sulfur compound emissions from Kīlauea Volcano (NPS ARD 2016a). A baseline water quality study reported pH values of less than 6.5 between 1972 and 1995 in the park's freshwater streams (NPS WRD 1999). Low pH may result from atmospheric deposition of S from Kīlauea's emissions, or from naturally-occurring S in bedrock (NPS ARD 2016a).

Hawaiian ecosystems are also known to respond to nutrient enrichment, sometimes resulting in the promotion of invasive plant species to the detriment of native plants (Vitousek and Walker 1989, Ostertag and Verville 2002). This is most likely to occur on younger, N-limited soils, such as those in subalpine shrubland and alpine communities (NPS ARD 2016a).

No recent data for N and S wet deposition are available for HALE. For example, there are no National Atmospheric Deposition Program monitoring sites on Maui. However, anthropogenic N and S emission levels are generally low for Maui County. Data from the EPA's National Emissions Inventory indicate that for the year 2011, emissions of SO_2 were 1-5 tons/mi^2/yr, emissions of nitrogen oxides (NO_x) were 5-25 tons/mi^2/yr, and emissions of ammonia (NH_3) were 2-8 tons/mi^2/yr (Sullivan 2016b). These emission levels are unlikely to lead to high rates of N and S deposition at HALE, and Sullivan (2016b) concluded that "There are no data suggesting that terrestrial or aquatic resources in the PACN region are affected at this time by nutrient enrichment caused by atmospheric N deposition at current atmospheric loading rates."

Despite the indications and expectations of low N and S deposition at HALE, there are currently no data with which to evaluate current condition and trend of this air quality measure (NPS ARD 2016b).

Visibility Condition

The 1977 CAA amendments set a specific goal for visibility protection in Class I areas, such as HALE, that mitigate and prevent visibility impairment from human sources (42 U.S.C. 7491). To further the visibility goal, in 1999, EPA promulgated the Regional Haze Rule (RHR), which requires each state to develop a plan to improve visibility in Class I areas, with the goal of returning visibility to natural conditions, i.e., no human-caused impairment, by 2064. The RHR requires improved visibility on the haziest days, with no degradation on the clearest days. If a state does not make reasonable progress in achieving visibility improvement, it may be required to implement additional pollution reduction strategies.

Visibility condition is reported with the Haze Index, in units of deciviews (dv), with lower values corresponding to better visibility. Visibility in National Parks is typically evaluated in terms of clear days (<20th percentile), hazy days (>80th percentile), and average days (40th-60th percentiles). Clear day visibility must not deteriorate from a recent baseline (2000-2004) value under the RHR (Sullivan 2016b).

NPS monitors visibility at HALE as part of the Interagency Monitoring of Protected Visual Environments (IMPROVE) Network. An IMPROVE monitor operated downslope and west of HALE (HALE1) from 2001 to 2012, and a second monitor (HACR1) was installed in the park in 2007 in the Crater District (IMPROVE 2016).

Based on the newer HACR1 monitor, the largest contributor to haze at HALE is ammonium sulfate, especially on hazy days (Figure 4.1.1-1). This compound forms in the atmosphere from SO_2 emissions, which at HALE likely originates mainly from Kīlauea Volcano on Hawaiʻi Island (Sullivan 2016b). The effects of this haze on the HALE crater viewshed are evident in Figure 4.1.1-2.

Taylor (2017) recommends assessing current visibility condition in terms of the difference between the observed visibility on average days and the estimated natural visibility on average days. The estimated natural visibility (assuming no human-caused pollutants) on average days at HACR1 is 1.8 dv (NPS ARD 2016b). The resulting visibility condition was calculated to range from 1.9 to 2.4 dv between 2009 and 2015 (Table 4.1.1-2), the years for which data were available (NPS ARD 2016b). The 2015 values indicate a good condition while 2009 to 2014 values all Warrant Moderate Concern (Table 4.1.1-1, NPS ARD 2013).

Table 4.1.1-2. Visibility condition at HALE measured at HACR1 monitor station, calculated for years 2009-2015.

Year	Visibility Condition (dv)*
2009	2.4
2010	2.4
2011	2.3
2012	2.4
2013	2.4
2014	2.2
2015	1.9

* Calculated as visibility on average days over 5 prior years minus estimated natural visibility on average days

Trends in visibility at HALE indicate no significant change in the Haze Index on clear days from 2007-2015, but a significant trend of improving visibility on hazy days from 2007-2015 (Figure 4.1.1-3; IMPROVE 2016). The latter indicates that haze in recent years has decreased subsequent to the relatively poor visibility conditions prevalent during 2008-2010, which were caused by high levels of emissions from a new vent that opened in Kīlauea at that time (Elias and Sutton 2012). An improving visibility trend on hazy days, combined with a lack of deteriorating trend on clear days, suggests an overall improving trend in visibility condition at HALE in recent years. It remains to be seen how the 2018 Kīlauea eruption will affect the trend.

These trends suggest that HALE appears to be on track to meet the goals of the RHR. However, it is important to note that ambient visibility conditions at HALE are largely determined by volcanic activity at Kīlauea (Figure 4.1.1-1), which is inherently variable and unpredictable. For example, assessments of trends in visibility conditions over the periods 2003-2012 and 2005-2014 found no significant changes (NPS ARD 2016b). While volcanic emissions are beyond the control of the state, NPS ARD recommended that SO_2 limits be established for several power generating plants on Maui (Sullivan 2016b).

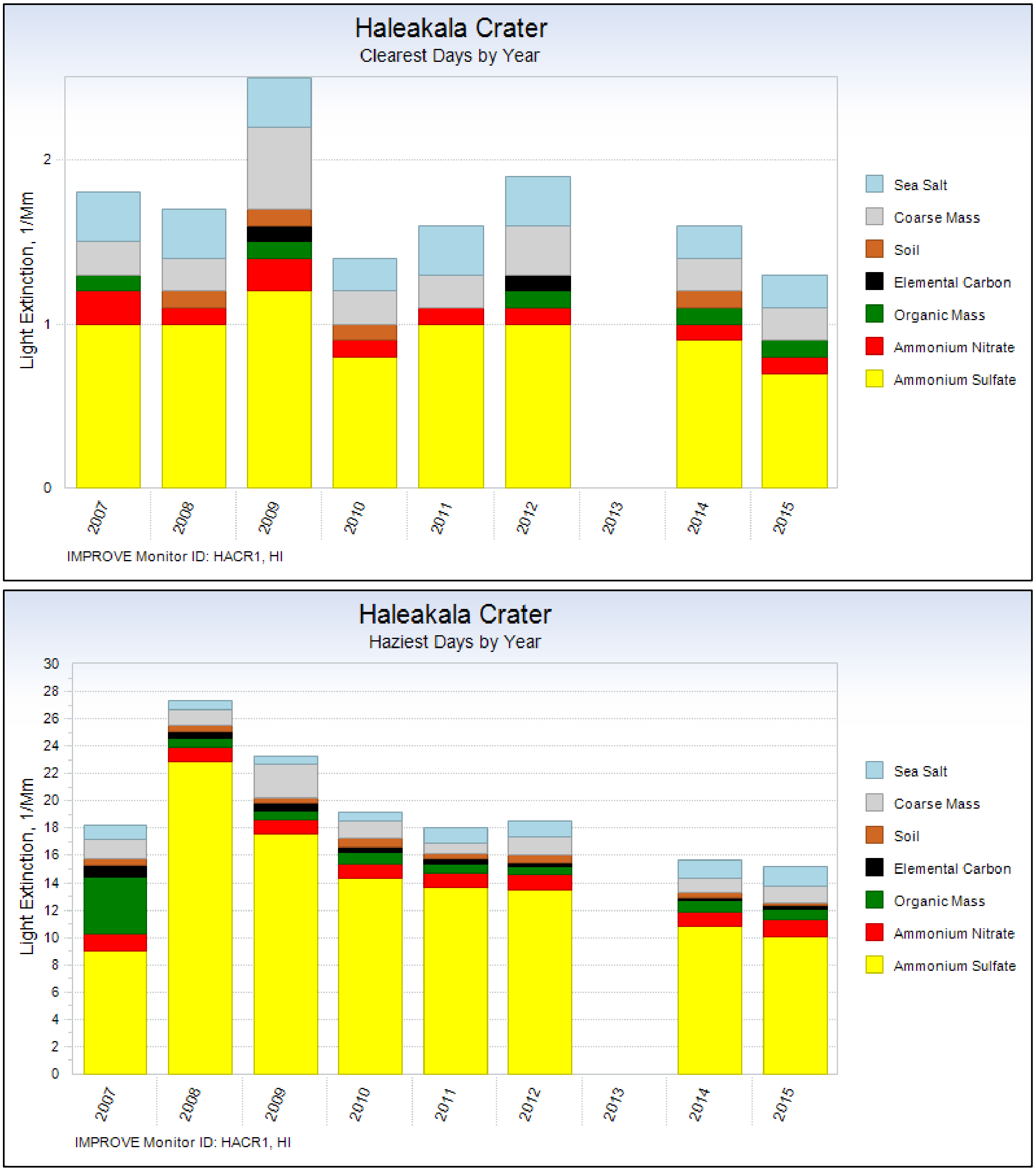

Figure 4.1.1-1. Composition of haze at HALE on clear days (top) and hazy days (bottom), measured at HACR1 monitor station. Haze measured in units of Light Extinction (b_{ext}), with higher values corresponding to more haze and poorer visibility; deciviews (dv) are proportional to log(b_{ext}). Source: IMPROVE (2016).

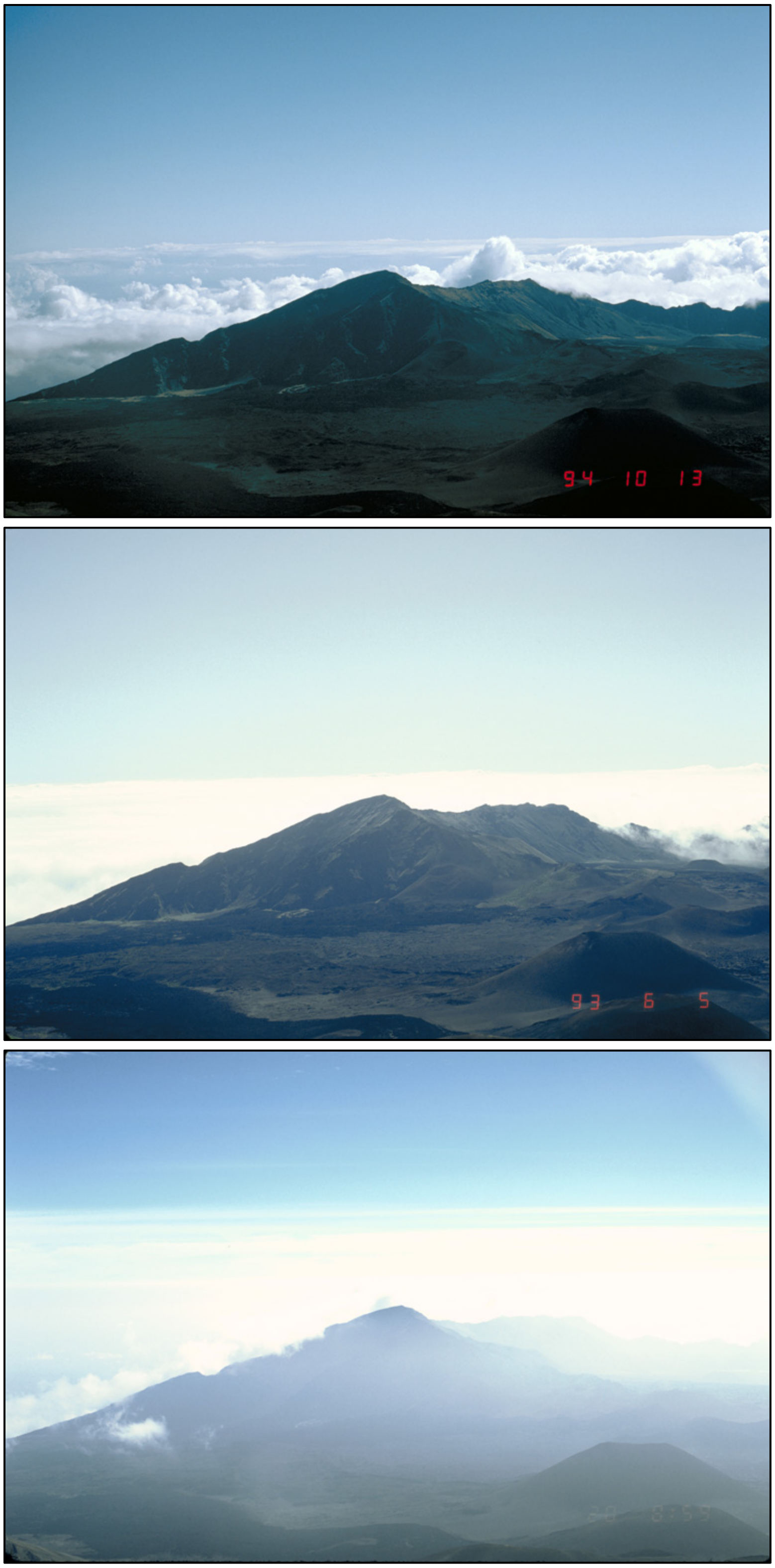

Figure 4.1.1-2. Representative views of Haleakalā crater on a clear day (top), average day (middle), and hazy day (bottom). Source: IMPROVE (2016).

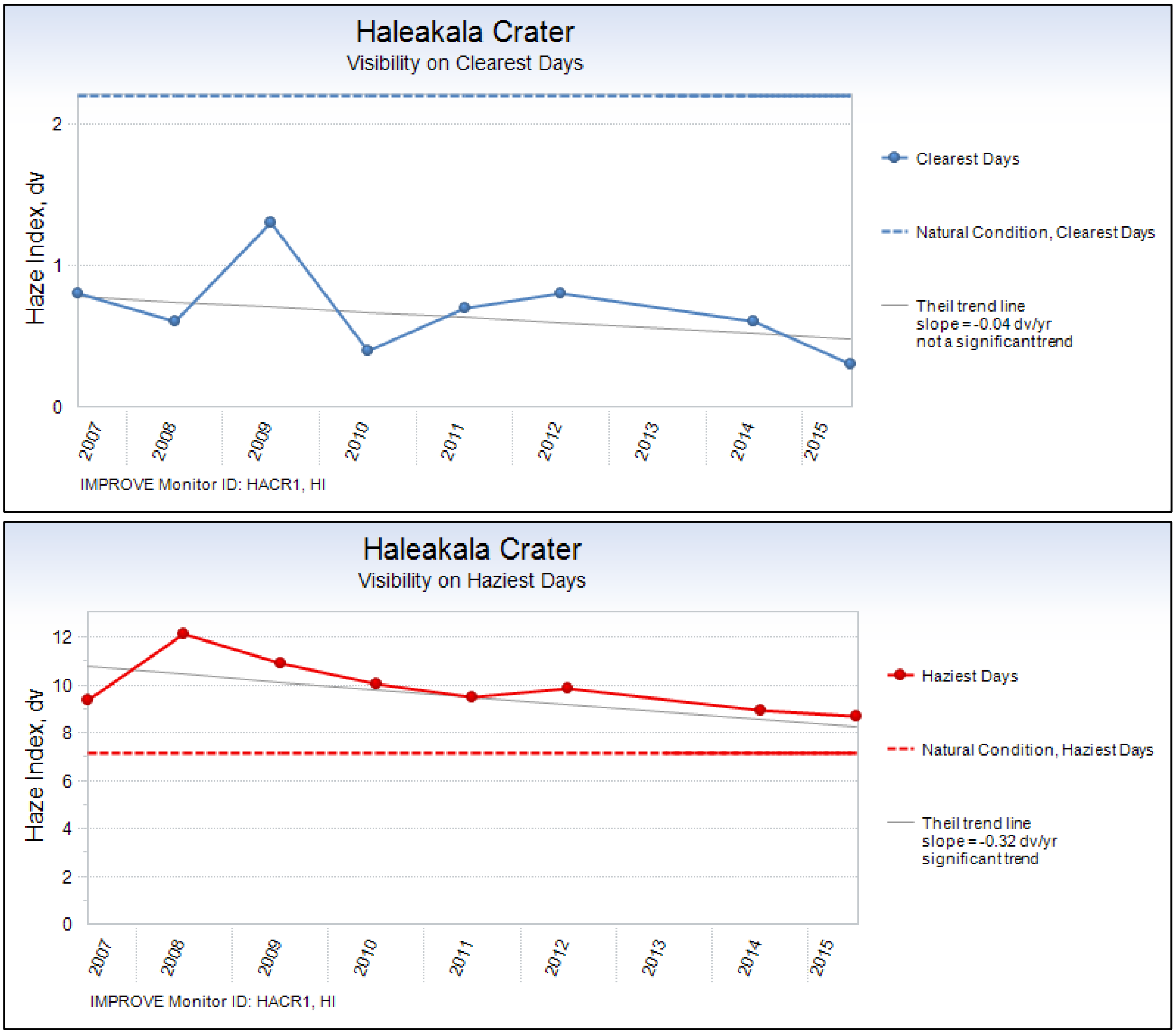

Figure 4.1.1-3. Trends in visibility at HACR1 monitoring site from 2007-2015 on clear days (top) and hazy days (bottom). There was no significant trend on clear days, but a significantly improving trend on hazy days. Source: IMPROVE (2016).

Overall assessment

Visibility condition is the only indicator of air quality that can be assessed for HALE at this time, resulting in a good overall current condition based on the latest available data, with an improving trend in air quality. The limited information available for the other two indicator categories, ozone risk and N and S deposition, generally does not raise concern that this assessed condition may need to be down-graded. For example, NPS ARD (2016a) stated that past monitoring found relatively low levels of ozone in the park that would be unlikely to cause ozone injury to plants.

Level of confidence

Confidence in the current air quality condition is high, because visibility data are collected on-site and therefore highly reliable. Similarly, we assign a high level of confidence to the trend assessment, but note that future visibility trends may change according to levels of volcanic activity on Hawai‘i

Island. Trends in ozone risk and N and S deposition are unknown for HALE. (return to Condition Summary)

Information gaps and research recommendations

Data on ozone concentrations at HALE are not collected, and would provide a clearer picture of threats to human health and plants. There is also very little information on N and S deposition at HALE. Ecosystem sensitivity to acidification was assessed on a relative ranking scale, which provides a very general indication of risk, but more comprehensive park-specific data on ecosystem sensitivity is needed for more precise estimation of risk from acidification (Sullivan 2016b). Given the high estimated ecosystem sensitivity, collection of on-site deposition data is recommended.

Sources of Expertise

Tonnie Cummings, Air Resources Specialist in the NPS, Pacific West Region, contributed to this chapter.

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4.1.2. Soil Quality

Condition Summary

Overall resource condition warrants moderate concern with an unchanging or stable trend; confidence in condition determination is low. Rationale is discussed in the "*Overall assessment*" section and following.

↔

Relevance and Context

Soils are the foundations of terrestrial ecosystems, and soil quality is an important determinant of the health of these ecosystems. NPS defines soil as "the unconsolidated portion of the earth's crust modified through physical, chemical, and biotic processes into a medium capable of supporting plant growth" (NPS 2004, NPS SRI 2017). The age of the basalt parent material varies considerably across HALE, which along with climatic gradients, strongly affects the degree of soil development and the biomass of vegetation that can be supported. Haleakalā Volcano was formed in three major phases of volcanism: the Honomanū shield-building phase (~1.1 to 0.75 million years ago), the Kula phase (930,000 to 150,000 years ago), and the Hāna phase (120,000 to present) (Thornberry-Ehrlich 2011). Regions of the park situated on the outer flanks of the volcano are much older, generally >140,000 years, while areas within the Crater and Kīpahulu Valley are characterized by more recent activity and are generally younger than 30,000 years, often considerably so (Figure 4.1.2-1, Sherrod et al. 2007).

Soil types have been mapped across East Maui by the US Department of Agriculture's (USDA) Natural Resources Conservation Service (NRCS), which have been adopted by the NPS Soil Resources Inventory (SRI) program for producing a Soil Survey Geographic (SSURGO) map for HALE (NPS 2005). An updated map, including the park's more recently acquired Nu'u Parcel, includes 17 recognized soil types (Table 4.1.2-1). Distributions for the seven most widespread soils, which together comprise over 96% of the park area, are shown in Figure 4.1.2-2.

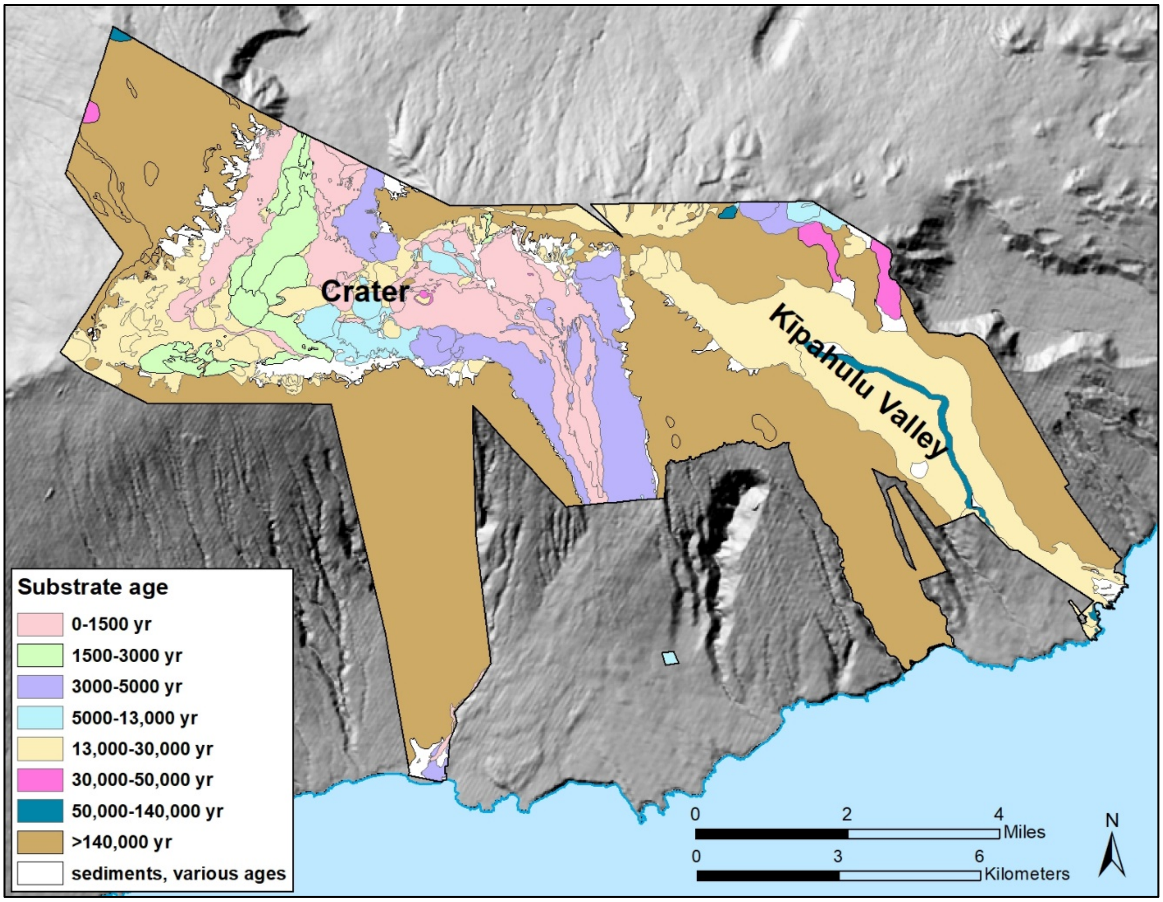

Figure 4.1.2-1. Ages of geological substrates at HALE. Source: Sherrod et al. (2007).

Table 4.1.2-1. Soil types within HALE. Source: SSS NRCA USDA (2017).

Soil type	**Area (ha)**	**% of park**
Very stony land	3,728.0	27.3
Hydrandepts-Tropaquods association	2,886.4	21.2
Rock outcrop	2,284.2	16.7
Cinder land	1,834.3	13.4
Rough mountainous land	1,767.2	13.0
Puu Pa very stony silt loam, 7 to 40 percent slopes	336.4	2.5
Lava flows, aa	288.5	2.1
Makaalae silty clay, 7 to 25 percent slopes	204.2	1.5
Waiakoa extremely stony silty clay loam, 3 to 25 percent slope, eroded	92.3	0.7
Makaalae extremely stony silty clay, 7 to 25 percent slopes	46.8	0.3
Stony alluvial land	46.6	0.3
Laumaia extremely stony loam, 7 to 40 percent slopes	45.0	0.3
Rock land	27.7	0.2

Table 4.1.2-1. Soil types within HALE. Source: SSS NRCA USDA (2017).

Soil type	**Area (ha)**	**% of park**
Laumaia loam, 40 to 70 percent slopes	26.8	0.2
Kaipoioi loam, 7 to 40 percent slopes	17.2	0.1
Kaupo very stony silty clay loam, 3 to 25 percent slopes	7.0	<0.1
Makaalae clay, 7 to 40 percent slopes	5.5	<0.1

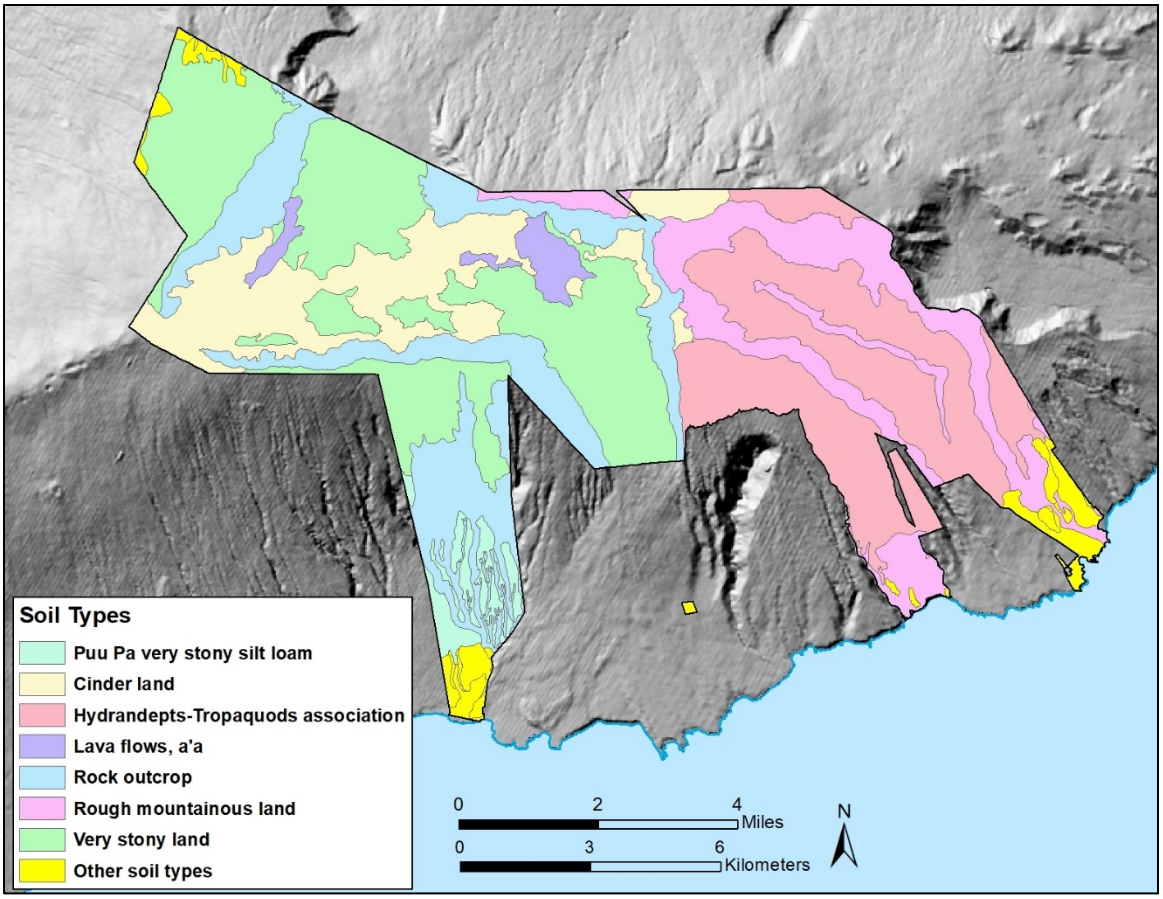

Figure 4.1.2-2. Major soil types in HALE. Source: SSS NRCA USDA (2017).

Much of the drier and higher elevation regions of the park, including the Crater, are classified as Very stony land, Rock outcrop, Cinder land or Lava flows (Figure 4.1.2-2). These areas have somewhat to very poorly developed soils, and generally support shrublands and very sparse vegetation. Cinder land areas in the crater and summit have experienced minimal weathering, and these soils are very porous and nutrient limited (Pérez 2001). Decomposition of sparse pioneer plants, such as Haleakalā ʻāhinahina (silverswords, *Argyroxiphium sandwicense* subsp. *macrocephalum*), substantially augments both the nutrient content and water-holding capacity of surrounding soils, promoting further vegetative growth in these otherwise barren areas (Pérez 2001).

Eastern regions of the park receive much higher rainfall (Giambelluca et al. 2013, Figure 2.2.1-1), and these generally older substrates have weathered to form deeper soils that support forests.

Soil quality can be defined as the continued capacity of soil to function as a vital living ecosystem that sustains plants, animals, and humans (USDA NRCS 2017). The NRCS further defines healthy soils as those which perform five essential functions: regulating water, sustaining plant and animal life, filtering and buffering potential pollutants, cycling nutrients, and providing physical stability and support (USDA NRCS 2017). A variety of different indicators can be used to assess soil quality, including measures of soil organic matter, as well as physical, chemical and biological attributes of soil (USDA NRCS 2017). Each of these categories of indicators, and individual indicators within them, will be more or less relevant to different settings. For example, many metrics relevant to highly modified and frequently disturbed agricultural soils will often be less important within National Parks.

Among the most important processes affecting the geology and soils of HALE is erosion, through both water- and wind-generated movement of soil (Thornberry-Ehrlich 2011). Land use practices, and the presence of non-native ungulates and plants, can influence the rates of both types of erosion (Street 1988, Thornberry-Ehrlich 2011), and thereby influence soil quality. Contamination of soils is another potential factor influencing soil quality at HALE. While the risk of point source contamination of soils by pollutants is generally low, owing to the remote nature of most of the park, atmospheric deposition of nitrogen (N) and sulfur (S) can lead to soil acidification and nutrient enrichment, which may promote growth of invasive plants or cause other undesired changes to biological community composition and ecosystem function (Sullivan et al. 2011). Such deposition results from airborne anthropogenic pollutants generated by power plants, vehicles and other activities. HALE is also sometimes affected by volcanic emissions from Kilauea Volcano on the adjacent Hawaiʻi Island, and this "vog" includes sulfur dioxide, sulfuric acid, and sulfate particles (NPS ARD 2016), increasing the risk of S deposition on HALE soils. However, the latter process is natural, and thus should not be considered contamination. Similarly, native animals are a natural source of N addition to the soil, while the same input from non-native animals may be viewed as contamination.

These two indicators of soil quality, erosion and contamination, are also highlighted in the NPS Natural Resource Management Reference Manual #77 guidelines for soil management policies (NPS 2004): "The Service will actively seek to understand and preserve the soil resources of parks, and to prevent, to the extent possible, the unnatural erosion, physical removal, or contamination of the soil, or its contamination of other resources." We therefore assess the current condition of soil quality at HALE using these two indicators.

Indicators

- Erosion
- Soil contamination

Data and Methods

A review of available data and information was used to assess conditions, and if possible, trends, in the indicators listed above.

Sources of information

Information on stream turbidity, as a metric of erosion, was obtained from Raikow and Farahi (2016), and information on the status of soil erosion in Nuʻu was taken from NPS (2016). Information on N and S wet deposition was obtained from Sullivan (2016).

Reference condition

Rates of erosion should ideally be within the natural range of variation for a given habitat type. Because such ranges are unknown at HALE, we consider good erosion dynamics to be an absence of apparently high or unusual degrees of soil loss that can be tied to unnatural causes or processes. Ideally, there should be no soil contamination stemming from anthropogenic pollutants or non-native animals.

Condition and Trend

Erosion

There are no programs, or associated data, to monitor rates of soil erosion at HALE (Rutherford and Kaye 2006). In most park regions, however, erosion outwardly appears to be a relatively minor problem, likely due in large part to the removal of non-native ungulates, whose digging and trampling of soil and vegetation greatly increases rates of erosion. Thornberry-Ehrlich (2011) judged that "On the windward, moisture-laden side of the island, flowing water is the primary erosive force. Erosion there has been exacerbated by anthropogenic changes to the landscape, including introduction of feral ungulates and invasive plant species." Non-native ungulate soil disturbance increases sediment loads in streams, reducing water quality and impacting aquatic ecosystems. Although information on the levels of total suspended solids in HALE streams is lacking, turbidity values in these streams are generally good (Raikow and Farahi 2016, see Chapter 4.1.3), suggesting that erosion in wet forest areas of the park is not currently a large problem.

In drier areas of HALE, wind is an important additional agent of soil transport. "In the arid portions of the park (high elevation areas and the leeward flanks), land use practices can increase the amount of sediment available to prevailing winds. Once disturbed, stabilizing vegetation is slow to reestablish itself' (Thornberry-Ehrlich 2011). The exclusion of non-native ungulates from the Crater and high elevation shrublands has generally resulted in recovery of native vegetation in these areas (see Chapters 4.2.6 and 4.2.8), and has eliminated trampling and digging of the ground by these animals. Although no measurements are available, erosion is not a major apparent problem throughout most of these areas.

One area that has clear and severe erosion problems is the newly acquired Nuʻu Parcel (NPS 2016, see Chapter 4.2.9). This area was long used for cattle ranching, and remains unfenced and still supports large numbers of feral ungulates, especially goats (*Capra hircus*) (NPS 2016). These animals, and the associated land-use practices, have eliminated nearly all of the forest that previously occurred there, and have also strongly degraded remaining areas of shrubland habitat. This reduction in vegetative cover and the continued ground disturbance caused by animals has led to the loss of massive amounts of soil from the steep slopes, likely through both water and wind transport (Figure 4.1.2-3).

Figure 4.1.2-3. Example of severe erosion on the leeward slopes of the Nu'u Parcel. Photo: C. Chimera.

In the 1980's, Medeiros et al. (1986) commented that of the entire southern flank of Haleakalā, "The middle and upper slopes of Nuʻu appear to present as spectacular an example of accelerated erosion due to goat browsing as exists anywhere." Not only does this soil loss inhibit regeneration of native vegetation, but also degrades nearshore marine environments. The Environmental Assessment for the management and removal of feral animals at Nuʻu (NPS 2016) stated that up to 5-8 ft of top soil, leaf litter and forest duff have been lost, although we could not find any original figures associated with that estimate. An ungulate exclusion fence is now being constructed at Nuʻu, after which goats and other non-native ungulates will be removed. It is anticipated that vegetation will recover relatively quickly (Chapter 4.2.9), which should greatly slow the rate of soil erosion in this area.

The Kaʻāpahu portion of the park is also currently unfenced and unmanaged. Although erosion in Kaʻāpahu is not as severe as in Nuʻu, feral pigs and goats have caused the loss of considerable quantities of topsoil in this area (P. Welton pers. comm. 2018). A small hog panel exclosure was built by the Vegetation Management Program in 2016, with an aim to stabilize the soil in preparation for outplanting of endangered *Cyanea asplenifolia* plants.

We assess the current status of erosion dynamics at HALE to warrant moderate concern. While the situation in the vast majority of the park appears to be good, erosion problems are currently severe in Nuʻu and substantial in Kaʻāpahu. We assign an unchanging trend in this condition, but anticipate that it will begin improving shortly after feral ungulates are removed from Nuʻu. Confidence in this assessment is low, owing to the lack of any quantitative data on rates of erosion in the park.

Soil contamination

There are no programs, or associated data, to monitor soil contaminants at HALE. However, there are few obvious point sources for contamination resulting from chemical or other anthropogenic pollutants. In addition to the built-up areas at park headquarters, the summit region, the road connecting them, and the Kīpahulū District visitor center, there are only two non-wilderness and three wilderness cabins and/or camping areas open to the public. Although regular monitoring of soils does not occur at these sites, these limited facilities pose little risk of contaminating soils beyond their immediate vicinities.

Similarly, there are no recent data on rates of atmospheric deposition of N and S for HALE (see Chapter 4.1.1). Sullivan (2016) determined that ecosystem sensitivity to acidification from such deposition was Very High at HALE (the highest category), owing to the relatively steep terrain slopes in the park, which increases potential for acidification of aquatic systems, and to the park's high diversity of mosses, a plant group known to be higher in acid sensitivity. However, anthropogenic N and S emission levels are generally low for Maui County, and are unlikely to lead to high rates of N and S deposition in the park (see Chapter 4.1.1). Sullivan (2016) concluded that there are no data suggesting that terrestrial or aquatic resources in the PACN region are currently affected by nutrient enrichment caused by atmospheric N deposition.

Because feral ungulates have now been excluded from large areas of the park, N inputs from these non-native animals should be low to zero in these areas. As noted above, the major exceptions to this are the Nuʻu and Kaʻāpahu regions, which is currently unfenced and is home to large numbers of goats and other feral animals. Pigs have also infiltrated certain wet forest areas in recent years (see Chapter 4.2.3). Fecal waste from these animals could lead to unnatural nutrient enrichment of soils in these areas, which can favor the growth of invasive plant species over native plants in some situations (Vitousek and Walker 1989, Ostertag and Verville 2002). However, soil nutrient levels are unmeasured at HALE, so it is unclear whether this is a significant threat to soil quality.

In summary, although it appears highly unlikely that soil contamination is a major issue at HALE, there is insufficient information with which to make an assessment of the current condition or trend of this indicator.

Overall assessment

Soil erosion condition is the only indicator of soil quality that can be assessed for HALE at this time, resulting in an overall current condition that warrants moderate concern, with an unchanging trend.

As stated above, this assessment results principally from the severe erosion issues at Nu‘u, which are expected to improve following completion of the ungulate exclusion fence and subsequent removal of animals. Otherwise, there are no reasons to suspect substantial problems with erosion, soil contamination, or overall soil quality in most of the park. (return to Condition Summary)

Level of confidence

Confidence in the current soil quality condition is low, because while there are no obvious reasons to suspect soil quality problems, actual data to confirm this are lacking.

Information gaps and research recommendations

There are no monitoring programs or data on rates of erosion, soil contamination, or other metrics of soil quality at HALE.

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4.1.3. Water Quality

Condition Summary

Overall resource condition is good with an unknown trend; confidence in condition determination is medium. Rationale is discussed in the "*Overall assessment*" section and following.

●

Relevance and Context

Water is a critical component of all natural ecosystems, being among the most important requirements for sustaining life. Abundant rainfall across the windward, eastern half of HALE (see Figure 2.2.1-1) produces numerous intermittent surface streams that feed into several major perennial streams in the park, including ʻAlelele, Pīpīwai and Palikea Streams (Figure 4.1.3-1). Altogether, HALE encompasses 36.0 miles (57.9 km) of intermittent streams, 9.2 miles (14.8 km) of perennial streams, and three small, perennial montane lakes (Waiʻānapanapa, Waiʻeleʻele and Wai nēnē) totaling 0.3 ac (0.12 ha) in area (NPS WRD 2016a).

In addition to the ecological importance of these surface waters, they contribute to the aesthetic and recreational values of HALE. For example, ʻOheʻo Gulch, the name typically used for the lower reaches of Palikea Stream below its confluence with Pīpīwai Stream, and which contains a series of dramatic pools near its entry to the Pacific Ocean, is a primary viewing and swimming attraction (Figure 4.1.3-2) for the 300,000-600,000 visitors each year to the park's Kīpahulu District (Figure 2.1.3-2).

The importance of water to the health of living organisms, including humans, has led to the passage of a variety of laws and regulations instituting its protection at levels ranging from local to federal. National Parks are subject to many of these laws, such as the federal Clean Water Act (CWA) of 1977, but also have promulgated a number of policies directed at protecting water resources of all types within parks (DeVerse and DiDonato 2006, NPS WRD 2016b). Water resources were recognized as a key element to be addressed in the NPS Natural Resources Challenge (NRC), and several Vital Signs related to water were chosen for monitoring as indicators of water resource condition within PACN (Jones et al. 2011). Two of these correspond to focal resources identified for assessment in this NRCA: water quality, and the biological communities supported by freshwater ecosystems.

Water quality, defined here as the physical, chemical and microbial condition of water resources, is addressed in the present section, while the condition of aquatic communities is covered in a later section devoted to freshwater ecosystems (Chapter 4.2.8). Furthermore, water quality can be assessed for ground water in addition to surface water. Ground water may re-emerge to feed surface hydrology, or may percolate down to recharge freshwater aquifers or marine waters (Jones et al. 2011). Although the quality of this ground water also influences ecological health in National Parks and surrounding lands, it is not monitored at HALE and this assessment is restricted to quality of surface waters.

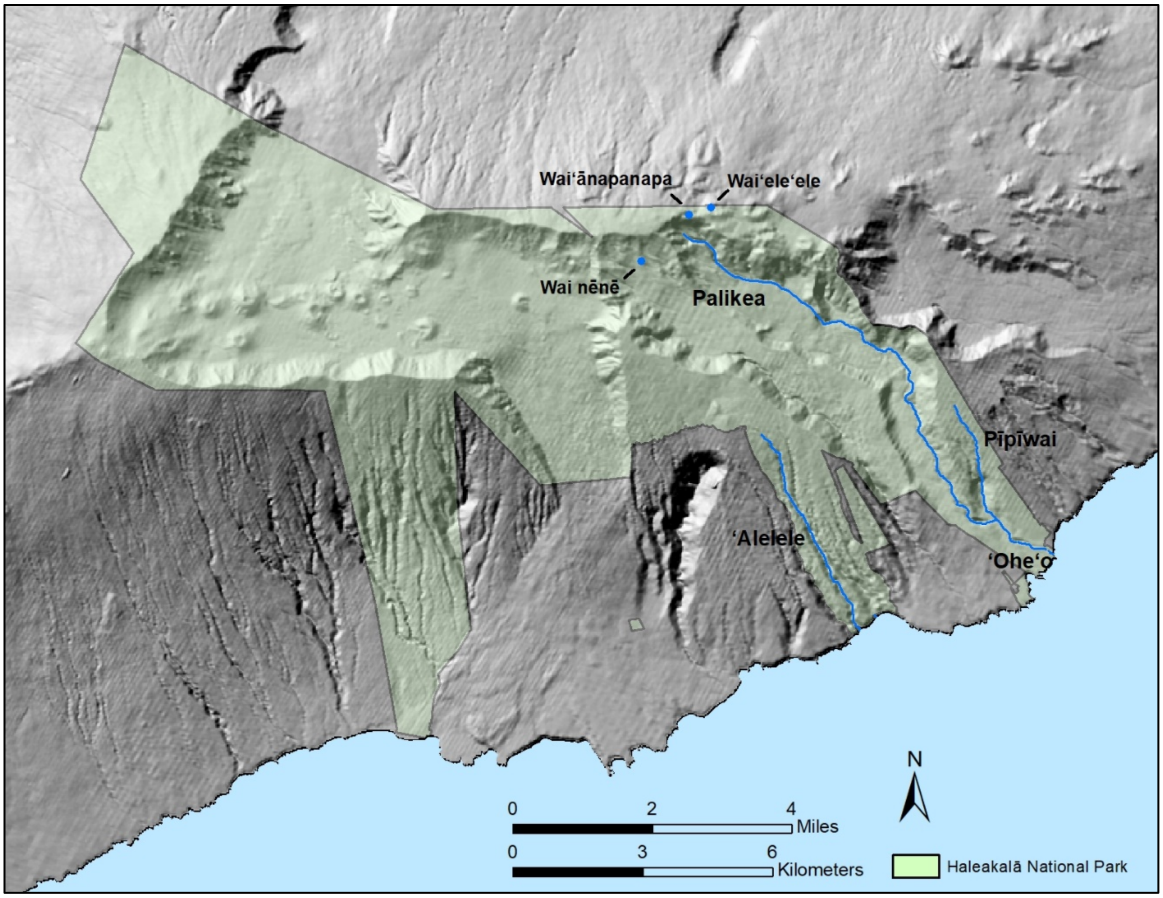

Figure 4.1.3-1. Major perennial surface waters in HALE. Shown are ʻAlelele, Pīpīwai and Palikea Streams. The lower reaches of Palikea Stream are commonly referred to as ʻOheʻo Gulch. Three small perennial lakes, Waiʻānapanapa, Waiʻeleʻele and Wai nēnē, are situated in the northeast section of the park. Source: State of Hawaiʻi Division of Aquatic Resources.

A wide variety of parameters can be used to characterize water quality. Under the authority of the CWA, the State of Hawaiʻi Department of Health (DOH) has issued Water Quality Standards (WQS) for the state that include specific criteria for Hawaiian streams (Hawaii DOH 2014). These standards specify average and extreme values not to be exceeded for total phosphorus (P), total nitrogen (N), nitrate + nitrite ($NO_3 + NO_2$), total suspended solids, and turbidity (Table 4.1.3-1). High levels of P and N, including NO_3 and NO_2, can impair water quality by promoting blooms of algae and bacteria. High levels of suspended solids may contain organic or inorganic pollutants, and reduce water clarity; high turbidity also reflects diminished water clarity. All of these parameters degrade habitat quality for native aquatic stream organisms, and may impact human health for park visitors that enter, swim in, or drink from streams.

Figure 4.1.3-2. The scenic pools in ʻOheʻo Gulch are popular for viewing and swimming. Photo: NPS.

Table 4.1.3-1. State of Hawaiʻi Water Quality Standards for streams (Hawaii DOH 2014).

Parameter	Season[1]	Geometric mean not to exceed	Not to exceed more than 10% of time	Not to exceed more than 2% of time
Total Phosphorus (µgP/L)	Wet	50.0	100.0	150.0
	Dry	30.0	60.0	80.0
Total Nitrogen (µgN/L)	Wet	250.0	520.0	800.0
	Dry	180.0	380.0	600.0
Nitrate + Nitrite (µgN/L)	Wet	70.0	180.0	300.0
	Dry	30.0	90.0	170.0
Total Suspended Solids (mg/L)	Wet	20.0	50.0	80.0
	Dry	10.0	30.0	55.0
Turbidity (NTU)2	Wet	5.0	15.0	25.0
	Dry	2.0	5.5	10.0

[1] Wet season: November 1 through April 30; Dry season: May 1 through October 31

[2] Nephelometric Turbidity Units

The State of Hawaiʻi has also issued WQS for additional stream parameters, including pH, dissolved oxygen saturation, specific conductance, and temperature (Hawaii DOH 2014). However, these standards are specified in terms of values not to be exceeded (Table 4.1.3-2), rather than averages or extremes not to be exceeded, and in some cases are defined relative to ambient conditions. This makes them more challenging to implement, as point measurements may occasionally exceed the standards even if most measurements do not, and ambient conditions would need to be characterized for any given stream.

NPS has also identified a set of parameters with which to monitor water quality within PACN as part of the Inventory & Monitoring (I&M) Program (HaySmith et al. 2006, Jones et al. 2011). Some of these overlap partially or completely with State of Hawaiʻi WQS parameters, and include Total Dissolved Phosphorus (TDP), Total Dissolved Nitrogen (TDN), Nitrate + Nitrite (NO_3 + NO_2), Chlorophyll, Dissolved Oxygen Concentration and Dissolved Oxygen Saturation, pH, Salinity, Specific Conductance, Temperature, and Turbidity. During 2013-2014 Total Phosphorus (TP) and Total Nitrogen (TN) measurements, which include measurements of particulate contributions of these elements, were also made. However, NPS has not set standards for these measures, and instead has identified an initial goal to determine range, spatial variance, temporal trends, and spatial trends of the parameters (Jones et al. 2011). The rationale is that the range of variation needs to be understood for each water body before appropriate standards can be formulated (DeVerse and DiDonato 2006, Jones et al. 2011).

Table 4.1.3-2. Additional State of Hawaiʻi Water Quality Standards for streams (Hawaii DOH 2014).

Parameter	Standard
pH	Shall not deviate more than 0.5 units from ambient conditions and shall not be lower than 5.5 nor higher than 8.0
Dissolved Oxygen	Not less than 80% saturation, determined as a function of ambient water temperature
Temperature	Shall not vary more than 1° C from ambient conditions
Specific Conductance	Not more than 300 micromhos/cm

Prior to the standardized monitoring program implemented by I&M in 2007 (Raikow and Farahi 2016), water quality measurements in HALE streams were conducted by a variety of agencies and individuals in a sporadic and intermittent fashion, summarized in the NPS Water Resources Division Baseline Water Quality Data report (NPS WRD 1999) and in DeVerse and DiDonato (2006). The vast majority of both historic and more recent standardized I&M measurements have been conducted in three HALE streams: ʻAlelele, Pīpīwai and Palikea. This assessment is therefore restricted to these three streams. Only preliminary water quality measurements have been conducted in the three remote, perennial montane lakes in HALE (Waiʻānapanapa, Waiʻeleʻele and Wai nēnē); currently NPS considers them unimpaired and presumes their condition to be “pristine” (DeVerse and DiDonato 2006). For indicators of stream water quality condition, we focus our assessment on the five parameters for which the State of Hawaiʻi has most clearly defined WQS criteria (Table 4.1.3-1), because, as noted above, the other parameters that have at times been measured do not as yet have clear and usable standards with which to assess condition or trend. However, we exclude the measure Total Suspended Solids, for which the state has defined WQS criteria (Table 4.1.3-1), because this parameter has rarely if ever been measured in HALE streams. In assessing overall condition of water quality, we discuss, where possible, data on some of the other water quality parameters that have periodically been measured.

Indicators

- Total Phosphorus
- Total Nitrogen
- Nitrate + Nitrite
- Turbidity

Data and Methods

A review of available data and information was used to assess conditions, and if possible, trends, in the indicators listed above. No additional data analysis was performed.

Sources of information

Information on recent and historic measurements of water quality and HALE were obtained from summaries presented in NPS WRD (1999), DeVerse and DiDonato (2006), and Raikow and Farahi (2016). Summaries of certified PACN measurements subsequent to those reported in Raikow and Farahi were provided by D. Raikow.

Reference condition

An ideal reference condition for water quality in HALE streams describes completely unimpaired waters, with no anthropogenic pollutants, unimpeded flow with high levels of oxygenation, and good clarity at baseflow. Because parameter values for completely unimpaired water quality conditions are unknown for HALE streams, we define reference conditions as those that, at a minimum, meet the State of Hawaiʻi WQS listed in Table 4.1.3-1.

Condition and Trend

Total Phosphorus

Data on total dissolved phosphorus (TDP), collected by the NPS I&M Program (Raikow and Farahi 2016), are available for two of the three HALE streams under consideration (Palikea and Pīpīwai) for the years 2007-2011, and for the third stream (ʻAlelele) for the years 2009-2011. These measurements (Figure 4.1.3-3) were collected on a quarterly basis at four fixed and four random sites across all streams (Raikow and Farahi 2016). The geometric means of all TDP measurements calculated over each season within each year are shown for the three streams in Table 4.1.3-3.

None of the mean values of TDP in Table 4.1.3-3 exceed the State of Hawaiʻi standards of total phosphorus (TP) concentrations of 50 µgP/L in the wet season and 30 µgP/L in the dry season. However, TP is comprised of both TDP and particulate phosphorus (PP). As a result, the NPS I&M data on TDP in Table 4.1.3-3 do not measure the full content of phosphorus in the streams, to the extent that any PP occurs in them. Because the PP concentration in HALE streams was unknown at the time of these measurements, it is possible that TP may have exceeded state standards even though TDP did not. Measurements of TP during 2013-2014, however, suggest that levels of PP in HALE streams may not generally be high. Although particulates contributed substantially to TP in some measurements, the median proportion of PP as a component of TP was 0.3 (NPS I&M unpublished data). While low particulate loads are carried by HALE streams at baseflow, floods mobilize large amounts of particulate matter including phosphorus.

Figure 4.1.3-3. NPS I&M staff collecting a water quality sample in HALE. Photo: NPS.

Table 4.1.3-3. Values of total dissolved phosphorus (µgP/L) in three HALE streams from 2007 to 2011 (Raikow and Farahi 2016). Shown are geometric means calculated over all monitoring sites for each season in each year.

Year	Season*	‘Alelele Stream geometric mean	Palikea Stream geometric mean	Pīpīwai Stream geometric mean
2007	Wet	na	28.3	36.0
	Dry	na	2.9	4.2
2008	Wet	na	14.4	13.9
	Dry	na	11.6	11.3
2009	Wet	na	15.0	15.0
	Dry	19.8	15.2	15.5
2010	Wet	15.2	15.0	15.4
	Dry	15.4	15.0	15.0
2011	Wet	15.8	15.0	15.0
	Dry	18.3	15.0	15.0

* Wet season: November 1 through April 30; Dry season: May 1 through October 31

Summaries of historic water quality measurements in and around HALE between 1972 and 1997 report only very limited measurements of phosphorus in the form of phosphate (PO_4), and do not indicate any known impairment of HALE streams due to phosphorus contamination (NPS WRD 1999, DeVerse and DiDonato 2006).

Based on the available information, the condition of HALE streams with respect to total phosphorus appears to be good, with no known violations of state standards. Confidence in this assessment is medium, because while TDP concentrations have been collected in a rigorous and standardized manner in recent years, concentrations of particulate phosphorus have only been measured for a short duration, and could result in unacceptably high concentrations of total phosphorus at certain times. The trend in phosphorus levels is unknown, as available data currently span too short of a time period to make an assessment.

Total Nitrogen

Data collected by the NPS I&M Program (Raikow and Farahi 2016) are also the best source for recent information on total dissolved nitrogen (TDN) in HALE streams, and are available over the same time period and were collected in the same manner summarized above for TDP. The geometric means of all TDN measurements calculated over each season within each year are shown for the three streams in Table 4.1.3-4.

Table 4.1.3-4. Values of total dissolved nitrogen (µgN/L) in three HALE streams from 2007 to 2011 (Raikow and Farahi 2016). Shown are geometric means calculated over all monitoring sites for each season in each year.

Year	Season*	‘Alelele Stream geometric mean	Palikea Stream geometric mean	Pīpīwai Stream geometric mean
2007	Wet	na	82	73
	Dry	na	80	84
2008	Wet	na	142	98
	Dry	na	77	78
2009	Wet	na	88	73
	Dry	165	137	133
2010	Wet	95	74	90
	Dry	178	123	96
2011	Wet	153	106	85
	Dry	149	148	135

* Wet season: November 1 through April 30; Dry season: May 1 through October 31

None of the mean values of TDN in Table 4.1.3-4 exceed the State of Hawai‘i standards of total nitrogen (TN) concentrations of 250 µgN/L in the wet season and 180 µgN/L in the dry season, although several measurements in ‘Alelele Stream approached the dry season standard. However, the same caveat regarding total versus dissolved concentrations of nitrogen applies: TN is comprised of both TDN and particulate nitrogen (PN). Because the PN concentration in HALE streams was unknown at the time of the measurements in Table 4.1.3-4, it is possible that TN may have exceeded state standards even though TDN did not. Measurements of TN during 2013-2014, however, suggest that levels of PN in HALE streams may not generally be high. Although particulates contributed substantially to TN in some measurements, the median proportion of PN as a component of TN at baseflow was very low: 0.08 (NPS I&M unpublished data).

Summaries of historic water quality measurements in and around HALE between 1972 and 1997 report only very limited measurements of nitrogen, in the form of ammonia (NH_4) and nitrate (NO_3), with no reported measurements of total nitrogen. These historic measurements therefore did not indicate any known impairment of HALE streams due to total nitrogen contamination (NPS WRD 1999, DeVerse and DiDonato 2006).

Based on the available information, the condition of HALE streams with respect to total nitrogen generally appears to be good, with no known violations of state standards. Several measurements in ʻAlelele Stream between 2009 and 2011, however, were very close to exceeding state standards during the dry season. Confidence in this assessment is medium, because while TDN concentrations have been collected in a rigorous and standardized manner in recent years, concentrations of particulate nitrogen have only been measured for a short duration, and could result in unacceptably high concentrations of total nitrogen at certain times. The current trend in nitrogen levels is unknown, as available data currently span too short of a time period to make an assessment.

Nitrate + Nitrite

NPS I&M Program measurements of nitrate + nitrite are the best source of recent data on this water quality parameter for HALE streams (Raikow and Farahi 2016). Data are available for the years 2007 through 2011, which are summarized in Table 4.1.3-5. Monitoring methods were the same as those summarized for TDP.

Several of the seasonal means exceeded State of Hawaiʻi standards for nitrate + nitrite concentrations of 70 µgN/L in the wet season and 30 µgN/L in the dry season (shown in red bold in Table 4.1.3-5), in both ʻAlelele and Palikea Streams. Standards were exceeded most commonly in the dry season, reaching two to four times the WQS threshold in ʻAlelele Stream. State standards were never exceeded in Pīpīwai Stream during the period of record, although most of the dry season means were very close to the state WQS threshold.

Summaries of historic water quality measurements in and around HALE between 1972 and 1997 report only limited measurements of nitrate, with no reported measurements of nitrate + nitrite. None of the historic nitrate measurements exceeded US EPA standards (NPS WRD 1999, DeVerse and DiDonato 2006).

Table 4.1.3-5. Values of nitrate+nitrite concentrations (µgN/L) in three HALE streams from 2007 to 2011 (Raikow and Farahi 2016). Shown are geometric means calculated over all monitoring sites for each season in each year. Means that exceed state standards are displayed in bold.

Year	Season[1]	‘Alelele Stream geometric mean	Palikea Stream geometric mean	Pīpīwai Stream geometric mean
2007	Wet	na	23.9	20.2
	Dry	na	26.3	28.0
2008	Wet	na	50.8	16.5
	Dry	na	12.3	13.3
2009	Wet	na	43.7	19.5
	Dry	**59.6[2]**	**36.4[2]**	27.8
2010	Wet	54.3	9.4	7.3
	Dry	**123.8[2]**	**60.4[2]**	26.0
2011	Wet	**107.5[2]**	43.2	11.6
	Dry	**71.7[2]**	**35.5[2]**	25.3

[1] Wet season: November 1 through April 30; Dry season: May 1 through October 31

[2] Means that exceed state standards, also displayed in bold.

Based on the available information, and assuming that the state standards are appropriate, the condition of HALE streams with respect to nitrate + nitrite contamination warrants moderate concern. Two of three monitored streams have exceeded state water quality standards in recent years, most commonly during the dry season when water flow is lower (NPS WRD 1999). Flow in these streams can become interrupted during dry periods (Kinzie and Ford 1977), potentially allowing pollutants to accumulate and concentrate. A variety of factors can lead to elevated nitrate and nitrite levels, potentially including nitrification of feral animal waste that may enter streams via leaching or by being carried directly with soils during rain events. Although most of the park is fenced to exclude invasive ungulates, at times feral animals such as pigs, goats and cows, which can alter soil properties in ways that promote erosion (Long et al. 2017), still infiltrate areas upstream of the water quality monitoring stations (T. Bailey, J. Mallinson pers. comm. 2017). While standards for mean concentrations were sometimes exceeded, values did not generally reach extreme levels (Raikow and Farahi 2016), which might warrant significant concern.

Confidence in this assessment is high, because nitrate and nitrite concentrations were monitored in a rigorous and standardized manner in recent years. The trend in nitrate + nitrite levels is unknown, as available data currently span too short of a time period to make an assessment.

Turbidity

NPS I&M Program measurements of turbidity are the best source of recent data on this water quality parameter for HALE streams (Raikow and Farahi 2016). Data are available for the years 2007 through 2011, which are summarized in Table 4.1.3-6. Monitoring methods were the same as those summarized for TDP, with the exception that turbidity measurements were made using a sonde.

Seasonal turbidity means exceeded State of Hawai‘i standards of 5.0 NTU in the wet season and 2.0 NTU in the dry season only in 2007, in both Palikea and Pīpīwai Streams. Turbidity levels were much higher in this year than in all subsequent years during the period of record, most likely due to heavy rainfall and high stream flow around monitoring dates in 2007.

Prior to I&M monitoring, turbidity appears to only have been measured at one location in Palikea Stream, from 1972 to 1977 (NPS WRD 1999). None of the 12 measurements at this location exceeded US EPA standards (NPS WRD 1999).

Table 4.1.3-6. Values of turbidity (NTU) in three HALE streams from 2007 to 2011 (Raikow and Farahi 2016). Shown are geometric means calculated over all monitoring sites for each season in each year. Means that exceed state standards are displayed in bold.

Year	Season*	‘Alelele Stream geometric mean	Palikea Stream geometric mean	Pīpīwai Stream geometric mean
2007	Wet	na	**8.5**[2]	**7.7**[2]
	Dry	na	**4.9**[2]	**4.3**[2]
2008	Wet	na	1.9	0.6
	Dry	na	0.5	0.5
2009	Wet	na	0.9	0.3
	Dry	0.409	0.7	0.4
2010	Wet	0.100	0.2	0.3
	Dry	0.100	0.1	0.3
2011	Wet	0.114	0.2	0.3
	Dry	0.210	0.5	0.4

[1] Wet season: November 1 through April 30; Dry season: May 1 through October 31.

[2] Means that exceed state standards, also displayed in bold.

Based on the available information, the condition of HALE streams with respect to turbidity appears to be good. Although state turbidity standards were exceeded in one of five years on record, Hawaiian streams such as Palikea are prone to severe flash flooding (Kinzie and Ford 1977, Hodges 1994) because of heavy and often localized rainfall, with greatly elevated turbidity during floods. For example, annual rainfall is estimated to average 5,000 to 10,000 mm over much of HALE stream watershed areas (Giambelluca et al. 2013). Occasional high turbidity can thus be expected to be a natural part of Hawaiian stream hydrology.

Confidence in this assessment is high, because turbidity was monitored in a rigorous and standardized manner in recent years. The trend in turbidity is unknown, as available data currently span too short of a time period to make an assessment.

Other parameters

A variety of other water quality parameters have been measured periodically, and generally fairly sporadically, in HALE streams. Of 75 different parameters measured and catalogued for surface waters in and around HALE, only five categories of parameters were measured to exceed US EPA standards (NPS WRD 1999). The most commonly measured of these is pH, which was found to frequently exceed US EPA standards (below 6.5 or above 9.0). For example, at one site on Palikea Stream measured by USGS between 1972 and 1983, 21 of 64 measurements (33%) registered below pH 6.5 (DeVerse and DiDonato 2006). At six sites in Palikea Stream, Pīpīwai Stream and ʻOheʻo Gulch measured in 1994-1995, one-third to one-half of pH measurements, depending on site, fell below 6.5 (DeVerse and DiDonato 2006). However, State of Hawaii WQS allow for stream pH to vary between 5.5 and 8.0 (Table 4.1.3-2), resulting in fewer historic violations of standards. More recently, NPS I&M monitoring has recorded pH values generally within the state WQS range in HALE streams (Raikow and Farahi 2016). Median pH, as well as the middle 50% of measurements, were usually between 6.5 and 8.0 between 2007 and 2011, with the exception of higher pH measurements in 2007, when median values were 8.01 and 8.04 in Palikea and Pīpīwai Streams, respectively (Raikow and Farahi 2016).

Dissolved oxygen (DO) was also reported to exceed US EPA standards in a very few historic measurements taken between 1972 and 1995: 4 out of 200 measurements in and around HALE fell below minimum standards, with all violations occurring in ʻAlelele Stream (NPS WRD 1999). One out of six measurements taken by the Hawaii DOH in 1999-2000 at the mouth of ʻOheʻo Gulch was below minimum DO percent saturation standards (DeVerse and DiDonato 2006). NPS I&M measurements at HALE found very good DO percent saturation conditions from 2007-2011, with median values near 100% in all cases, and only very few individual measurements falling below the state WQS of 80% (Raikow and Farahi 2016).

Two out of 71 total chloride concentration measurements taken from 1972 to 1994 in and around HALE exceeded US EPA drinking water standards (NPS WRD 1999). However, both of these violations occurred at Punahoa Springs, which is located outside of HALE boundaries. None of the measurements taken in Palikea Stream exceeded US EPA standards. More recent measurements of chloride levels in HALE streams are unavailable.

Finally, despite hazards in ʻOheʻo Gulch or Palikea or Pīpīwai Streams due to falling rocks and flash flooding (NPS 2017), these nevertheless remain popular swimming locations, especially the lower pools (Figure 4.1.3-4). The State of Hawaii DOH has therefore monitored the pools at ʻOheʻo Gulch at various times between 1973 and 2000 as part of its efforts to monitor recreational swimming areas for levels of public health pathogens (DeVerse and DiDonato 2006). After assays measuring *Enterococci* and *Clostridium perfringins* were initiated in 1989 and 1993, respectively, US EPA standards were never exceeded at ʻOheʻo Gulch through the year 2000 (DeVerse and DiDonato 2006). Standards for total coliform and fecal coliform were exceeded seven times between 1973 and 1991, however it has been questioned whether these pathogens are appropriate indicators in the tropics (DeVerse and DiDonato 2006). Taken as a whole, this evidence suggests that pathogenic contamination is not a chronic issue at ʻOheʻo Gulch. Swimming visitors may also introduce harmful substances to water through the use of sunscreen and other personal care products, but the magnitude and importance of this potential source of contamination is unstudied.

Figure 4.1.3-4. The pools in ʻOheʻo Gulch attract large numbers of tourists on busy days. Photo: Forest and Kim Starr.

Overall assessment

Integration of the conditions of the four main indicators/measures considered (total phosphorus, total nitrogen, nitrate + nitrite, turbidity) yields an overall good condition, with unknown trend, for water quality in HALE streams. This assessment might be modified with consideration of information known about the secondary water quality parameters discussed above. Although some of these parameters, most commonly pH, exceeded either state or US EPA standards, these violations were generally infrequent and moderate in magnitude. Considering the body of available measurements in and around HALE streams, the NPS Water Resources Division (NPS WRD 1999) concluded that "…surface waters near the southern coast appear to be generally of good quality with some impact from human activities. Potential anthropogenic sources of contaminants include recreational use; stormwater runoff; agricultural operations; marine traffic; and atmospheric deposition." Similarly, DiVerse and DiDonato (2006) argued that "Streams in HALE are in good condition relative to the State of Hawaii WQS, though they vary in natural attributes, such as drainage area, slope, and geology, and the degree of stress caused by feral animals and diversion up-slope." We concur with these assessments, noting that periodically higher than desired levels of N appear to be the most significant current source of impairment. (return to Condition Summary).

Level of confidence

Confidence in the current water quality condition is medium. Although recent NPS I&M Program monitoring data are rigorous, currently they are available for a relatively short time period. Some of the parameters measured fluctuated relatively strongly over the five-year period of record, imparting some uncertainty about their representativeness. The continuation of this monitoring protocol will clarify the central tendencies and ranges of variation of a number of important water quality parameters. It will ultimately allow appropriate standards to be defined, and will enable evaluation of directional trends in these selected water quality indicators.

Information gaps and research recommendations

Various other water quality parameters have not been measured recently, or with sufficient frequency, to assess their impacts on water quality at HALE. Some of these, such as concentrations of toxic compounds, can be highly detrimental to aquatic life, however, there are few obvious point sources for such contaminants in HALE. In addition, relatively little information exists regarding human health pathogens in HALE streams. Finally, the remote nature of the three montane lakes in the park protects them from most sources of contamination, but has also undoubtedly contributed to the lack of information on the quality of their waters.

Sources of Expertise

David Raikow and Anne Farahi, NPS I&M Program, PACN, provided data and expert commentary on this section.

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4.2. Biological Integrity

4.2.1. Coastal Ecosystems

Condition Summary

Overall resource condition warrants significant concern with an unchanging or stable trend; confidence in condition determination is high. Rationale is discussed in the "*Overall assessment*" section and following.

Relevance and Context

The boundaries of HALE reach the coast in three locations, including a strand between Nuʻu and Huakini Bays, at Kaʻāpahu Bay, and between Awapaewaʻa and Pepeiaolepo Bays (Figure 4.2.1-1). Approximately 5.2 km of coastline sit within the park (Natividad Bailey 2009). In this assessment coastal ecosystems will be broadly defined as those that occur between the high-tide line and 300 m elevation, in accordance with coastal plant communities described by Gagné and Cuddihy (1990).

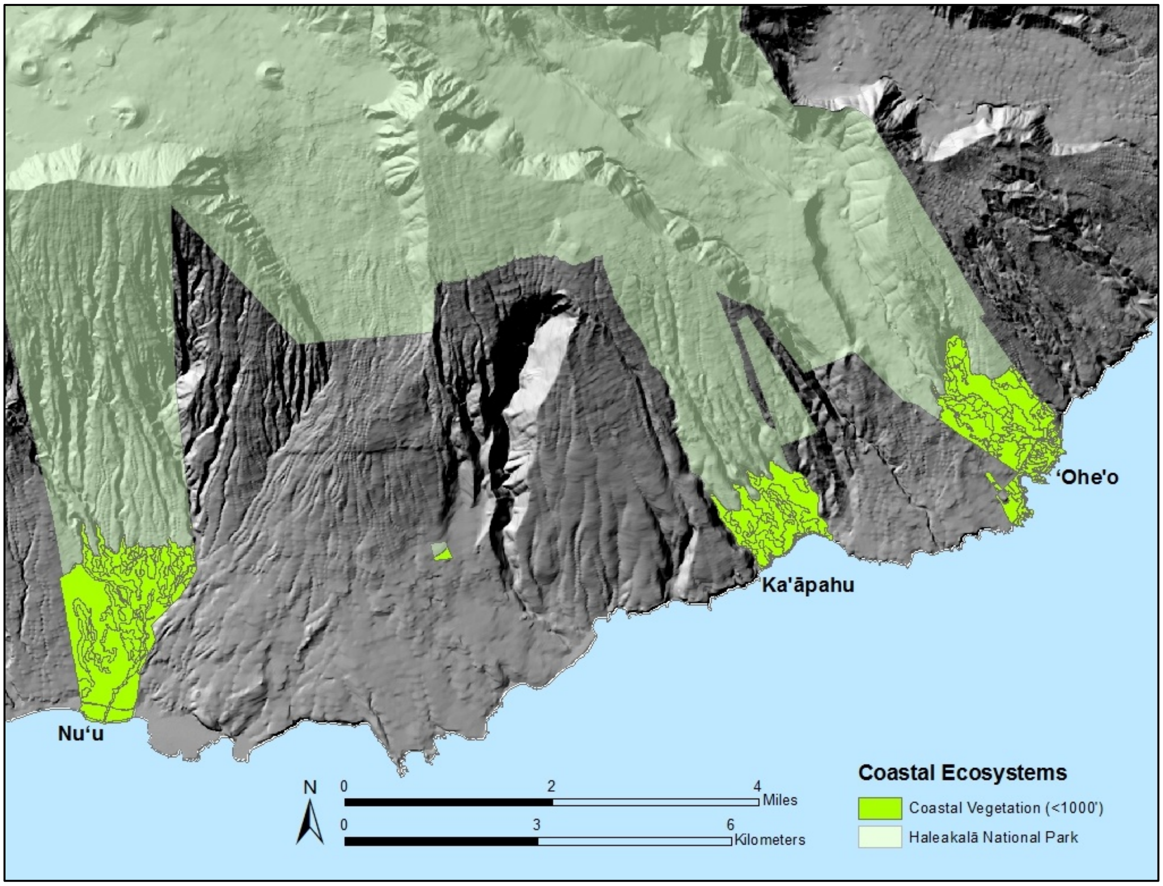

Figure 4.2.1-1. Coastal ecosystems in HALE. Vegetation communities below 1,000 ft (305 m) elevation are highlighted. Shapefile source: Green et al. 2015.

In much of the Hawaiian Islands, coastal ecosystems have been so heavily modified that it is difficult to determine original plant cover and composition, but some relatively good examples still exist throughout the archipelago (Mueller-Dombois and Fosberg 1998). Coastal ecosystems in HALE have been dramatically altered by both agricultural practices of early Polynesians and those that continued after European contact. Although non-native vegetation now dominates most locations, a narrow strip of native strand vegetation lines the coast. On flat areas, previous grazing by cattle had degraded this strand community. Characteristic native plant species include naupaka-kahakai (*Scaevola taccada*), hala (*Pandanus tectorius*), nanea (*Vigna marina*), and the dwarf mat-forming sedge *Fimbristylis cymosa*. Several rare species recently occurred here but are now extirpated, including the endangered *Ischaemum byrone*, a rare grass that was formerly common along windward coastlines throughout the islands (Figure 4.2.1-2), and koʻokoʻolau (*Bidens hillebrandiana* subsp. *polycephala*). This narrow zone of native plants is threatened by trampling by visitors as well as by displacement by non-native plant species (Medeiros et al. 1998). Enforcement of the Park's Stay-On-Trail policy in the Kīpahulu District is needed to protect rare species (NPS 2012b).

Figure 4.2.1-2. *Ischaemum byrone*, an endangered grass formerly found along the coastal regions of ʻOheʻo, Haleakalā National Park. Photo: Forest & Kim Starr.

Coastal ecosystems in Hawaiʻi may also support unique native animal communities, provided that they are not too degraded. These include coastal nesting seabirds; shorebirds that forage along coastlines and lowland areas; marine animals, including sea turtles that nest in sandy substrates and monk seals that haul out above the waterline; and crabs, insects, and related invertebrates that utilize coastal habitats.

Coastal Birds

A variety of seabirds once nested in coastal ecosystems in the main Hawaiian Islands, but many of these have now been extirpated or excluded from these areas due to habitat destruction and conversion by people, disturbance from human activities, and predation by introduced mammals; large and diverse seabird colonies are now mostly restricted to the Northwest Hawaiian Islands and a few small islets off the main islands (Kozar et al. 2007, VanderWerf et al. 2007). Despite this, several seabird species still use coastal habitats for nesting in the main islands, including moli (Laysan Albatross, *Phoebastria immutabilis*), ʻuaʻu kani (Wedge-tailed Shearwater, *Puffinus pacificus*), koaʻe kea (White-tailed Tropicbird, *Phaethon lepturus dorotheae*), koaʻe ʻula (Red-tailed Tropicbird (*Phaethon rubricauda*), ʻiwa (Great Frigatebird, *Fregata minor palmerstoni*), ʻā (Red-footed Booby, *Sula sula*), ʻā (Brown Booby, *Sula leucogaster*), noio (Hawaiian or Black Noddy, *Anous minutus melanogenys*), and noio kōhā (Brown Noddy, *Anous stolidus*) (Kozar et al. 2007, VanderWerf et al. 2007, Natividad Bailey 2009, Young et al. 2009, VanderWerf et al. 2014).

Several migratory shorebird species winter in the main Hawaiian Islands, and commonly use coastal ecosystems as foraging grounds; they breed in the arctic during summer months. These include kōlea (Pacific Golden Plover, *Pluvialis fulva*), ʻūlili (Wandering Tattler, *Heteroscelus incanus*), ʻakekeke (Ruddy Turnstone, *Arenaria interpres*), and hunakai (Sanderling, *Calidris alba*); kioea (Bristle-thighed Curlew, *Numenius tahitiensis*) is rare in the main islands, but somewhat more common in the Northwestern Hawaiian Islands (Kozar et al. 2007).

Coastal Vertebrates

Īlio-holo-i-ka-uaua, or the Hawaiian monk seal (*Monachus schauinslandi*), is a federally endangered marine mammal that occurs in Hawaiian waters. These primarily solitary seals forage in the marine environment, but haul out on sandy and rocky shorelines to pup, nurse and rest (Hawaiʻi DLNR 2005c). Most of the Hawaiian population is located in the Northwest Hawaiian Islands, but a small number (~50 individuals) are known to occur in the main islands (Hawaiʻi DLNR 2005c).

Five species of sea turtles are known to occur in or around Hawaiian waters: honu, or green sea turtle (*Chelonia mydas*), hawksbill sea turtle (*Eretmochelys imbricata*), leatherback sea turtle (*Dermochelys coriacea*), loggerhead sea turtle (*Caretta caretta*), and olive ridley sea turtle (*Lepidochelys olivacea*) (Hawaiʻi DLNR 2005a, b, d, e, f). However, while all of these may have once nested in the Hawaiian Islands, only honu and hawksbill sea turtles are known to nest regularly in Hawaiʻi today; the other species may do so only extremely rarely if at all (Hawaiʻi DLNR 2005a, b, d, e, f). Both honu and hawksbill turtles prefer sandy beaches for nesting, although hawksbill turtles may also nest in soil, usually within 5 m of the high water line (Hawaiʻi DLNR 2005a, b). Honu also periodically haul out on beaches and other shorelines to bask (Hawaiʻi DLNR 2005b).

Coastal Invertebrates

Several species of rock crabs in the family Grapsidae are active on wet coastal rocks in the splash zone in Hawaiʻi, and as in other terrestrial ecosystems, gastropods, insects and their arthropod relatives are abundant, diverse and important ecological components of coastal ecosystems. However, contemporary arthropod communities in Hawaiʻi are generally thought to be very highly degraded in coastal and lowland ecosystems, resulting from the wholesale conversion of these ecosystems to agriculture or urban development, or to novel plant communities harboring few if any native host plants; in addition, these areas are also subject to invasion from large numbers of non-native arthropod species (Zimmerman 1970, Howarth 1990). As an example, a survey of arthropods in the coastal areas surrounding Kahului Airport, Maui, found that only 11-14% of the species sampled were native (Howarth et al. 2012). Even Ka‘ena Point, a protected and relatively remote coastal Natural Area Reserve on O‘ahu that supports a fairly diverse native vegetation community, is today characterized by an arthropod community comprised of only ~6% native species (P. Krushelnycky unpub. 2015 data).

Most available information regarding HALE coastal ecosystems concerns vegetation, including surveys, checklists, monitoring plots, and data on rare native species and invasive species control. We therefore focus on coastal vegetation as the primary indicator of coastal ecosystems condition. In addition, we consider the relatively limited information regarding coastal vertebrates, including marine animals that use coastal lands (sea turtles and monk seals), seabirds, shorebirds, and other birds. We also assess the status of coastal invertebrates, using limited information on terrestrial insects and arthropod relatives. We discuss the effects of invasive species on the condition of each indicator, rather than assessing invasive species as a separate indicator of ecosystem condition.

Indicators

- Coastal vegetation
- Coastal vertebrates
- Coastal invertebrates

Data and Methods

A review of available data and information was used to assess conditions, and if possible, trends, in the indicators listed above. No additional data analysis was performed.

Sources of information

Monitoring, outplanting, and propagule collection data for common, rare, and endangered native taxa have been recorded by HALE Vegetation Management staff (primarily Patti Welton, Bill Haus, Matt Shultz, Regan Emmons, Chuck Chimera, David Palumbo and Michelle Osgood) for the entire park from 1992 through present. Data for the coastal vegetation for ‘Ohe‘o and the vicinity are available from December 2000 through present (NPS 2017a).

Control data for invasive, non-native plants have been collected by HALE Vegetation Management staff for the entire park from 1986 through present. Data for the coastal vegetation for ‘Ohe‘o and the vicinity have been compiled by Bill Haus and Patti Welton and are available from 1992 through present (NPS 2017b).

Vegetation classification data were also collected by HALE and Pacific Island Network (PACN) I&M staff in 220 circular plots (400 m^2) and observation points in 2011 (Green et al. 2015). Of these, 40 plots fell within the coastal elevation zone.

Medeiros et al. (1998) provided an annotated checklist of all flowering plants of the park, which includes information on coastal vegetation of ʻOheʻo. Welton and Haus (2008) provided descriptive information of the vascular plants of the Kaʻāpahu section of the park, with notes on some of the more prominent plants and vegetation of the coastal portion below 1000 ft (305 m) elevation.

Kozar et al. (2007) surveyed two sections of coastline for the presence of shorebirds, seabirds and other birds in HALE in March of 2005. Natividad Bailey (2009) surveyed the same two sections of coastline for the presence of seabirds in July of 2005.

Stemmermann (1980) surveyed lower elevations of Kīpahulu Valley below 2,000 ft, and provided some information about occurrence of seabirds and migratory shorebirds in coastal areas.

The HALE Resources Management Plan (NPS 1999) and information for Section 7 Endangered Species Consultation (NPS 2012b) mention sightings of sea turtles and monk seals, and Pacific Islands Fisheries Science Center, National Marine Fisheries Service, provided a report on sightings of monk seals at HALE from 2007-2016 (PIFSC 2017).

Information on coastal invertebrate resources was obtained from surveys conducted by Gagné (1980), Gon and Pinter (1980), Severns (1980) and Takumi (2001).

Reference condition

Although no examples of entirely unmodified coastal vegetation currently exist, Warshauer et al. (2009) surveyed 133 coastal sites of note throughout the state (primarily on the islands of Molokaʻi, Maui, and Oʻahu) and categorized them based on native species diversity and, in some cases, connection with lowland vegetation. Highest species diversity occurred on Molokaʻi and Maui (with 30 and 32 species respectively), although 95% of Maui sites had 20 or fewer species. An average of 11 species was recorded for all coastal sites on Maui (n = 55). Among the East Maui sites surveyed, five occur within or near coastal regions of the park. Ideal reference conditions for coastal vegetation should therefore consist of intact, unmodified communities of relatively high diversity, possessing at least 20 native species per site. These communities should be minimally invaded by non-native plants, and certainly not dominated by them. Coastal vertebrate communities should include healthy colonies of nesting seabirds, relatively abundant migratory shorebirds, and other native bird species in inland coastal habitats. Sea turtles and monk seals should utilize coastal habitats, if the naturally available habitats are suitable. Coastal invertebrate communities should be diverse and possess a high percentage of native species relative to non-native species.

Condition and Trend

Coastal vegetation

Although not a comprehensive list of all plant taxa found in the coastal districts of HALE, Green et al. (2015) provide a broad sampling of species occurring in the Nuʻu, Kaʻāpahu and ʻOheʻo regions of the park through a series of 31 classification and 9 observation plots (Table 4.2.1-1; Figure 4.2.1-1).

Table 4.2.1-1. Total and mean species diversity recorded in coastal plots below 1,000 ft (305 m) elevation. Source: Green et al. (2015). Nuʻu (n=12); Kaʻapahu (n=7); ʻOheʻo (n=12); total (n=31).

Native or non-native species	**Nuʻu**	**Kaʻapahu**	**ʻOheʻo**	**Total**
Native Species (Total)	5	10	12	24
Non-Native Species (Total)	46	35	55	108
Native Species (Mean)	1.4	2.0	1.4	1.5
Non-Native Species (Mean)	12.0	11.4	11.8	11.8

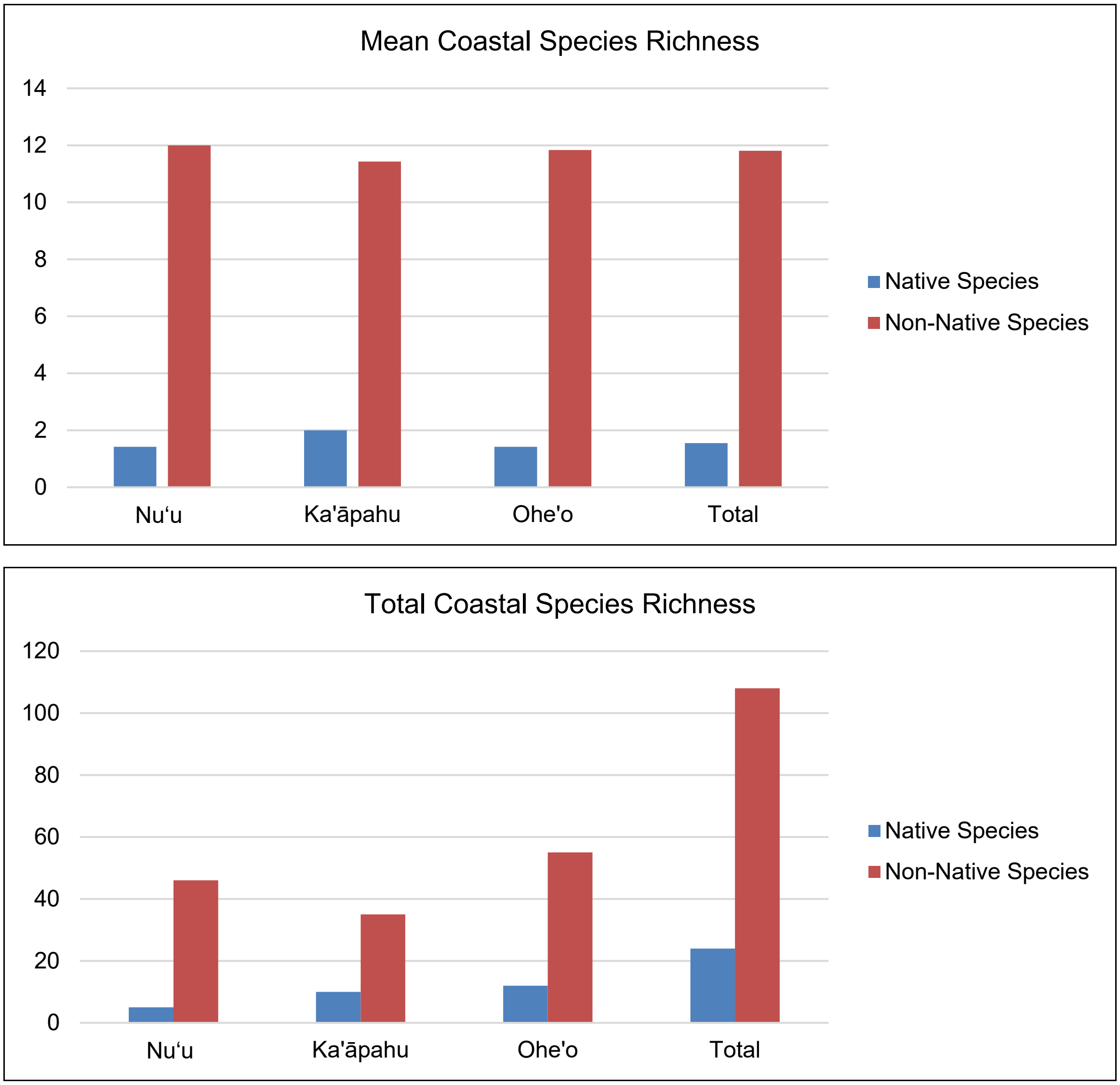

Figure 4.2.1-3. Total and mean species richness recorded in plots within three coastal regions of HALE. Mean diversity was calculated from 400 m^2 circular monitoring plots (Green et al. 2015). Nuʻu (n = 12); Ka'āpahu (n = 7); Ohe'o (n = 12); Total (n = 31).

Nuʻu Parcel

The lowland, dry zone of the Nuʻu Parcel is dominated by non-native *Prosopis pallida* Coastal Dry Semi-natural Woodlands (Green et al. 2015). This community, from approximately 30-150 m elevation and roughly 229 ac (93 ha) in area, is arid, with typically less than 1270 mm (50 in) of precipitation per year, and often affected by sea spray and brackish water tables. Vegetation in these areas has been both historically and currently grazed, and is relatively lacking in native species diversity and abundance (Figure 4.2.1-4).

Figure 4.2.1-4. *Prosopis pallida* (indicated with arrow) dominated coastal community, Nuʻu Parcel. Photo: Forest & Kim Starr.

A remnant native *Erythrina sandwicensis* Dry Lowland Woodland also occurs in the Nuʻu Parcel slightly upslope of the *Prosopis pallida* Coastal Dry Semi-natural Woodlands. This approximately 2.9 ac (1.2 ha) coastal community is characterized by an open tree canopy dominated by *Erythrina sandwicensis*. *Prosopis pallida* is present to codominant, with a subcanopy or tall-shrub layer dominated by *Leucaena leucocephala*. The understory is characterized by a moderate to dense herbaceous layer dominated by non-native grasses including *Megathyrsus maximus* and *Cenchrus ciliaris*.

The remainder of the coastal Nuʻu Parcel below 1000 ft (305 m) elevation is dominated by non-native vegetation. This includes large disturbed and goat-browsed areas covered by *Cenchrus ciliaris*

- *Bothriochloa pertusa* Semi-natural Herbaceous Vegetation, *Lantana camara* Semi-natural Shrubland and *Leucaena leucocephala* Semi-natural Shrubland (Green et al. 2015). The endangered plant *Portulaca villosa* was recorded historically, in 1920, from coastal lowland and grassland ecosystems of Nuʻu (chapter 4.2.10), but has not been observed recently.

Because of the heavily-modified state of this area, the condition along the Nuʻu coastline is unlikely to improve without active intervention through non-native plant control, native outplanting and restoration, and protection from ungulates. Currently, ungulate browsing (e.g. goats, deer, and pigs) favors grasses and woody vegetation with adaptations to tolerate such pressure, and this region shows evidence of disturbance from its ranching history as well. To date, no outplanting or weed control has been conducted in this coastal area of the park, but two community-dominant weeds, *L. leucocephala* and *P. pallida*, have been slated for control (Table 4.2.1-2, NPS 2017b). Warshauer et al. (2009) recorded 6 native species in the nearby Nuʻu Salt Pond, 5 fewer than the average native diversity recorded for other coastal strand areas on Maui.

Table 4.2.1-2. Notable non-native vegetation controlled below 1000 ft in coastal ʻOheʻo, Kaʻāpahu and Nuʻu (Source: NPS 2017b).

Taxon	Total Controlled (individuals)	Description of Impacts	Region
Ardesia elliptica	138,303	Invasive tree establishing by seed in understory	ʻOheʻo, Kaʻāpahu
Cenchrus purpureus	75,426	Forms dense stands that exclude most other native and non-native plants	ʻOheʻo
Clidemia hirta	4,159	Aggressive, thicket-forming shrub of trails and wet forest	ʻOheʻo
Coffea arabica	4,773	Capable of forming dense cover along trail to Waimoku Falls	ʻOheʻo
Furcraea foetida	2,645	Naturalized on cliffs near pools and lower pastures; competes with native strand vegetation	ʻOheʻo, Kaʻāpahu
Leucaena leucocephala	Not Controlled Yet	Dominates shrub layer in the *Erythrina sandwicensis* Lowland Dry Woodland; may compete with and prevent native recruitment	Nuʻu
Megathyrsus maximus	2,815	Robust perennial grass of disturbed and human-modified sites	ʻOheʻo
Psidium guajava	9,071	Edible fruit and hard seed enable spread by animals; forms dense stands and shades out understory	ʻOheʻo, Kaʻāpahu
Prosopis pallida	Not Controlled Yet	Present to codominant in *Erythrina sandwicensis* Lowland Dry Woodland; may compete with and prevent native recruitment	Nuʻu
Ricinus communis	2,627	Weed of roadsides and disturbed habitat	ʻOheʻo

Table 4.2.1-2 (continued). Notable non-native vegetation controlled below 1000 ft in coastal ʻOheʻo, Kaʻāpahu and Nuʻu (Source: NPS 2017b).

Taxon	Total Controlled (individuals)	Description of Impacts	Region
Schinus terebinthifolius	2,211	Forms thickets along coastal cliffs, excluding most other vegetation	ʻOheʻo
Spathodea campanulata	1,971	Capable of forming dense canopies that can shade out native vegetation and prevent regeneration	ʻOheʻo, Kaʻāpahu
Sphagneticola trilobata	5,717	Vigorous mat-forming perennial herb can smother and compete with low-stature vegetation	ʻOheʻo
Syzygium cumini	887	Capable of forming dense canopies that can shade out native vegetation and prevent regeneration	ʻOheʻo, Kaʻāpahu
Syzygium jambos	1,594	Capable of forming dense cover along trail to Waimoku Falls	ʻOheʻo, Kaʻāpahu
Terminalia catappa	6,526	Forms dense stands that exclude most other native and non-native plants	ʻOheʻo
Urochloa mutica	13,380	Forms dense mats in areas of high soil moisture or nearby open water	ʻOheʻo

Kaʻāpahu

The coastal zone of Kaʻāpahu below 1000 ft (305 m) elevation consists of a mix of predominantly non-native vegetation communities, with some notable native elements contained therein (Welton and Haus 2008, Green et al. 2015). Adjacent to and below the road (Highway 360) is a small (ca. 0.14 ac) strip of *Scaevola taccada* Coastal Dry Shrubland, which, as the names implies, is dominated by a dense cover of the indigenous *S. taccada*. Other native vegetation at sea level includes a *Pandanus tectorius* Coastal Mesic Forest along lower ʻAlelele Stream near Kaʻāpahu Bay. This community includes a canopy of the indigenous tree *P. tectorius* with *Scaevola taccada* as a notable native component of the understory (Green et al. 2015; Figure 4.2.1-5). A dense *Hibiscus tiliaceus* Lowland Wet Forest also occupies a portion of the coast between Lelēkea and Kaʻāpahu Streams and is characterized by a dense tree canopy of *H. tiliaceus*, possibly an early Polynesian introduction, with minimal or no understory vegetation.

Warshauer et al. (2009) recorded 11 native species in surveys of both Lelēkea Bay, and the stretch of coastline between ʻAlelele and Kālepa, which equals the average native diversity recorded for other coastal strand areas on Maui.

Significant communities further inland include *Mangifera indica* Semi-natural Forest, *Schinus terebinthifolius* Semi-natural Scrub Woodland, *Syzygium cumini* Lowland Mesic Semi-natural Forest, *Psidium cattleianum* Lowland Wet Semi-natural Forest and *Aleurites moluccana* Lowland Wet Forest (Green et al. 2015). Although these communities are dominated by non-native vegetation, Welton and Haus (2008) describe some important native components, including lama (*Diospyros* *sandwicensis*), neneleau (*Rhus sandwicensis*) and alaheʻe (*Psydrax odoratum*) scattered sporadically on ridges and along a historic trail, a small remnant patch of ʻaʻaliʻi (*Dodonaea viscosa*) shrubland persisting on an open rocky ridge above the southwestern slope, and the only known wiliwili (*Erythrina sandwicensis*) tree below 30 m (100 ft) elevation at the lower southwest corner of Kaʻāpahu.

Figure 4.2.1-5. *Pandanus tectorius* along Kaʻāpahu coastline. Photo: Forest & Kim Starr.

Minimal restoration and native outplanting activity has occurred in the coastal region of Kaʻāpahu. In 2006, 72 *Pandanus tectorius* juveniles were planted along a lower elevation access trail (NPS 2015a). Six non-native plant species have been targeted for control to date (Table 4.2.1-2, NPS 2017b). Because feral ungulate damage has greatly reduced much of the ground cover throughout the coastal parts of this region (Welton and Haus 2008), native vegetation is unlikely to recover without fencing, ungulate removal, and supplemental restoration actions. However, continued non-native plant control may contain and suppress the expansion of priority habitat-modifying invasive species.

‘Ohe‘o

Although much of the vegetation in the coastal portion of ‘Ohe‘o is a matrix of predominantly non-native communities, narrow bands of *Scaevola taccada* Coastal Dry Shrubland, characterized by a moderately dense shrub layer of *S. taccada*, occupy a stretch of the coastline from Awapaewa‘a and Pepeiaolepo Bays (Green et al. 2015; Figure 4.2.1-6). Small patches of *Pandanus tectorius* Coastal Mesic Forest, with an open to closed *P. tectorius* tree canopy, occur slightly inland from elevations of 5-60 m near the Kīpahulu Visitor Center, along the Kuloa Point Loop Trail, and near the road to the heiau at Puhilele (Green et al. 2015). A non-native scale insect, *Thysanococcus pandani*, first detected on the grounds of the National Tropical Botanical Gardens on Maui in 1995, has since spread to infest *Pandanus* trees throughout the island, and could threaten the continued health and future viability of this forest type (Kumashiro & Heu 1997, Matsunaga 2015).

Figure 4.2.1-6. *Scaevola taccada* Coastal Dry Shrubland along ‘Ohe‘o coastline. Photo: Forest & Kim Starr.

Warshauer et al. (2009) recorded 15 native species in surveys of ‘Ohe‘o Gulch, which exceeds the average native diversity of 11 species recorded for other coastal strand areas on Maui, but found only 4 native species in nearby Ka‘ū Bay, demonstrating the patchy nature of remnant native vegetation throughout the heavily modified coastal region.

Other prominent vegetation communities below 1000 ft (305 m) elevation in coastal ʻOheʻo include *Syzygium cumini* Lowland Mesic Semi-natural Forest, *Cenchrus purpureus* Semi-natural Herbaceous Vegetation, *Paspalum conjugatum* - *Sacciolepis indica* - *Axonopus fissifolius* Mixed Semi-natural Herbaceous Vegetation, Mixed Non-native Disturbed Herbaceous Vegetation, *Schinus terebinthifolius* Semi-natural Scrub Woodland, *Mangifera indica* Semi-natural Forest, *Psidium cattleianum* Lowland Wet Semi-natural Forest, *Terminalia catappa* Semi-natural Woodland, *Schinus terebinthifolius* Semi-natural Scrub Woodland, *Psidium guajava* Disturbed Understory Semi-natural Forest, *Aleurites moluccana* Lowland Wet Forest and *Phyllostachys nigra* Semi-natural Thicket (Green et al. 2015). Although these communities may contain small patches of or scattered individual native plants in the understories, they are otherwise comprised of almost entirely non-native species.

Outplanting and restoration activities have supplemented, and in some cases, expanded native plant numbers and communities in coastal ʻOheʻo. From 1993 through the present, 49 species, totaling 6,516 individuals, have been outplanted in the area (Table 4.2.1-3, NPS 2017a). This total includes 808 endangered *Ischaemum byrone* individuals outplanted at 8 sites along the coast in Kīpahulu and Puhilele. Seedlings and flowering plants have been observed below the outplanted individuals at Puhilele (NPS 2012a).

Table 4.2.1-3. Numbers of native plant taxa outplanted in coastal areas of ʻOheʻo. Source: NPS (2017a).

Taxon	Plants	T&E
Acacia koa	199	NA
Aleurites moluccana	70	NA
Alyxia stellata	3	NA
Antidesma platyphyllum	139	NA
Artocarpus altilis	1	NA
Bacopa monnieri	3	NA
Bidens hillebrandiana subsp*. polycephala*	101	NA
Calophyllum inophyllum	113	NA
Clermontia arborescens	14	NA
Cordyline fruiticosa	10	NA
Cordia subcordata	4	NA
Cyperus javanicus	246	NA
Cyperus phleoides	4	NA
Diospyros sandwicensis	9	NA
Dodonaea viscosa	146	NA
Erythrina sandwicensis	3	NA
Euphorbia celastroides	12	NA
Heteropogon contortus	75	NA
Hibiscus kokio ssp. *kokio*	75	NA

Table 4.2.1-3 (continued). Numbers of native plant taxa outplanted in coastal areas of ʻOheʻo. Source: NPS (2017a).

Taxon	**Plants**	**T&E**
Ipomoea pes-caprae	2	NA
Ishaemum byrone	822	E
Machaerina mariscoides	5	NA
Melicope clusifolia	1	NA
Melicope ovalis	1	NA
Metrosideros polymorpha	1	NA
Myoporum sandwicense	2	NA
Nototrichium sandwicense	24	NA
Osteomeles anthyllidifolia	32	NA
Pandanus tectorius	1,022	NA
Perottetia sandwicensis	25	NA
Phyllostegia macrophylla	1	NA
Phytolacca sandwicensis	12	NA
Pisonia umbellifera	195	NA
Pittosporum glabrum	1	NA
Portulaca lutea	23	NA
Portulaca villosa	234	E
Pritchardia woodii	4	NA
Psydrax odoratum	15	NA
Psychotria mariniana	3	NA
Polyscias sandwicensis	8	NA
Rhus sandwicensis	13	NA
Santalum ellipticum	7	NA
Scaevola taccada	2,513	NA
Sesuvium portulavacastrum	4	NA
Strongylodon ruber	6	NA
Syzgium malaccense	6	NA
Thespesia populnea	62	NA
Vigna marina	65	NA
Wikstromia uva-ursa	180	NA
Total	6,516	–

Because much of the coastal area has been heavily impacted and modified by human activities, efforts to control non-native plants are typically done on a case by case basis, and are directed towards those which may be affecting the visitor experience, or interfering with restoration or maintenance of rare coastal plants or communities. None of the non-native plants in coastal areas are currently targeted for eradication. From 1992 through the present, eighty-two species, totaling 184,726 individuals, were controlled in the ʻOheʻo region of the park (NPS 2017b). Some of the most important non-native species controlled at ʻOheʻo are listed in Table 4.2.1-2.

Condition and trend determination for coastal vegetation

In summary, the condition of the coastal vegetation of HALE has been heavily modified and consists of predominantly non-native dominated communities throughout the three coastal regions of the park. Coastal Nuʻu is the most degraded of the three, with the lowest native diversity and only small pockets of native species surrounded by non-native grasslands and thickets. Coastal Kaʻāpahu is also dominated by non-native vegetation, but has some remnant native strand communities and notable forest species further inland. Coastal ʻOheʻo retains some small, but relatively good quality coastal shrubland and forest. It has the highest native diversity of the three sites, and is also relatively speciose compared to other coastal sites throughout the island. Relative to ideal reference conditions, the current overall state of coastal vegetation is poor and warrants significant concern. The trend in this condition appears unchanging: due to the already heavily impacted nature of the coastal communities, and the resilience of the remnant native vegetation, an equilibrium may have been reached in which remaining native communities persist with minimal management intervention. Ongoing weed control efforts can maintain this status, but future ungulate management and concerted restoration efforts will be needed to enhance or expand native-dominated communities and plant populations. Confidence in this assessment is high. Although additional vegetation monitoring and survey plots would be desirable, and might reveal the presence of additional native species that would benefit from management, it is unlikely that additional data would substantially modify the overall assessment of coastal vegetation communities in the park.

Coastal vertebrates

Honu (Green sea turtles) are commonly viewed in waters off HALE coastlines (NPS 2012b), but no sea turtles are known to nest in HALE, likely owing to an absence of sandy beaches within the park. Īlio-holo-i-ka-uaua (Hawaiian monk seals) are occasionally observed resting on rocky beaches within HALE, such as at Kukui Bay and other locations at ʻOheʻo (NPS 1999, 2012b). According to official records collected by the Pacific Islands Fisheries Science Center, of the National Marine Fisheries Service, a total of seven seal sightings have been reported at HALE between 2007 and 2016 (PIFSC 2017), six of which were at ʻOheʻo Gulch, and the location of the seventh was unspecified but also likely at ʻOheʻo. One sighting occurred in 2011, and three each were reported in 2012 and 2013. Only one sighting was confirmed, and was identified as a tagged adult male. No seal births have been documented within the park (PIFSC 2017). Because numbers of seals are low in the main Hawaiian Islands (Hawaiʻi DLNR 2005c), the remote and relatively undisturbed nature of portions of the HALE coastline should be preserved.

Information on the status of birds in HALE coastal ecosystems is largely limited to two surveys of two of the park's coastal units (ʻOheʻo to Puhilele Point, and Kaʻāpahu) that took place in March and July of 2005 (Kozar et al. 2007, Natividad Bailey 2009). A survey of lower Kīpahulu Valley was also conducted in 1980 (Stemmermann 1980). Of seabirds that have been detected within HALE, those that might nest in coastal zones include ʻakeʻake (Band-rumped Storm-Petrel, *Oceanodroma castro*), although no nests have been located in the park (Natividad Bailey 2009). Noio (Hawaiian or Black Noddy) are commonly seen flying along HALE coastlines, and have been observed flying to nest sites on coastal cliffs at ʻOheʻo (Kozar et al. 2007, Natividad Bailey 2009). These nests are otherwise undocumented and not monitored. Koaʻe kea (White-tailed Tropicbird) are regularly seen flying over HALE coastal areas, but are thought to nest at higher elevations outside the coastal zone, though they do nest in coastal areas on other islands (Kozar et al. 2007, Natividad Bailey 2009). ʻIwa (Great Frigatebird) also are regularly seen flying over HALE coastal areas, but are not known to nest within the park (Kozar et al. 2007, Natividad Bailey 2009). The threatened ʻaʻo or Newell's shearwater (*Puffinus auricular newelli*) was heard calling at 1,750 ft elevation in Kīpahulu Valley in 1980 (Stemmermann 1980), and conceivably could have been nesting in the area or commuting to nesting areas in higher elevation areas of the park. One of the most common shoreline nesting seabirds in Hawaiʻi, ʻuaʻu kani (Wedge-tailed Shearwater), is not known to nest within HALE, although colonies are known to occur on off-shore islets approximately 10 km from ʻOheʻo (Natividad Bailey 2009). ʻOheʻo coastal areas could provide suitable nesting habitat for ʻuaʻu kani with appropriate vegetation restoration and feral predator management (Natividad Bailey 2009). Similarly, Kozar et al. (2009) observed that while habitat for seabird nesting in coastal areas is generally good in Hawaiʻi National Parks, including HALE, seabird numbers are relatively low, likely owing to predation pressures from a variety of introduced mammals. Control or eradication of non-native predators has resulted in growth of ʻuaʻu kani colonies on other islands in Hawai'i (Marie et al. 2014, VanderWerf and Young 2014, VanderWerf et al. 2014).

Three species of shorebirds have been documented to occur in HALE coastal ecosystems: kōlea (Pacific Golden Plover), ʻakekeke (Ruddy Turnstone), and ʻūlili (Wandering Tattler) (Kozar et al. 2007). Kōlea and ʻakekeke were observed foraging in grassy, shrubby and rocky areas slightly inland from the coast, while ʻūlili foraged for marine resources along wave-washed habitats nearer the waterline; these coastal areas appear to provide generally good habitat for migratory shorebirds (Kozar et al. 2007). Kōlea and ʻūlili also were observed in coastal areas of Kīpahulu Valley during surveys in 1980 (Stemmermann 1980).

No waterfowl or raptors were reported from either survey (Kozar et al. 2007, Natividad Bailey 2009). The only waterbird reported was the non-native cattle egret (*Bubulcus ibis*), which was the most common of the birds counted (63% of individuals) during the March, 2005 surveys (Kozar et al. 2007). This species could impact native seabirds via predation of nestlings or transmission of disease (Kozar et al. 2007). All passerine species observed in the March, 2005 coastal survey were non-native species (Kozar et al. 2007).

Both coastal bird surveys at HALE focused on documenting the presence of species, but because only 4 survey days during two periods in a single year were employed across both efforts, it is possible that additional bird species utilizing these coastal ecosystems went undetected. However, the likelihood that additional nesting seabird species inside HALE were missed seems low. Surface-nesting seabirds, such as noio and ʻiwa, are relatively conspicuous and likely would have been noticed both during the surveys and during the course of other resources management work in the area. Burrow-nesting seabirds, such as ʻuaʻu kani, are vulnerable to predators and generally have not persisted in Hawaiian coastal areas because of the abundance of predators. As noted earlier, shorebird species do not nest in Hawaii. No estimates of population sizes, breeding pairs, or other measures of abundance are available for coastal birds in HALE.

Condition and trend determination for coastal vertebrates

Relative to ideal reference conditions, the current condition of coastal vertebrates at HALE warrants significant concern. While substantial areas of suitable habitat are naturally lacking for sea turtles and monk seals, habitat capable of supporting seabird colonies along HALE coastlines has either been altered and made unsuitable, or exists but is largely unoccupied owing to predation from non-native mammals. Furthermore, while migratory shorebirds occur in HALE, coastal shrubland and forested ecosystems in HALE are completely devoid of native passerine or other birds, as is the case across the Hawaiian Islands. This absence results from a combination of factors including predation by non-native mammals, novel diseases vectored by non-native mosquitos, and nearly complete loss or conversion of native plant communities that may provide food resources (including native insect communities) unavailable in non-native plant communities (Pratt et al 2009). The trend in this condition is unchanging, as the primary causes of degradation took effect, in most cases, many decades or centuries ago. Much active management would be needed to ameliorate most of these conditions, such as intensive predator suppression or exclusion to recover coastal seabird colonies (Young et al. 2013, VanderWerf and Young 2014, VanderWerf et al. 2014). Confidence in this assessment is high, as it is unlikely that significant coastal vertebrate resources have gone undetected.

Coastal invertebrates

Aside from surveys of aquatic insects within the coastal zone (see Chapter 4.2.2), only two survey efforts have focused on arthropods, including insects, in coastal areas of HALE. The first includes a survey of spiders and relatives (Gon and Pinter 1980), conducted over four days in September 1980, and a survey of insects, millipedes and centipedes (Gagné 1980), conducted over an unspecified number of days in July, 1980. Both were components of an effort to document the resources of lower Kīpahulu Valley below 2000 ft (610 m) elevation, and focused survey efforts along the riparian corridors of Pua‘alu‘u Stream, ‘Ohe‘o Gulch, Palikea Stream, Kou‘kou‘ai Stream, and a non-riparian tract from Palikea to Pu‘u ‘Ahu‘ula (Gagné 1980, Gon and Pinter 1980). This effort also included a limited survey of land snails in the area (Severns 1980). The second survey was also a relatively rapid assessment of insects in portions of ‘Ohe‘o, in which areas mainly below the road (Highway 31) and between ‘Ohe‘o Gulch and the RM baseyard were surveyed over five days in June, November and December of 2000 (Takumi 2001). Arthropod communities in coastal Ka‘āpahu and Nu‘u have not been surveyed.

Native spiders in lower Kīpahulu Valley were found to be “associated with areas of more or less intact native vegetation” (Gon and Pinter 1980). As discussed in the section on coastal vegetation, few native plant communities persist below 1000 ft (305 m) in HALE. Surprisingly, Gon and Pinter (1980) found that 60% of species collected in areas below 1000 ft elevation were native. Nevertheless, the abundances of native individuals appeared to drop below this elevation, relative to areas above it. One exception to this was a riparian *Tetragnatha* species that was common even at low elevations (see Chapter 4.2.2).

Of the 398 species of insects, millipedes and centipedes collected by Gagné (1980), 54% were endemic. However, this figure does not distinguish between areas above and below 1000 ft (305 m) elevation, making it difficult to assess the nature of insect communities in lower coastal ecosystems. In general, Gagné (1980) stated that most of the endemic arthropods are now restricted to aquatic, shoreline, and riparian ecosystems, and in remnants of native vegetation. He further reported that most endemic species were found in the primarily native vegetation between Palikea Stream at 1550 ft (473 m) elevation and Palikea peak (at 2225 ft (678 m) elevation), outside the coastal zone. He concluded that "The current status of the arthropod fauna of the area is not satisfactory."

Of 206 species of insects collected at ʻOheʻo by Takumi (2001), mainly in non-riparian habitats ranging from 33-98 ft (10-30 m) elevation, only 15% were native. The majority of these were flies (Diptera) in a variety of families, but also included several species of moths (Lepidoptera), true bugs (Hemiptera), beetles (Coleoptera), thrips (Thysanoptera) and dragonflies (Odonata).

Severns (1980) reported that 5 of 11 terrestrial snail species found in lower Kīpahulu Valley were native. However, only one of these, an unidentified *Elasmias* species (family Achatinellidae), appears to have been found in the lower, more disturbed coastal zone (below 1000 ft). Severns (1980) reported it as "common on Palikea." Main causes for the very low diversity and abundance of native snails were stated to be predation from the invasive, predatory rosy wolf snail (*Euglandina rosea*), and destruction of habitat for ground-dwelling species by feral pigs and cattle (Severns 1980).

The impacts of invasive species on native invertebrates in coastal ecosystems stem from both loss of native plant-dominated habitats, and from competition and predation from invasive invertebrates. Although some native arthropods are generalists that can make use of non-native plants, such as some thomisid spiders (Gon and Pinter 1980), many are reliant on native host plants and disappear from exotic plant communities. A large number non-native arthropods were reported in the above surveys, and these likely underestimate the actual diversity of non-native species in these ecosystems. The presence of some of these, such as yellow crazy ants (*Anoplolepis gracilipes*) and some of the nine other ant species encountered (Gagné 1980, Takumi 2001), are likely to exert well-known and outsized impacts on native species (Krushelnycky et al. 2005). The effects of competition and predation from the wide diversity of other exotic arthropods are more poorly understood, but likely also have a large cumulative impact. For native snails, the presence of the rosy wolf snail is thought to be an important threat (Holland et al. 2012), and for ground-dwelling native invertebrates, both arthropods and molluscs, disturbance of habitat by feral ungulates and rodent predation are important additional factors.

Condition and trend determination for coastal invertebrates

Relative to ideal reference conditions, the current condition of coastal invertebrates at HALE warrants significant concern. In low-lying habitats, very few native species appear to persist. At somewhat higher elevations (up to 1000 ft), and in some riparian corridors, native species composition appears to increase, but all indications are that most native species are mainly restricted to pockets of habitat supporting native plants. Such native plant refuges are rare in coastal ecosystems at HALE (Green et al. 2015). Confidence in this assessment is low, because data on invertebrate communities in coastal ecosystems are very limited, both spatially and temporally.

Coastal Nuʻu and Kaʻāpahu, for example, are completely unsurveyed. Given the condition of vegetation communities in those areas, however, it is likely that invertebrate communities are also highly degraded. We are unable to estimate a trend in this condition, as the available information is too sparse. One might nevertheless predict that invasive invertebrate species have likely been increasing in diversity over time.

Overall assessment

Integration of the three main indicators of coastal ecosystems yields an overall condition warranting significant concern. The trend in this condition was assessed as unchanging, because the condition of both coastal vegetation and coastal vertebrates appears stable.

This condition assessment reflects the level of contemporary divergence from ideal reference conditions. It is important to note that, today, these ideal conditions for coastal ecosystems exist almost nowhere in Hawaiʻi, and indeed on very few oceanic islands. Coastal areas are typically the first to be modified by humans, and few insular natural communities can tolerate such conversion. This should not be taken to mean that coastal ecosystems in HALE do not retain important elements that should continue to be protected and managed, or that these ecosystems no longer hold ecological value. (return to Condition Summary)

Level of confidence

Overall level of confidence in this assessment is high. Although rigorous, standardized, and quantitative data on natural resources in these ecosystems is quite limited, the nature and extent of modifications to the original state are so pervasive and conspicuous that alternate conclusions are highly unlikely, even with additional data.

Information gaps and research recommendations

Repeated quantitative vegetation data are generally lacking. No NPS I&M Focal Terrestrial Plant Community or Established Invasive Plant Species monitoring plots are located in coastal ecosystems at HALE, yet additional vegetation plot data would greatly enhance future vegetation trend analyses. No surveys of invertebrates have been conducted in coastal ecosystems at Nuʻu or Kaʻāpahu, and no bird surveys from Nuʻu have been reported. Information on invertebrates at ʻOheʻo is also very limited in geographic and temporal scope.

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4.2.2. Freshwater Ecosystems

Condition Summary

Overall resource condition warrants moderate concern with an unchanging or stable trend; confidence in condition determination is medium. Rationale is discussed in the "*Overall assessment*" section and following.

↔

Relevance and Context

HALE's freshwater ecosystems are mainly represented by numerous intermittent and perennial streams, and three small montane lakes, in the eastern portion of the park (Figure 4.2.2-1, see also Chapter 4.1.3). Additional drainages on the western and southern faces of Haleakalā volcano flow only rarely, during extreme rainfall events, but may hold standing water in pools for longer periods.

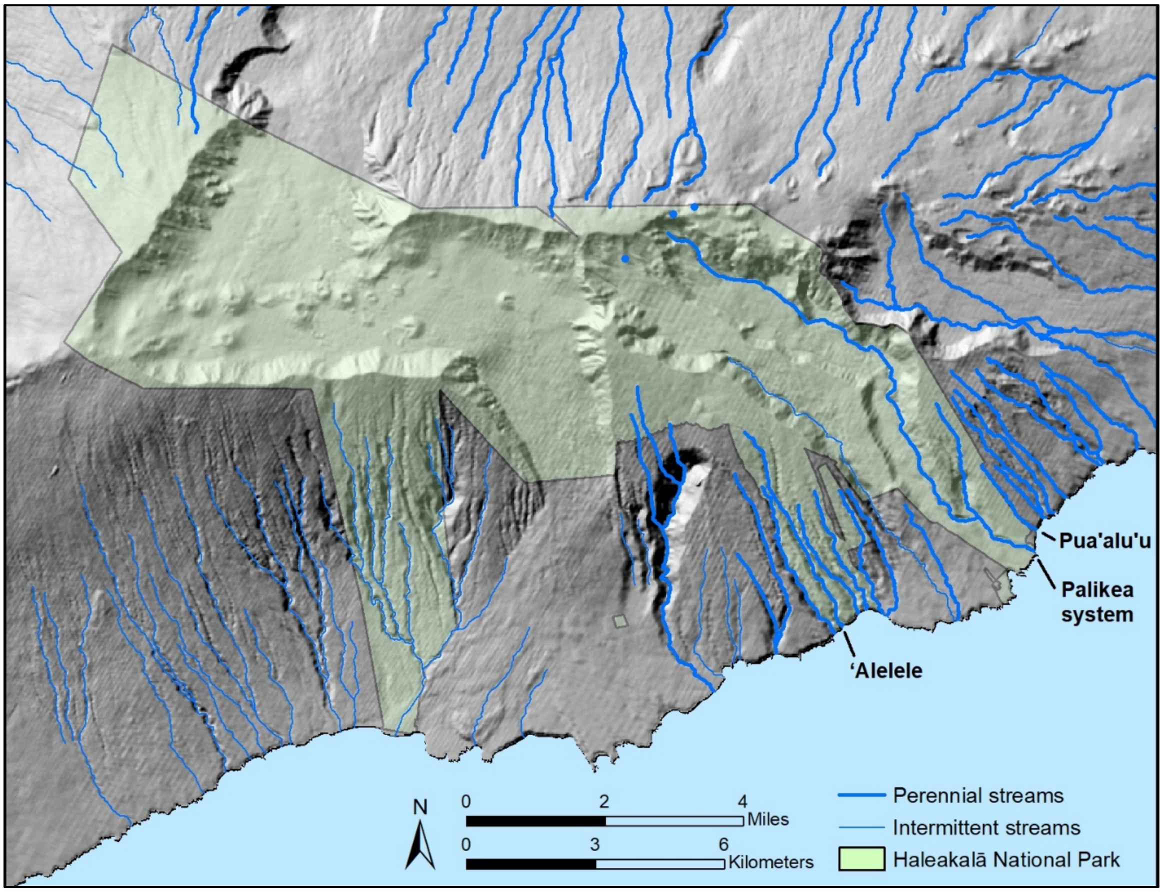

Figure 4.2.2-1. Surface waters in and around HALE. Shown are perennial and intermittent streams, and three small lakes. Streams referenced in this chapter are the Palikea Stream system (including Palikea Stream, its tributary Pīpīwai Stream, and ʻOheʻo Gulch below the confluence of the two), Puaʻaluʻu Stream just to the north of the Palikea system, and ʻAlelele Stream to the west. Source: State of Hawaiʻi Division of Aquatic Resources.

The condition of water quality parameters for freshwater ecosystems in HALE is evaluated in Chapter 4.1.3. In the present chapter, the status of biological communities in these ecosystems is assessed.

Investigations of Hawaiian freshwater communities have concentrated on the aquatic “macrofauna,” including fish, freshwater shrimps and prawns, and freshwater molluscs (Kinzie and Ford 1977, Hodges 1994, Brasher et al. 2011). Aquatic insects are also relatively well studied, especially the large radiation of charismatic damselflies (Polhemus and Asquith 1996).

Freshwater Fish

Hawai‘i has five native species of gobies (family Gobiidae) and sleepers (family Eleotridae), four of which are endemic (Fizsimons et al. 2007). These fish, known collectively as ‘o‘opu, are all diadromous, spending the majority of their lives in freshwater streams, but also requiring a several month dispersal period as planktonic larvae at sea (Brasher et al. 2011). These species tend to partition streams by reach and habitat type. *Eleotris sandwicensis* (‘o‘opu ‘akupa) lacks the fused pelvic fins of the four gobiid species, and *Stenogobius hawaiiensis* (‘o‘opu naniha) has only a delicately formed pelvic disk (Fitzsimons et al. 2007). Both species therefore are unable to climb waterfalls, and are typically restricted to the lowest stream reaches before the first steep waterfall. *Eleotris sandwicensis* is predatory, while *S. hawaiiensis* is omnivorous (Fitzsimons et al. 2007). *Awaous guamensis* (‘o‘opu nākea) and *Sicyopterus stimpsoni* (‘o‘opu nōpili) are usually found in the middle reaches of streams, but *A. guamensis* prefers bottoms of pools and sheltering behind boulders, while *S. stimpsoni* prefers rapidly flowing runs and riffles and the sides and tops of large rocks and boulders. *Awaous guamensis* is omnivorous, while *S. stimpsoni* is herbivorous (Fitzsimons et al. 2007). *Lentipes concolor* (‘o‘opu alamo‘o) is most common in upper reaches, often climbing tall waterfalls, but can also be found in lower stream reaches. It is omnivorous (Fitzsimons et al. 2007).

Freshwater Invertebrates

Hawaiian streams support two types of endemic crustaceans: the mountain shrimp, *Atyoida bisulcata* (‘ōpae kala‘ole), and a freshwater prawn, *Macrobrachium grandimanus* (‘ōpae ‘oehaa) (Hodges 1994, Tango et al. 2005). The mountain shrimp can occur through upper stream reaches, while the prawn is less common, and typically restricted to lower reaches and estuary areas (Tango et al. 2005). Both appear to feed on algae and detritus (Hodges 1994), and are diadromous, and thus disperse freely among streams.

Two native freshwater molluscs in the family Neritidae occur in Hawaiian streams. *Neritina granosa* (hīhīwai) occurs in lower and middle stream reaches, but is apparently restricted today to relatively few high-quality streams in the state (Brasher 1997, Brasher et al. 2011). *Theodoxus vespertinus* (hapawai) occupies brackish waters of estuaries and lower stream reaches (Kinzie and Ford 1977, Brasher 2003). Both neritid snail species are diadromous. Several native freshwater snails in the family Lymnaeidae occur along stream banks, often near seeps and waterfalls (Kinzie and Ford 1977, Brasher 2003). Of these native molluscs, *N. granosa* has been the primary focus in the majority of stream surveying and monitoring efforts.

Indicative of the overall disharmonic nature of Hawaiʻi's native insect fauna, several important groups of aquatic insects, like caddisflies (Trichoptera) and mayflies (Ephemeroptera) are missing from the native aquatic fauna. Despite this, a wide variety of endemic aquatic insects have evolved in the islands, and are well represented in pristine Hawaiian streams (Englund and Polhemus 2001). Apparently unique to Hawaiʻi, several groups of moths in the genus *Hyposmocoma* have evolved amphibious larvae that can feed on algae completely submerged in streams (Rubinoff and Schmitz 2010). Showy damselflies in the genus *Megalagrion* are among the most prominent representatives of the native aquatic insect fauna, with 23 species statewide, and 7 known from east Maui freshwater habitats (Polhemus and Asquith 1996, Jordan et al. 2003).

Impacts to Freshwater Systems

A large number of invasive species have been introduced, either purposely or inadvertently, to Hawaiian streams (Brasher 2003). The Tahitian prawn, *Macrobrachium lar*, was first released in Hawaiʻi in 1956 for food, and quickly spread throughout the state (Brasher 2003). It is omnivorous, having been observed to feed on ʻoʻopu and their egg masses, and on native molluscs (hīhīwai) (Hodges 1994). They may compete with native *A. bisulcata* for food and space, and may directly prey on them as well (Hodges 1994). A wide variety of non-native fish have been introduced for food and sport, in efforts to control mosquitos, or as releases of aquarium species (Brasher 2003). These are thought to have devastated native aquatic insects, such as *Megalagrion* damselflies (Polhemus and Asquith 1996, Englund 1999), as well as native ʻoʻopu and ʻōpae (Brasher 2003). In addition to exerting predatory and competitive pressures, they may serve as sources of new parasites on native fish (Font and Tate 1994). Invasive insects, such as ants, may impact native aquatic and riparian insects along riparian corridors (Hardy 1979).

Hawaiian freshwater ecosystems, especially streams, have suffered much degradation from anthropogenic pressures, in addition to those caused by the introduction of invasive species (Kinzie and Ford 1977, Polhemus and Asquith 1996, Brasher 2003, Brasher et al. 2011). These include impacts caused by diversion for agriculture, channelization and other morphological alteration resulting from development, and contamination from urban and agricultural pollutants. Alteration of stream flow can be especially damaging to Hawaiian aquatic communities, because, as noted above, all of Hawaiʻi's native macrofauna species are diadromous, and thus require unimpeded movement between freshwater and marine environments to complete their lifecycles. Relatively few streams in Hawaiʻi remain in good condition today, and as long ago as the 1970's Kinzie and Ford (1977) recognized the exceptional value of the Palikea Stream system in HALE (comprised of Palikea Stream, its tributary Pīpīwai Stream, and the reaches below their confluence usually referred to as ʻOheʻo Gulch), writing:

> *"Palikea is one of the largest streams on Maui (by length and discharge) which is not diverted, and therefore it may provide a suitable reservoir for significant communities of endemic aquatic organisms. Because most freshwater ecosystems in Hawaiʻi are degraded, and many of the remainder are threatened by water exploitation and development, it is important to realize that Palikea-Pīpīwai is the*

> *only major perennial stream system of high natural quality that is currently under jurisdiction of the National Park Service."*

Since the addition of the Kaʻapahu parcel to HALE, the perennial ʻAlelele Stream also falls completely within NPS boundaries.

In this assessment, we focus on the condition of HALE biotic communities in ʻAlelele Stream and in the Palikea Stream system. These represent the major freshwater habitats in the park, as well as those that have undergone the most study. For indicators or measures of freshwater ecosystem condition, we focus on fish and aquatic invertebrates (including molluscs, crustaceans and insects). We comment on other habitats, such as Puaʻaluʻu Stream, and other potential indicators of condition where possible.

Indicators

- Fish
- Aquatic invertebrates

Data and Methods

A review of available data and information was used to assess conditions, and if possible, trends, in the indicators listed above. Aside from raw data review and tabulation, no new data analysis was performed.

Sources of information

The majority of information on the condition of HALE freshwater biotic communities has been collected in the Palikea Stream system. Kinzie and Ford (1977) conducted an initial survey in 1975-76 that was mainly qualitative in nature, mostly reporting the presence and absence of the main macrofauna taxa, as well as some information on aquatic insects and riparian flora. Additional surveys of aquatic and riparian insects (and arthropod relatives) were conducted in and around the Palikea Stream system and nearby Puaʻaluʻu Stream in 1979 (Hardy 1979), 1980 (Gagné 1980, Gon 1980) and 1993 (Polhemus 1993). Hodges (1994) conducted the first quantitative survey of the Palikea Stream system in 1993, focusing on distribution and abundances of fish, the native shrimp *A. bisulcata*, the native mollusc *N. granosa*, and the invasive Tahitian prawn *M. lar*. He also provided a less extensive and more qualitative survey of the lower reaches of Puaʻaluʻu Stream. More recently, the NPS I&M program conducted an initial inventory of macrofauna in ʻAlelele Stream in 2005 (Tango et al. 2005), and the NPS I&M program initiated standardized monitoring of instream aquatic macrofauna in HALE in 2009. This consists of regular quantification of native and introduced fish, shrimp and snail abundances at fixed and randomly selected stations along the Palikea Stream system (including Pīpīwai and ʻOheʻo Gulch) and in ʻAlelele Stream (Brasher et al. 2011, Figure 4.2.2-2). This monitoring also collects data on stream habitat characteristics (separate from water quality). Results of this monitoring have not yet been formally reported, but the NPS I&M program has shared unpublished data from 2009-2013 for this assessment.

Figure 4.2.2-2. NPS I&M staff surveying aquatic macrofauna in a pool in lower ‘Ohe‘o Gulch. Photo: NPS.

Reference condition

An ideal reference condition for freshwater ecosystems in HALE streams consists of completely unaltered stream habitats, with fully intact biotic communities and no invasive species. Because pre-human species diversities and population abundances in HALE streams are unknown, we consider good conditions as possessing the full complement of native aquatic macrofauna in densities comparable to other Hawaiian streams thought to be in good condition, a diverse assemblage of native aquatic insects, and an absence of damaging invasive species.

Condition and Trend

Fish

Kinzie and Ford (1977) and Hodges (1994) found qualitatively similar fish communities in the Palikea Stream system in 1975-76 and 1993, respectively: both encountered the same three native goby species. Kinzie and Ford (1977) conducted visual observations at 13 stations along the stream system, and found *A. guamensis* at 4 of 9 stations along the ‘Ohe‘o Gulch section and none of the 4 stations on Palikea and Pīpīwai Streams; similarly found *S stimpsoni* at 2 of 9 stations along ‘Ohe‘o Gulch and none on Palikea and Pīpīwai Streams; and found *L. concolor* at 2 of 9 stations along ‘Ohe‘o Gulch, at 1 of 2 stations on Pīpīwai Stream, and 0 of 2 stations on Palikea Stream. Hodges (1994) conducted visual observations in ten 1 x 1 m quadrats at each of 18 stations along the stream system during each of two events, and observed *A. guamensis* at 6 of 10 stations along the ‘Ohe‘o

Gulch section and none of the stations on Palikea and Pīpīwai Streams; observed *S. stimpsoni* at 1 of 10 stations along ʻOheʻo Gulch and none on Palikea and Pīpīwai Streams; and found *L. concolor* at 7 of 10 stations along ʻOheʻo Gulch, at 3 of 3 stations on Pīpīwai Stream, and 3 of 5 stations on Palikea Stream. Therefore, both surveys found *A. guamensis* to be widely distributed along lower to middle reaches of the stream system, *S. stimpsoni* to be sparsely to very sparsely distributed along middle reaches of the stream, and *L. concolor* to be the most widely distributed species, occurring from the lowest to the highest stream reaches. An absence of quantitative data in the earlier survey prevents comparison of relative densities of these species across the two time periods.

Two and a half decades after the Hodges (1994) survey, the NPS I&M program monitoring detected similar fish community composition in the Palikea Stream system. The latter monitoring effort made observations in ten 1 x 1 m quadrats at each of 6 to 14 stations along the stream system each year from 2009-2013, and again detected the same three species, plus one individual of *S. hawaiiensis* during the entire monitoring period; the latter was observed in the first pool just before the stream terminus at the ocean (NPS I&M unpub. data). Quantitative fish counts in this effort and the Hodges (1994) effort permit comparison of proportional composition of these fish species in 1993 and 2009-2013 (Figure 4.2.8-3). During both time periods, *L. concolor* was the most abundant species by far, but the relative abundances of *A. guamensis* and *S. stimpsoni* appears to have shifted somewhat over time. The nearly complete absence of *E. sandwicensis* and *S. hawaiiensis* from the Palikea Stream system over all three surveys spanning 40 years is readily explained by the morphology of the stream terminus: a small but apparently significant waterfall at the marine boundary usually prevents colonization of the stream by these two species, which have only poor or no climbing abilities (Kinzie and Ford 1977, Hodges 1993, Tango et al. 2005). Occasional access may be provided by unusually high tides or storm surges, when waves can wash into the first pools (Kinzie and Ford 1977).

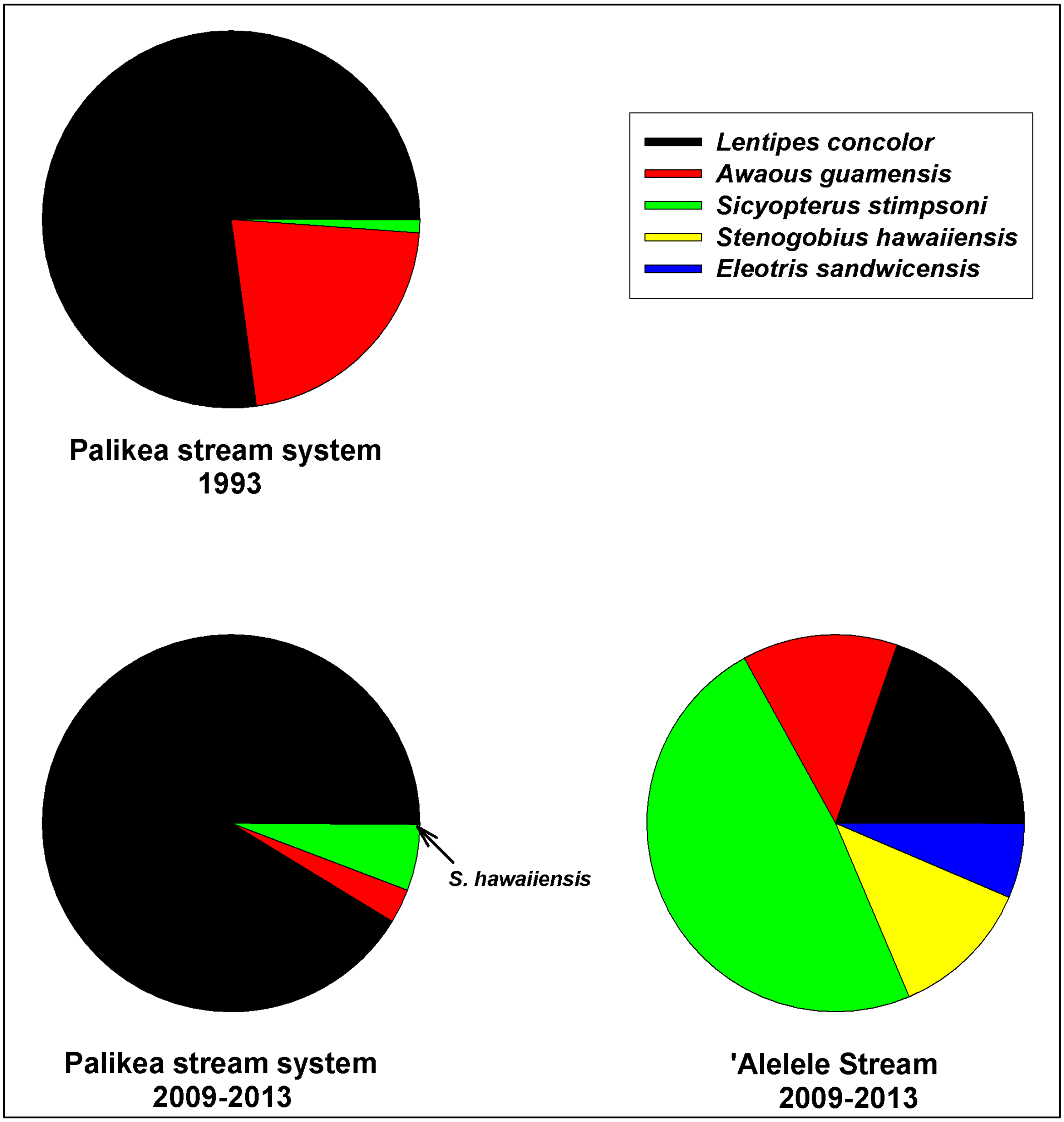

Figure 4.2.8-3. Proportional composition of goby species observed in HALE streams during quantitative surveys in 1993 and 2009-2013. One individual of *S. hawaiiensis* was observed in the Palikea stream system from 2009-2013, which cannot be seen in the pie chart. Source: Hodges (1994), NPS I&M unpublished data.

Densities of all ‘o‘opu species (excluding unidentifiable juveniles, termed hinana) in the Palikea Stream system averaged 0.5 fish/quadrat in 1993 (Hodges 1994) and averaged 1.8 fish/quadrat from 2009-2013 (NPS I&M unpub. data). Although densities were higher in the latter period, they fluctuated somewhat from year to year, with some years exhibiting densities more similar to that observed in 1993 (Figure 4.2.8-4).

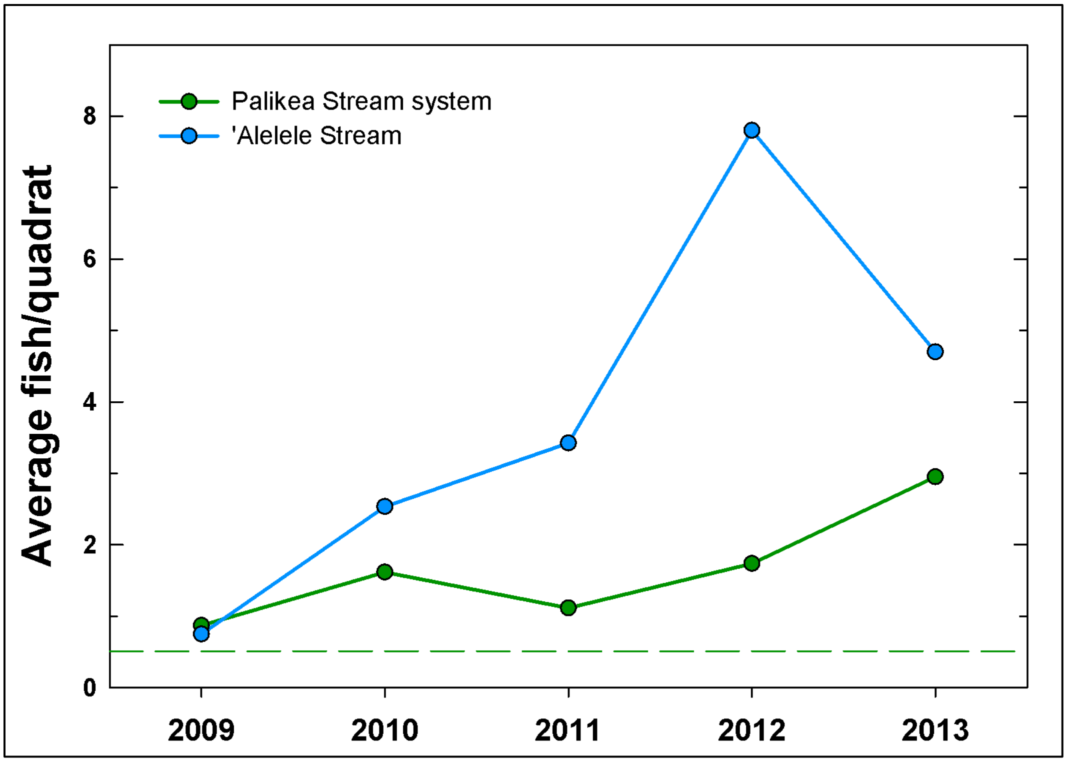

Figure 4.2.8-4. Average numbers of native gobies (of all species) per 1 m^2 quadrat from 2009 to 2013 in the Palikea Stream system (green circles and solid lines) and ʻAlelele Stream (blue circles and solid lines). The green dashed line indicates the average density measured in the Palikea Stream system in 1993. Source: Hodges (1994), NPS I&M unpublished data.

Both Kinzie and Ford (1977) and Hodges (1994) commented on the perceived relatively low densities of ʻoʻopu in this stream system, particularly in the ʻOheʻo Gulch section (below the confluence of Palikea and Pīpīwai), compared to other high-quality streams in Hawaiʻi. It is possible that this condition is related to the extreme variation in discharge volume, current velocity and water turbidity that naturally characterizes this stream system (Kinzie and Ford 1977). During dry periods, flow can stop altogether along certain reaches, creating relatively stagnant conditions in some pools. In contrast, during wetter periods, torrential flooding can occur, stemming from the lengthy course of Palikea Stream and its large area of drainage. The effects of extreme spates are evident in the scoured bedrock runs and channels in sections of the ʻOheʻo streambed (Polhemus 1993). Hodges (1994) noted the divergence of this morphology from other streams that provide good freshwater habitat on East Maui, such as Hanawī and Honomanū, stating that current and boulder riffles that are common in streams hosting large populations of the amphidromous fauna are uncommon in ʻOheʻo Stream. He further noted that turbidity was often high, and that extremely large spates were common during his study, which may impact the faunal populations.

However, Hodges (1994) also noted that densities of ʻalamoʻo [*L. concolor*] in certain parts of ʻOheʻo were higher than in any other streams observed in Hawaiʻi, and were also larger in size. Given that this species has been characterized as the least common of the native gobies statewide (Hodges 1994, Brasher et al. 2011), the Palikea Stream system should be recognized as an important refuge for native fish in general and this species in particular. Furthermore, generally low densities of gobies may result largely from natural stream characteristics, rather than principally from human-caused degradation. It is also possible that the assessments of low fish densities by both Kinzie and Ford (1977) and Hodges (1994) were influenced by high stream levels during their study periods.

Much less information exists for the fish fauna of ‘Alelele Stream, although surveys in recent years suggest a diverse and healthy community. NPS I&M monitoring made visual observations at ten 1 x 1 m quadrats at each of 2 to 4 stations along the stream system each year from 2009-2013, detecting all five of the native goby species (Figure 4.2.8-3). A separate survey of ‘Alelele in 2005 also detected all five species (Tango et al. 2005). Because ‘Alelele possesses a well-developed estuary section, both *E. sandwicensis* and *S. hawaiiensis* find suitable habitat in this stream. Unlike the Palikea Stream system, *S. stimpsoni* is a much more common member of the community, and *L. concolor* is not numerically dominant. However, upper reaches of ‘Alelele are harder to access, and a greater number of sampling stations in upper sections of the stream might produce a higher proportion of *L. concolor*. Densities of all ‘o‘opu species averaged 2.8 fish/quadrat from 2009-2013, with substantial variation in density between years (Figure 4.2.8-4); somewhat higher densities of gobies may exist in ‘Alelele Stream compared to the Palikea Stream system.

In addition to ‘Alelele Stream and the Palikea Stream system, the small perennial Pua‘alu‘u Stream, near ‘Ohe‘o Gulch, has been investigated several times. Although this stream is partially diverted for domestic and agricultural use, Kinzie and Ford (1979) reported that, in 1977, *A. guamensis*, *S. stimpsoni*, and *L. concolor* were all present, with large numbers of the latter species distributed throughout 9 of 10 sampling stations. Hodges (1994) collected quantitative data at ten 1 m^2 quadrats at each of three stations in the lower reaches of Pua‘alu‘u on four separate dates in 1993, and found much higher densities of ‘o‘opu (average of 3.5 fish/quadrat) than in the Palikea Stream system during the same year (average of 0.5 fish/quadrat). The vast majority of these were *L. concolor*, with a few *A. guamensis* and *S. stimpsoni* also observed. Several other HALE streams in the vicinity of ‘Alelele were scouted in 2005, but they did not hold water at that time and were not surveyed (Tango et al. 2005).

Condition and trend determination for fish

In summary, the condition of fish in HALE streams appears to be good. All five native gobies occur in the park, including apparently large populations of the least common native species, *L. concolor*. In addition to the Palikea Stream system, ‘Alelele, Pua‘alu‘u, and possibly other streams in the park support native fish populations. Unlike many Hawaiian streams, no non-native fish have been reported from HALE streams. Based on similar patterns in species composition and stable or possibly increasing densities over a period of several decades in the Palikea Stream system, the trend in fish resource condition appears stable. Confidence in this assessment is medium, because information for assessing trends is limited, with information lacking over large periods of time, and magnitude of inter-annual variability uncertain. Ability to evaluate trends more robustly will improve as the NPS I&M dataset lengthens.

Aquatic invertebrates

Molluscs

The same surveys that recorded fish distributions in HALE streams, reviewed above, also recorded observations of the native freshwater neritid mollusc, *N. granosa* (hīhīwai). Kinzie and Ford (1977) found *N. granosa* at 4 of 9 stations sampled in the ʻOheʻo Gulch section of the Palikea Stream system in 1975-76, but none at the 4 stations along Palikea and Pīpīwai Streams. Their survey also reported the presence of the native estuarine neritid, *T. vespertinus*, at the lowest pool near the terminus, and two endemic freshwater snails in family Lymnaeidae, *Pseudisidora rubella* and *Erinna aulacospira*, at two middle to upper reach stations. In 1993, Hodges (1994) observed only 2 individuals of *N. granosa* in the Palikea Stream system, even though he surveyed 10 quadrats (0.25 x 0.25 m) at each of 17 stations during each of three events. Both snails were at a single station in the middle reaches of ʻOheʻo Gulch, which was also the only location where egg cases were seen. Between 2009-2013, NPS I&M surveys failed to detect any *N. granosa* in the Palikea Stream system (NPS I&M unpub. data). These surveys made observations at ten quadrats (0.25 x 0.25 m) at each of 6 to 14 stations along the stream system each year. Although Kinzie and Ford (1977) provided no statements as to the numbers of *N. granosa* that they observed, the presence of this snail at 30% of stations surveyed in 1975-76 strongly suggests that its population has declined substantially in subsequent years.

In ʻAlelele Stream, a survey in 2005 found *N. granosa* in the upper reaches, and in the plunge pool at the base of the first falls (Tango et al. 2005). Between 2009-2013, NPS I&M surveys observed only a single individual of this species in ʻAlelele Stream, despite surveying 10 quadrats (0.25 x 0.25 m) at each of 2 to 4 stations each year (NPS I&M unpub. data).

In Puaʻaluʻu Stream, *N. granosa* was reported to be "abundant" at each of the three lower reach stations surveyed in 1977, and present at 6 of 8 stations overall (Kinzie and Ford 1979). Kinzie and Ford (1979) further reported the presence of four additional native freshwater mollusc species, describing the molluscan community as "both abundant and diverse." In 1993, Hodges (1994) observed a total of 2, 8, 2 and 3 individual *N. granosa* in four separate surveys of Puaʻaluʻu Stream, each of which utilized 10 quadrats (0.25 x 0.25 m) at each of 3 stations. Although these densities were higher than those observed in the Palikea Stream system in the same year (see above), they would appear to be substantially lower than the qualitative descriptions of hīhīwai densities in Puaʻaluʻu Stream in 1977 (Kinzie and Ford 1979).

On the balance of available information, populations of *N. granosa* in HALE streams appear to have declined between the 1970's and 1990's, and have not subsequently recovered. Currently, this species appears to be very rare in the park. No obvious changes in habitat or water quality in HALE streams readily explain these declines, with the exception of the establishment of the invasive Tahitian prawn (*M. lar*), which appears to be a prominent feature of HALE stream communities. Kinzie and Ford (1977) reported it from 6 of 9 observation stations along ʻOheʻo Gulch, but not from Palikea and Pīpīwai Streams, in 1975-76. Similarly, Hodges (1994) found it throughout the Palikea system in 1993, but most abundantly in the lower and upper reaches of the ʻOheʻo section, below the confluence of Palikea and Pīpīwai. His qualitative judgement was that densities of *M. lar* were relatively high compared to other streams he had surveyed. NPS I&M surveys also found *M. lar* throughout the Palikea Stream system from 2009-2013 (NPS I&M unpub. data). *Macrobrachium lar* is also present in ʻAlelele Stream, but NPS I&M surveys encountered only 5 individuals from a total of 160 quadrats surveyed from 2009-2013.

Macrobrachium lar is reported to prefer deep, shaded pools (Hodges 1994, Tango et al. 2005), which may explain their abundance in ʻOheʻo Gulch and relative scarcity in ʻAlelele Stream. This relatively large prawn has been observed to feed on hīhīwai (Hodges 1994, Brasher et al. 2011). However, without direct study of the effects of *M. lar* on hīhīwai populations, it is difficult to be certain of its role in the decline of hīhīwai populations. It is possible, for example, that populations of hīhīwai in HALE could be affected by demographic factors caused by deteriorating conditions in other streams outside the park. Because their larvae disperse through the ocean and often subsequently establish in other streams (Brasher et al. 2011), the overall source pool of larval colonists may be severely reduced as more streams across the state become inhospitable to adult stages.

Crustaceans

Kinzie and Ford (1977) found the native mountain shrimp (ʻōpae kalaʻole), *A. bisulcata*, to be widespread in their 1975-76 faunal surveys of the Palikea Stream system, described above. *Atyoida bisulcata* was present at 6 of 9 stations along the ʻOheʻo Gulch section of the stream, and all four of the Palikea and Pīpīwai stations. They also recorded the endemic prawn (ʻōpae ʻoehaa), *M. grandimanus*, in one of the lower ʻOheʻo pools. In 1993, Hodges (1994) also encountered a relatively abundant population of *A. bisulcata* in the Palikea Stream system. *Atyoida bisulcata* was present at 14 of 16 stations surveyed, and he observed a total of 1,399 individuals over the course of two survey events, for an average of 4.4 ʻōpae/quadrat surveyed (quadrats were 1 x 1 m). In contrast, from 2009 to 2013, the NPS I&M surveys only observed a total of 5 individuals of *A. bisulcata* in the Palikea Stream system, despite surveying a total of 589 quadrats (1 x 1 m) during this period, which were spread across 6 to 14 stations each year (NPS I&M unpub. data). This translates to an average density of only 0.008 ʻōpae/quadrat surveyed, which is over 500 times lower than the density observed in 1993. *Macrobrachium grandimanus* was not reportedly observed by Hodges (1994), and was also not encountered in the 2009-2013 NPS I&M surveys.

Atyoida bisulcata was observed in the plunge pool below the first large falls in ʻAlelele Stream in 2005 (Tango et al. 2005), but no individuals of this species were detected in NPS I&M surveys from 2009-2013, which surveyed 10 quadrats (1 x 1 m) at each of 2 to 4 stations each year (NPS I&M unpub. data).

Both Kinzie and Ford (1979) and Hodges (1994) reported *A. bisulcata* to be abundant in Puaʻaluʻu Stream in 1977 and 1993, respectively. Hodges' survey indicated higher densities of ʻōpae in Puaʻaluʻu Stream than in the Palikea Stream system in 1993.

Similar to the situation with the mollusc *N. granosa*, populations of the crustacean *A. bisulcata* appear to have declined dramatically in recent decades in HALE streams. Compared to *N. granosa*, the decline in *A. bisulcata* may have begun later, as it appeared to still be relatively abundant in the 1990's. The invasive Tahitian prawn has been hypothesized to impact native ʻōpae through interference competition for space (Hodges 1994, Brasher et al. 2011), and may therefore be at least partly responsible for their apparent decline. NPS I&M surveys found *M. lar* throughout the Palikea Stream system from 2009-2013 in densities far higher than native crustaceans: they observed a total of 242 individuals of *M. lar* compared to only 5 *A. bisulcata* in their sampling quadrats (NPS I&M unpub. data). As with *N. granosa*, however, it is possible that populations of *A. bisulcata* in HALE could be affected by deteriorating conditions in other streams outside the park.

Insects

Insect surveys of HALE freshwater ecosystems from 1979-1993, restricted to elevations below 2000 ft (610 m), documented the presence of a wide variety of native aquatic and riparian insects and arthropod relatives (Hardy 1979, Gagné 1980, Gon and Pinter 1980, Polhemus 1993). These included numerous species in several principal orders: Coleoptera (families Dytiscidae and Hydrophylidae), Diptera (families Canaceidae, Cecidomyiidae, Ceratopogonidae, Chironomidae, Dolichopodidae, Ephydridae, Psychodidae, Sciaridae and Tipulidae), Hemiptera (families Saldidae and Veliidae), and Odonata (families Aeshnidae, Libellulidae and Coenagrionidae). Of the odonates, five of the seven endemic species of *Megalagrion* damselflies known to occur in East Maui aquatic habitats were recorded from HALE streams in the surveys listed above: *M. blackburni, M. calliphya, M. hawaiiense, M. nigrohamatum,* and *M. pacificum*. At least one species, *M. nigrohamatum nigrohamatum*, was relatively common around Palikea Stream at higher elevations, around 3,000 ft (915 m), as of the late 1990's (P. Krushelnycky pers. obs.), but its current status is unknown.

The lowland species *M. pacificum* is of particular interest, because while its historic distribution included all of the main islands, its range shrank in recent decades to include only windward areas of Maui and Moloka‘i (Polhemus 1993), and is now listed as federally endangered (NPS 2012). In 1993, adults were "regularly observed along the course of Palikea Stream between 450 and 1200 ft, indicating that a large and healthy breeding population is present" (Polhemus 1993); the current status of populations in these areas is unknown. Gon and Pinter (1980) also reported one native spider (*Tetragnatha uncifera*) building webs along the riparian corridor of the Palikea Stream system nearly down to sea level. Numerous other species were collected in riparian corridors, but are not strictly aquatic or restricted to riparian habitat. In recent decades, one species of amphibious caterpillar was collected from Palikea Stream around 3,000 ft (915 m) elevation in 2004, and subsequently described as a new species, *H. moopalikea*, known only from that locality and likely endemic to Palikea Stream (Schmitz and Rubinoff 2011). There appear to be few reported collections of aquatic insects from higher elevations of Palikea Stream, but this habitat is generally in very good condition and undoubtedly supports many additional endemic species.

Polhemus' 1993 survey provides the best overall characterization of the aquatic insect fauna of HALE's lower streams, even though he considered it to be of relatively low intensity (14 hours of sampling over 2 days) and therefore preliminary (Polhemus 1993). A total of 34 aquatic insect species were collected, 29 of which (85%) were native. This high percentage stands in contrast to the composition of the surrounding terrestrial insect fauna. Gagné (1980), for example, reported an overall endemic composition of 54% among (non-spider) arthropods surveyed below 2,000 ft (610 m). This also included areas with more intact native vegetation; lower areas dominated by alien vegetation, such as the environs of the lower Palikea Stream system, would undoubtedly exhibit a lower percentage of native arthropods. Gon and Pinter (1980) reported few native spiders below approximately 1,000 ft (305 m) elevation. Takumi (2001) found only 14% of terrestrial (non-riparian) insects in the lower ʻOheʻo area to be native in a 2000 survey.

Many non-native insects have established in Hawaiʻi, including species in and around streams. The caddisfly, *Cheumatopsyche pettiti*, was apparently purposely introduced as food for game fish in the 1950's, and subsequently became one of the "most ubiquitous exotic species occurring in Hawaiian streams" (Kinzie and Ford 1979). It has been present in HALE streams since at least the 1970's (Hardy 1979, Kinzie and Ford 1979, Polhemus 1993), but its impact on native food webs is unknown. One of the most worrisome group of invasive insects is ants, some species of which, while not aquatic, may become very abundant along stream banks. The yellow crazy ant, *Anoplolepis gracilipes*, is a good example, and has been present in the lower Kīpahulu Valley area, including along Palikea Stream, ʻOheʻo Gulch, and Puaʻaluʻu Stream, since at least the 1970's (Hardy 1979, Gagné 1980). Based on a pattern of very low abundance and diversity of native aquatic insect species along a stretch of Puaʻaluʻu Stream that supported high densities of *A. gracilipes*, Hardy (1979) argued that this ant was likely responsible for eliminating or greatly reducing the native aquatic fauna at this location, and further hypothesized that it had produced a similar effect several years earlier along ʻOheʻo Gulch. While the evidence was circumstantial, Hardy's conclusion was consistent with what is generally known about the impacts of invasive ants on native Hawaiian insects (Krushelnycky et al. 2005). These impacts, however, are density dependent, and invasive ant populations can fluctuate dramatically over time; current densities of yellow crazy ants in the vicinity of HALE streams are unknown. Aside from *A. gracilipes*, Takumi (2001) documented the presence of an additional 9 introduced ant species in the lower ʻOheʻo area, although the riparian corridor was not surveyed at that time.

The relatively intact nature of the aquatic insect fauna in the Palikea Stream system, at least as measured prior to 1994, is likely attributable in large part to the absence of non-native fish, which are thought to have devastated native aquatic insects in many other streams across the state (Englund 1999, Englund and Polhemus 2001). It is unknown if aquatic insect communities currently remain diverse and relatively healthy in HALE freshwater ecosystems, as no known systematic surveys have focused on aquatic and riparian insects since 1993. Regardless, the prevention of invasive fish establishment is critical to maintaining populations of whatever native species persist. Other major threats include invasive riparian vegetation that can alter microhabitats, both in the water and along the banks, for both foraging and breeding. Riparian vegetation is almost completely non-native and often highly altered in structure along most sections of HALE streams below about 1,600 ft (488 m) (Kinzie and Ford 1977, 1979; Green et al. 2015). The composition and structure of riparian vegetation can affect water temperature, degree of stream bank erosion, and the nature and amount of organic material input into aquatic trophic webs (Kinzie and Ford 1977). Some types of non-native vegetation may be especially problematic for native freshwater ecosystems. For example, large stands of bamboo now dominate the banks of sections of ʻOheʻo Gulch, Pīpīwai Stream and Puaʻaluʻu Stream (Green et al. 2015). Hardy (1979) found very few native aquatic insects in sections of Puaʻaluʻu Stream that were dominated by bamboo, and felt that the presence of the bamboo was responsible for this pattern, although mechanisms were not elaborated.

Condition and trend determination for aquatic invertebrates

Although aquatic insect communities appear to be in relatively good condition, at least as of the early 1990's, freshwater molluscs (hīhīwai) and shrimp (ʻōpae) appear to have undergone dramatic declines since earlier surveys in the 1970's and/or 1990's. Hence, we conclude that the current condition of aquatic invertebrates, overall, warrants significant concern. Mollusc and shrimp populations declined in prior decades, but since both are now almost entirely absent from HALE streams, the more recent trend in the condition appears unchanging. Confidence in this assessment is medium, because while all data were collected on site, usually with standardized sampling methods, large temporal gaps in data exist, potentially obscuring cyclical or other patterns.

Overall assessment

Integration of the two indicators/measures of freshwater ecosystems yields an overall current condition that warrants moderate concern, with an unchanging trend. Habitat within HALE streams appears generally unimpaired. Although two dams exist in the Palikea Stream system, one on lower Palikea Stream and one in ʻOheʻo Gulch, water flows over or around these dams at both high and low stream levels, and neither dam presents obstacles to migration of native aquatic animals. No water is diverted from these or any other streams in HALE, with the exception of Puaʻaluʻu Stream. The native fish fauna is well represented in HALE streams, and fish communities appear to be in good condition and fairly stable. Native aquatic insect communities also appear to be in relatively good condition, although information here is much more limited. In contrast, native freshwater molluscs and crustaceans have almost completely disappeared in recent decades. Invasive species such as the Tahitian prawn may be responsible, in part, for these declines, but precise causes have not been directly investigated and therefore remain unclear. Declines in larval source pools outside the park, for example, could also play a role.

Non-native ungulates, including cattle, pigs and goats, impact HALE streams by disturbing stream banks, leading to increased erosion and sedimentation, and by increasing nitrogen levels via their waste. Much of the lower portion of Kīpahulu Valley was open cattle pasture into the early 1990's, with cows able to access the stream directly in locations (Gagné 1980, Hodges 1994). Gagné (1980) stated that "The continued disruptive activity of feral pigs in upper areas and the encroachment of cattle on riparian environments in lower areas are serious problems." Today, most of the lower valley area is fenced, with cattle and most pigs and goats removed. However, feral cattle, pigs and goats still infiltrate the fences in the vicinity of the Palikea Stream system (J. Mallinson, T. Bailey pers. comm. 2017), although typically no longer enter the streams directly. The lower portion of the Kaʻapahu parcel remains completely unfenced, so streams in this area, including ʻAlelele, are accessible to feral goats and pigs.

Alien riparian vegetation below roughly 1,600 ft (488 m) elevation may also affect in-stream habitat quality in some areas, particularly those characterized by alien plant monocultures; however, in most areas vegetated banks, and the exclusion of feral ungulates, likely result in desirable erosion control.

An additional potential threat to habitat quality concerns the use of ʻOheʻo Gulch by park visitors. The Kīpahulu District receives 300,000-600,000 visitors each year (Figure 2.1.3-2), with numerous people swimming in the lower pools of ʻOheʻo Gulch at times (Figure 4.1.3-2). Although human activity around HALE streams is concentrated in the lower reaches of ‘Ohe‘o, and would not appear to strongly affect many of the higher stream areas, the lower terminal reaches are critically important for maintaining the necessary link between freshwater and marine environments for the diadromous native macrofaunal species. Effects of human use in this habitat are unknown. (return to Condition Summary).

Level of confidence

Confidence in the current condition of freshwater ecosystems is medium. Although recent NPS I&M Program monitoring data are rigorous, they are so far available for a relatively short time period. No information exists for large periods of time over the past decades, since surveys of HALE streams began. In addition, some biotic components of freshwater ecosystems, like insects and many invasive species, are only minimally studied or characterized.

Information gaps and research recommendations

Continuous data on trends in aquatic biota are lacking for long time periods, but should be rectified with continued monitoring as part of the NPS I&M program. Information on aquatic insects, especially at higher elevations, including the three montane lakes in the park, is very limited. Also limited is information on status, trends, and ecological effects of invasive species in and around freshwater ecosystems.

Sources of Expertise

David Raikow and Anne Farahi, NPS I&M Program, PACN, provided information and reviewed this chapter.

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4.2.3. Forest Ecosystems

Condition Summary

Overall resource condition warrants moderate concern with an unchanging or stable trend; confidence in condition determination is medium. Rationale is discussed in the "*Overall assessment*" section and following.

↔

Relevance and Context

Forest ecosystems of the Hawaiian Islands occur across a broad range of elevation and rainfall gradients, from lowland dry forests which occur below about 4,900 ft (1,500 m) elevation and receive less than 50 inches (ca. 1,300 mm) of annual rainfall, up to montane wet forests which occur at elevations of about 3,280-6,230 ft (1,000–1,900 m) and receive rainfall amounts exceeding 98 inches (2,500 mm) and reaching as high as 400+ inches (>10,000 mm) on Kauaʻi and Maui (Giambelluca et al. 2013, Gustafson et al. 2014). Within HALE, the majority of forest ecosystems occur in Kīpahulu Valley, within Hāna Rainforest on the northeast rift above Kīpahulu, the Kaʻāpahu section of the park, and the Manawainui Planeze, with additional smaller patches of forest and woodland in Kaupō, upper Nuʻu, and the West Slope frontcountry (Figure 4.2.3-1).

These ecosystems provide important habitat for many rare and endangered plants and animals, and collectively harbor the majority of the biodiversity in the park (Loope and Medeiros 1995). A total of 25 endangered plant taxa are known to occur in HALE forests, and another eight occurred historically but have not been recently observed (chapter 4.2.10). In addition, at least three threatened or endangered bird species live in HALE forests (chapter 4.2.10 and below). Excluding the coastal zone below 1,000 ft (305 m) elevation, which is addressed in the coastal ecosystems section (Chapter 4.2.1), forests and woodlands of all types collectively cover 10,993 ac (4,449 ha), or 32.6% of the park. Important differences exist in HALE between high elevation forests (above approximately 4,000 ft (1,220 m)) and low elevation forests (1,000-4,000 ft (305-1,220 m)), in terms of their structure and composition, as well as the magnitude and nature of stressors affecting them. We therefore assess these forest zones separately (Figure 4.2.3-1).

Wet forests, also commonly called rain forests, occupy the eastern half of the park, from the central Manawainui Planeze through Kīpahulu Valley and up onto the northeast outer rift in Hāna Rainforest. In upper elevation forest zones (>4,000 ft), *Metrosideros polymorpha* (ʻōhiʻa lehua, or ʻōhiʻa) is the dominant wet forest tree. Although these forests, like nearly all Hawaiian ecosystems, are threatened by non-native species invasion, they currently remain relatively intact refugia of native biodiversity. They shelter large remaining populations of six species of native forest birds (Judge et al. 2013), and as in wet forests elsewhere (Gagné 1979), can be expected to also support high diversities of native invertebrates. Wet forests on Haleakalā volcano grade into mesic forests on their peripheries, and as precipitation decreases on leeward exposures these communities transition to dry forests and woodlands interspersed within shrublands. The most important of the mesic and dry forest communities in HALE exist on the eastern wall and eastern to central floor of Kaupō Gap

(Figure 4.2.3-1). Although small in area, these forest communities harbor unique species and vegetation associations not found elsewhere in the park. They also represent important remnants of leeward Haleakalā vegetation communities that have been almost completely lost from the southern flank of the volcano (Medeiros et al. 1986). Additional mesic to dry forest and woodland patches occur in the upper portion of the Nuʻu parcel, and in the west slope frontcountry area of the park (Figure 4.2.3-1). The latter area includes stands of introduced *Eucalyptus* trees.

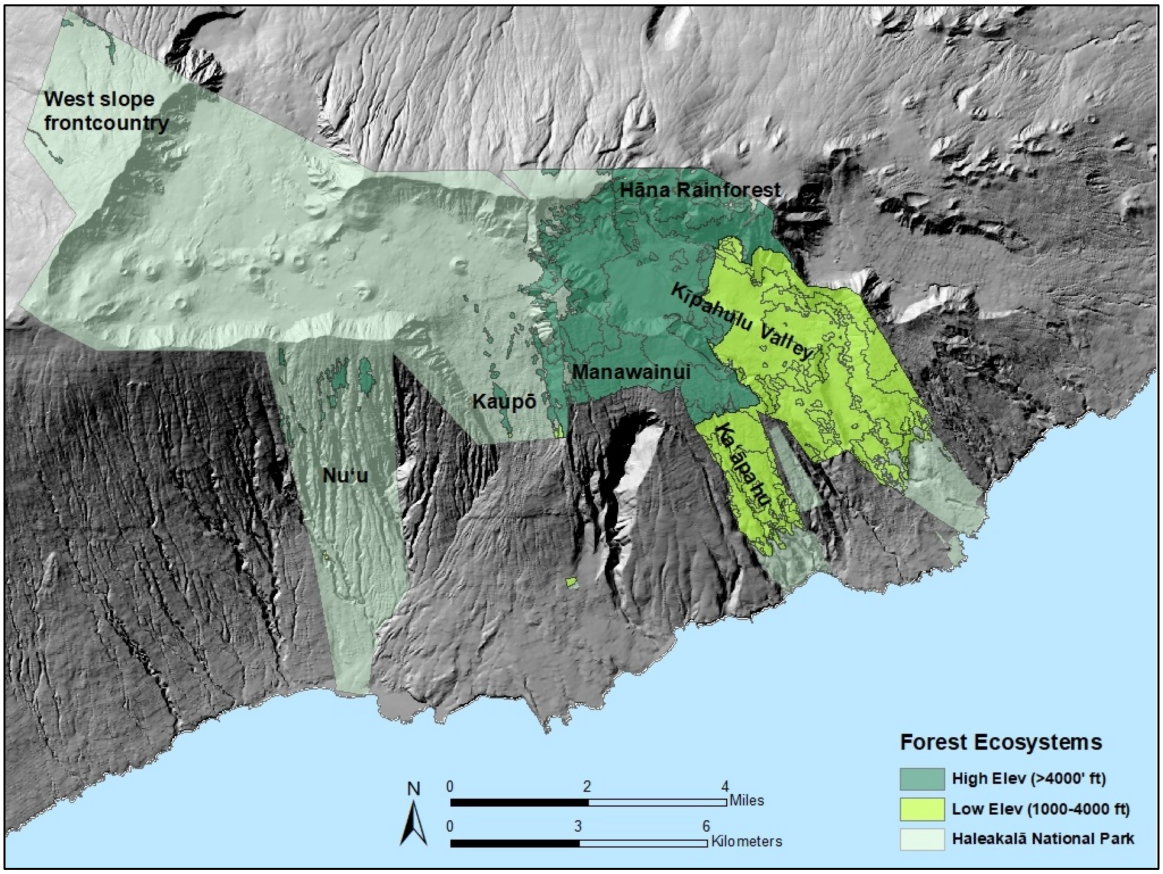


Figure 4.2.3-2. Forest and woodland ecosystems within HALE, differentiated by low elevation (1,000-4,000 ft) and high elevation (>4,000 ft) zones. Source: Green et al. (2015).

In low elevation forest zones (1,000-4,000 ft (305-1,220 m)) of Kīpahulu Valley and Kaʻāpahu, *Acacia koa* (koa) trees are common, either intermixed with ʻōhiʻa or overtopping them, sometimes interlocking and forming a distinct upper canopy layer (Medeiros et al. 1998). A variety of native tree species 10-20 m tall occur in the subcanopy (Gagné and Cuddihy 1990). Hapuʻu tree ferns (*Cibotium* spp.) also contribute to the understory canopy, and a diversity of shade-tolerant shrubs, ferns, mosses, and herbs comprise the lowest layer. Compared to high elevation forests, non-native plant species are a much more serious threat to native forest ecosystems below 4,000 ft (1,220 m). A greater presence of feral pigs in low elevation forests exacerbates this non-native plant invasion, because they are important agents for weed dispersal and create disturbance that further promotes weed establishment (Loope et al. 1992, Loh and Tunnison 1999, Nogueira et al. 2007, Nogueira-

Filho et al. 2009, Cole and Litton 2014, Wehr et al. 2018). The conversion from native to alien plant composition in low elevation forests has cascading impacts on native invertebrate communities (Villegas 1976, Gagné 1980, Howarth 1990), and may also negatively affect native bird populations through alterations to their food resource base (Foster 2005). Finally, the presence of mosquitos and their vectoring of avian diseases in low elevation forests adds another layer of stressors on native forest birds in these ecosystems (Aruch et al. 2007).

Because of the divergent attributes of high and low elevation forests summarized above, we assess the current condition and trend of forest ecosystems using the following five indicators: high elevation vegetation, low elevation vegetation, high elevation bird communities, low elevation bird communities, and invertebrate communities. Although there is no abrupt boundary between high and low elevation forests, we use the 4,000 ft (1,220 m) elevation contour to separate vegetation communities, which serves as a good approximate line of division as explained above. For forest bird communities, the most current information for HALE comes from the Pacific Island landbird monitoring annual report (Judge et al. 2013), which divided analyses into two elevation strata (above and below 4,200 ft (1,280 m)) based on three factors: the presence and absence of koa, the upper range of mosquitoes and transmission of avian malaria, and the approximate upper limit of substantial weed encroachment in Kīpahulu Valley. We therefore use this very similar elevation cutoff to assess high and low elevation forest bird communities. Although invertebrate communities also likely differ in condition between high and low elevation forest ecosystems, the available information is so limited that we make only a single combined assessment for this group of organisms.

Indicators

- High elevation vegetation (>4,000 ft)
- Low elevation vegetation (1,000-4,000 ft)
- High elevation bird communities (>4,200 ft)
- Low elevation bird communities (1,000-4,200 ft)
- Invertebrate communities

Data and Methods

A review of available data and information was used to assess conditions, and if possible, trends, in the indicators listed above. Some additional analyses were performed on a subset of NPS I&M geospatial and vegetation classification and monitoring data collected by HALE and PACN staff specific to forest ecosystems (Green et al. 2015, PACN and Ainsworth 2016). Basic summary analyses were also performed on Kaʻāpahu bird data reported in Natividad Bailey (2007). Elevational and ecosystem-based summaries of invertebrate communities were extracted from data presented in Beardsley (1980), Gagné (1980), and Gon and Pinter (1980).

Sources of information

Monitoring, outplanting, and propagule collection data for common, rare, and endangered native taxa have been recorded by HALE Vegetation Management staff for the entire park from 1993 through present (NPS 2017a).

Control data for invasive, non-native plants have been collected by HALE Vegetation Management staff for the entire park from 1986 through present (NPS 2017b).

Data collected by Anderson et al. (1992), and a comparable subset collected by Meston and Mallinson (2011) have been used to analyze the status of native and invasive plants in the wet forests of Kīpahulu Valley. Data collected along two weed transects in Kīpahulu Valley by NPS staff (NPS 2017d) were also used to assess the status of invasive plants in wet forests.

NPS I&M vegetation classification data were also collected by HALE and PACN staff in 220 circular plots (400 m^2) and observation points in 2011, 64 of which were in forest ecosystems (Green et al. 2015). NPS I&M also monitored 30 wet forest plots (20 x 50 m) in 2012-2013 as part of the Focal Terrestrial Plant Community monitoring system (Ainsworth et al. 2011, PACN and Ainsworth 2016), and 16 1,000 m long belt transects, which consisted of contiguous 5 x 20 m plots, in wet forest in 2012 as part of the Established Invasive Plant Species monitoring system (Gross et al. 2017).

Feral animal control data for priority non-native mammals have been provided by the Haleakalā Feral Animal Removal and Management staff from 1986 through January 18, 2017 (NPS 2017c).

Forest bird survey data collected using point-transect distance sampling in Kīpahulu Valley, Hāna Rainforest (Northeast Rift), and Manawainui from March-July, 2012, were used to estimate abundance of native and non-native species in the survey area (Judge et al. 2013). Prior surveys comprising the Hawaiʻi Forest Bird Survey were provided in Camp et al. (2009) and Gorreson et al. (2009). Surveys for the Maui parrotbill and assessment of forest habitat suitability for the parrotbill were conducted in the Manawainui area above 5,200 ft elevation (Stein 2007), and forest bird survey data were collected in the Kaʻāpahu area in 2002 and 2005 (Natividad Bailey 2007). Surveys of forest birds in lower Kīpahulu Valley below 2,000 ft were made in the summer of 1980 (Stemmerman, 1980). Bird survey data were collected on transects in forest and other habitats in the crater district of the park from 1976-1978 (Conant and Stemmerman 1979), and forest bird surveys were conducted on five transects in the Manawainui area in 1976 (Stemmerman 1976). Notes on forest birds were made during an expedition to Kīpahulu Valley in 1967, including the rediscovery of the Maui nukupuʻu and observations of the Maui parrotbill (Warner 1968b, Banko 1968).

Data on invertebrate diversity and distributions were obtained from surveys of arthropods in Kīpahulu Valley in 1967 (Carson 1968, Wilson 1968), insects in forested sections of the Crater District in 1975-77 (Beardsley 1980), arthropods and snails in Manawainui in 1976 (Villegas 1976, Meyer 1976), and arthropods and snails in lower Kīpahulu Valley in 1980 (Gagné 1980, Gon and Pinter 1980, Severns 1980).

Reference condition

Ideal reference conditions for the chosen indicators of forest ecosystems would include fully intact native vegetation, uninvaded by non-native weeds, at both high and low elevations. Similarly, bird communities would be diverse and abundant in both zones, with no transmission of diseases by introduced mosquitoes and no extinctions of forest bird species. Invertebrate communities would be highly diverse in species and function, and uninvaded by non-native invertebrate species and unaffected by non-native vertebrates. However, since the arrival of humans and the subsequent introduction of non-native plants and animals to the Hawaiian Islands, no forest community type currently exists in an entirely pristine state. Similarly, a multitude of bird species went extinct in the Hawaiian Islands following the arrival of humans, making the original condition of the Hawaiian avifauna difficult to determine (Olson and James 1982). Invertebrate communities have also undoubtedly experienced many extinctions from a combination of anthropogenic forces (Zimmerman 1948), and have been relatively poorly studied in montane wet forests.

Nevertheless, examples of relatively high-quality Hawaiian forests, in which native plant species richness and abundance exceeds that of non-natives, can still be found, particularly at higher elevations and remote locations throughout the islands. A desirable reference condition for HALE's high and low elevation forest vegetation is therefore that the richness (diversity) and cover of native species exceeds that of nonnative species. The first large-scale quantitative assessment of the status of Hawaiian forest birds was the Hawai`i Forest Bird Survey (HFBS), a series of extensive surveys throughout the main islands conducted from 1976 to 1983 (Scott et al. 1986). Several species became extinct several decades before those surveys began, but the HFBS represents the best modern-day reference condition against which to measure the current status of six remaining forest bird species. As mentioned above, original conditions of native invertebrate communities are difficult to estimate, but a desirable reference condition is that diversity of native species exceeds that of non-native species.

Condition and Trend

High elevation vegetation (>4,000 ft)

Green et al. (2015) have classified 16 vegetation community types as forest or woodland above 4,000 ft (1,220 m) elevation. Forest cover in this zone totals 5,885 ac (2,381 ha). Of these 16 forest communities, only one, *Eucalyptus* spp. - Mixed Semi-natural Forest, has non-native plants as the most abundant species within that forest type. Although this artificial community accounts for only 0.4 percent of upper elevation forest ecosystems, it is notable in that planted groves occur in otherwise native-dominated shrubland in the west slope frontcountry near the park entrance between 6,700 and 6,800 ft (2,040-2,070 m) elevation, and separately at 8,500 ft (2,590 m). The physiognomy of the non-native planted *Eucalyptus* and *Pinus* spp., and the ability of *Eucalyptus globulus*, *Pinus radiata, Pinus patula,* and other *Pinus* species to spread into and overtop lower stature native vegetation, suggest that this semi-natural community type will continue to encroach upon and transform native communities without active and sustained management (Medeiros et al. 1998).

The majority of high elevation forest communities are dominated by native species, and occur in the eastern portions of the park, including the Manawainui Planeze, Kīpahulu Valley, Hāna Rainforest on the northeast outer rift, and the eastern portions of Kaupō Gap (Figure 4.2.3-1). The dominant species is *M. polymorpha*, which is the main canopy tree in the eight largest forest community types (Table 4.2.3-1), and in 13 of 16 forest communities, at higher elevations (Green et al. 2015). Although not a comprehensive list of all plant taxa found in the high elevation forests of HALE, Green et al. (2015) provide a broad sampling of species through a series of 23 vegetation inventory and mapping plots located mainly in Kīpahulu Valley (n=15), but also in Kaupō (n=6), Manawainui (n=2), and Hāna Rainforest (n=3). Species identity and cover was recorded in multiple vegetation layers, or strata, including herbs, shrubs and trees (after Tart et al. 2005). A total of 94 native and 12 non-native plant species were recorded in these plots, with a mean species richness of 21.9 ± 2.1 natives and 1.2 ± 0.4 non-natives per plot (Figure 4.2.3-2). Mean native plant cover in these plots is more than 25 times greater than non-native plant cover in all strata (Figure 4.2.3-3). Among all strata, highest mean non-native plant cover is only 6.9% ± 3.9%, whereas mean non-native tree and shrub cover are both under 1% (Figure 4.2.3-3). These data suggest that upper elevation forests are relatively intact and generally free from shrubs and trees that can modify the subcanopy and canopy layers of these native-dominated forests.

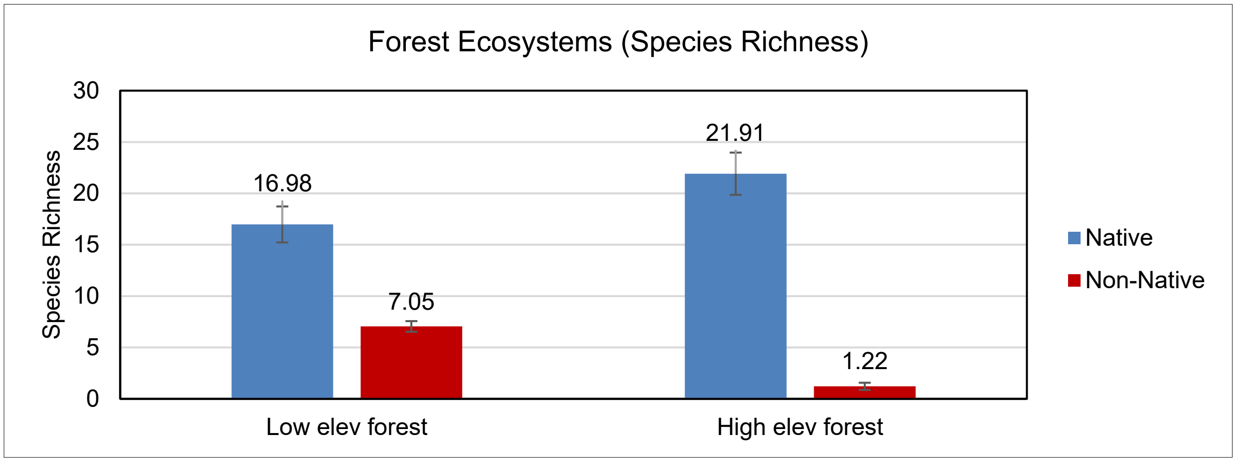


Figure 4.2.3-2. Mean (±SE) native and non-native plant species richness recorded within 400 m^2 inventory and mapping plots in low elevation forests (1,000 - 4,000 ft, n = 41) and high elevation forests (> 4,000 ft, n = 23). Source: Green et al. (2015).

Table 4.2.3-1. Ten largest high elevation forest communities by area within HALE. Areas calculated for land above 4,000 ft (1,220 m) only; however, some communities extend below the high elevation forest zone, as indicated by elevation ranges in the table. (Total park area = 33,718.8 ac). Source: Green et al. (2015).

Forest Community Classification	**Area (acres)**	**% of HALE**	**Elevation (ft)**	
			Lower	**Upper**
Metrosideros polymorpha / Vaccinium calycinum - (Broussaisia arguta, Coprosma foliosa) / Athyrium microphyllum Montane Wet Forest	1814.5	5.4	4495	6430
Metrosideros polymorpha / (Dicranopteris linearis - Sticherus owhyhensis - Diplopterygium pinnatum) Montane Wet Woodland	1118.4	3.3	3379	6562
Metrosideros polymorpha - Cheirodendron trigynum / (Cibotium spp.) Montane Wet Forest	657.5	2.0	3018	6890
Metrosideros polymorpha / Vaccinium calycinum - Rubus hawaiensis / (Coprosma foliosa - Leptecophylla tameiameiae) Montane Wet Forest	613.3	1.8	4921	8202
Metrosideros polymorpha / Dryopteris wallichiana Mesic Forest	452.2	1.3	6562	7054
Metrosideros polymorpha / Leptecophylla tameiameiae - Dodonaea viscosa Montane Woodland	282.4	0.8	4921	6562
Metrosideros polymorpha / Sadleria cyatheoides Forest	232.5	0.7	4921	7382
Metrosideros polymorpha / Rubus hawaiensis Montane Wet Forest	213.7	0.6	6070	6824
Acacia koa - Metrosideros polymorpha Wet Montane Woodland	160.4	0.5	3150	4757
Acacia koa / Coprosma foliosa - Dodonaea viscosa Montane Mesic Woodland	153.7	0.5	4232	6070

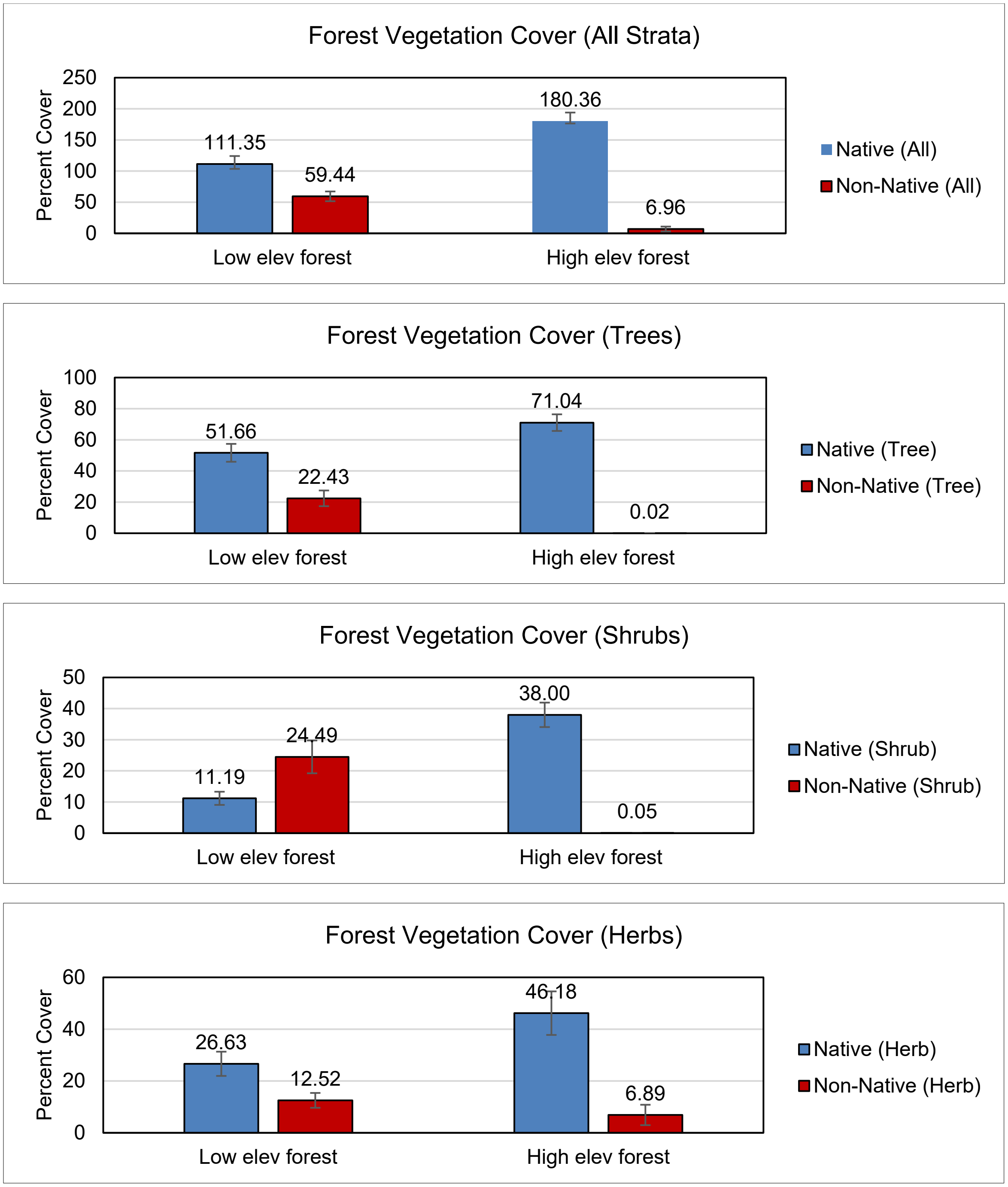


Figure 4.2.3-3. Mean percent cover (±SE) of native and non-native plants in all strata combined, and in individual tree, shrub and herb strata, within 400 m2 inventory and mapping plots in low elevation forests (1,000 - 4,000 ft, n = 41) and high elevation forests (> 4,000 ft, n = 23). Source: Green et al. (2015).

A similar pattern is evident from the first round of monitoring in the NPS I&M Focal Terrestrial Plant Community (FTPC) wet forest plots (PACN and Ainsworth 2016). These recorded a mean species richness of 42.8 ± 2.1 natives and 1.2 ± 0.4 non-natives per plot in high elevation wet forests (Figure 4.2.3-4). The richness values are substantially higher, especially for native species, for these

FTPC plots relative to the inventory and mapping plots because they are 2.5 times as large as the latter plots. Mean native understory (<2 m high) cover in the FTPC plots was 89.0% ± 1.6%, compared to only 0.8% ± 0.4% cover for non-natives species in high elevation wet forests (Figure 4.2.3-5).

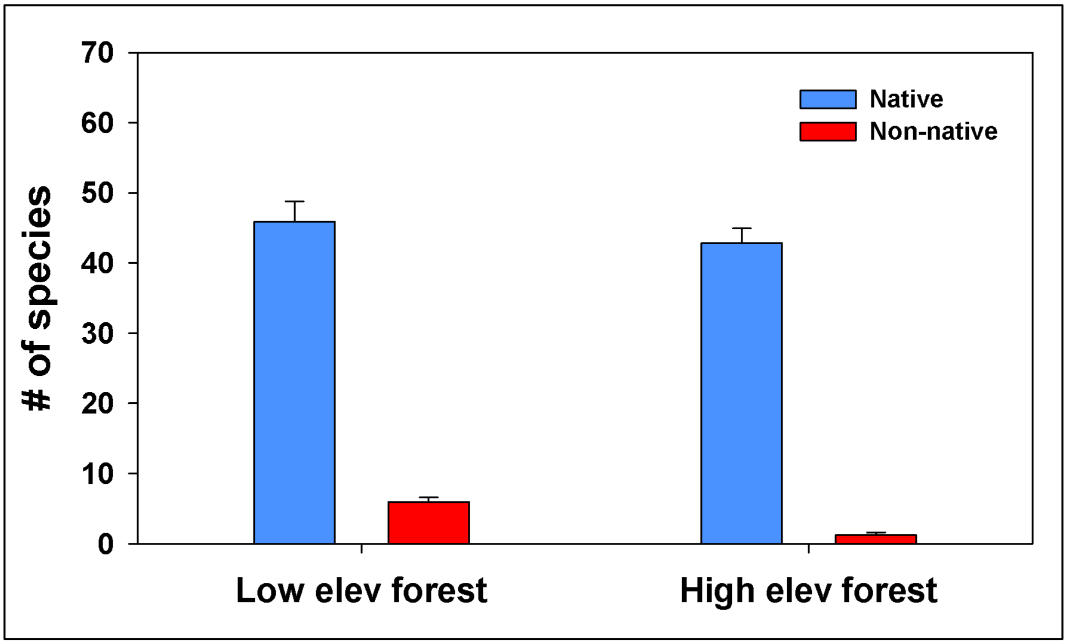


Figure 4.2.3-4. Mean (±SE) native and non-native plant species richness recorded within 1,000 m^2 FTPC monitoring plots in low elevation wet forests (1,000 – 4,000 ft, n = 17) and high elevation wet forests (> 4,000 ft, n = 13). Source: PACN and Ainsworth (2016).

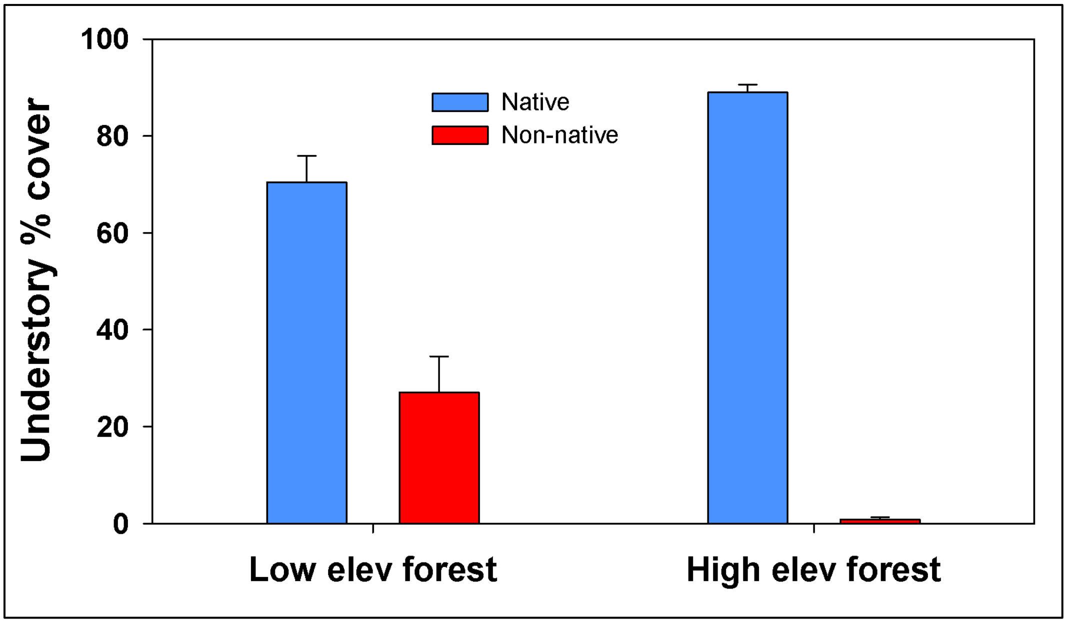


Figure 4.2.3-5. Mean (±SE) percent cover of native and non-native plant species in the understory (<2 m) recorded within 1,000 m^2 FTPC monitoring plots in low elevation wet forests (1,000 – 4,000 ft, n = 17) and high elevation wet forests (> 4,000 ft, n = 13). Source: PACN and Ainsworth (2016).

In addition, the initial round of monitoring of the NPS I&M Established Invasive Plant Species (EIPS) transects also suggest a similar pattern of minimal invasion in high elevation wet forests of HALE (Gross et al. 2017). Sixteen 1,000 m long belt transects (consisting of 50 contiguous 5 x 20 m plots) were established within wet forest in the park, nine of which were located mostly or completely above 4,000 ft (1,220 m) elevation, and baseline monitoring of non-native plants along the transects was completed in 2012 (Gross et al. 2017). On eight of the nine transects, cover of non-native plants was less than 1%, and was between 1 and 5% on the ninth transect. The number of non-native plant species per transect ranged from 0 to 8, and averaged 2.6 species across the nine high-elevation wet forest transects (Gross et al. 2017).

Three of the most damaging invasive plant species in wet forests are *Psidium cattleianum* (strawberry guava), *Clidemia hirta* (Koster's curse), and *Hedychium gardnerianum* (kahili ginger) (Anderson et al. 1992, Loope et al. 1992, Medeiros et al. 1998). Because of their ability to transform native forest ecosystems, these three weeds have been priority management targets for park vegetation management staff in Kīpahulu Valley for the past 25+ years (NPS 2017b). Numbers of individuals controlled for each species, and for invasive plants overall, are shown in Table 4.2.3-2. These data indicate the large difference in control efforts between high elevation and low elevation wet forests, owing to the large difference in degree of invasion between these forest zones. Gross et al. (2017), for example, found each of these three invasive plant species to be only sparingly present in wet forest plots above 4,000 ft (1,220 m) elevation in 2012. HALE Vegetation Management staff, however, have anecdotally noticed increases in the presence of weeds in high elevation forests since 2012 (P. Welton pers. comm. 2018). *Hedychium gardnerianum*, in particular, has been a persistent and increasing concern there. Kahili ginger was first found above 4,000 ft in Kīpahulu Valley in 1997, and much weed management effort in recent years has been focused on the control and containment of this weed in high elevation forests. The goal is to halt its spread into otherwise native-dominated communities, before infestations reach levels and densities that can preclude native species recruitment and community function.

Table 4.2.3-2. Total numbers of high priority invasive weeds controlled from 1992 to 2017 in Wet Forest Ecosystems of HALE including Manawainui, Ka'āpahu, Kīpahulu Valley and Kaumakani. Source: NPS (2017b).

Taxon	Low Elevation (1,000 – 4,000 ft)	High Elevation (>4,000 ft)
Clidemia hirta	198,388	21
Hedychium gardnerianum	~348,000	~6,000
Psidium cattleianum	18,772	14
Miconia calvescens	3,286	Not present
All Non-Native Plant Species	~588,100	~13,800

Although the most invasive non-native plants frequently spread into undisturbed forest, often dispersed by birds (Medeiros 2004), non-native mammals strongly facilitate the conversion of native forest to non-native forest through disturbance, browsing and spread of invasive plant seeds (Yocom

1967, Chynoweth et al. 2013, Wehr et al. 2018). Feral goats (*Capra hircus*) have been present along the headwalls of Kīpahulu Valley for many decades, damaging and destroying mainly montane native shrubland, whereas feral pigs (*Sus scrofa*) are a more recent addition to the valley's fauna (NPS 1999). Upper Kīpahulu Valley had been considered the most intact rain forest in the park, but feral pigs may have entered from the Kalapawili Grassland to the northwest and/or from the middle Kīpahulu area from the east (NPS 1999). To manage and prevent the further degradation of these communities by feral pigs, goats and other herbivorous mammals, the National Park Service has employed a multi-faceted mitigation strategy involving a network of barrier and boundary fences (Figures 2.2.3-1 and 4.2.3-6) to create feral animal control management zones, continued maintenance of the fence network, and removal of feral animals within the management zones (NPS 1999). Feral pig removal for the upper half of Kīpahulu Valley began in 1986 to determine efficacy of removal efforts, and by 1994, all known pigs were removed from the upper management zone and numbers of pigs in the mid-valley zone were down to remnant levels (NPS 1999). However, pigs have continued to infiltrate the fence barriers; while most of these incursions affect lower elevation wet forest areas, pigs have also been seen in a few more intact higher elevation forest locations in recent years (HALE staff, pers. comm. 2017). Axis deer (*Axis axis*) also infrequently infiltrate fences, especially near Manawainui and Kaupō. Non-native rodents such as the Polynesian rat (*Rattus exulans*), and the black or ship rat (*R. rattus*), are notorious seed predators, but can also feed on other plant parts and may also disperse seeds of some native and non-native plants (Sugihara 1997, Shiels and Drake 2011). Rats are known to occur in upper elevation forests at HALE (HALE unpub. data), but their numbers are not monitored and their impacts on native plant regeneration are unquantified.

Using the only data set that includes repeated sampling over time in high elevation forests at HALE, Meston and Mallinson (2011) documented changes in species richness and cover in a series of nine 400 m^2 monitoring plots first established in 1983-86 in Kīpahulu Valley (Anderson et al. 1992; Figure 4.2.3-6). From 1986 to 2011, mean native species richness slightly increased from 19.9 ± 2.0 to 22.3 ± 2.3, while non-native richness decreased from 0.7 ± 0.4 to 0.0 (Figure 4.2.3-7). Native cover in the plots remained relatively stable, increasing slightly from just under 100% to 100% native in the 25-year period (Meston and Mallinson 2011). This result was similar to that found for a high elevation forest on the north slope of Haleakalā within Waikamoi Preserve, immediately adjacent to HALE, where cover of native bryophytes and lichens increased 14 years after feral goat and pig removal (Hughes et al 2014). The HALE monitoring plots also reaffirm the findings of Green et al. (2015) and Gross et al. (2017) that high elevation forests are relatively intact and remain native-dominated communities despite past and present pressure from invasive non-native plants and animals.

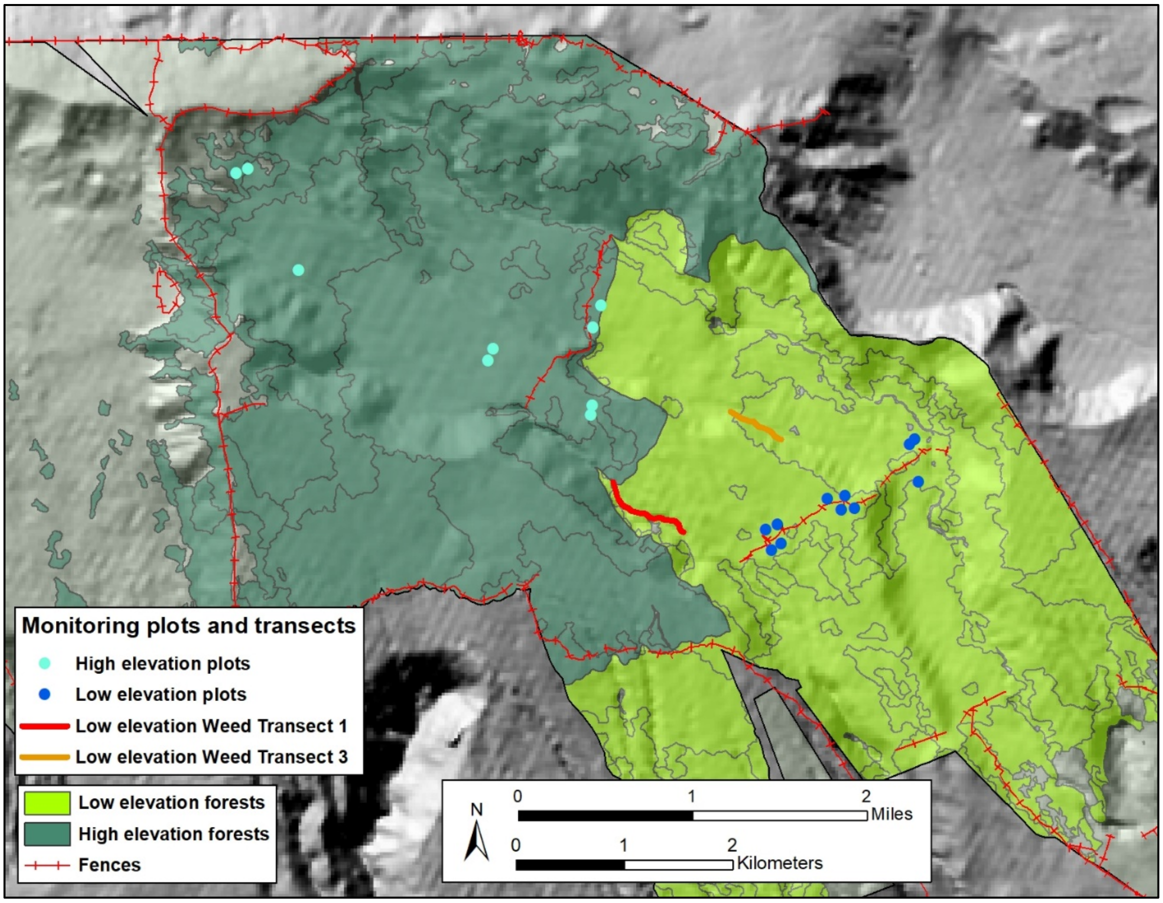


Figure 4.2.3-6. Monitoring plots and transects in Kīpahulu Valley used to document changes in forest species richness and cover, and degree of non-native plant invasion, over time. Source: Anderson et al. (1992), Meston and Mallinson (2011), NPS (2017d).

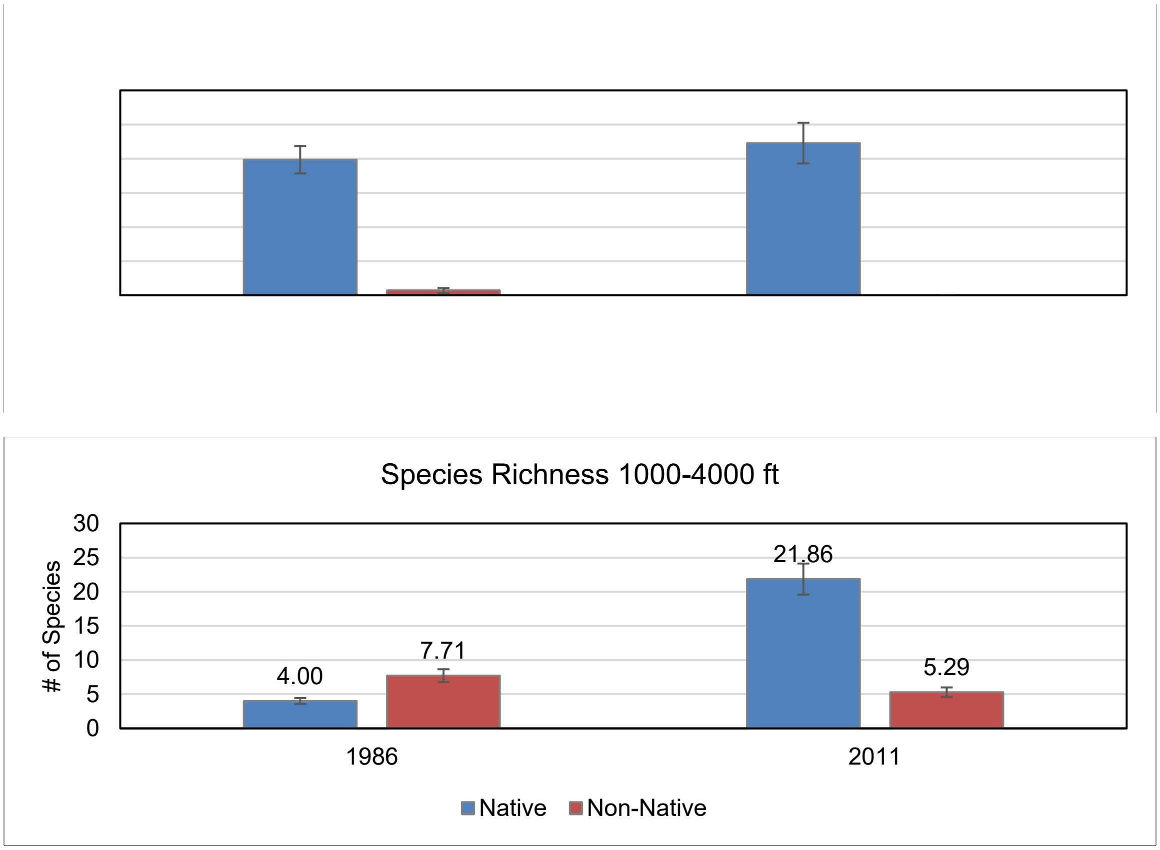


Figure 4.2.3-7. Changes in species richness from 1986 to 2011 in 400 m^2 monitoring plots within low elevation (1,000 – 4,000 ft, n = 11) and high elevation (> 4,000 ft, n = 9) wet forest in Kīpahulu Valley. Source: Anderson et al. (1992) and Mallinson and Meston (2011).

In addition to managing feral animals and controlling weeds, HALE RM staff has attempted to maintain and restore native vegetation diversity through outplanting of predominantly rare plant species, including many federally listed taxa. Cumulative totals of individuals planted in wet forest ecosystems are shown in (Table 4.2.3-3).

Table 4.2.3-3. Numbers of native plant taxa outplanted in lower (1,000 - 4,000 ft) and upper (>4,000 ft) elevation wet forest ecosystems in HALE, as of mid-2017. Source: NPS (2017a).

Taxon	Low Elevation (1000 – 4000 ft)	High Elevation (>4000 ft)	T&E
Anoectochilus sandvicensis	1	0	NA
Argyroxiphium grayanum	0	39	NA
Bidens campylotheca ssp. *pentamera*	19	0	E
Bidens campylotheca ssp. *waihoiensis*	37	16	E
Clermontia arborescens	15	0	NA
Clermontia samuelii ssp. *samuelii*	5	0	E
Coprosma pubens	4	0	NA

Table 4.2.3-3 (continued). Numbers of native plant taxa outplanted in lower (1,000 - 4,000 ft) and upper (>4,000 ft) elevation wet forest ecosystems in HALE, as of mid-2017. Source: NPS (2017a).

Taxon	Low Elevation (1000 – 4000 ft)	High Elevation (>4000 ft)	T&E
Cyanea asplenifolia	6	0	E
Cyanea copelandii ssp. *haleakalaensis*	45	0	E
Cyanea hamatiflora ssp. *hamatiflora*	19	0	E
Cyanea maritae	82	3	E
Cyrtandra ferripilosa	0	40	E
Dubautia plantaginea ssp. *plantaginea*	7	0	NA
Huperzia mannii	82	0	E
Joinvillea ascendens ssp. *ascendens*	62	0	E
Labordia venosa	1	0	NA
Melicope ovalis	59	0	E
Nothocestrum longifolium	3	0	NA
Phyllostegia bracteata	56	22	E
Phyllostegia brevidens	62	30	E
Phyllostegia haliakalae	148	32	E
Phyllostegia macrophylla	67	0	NA
Phytolacca sandwicensis	36	0	NA
Pittosporum glabrum	4	0	NA
Plantago princeps var. *laxiflora*	250	23	E
Schiedea diffusa ssp. *diffusa*	274	0	E
Strongylodon ruber	46	0	NA
Trematolobelia macrostachys	33	0	NA
Wikstroemia oahuensis var. *oahuensis*	3	0	NA
Total	**1,426**	**205**	**–**

All of the patterns and trends presented above are driven largely by dynamics within wet forests, which make up the majority of high elevation forest communities in HALE. However, as noted earlier, the mesic forest and woodland communities in Kaupō represent distinct vegetation community types, and are experiencing different dynamics worth detailing. Green et al. (2015) classified eight forest and woodland community types on the floor and eastern wall of Kaupō Gap (Figure 4.2.3-8, Table 4.2.3-4). Unlike wet forests, two of the three largest community types are dominated by koa instead of ʻōhiʻa (Table 4.2.5-4). These communities also support distinct and in some cases rare understory trees and shrubs, including āulu (*Planchonella sandwicensis*), naio (*Myoporum sandwicense*), hala pepe (*Pleomele auwahiensis*), olomea (*Perrottetia sandwicensis*), olopua (*Nestegis sandwicensis*), alani (*Melicope hawaiensis*), and kōpiko (*Psychotria mauiensis*) (Welton and Haus 2003).

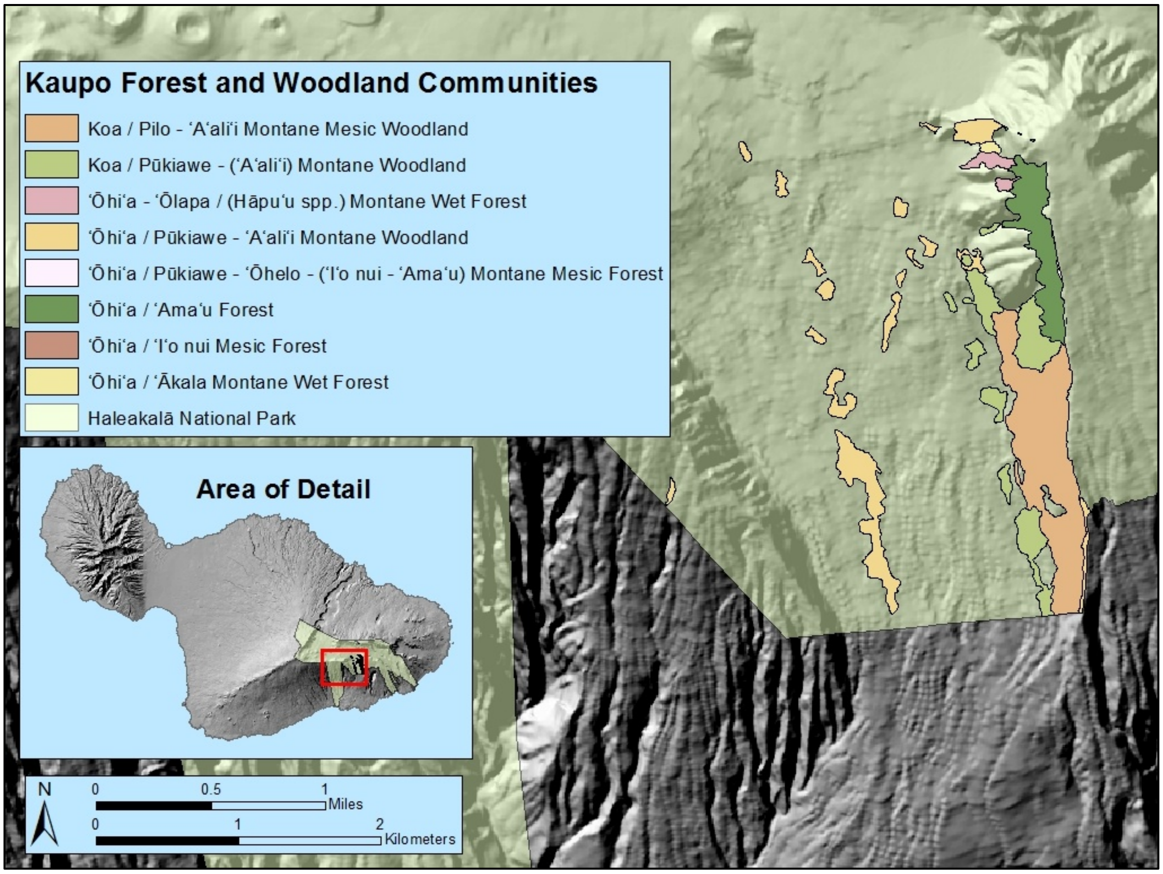


Figure 4.2.3-8. Kaupō forest and woodland communities, within the high elevation (> 4,000 ft) zone. Source: Green et al. (2015).

Table 4.2.3-4. Forest and woodland community types located within HALE Kaupō management units. Source: Green et al. (2015).

Community Classification	**Area (ac)**
Acacia koa / *Coprosma foliosa* - *Dodonaea viscosa* Montane Mesic Woodland	156.7
Metrosideros polymorpha / *Leptecophylla tameiameiae* - *Dodonaea viscosa* Montane Woodland	97.8
Acacia koa / *Leptecophylla tameiameiae* - (*Dodonaea viscosa*) Montane Woodland	78.6
Metrosideros polymorpha / *Sadleria cyatheoides* Forest	60.0
Metrosideros polymorpha - *Cheirodendron trigynum* / (*Cibotium* spp.) Montane Wet Forest	9.4
Metrosideros polymorpha / *Rubus hawaiensis* Montane Wet Forest	2.5
Metrosideros polymorpha / *Leptecophylla tameiameiae* - *Vaccinium reticulatum* - (*Dryopteris wallichiana* - *Sadleria cyatheoides*) Montane Mesic Forest	0.5
Metrosideros polymorpha / *Dryopteris wallichiana* Mesic Forest	0.2

In contrast to patterns in high elevation wet forests, invasion by non-native plants is substantially greater in high elevation Kaupō forests and woodlands. In the six NPS I&M inventory and mapping plots located in these communities, non-native species richness (mean = 3.2 ± 0.8) was approximately a third as high as native species richness (mean = 9.1 ± 1.6), and the cover of non-native plants (mean = 34.8 ± 12.0%) was nearly half as high as cover of native plants (mean = 83.7 ± 12.9%) (Figure 4.2.3-9, Green et al. 2015).

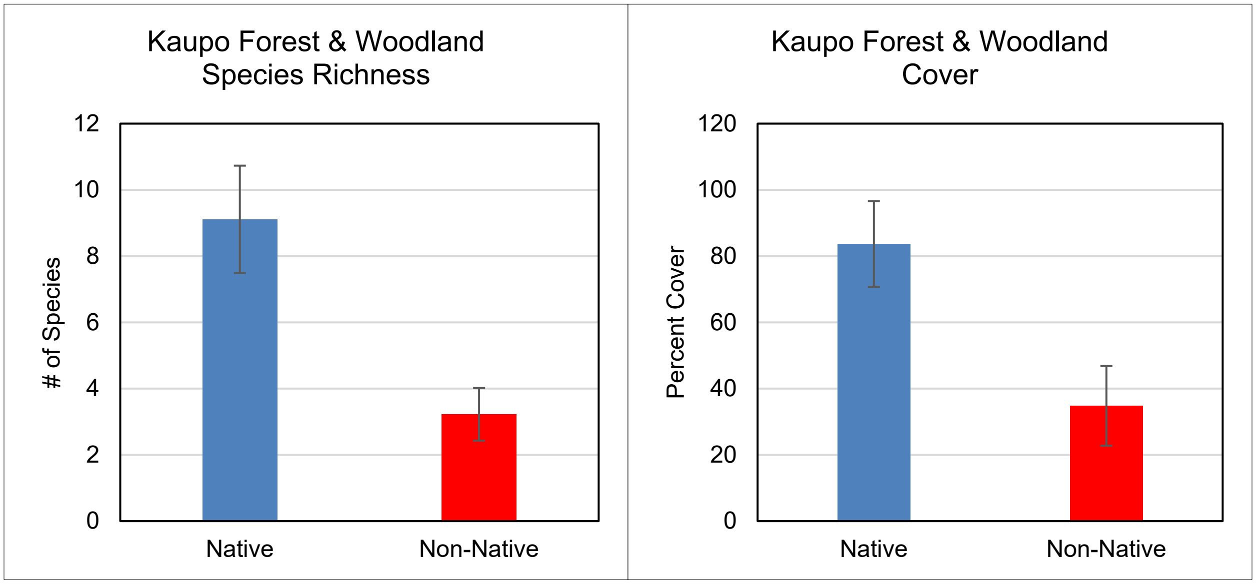


Figure 4.2.3-9. Mean (± SE) native and non-native plant species richness and percent cover recorded within 400 m2 inventory and mapping plots (n = 6) in Kaupō forest and woodland communities. Source: Green et al. (2015).

Important invasive plants in Kaupō forest and woodland ecosystems include *P. cattleianum*, *Schinus terebinthifolius* (Christmasberry), and *Senna septemtrionalis* (smooth senna or arsenic bush) among woody weeds. In addition, the mat-forming *Cenchrus clandestinus* (kikuyu grass), which inhibits recruitment of most native plant species (Medeiros et al. 1986, Welton and Haus 2003), *Melinus minutiflora* (molasses grass), the vine *Sechium edule*, and the thistle *Cirsium vulgare* have been problematic (NPS 2017b). All of these species, and others, have been actively managed by HALE RM staff, in some cases since the early 1980's (Table 4.2.3-5).

Table 4.2.3-5. Total numbers of individuals controlled, or acres treated, for high priority invasive weeds in Kaupō forest and woodland ecosystems of HALE. Source: NPS (2017b).

Taxon	Numbers or area
Cenchrus clandestinus (1994-2013)	32.4 acres (13.1 ha)
Cirsium vulgare (1983-2015)	10,310
Melinus minutiflora (1993-2009)	29 acres (11.7 ha)
Psidium cattleianum (1990-2017)	79
Ricinus communis (1982 – 2016)	50,363
Schinus terebinthifolius (1990 – 2014)	645
Sechium edule (1995-2017)	8,234

Table 4.2.3-5 (continued). Total numbers of individuals controlled, or acres treated, for high priority invasive weeds in Kaupō forest and woodland ecosystems of HALE. Source: NPS (2017b).

Taxon	Numbers or area
Senna septemtrionalis (1995 – 2017)	32,429
All Non-Native Plant Species (1982 – 2017)	105,568 individuals

Although natural recruitment of koa and ʻaʻaliʻi (*Dodonaea viscosa*) seedlings was observed following fencing and removal of feral ungulates from the area in the late 1980's to early 1990's, additional control of kikuyu grass, which also rebounded following fencing and ungulate removal, was implemented to increase survival of outplants and encourage the further recruitment of native plants (Welton and Haus 2003). Outplanting and sowing of seeds of a wide range of both common and rare native species has been conducted (Table 4.2.3-6), often in combination with control of kikuyu grass (Welton and Haus 2003). Active management of this type appears to be necessary to restore the mesic forest and woodland communities of Kaupō, although initial management efforts may lead to at least some spontaneous regeneration and succession: establishment of koa and ʻaʻaliʻi saplings in the restoration sites was found to promote the recruitment of other native species such as māmane (*Sophora chrysophylla*), pilo (*Coprosma foliosa*) and māmaki (*Pipturus albidus*), presumably owing to amelioration of microclimatic conditions (Welton and Haus 2003). More recently, native ferns and sedges have been observed to recruit naturally in the dark understory (P. Welton pers. comm. 2018). While still suffering from weed problems, the extent of mesic forest in Kaupō has expanded considerably since management efforts began there in the 1980's (P. Welton, pers. comm.).

Table 4.2.3-6. Numbers of native plant taxa outplanted and seeds sown in Kaupō forest ecosystems of HALE. Source: NPS (2017a).

Taxon	Plants	Seed	T&E
Acacia koa	825	9,560	NA
Alyxia stellata	229	–	NA
Bidens micanthra ssp.*kalealaha*	14	–	E
Cocculus orbiculatus	14	–	NA
Coprosma ernodeoides	357	60	NA
Coprosma foliosa	1,042	249	NA
Dodonaea viscosa	1,287	3,240	NA
Cyperus hillebrandii ssp. *hillebrandii*	8	–	NA
Eragrostis grandis	30	–	NA
Kadua centranthoides	4	–	NA
Lysimachia remyi	17	–	NA
Myoporum sandwicense	85	–	NA
Myrsine lessertiana	954	300	NA

Table 4.2.3-6 (continued). Numbers of native plant taxa outplanted and seeds sown in Kaupō forest ecosystems of HALE. Source: NPS (2017a).

Taxon	Plants	Seed	T&E
Nothocestrum latifolium	20	–	E
Osteomeles anthyllidifolia	777	1,090	NA
Peperomia cookiana	23	–	NA
Perrottetia sandwicensis	3	–	NA
Phyllostegia haliakalae	67	–	E
Pipturus albidus	2	–	NA
Pisonia brunoniana	399	–	NA
Pleomele auwahahiensis	4	–	NA
Planchonella sandwicensis	60	–	NA
Sophora chrysophylla	1,035	240	NA
Viola chamissoniana ssp.*tracheliifolia*	111	–	NA
Total	**7,367**	**14,739**	–

Condition summary of high elevation forest vegetation

In summary, the condition of high elevation forest vegetation is judged to generally be good. Native plant richness and cover in high elevation wet forests have increased slightly following the control of feral animals, and native vegetation there remains relatively intact. In mesic forests of Kaupō, the situation is less favorable, with a substantially higher degree of non-native plant invasion; notwithstanding, the condition of Kaupō mesic forests and woodlands has improved over the past half-century due to management, such as the removal of much of the kikuyu grass that had suppressed native plant regeneration. In all areas, the continued exclusion and/or management of feral ungulates is important to maintaining native vegetation diversity and inhibiting invasive plant spread. Persistent problematic weeds include *H. gardnerianum* in wet forests, and *P. cattleianum* and *S. terebinthefolius* in Kaupō mesic forests and woodlands. Diligent control of incipient invasions by these weeds will be needed to maintain current conditions or ideally to further recovery. Confidence in this assessment is medium, owing to a shortage of repeated monitoring data, limited information on the population status of most rare plant species, and the potentially increasing presence of pigs in some high elevation wet forest areas. The trend in this condition currently appears relatively stable, but the encroachment of weeds from lower elevations, and the renewed incursion of invasive mammals, is a persistent concern.

Low elevation vegetation (1,000 -4,000 ft)

Green et al. (2015) have classified 23 vegetation community types as forest or woodland between 1,000 and 4,000 ft (305 – 1,219 m) elevation. Forest cover in this zone totals 5,108 ac (2,067 ha), with *A. koa* as the dominant or co-dominant canopy tree in five of the six largest community types by area (Table 4.2.3-7). Overall, communities classified as having a native plant as the most abundant or dominant species account for 62% of low elevation forest cover (Figure 4.2.3-10). In comparison, >99% of high elevation forest cover is comprised of communities classified as having a native plant as the most abundant or dominant species. Furthermore, 16 of the 23 low elevation forest communities have non-native plants as the most abundant, or among the most abundant, species within that forest type. Although the latter represent only about a third of the low elevation forest area, these non-native communities reflect the more heavily invaded state of the lower elevation portions of the park compared to forests above 4,000 ft.

Table 4.2.3-7. Ten largest low elevation forest community types by area within HALE. Areas calculated for land between 1,000 and 4,000 ft (305-1,220 m) only; however, some communities extend above the low elevation forest zone, as indicated by elevation ranges in the table. (Total park area = 33,718.8 ha). Source: Green et al. (2015).

Forest Community Classification	**Area (acres)**	**% of HALE**	**Elevation (ft)**	
			Lower	**Upper**
Acacia koa - Metrosideros polymorpha Wet Montane Woodland	1204.4	3.6	3150	4757
Acacia koa - Psidium cattleianum Semi-natural Forest	1024.5	3.0	1198	3806
Acacia koa - Metrosideros polymorpha / Dicranopteris linearis Lowland Wet Forest	673.8	2.0	2657	3084
Metrosideros polymorpha / (Dicranopteris linearis - Sticherus owhyhensis - Diplopterygium pinnatum) Montane Wet Woodland	586.4	1.7	3379	6562
Acacia koa - Cheirodendron trigynum - (Antidesma platyphyllum, Syzygium sandwicense) / (Clidemia hirta) Lowland Wet Forest	385.7	1.1	2365	3186
Acacia koa - Metrosideros polymorpha Lowland Mesic Forest	318.7	0.9	984	2953
Psidium cattleianum Lowland Wet Semi-natural Forest	299.2	0.9	656	3609
Metrosideros polymorpha / Dicranopteris linearis Lowland Mesic Woodland	237.0	0.7	2575	3281
Syzygium jambos Lowland Mesic Semi-natural Woodland	95.4	0.3	1148	2296
Psidium guajava / Disturbed Understory Semi-natural Forest	84.0	0.2	1312	1804

Although not a comprehensive list of all plant taxa found in the low elevation forests of Haleakalā, Green et al. (2015) provide a broad sampling of species through a series of 41 vegetation inventory and mapping plots. Species identity and cover was recorded in multiple vegetation layers, or strata, including herbs, shrubs and trees (after Tart et al. 2005). A total of 105 native and 49 non-native plant species were recorded in these inventory plots, with a mean species richness of 17.0 ± 1.8 natives and 7.0 ± 0.5 non-natives per plot (Figure 4.2.3-2). Total native plant cover in these plots is greater than non-native plant cover in all strata with one exception: mean non-native shrub cover was 24.6% ± 5.3%, more than double the 11.2% ± 2.1% mean native species cover (Figure 4.2.3-3).

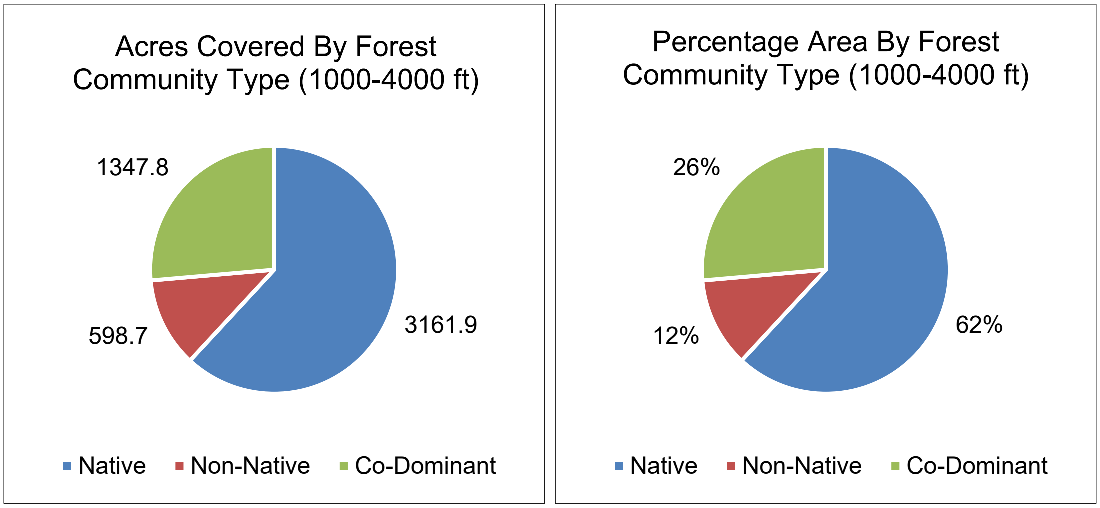


Figure 4.2.3-10. Area and percentage of low elevation forest (1,000 – 4,000 ft) with a native plant, non-native plant, or co-dominant native and non-native plant species as the dominant species. Source: Green et al. (2015).

The above NPS I&M inventory plots, combined with the two additional, aforementioned I&M monitoring efforts (FTPC wet forest plots and EIPS belt transects), provide consistent data indicating that non-native plants currently comprise a substantially larger fraction of both the overall diversity and cover of low elevation forests, as compared to high elevation forests at HALE. For example, mean non-native species values are consistently higher in low elevation forests than in high elevation forests in Figures 4.2.3-2 through 4.2.3-5. The low elevation FTPC plots recorded lower non-native percent cover than the I&M inventory plots (compare Figures 4.2.3-3 and 4.2.3-5), but this is likely because many of the FTPC plots were actually situated in more intact, higher elevation portions of the low elevation forest zone: 11 of the 17 low elevation forest plots were located between 3,000 and 4,000 ft elevation, and only one was below 2,000 ft elevation (PACN and Ainsworth 2016). Regardless, both sets of plots clearly indicate a higher degree of non-native plant invasion in low elevation forest zones compared to high elevation forest zones. Similarly, the average of the midpoint percent cover values of invasive plants along seven EIPS belt transects in low elevation wet forests of Kīpahulu Valley and Ka‘āpahu was just under 26%, whereas the equivalent average for high elevation forest transects was <1% (Gross et al. 2017). The number of non-native plant species per transect ranged from 5 to 20 on the seven low elevation EIPS transects, with an overall average of 12.3 non-native plant species per transect; as noted in the high elevation forest section above, the average was 2.6 non-native plant species per transect in high elevation forests.

The causes and dynamics leading to the present highly-invaded situation of low elevation forests appear to be complex. Forest communities below 2,000 ft (610 m) elevation in Kīpahulu Valley had already been highly altered and invaded by alien plants and feral pigs by the 1970’s, and probably much earlier (Canfield and Stemmerman 1980). Koa forests above 2,000 ft (610 m), however, appeared to rapidly transition from a near pristine to increasingly invaded state between the 1940’s and 1970 (Yoshinaga 1980a,b). In the 1980’s, forests between 2,200 and 3,000 ft (670-915 m), and even up to 4,100 ft (1,250 m) in places, were found to have high cover of alien plants, particularly in terms of ground cover, along with frequent pig activity (Anderson et al. 1992). This is reflected in 11 low elevation forest plots first monitored in 1983-1986 in the Dogleg area of Kīpahulu Valley (Figure 4.2.3-6), which measured ~62% cover of non-native plants at the time (Figure 4.2.3-11, 1986 data). In particular, there existed a very high percentage of ground covered by *Paspalum conjugatum* (Hilo grass), a Neotropical grass first documented in the Park as a component of the koa forest understory in 1919 (Loope et al. 1992; Figure 4.2.3-12, 1986 data). High levels of pig rooting and disturbance, evident from the considerable amount of bare ground existing in 1986 (Figure 4.2.3-11), favored the establishment and maintenance of dense mats of this grass, which inhibited native understory plant regeneration.

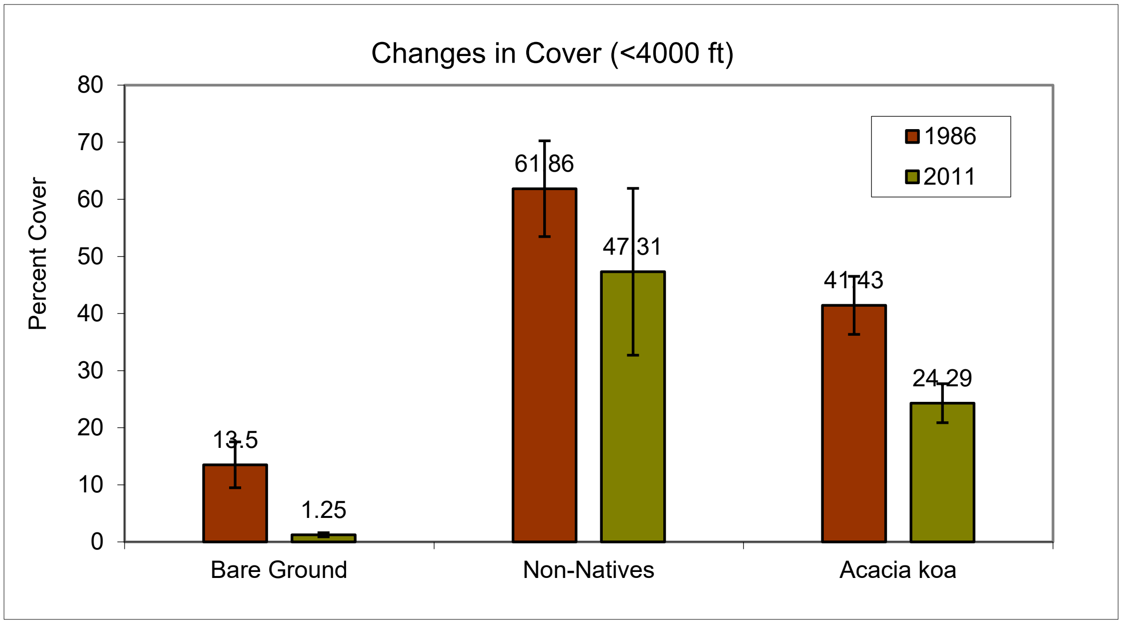
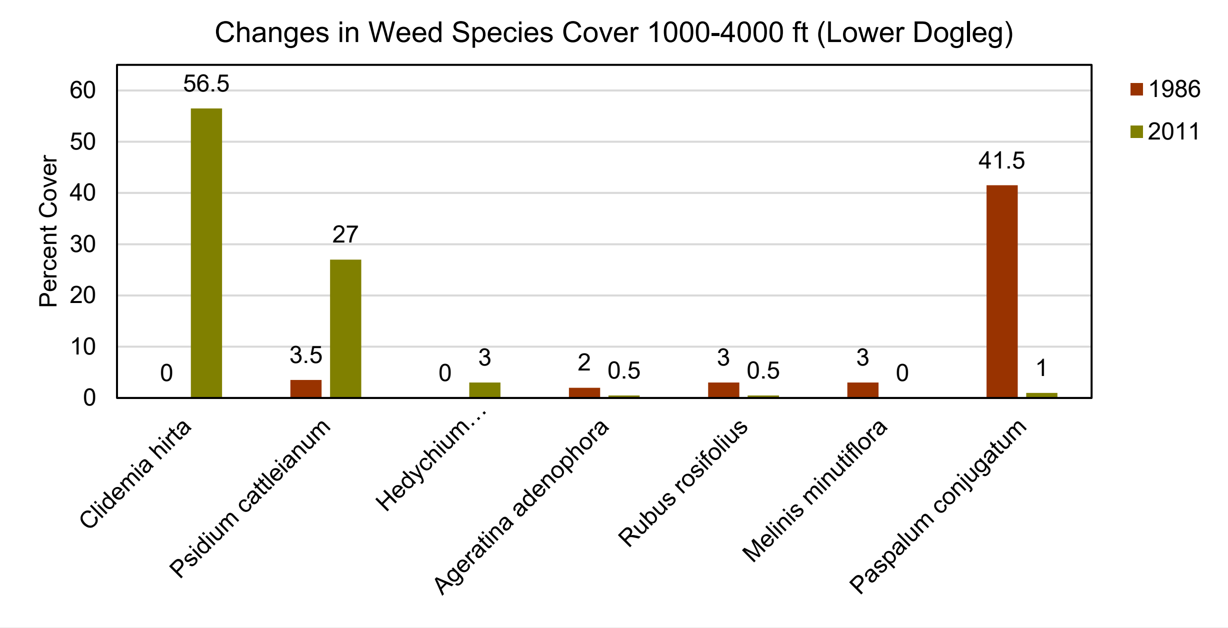


Figure 4.2.3-11. Changes in bare ground, all non-native plants and *Acacia koa* in 400 m^2 monitoring plots within wet forest along Dogleg fence (<4000 ft), Kīpahulu Valley (1986 to 2011). Source: Anderson et al. (1992) and Meston and Mallinson (2011).

Pigs were excluded from most of the lower forest areas of Kīpahulu shortly thereafter, by 1988 (Anderson and Welton 1993), and a variety of vegetation changes ensued over the following two decades. From 1986 to 2011, when the 11 vegetation plots were revisited, native species richness increased from 4.0 ± 0.4 to 21.9 ± 2.3, while mean non-native richness decreased from 7.7 ± 0.9 to 5.3 ± 0.7 (Figure 4.2.3-7). Percent cover of bare ground, non-native plants, and *Acacia koa* also declined during this period (Figure 4.2.3-11). The decrease in bare ground and non-native plant cover, especially the dramatic decline in *P. conjugatum* ground cover (Figure 4.2.3-12), may be attributed to the suppression of feral pig numbers, and these changes also seem likely to be responsible for the dramatic recovery of native understory plant diversity (e.g., Cole and Litton 2014).



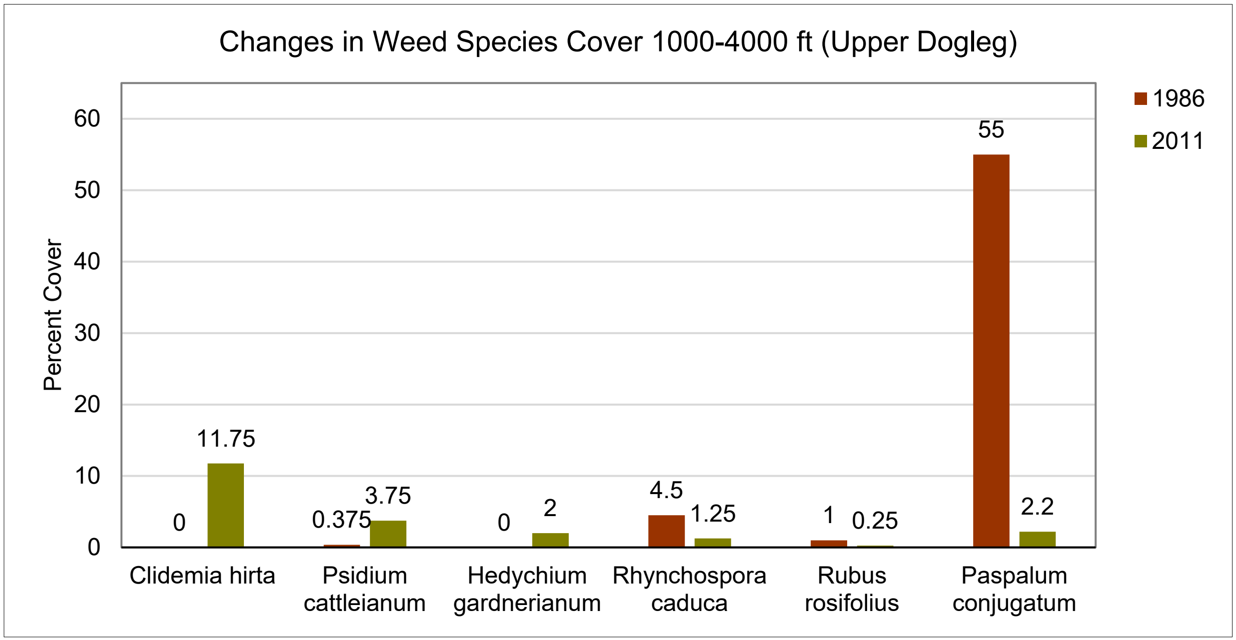
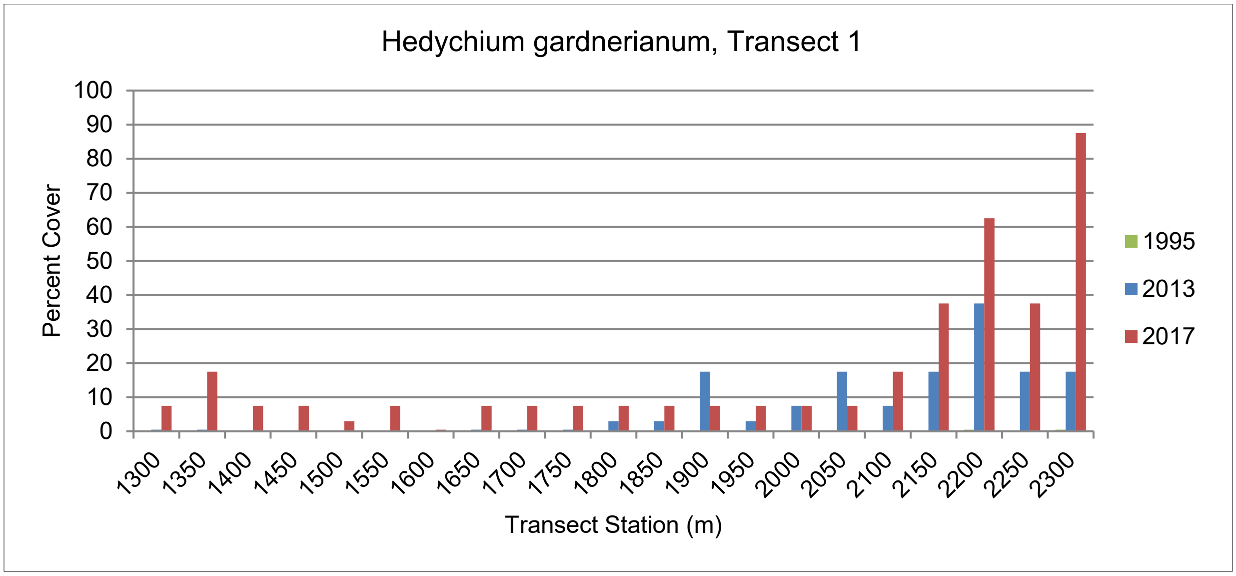


Figure 4.2.3-12. Changes in cover of prominent weed species in eleven 400 m^2 monitoring plots within wet forest along Dogleg fence (<4,000 ft), Kīpahulu Valley (1983-86 to 2011). Source: Anderson et al. (1992) and Meston and Mallinson (2011).

For example, monitoring of some of the plots in the Dogleg area of Kīpahulu in 1993 revealed that a wide diversity of native seedlings were establishing on fallen nurse logs of koa and hapu‘u (*Cibotium* spp.), where these logs acted as favorable islands for germination within areas of thick cover of *P. conjugatum* (Anderson and Welton 1993). Furthermore, a variety of native understory shrub species recovered in size and abundance within five years of pig suppression (Anderson and Welton 1993). The decrease in koa cover between 1986 and 2011, meanwhile, could be attributed to two defoliation events in 2003 and 2008 following outbreaks of the native koa moth (*Scotorythra paludicola*), and the canopy may not yet have fully recovered (Haines et al. 2009, Meston and Mallinson 2011). The decline in koa might also be the result of an increase in other invasive weeds which may have prevented native seedling regeneration: although ground cover of *P. conjugatum* declined sharply since fencing and pig removal, this has been accompanied by a large increase in cover of the invasive understory shrub *C. hirta* and tree *P. cattleianum* between 1986 and 2011 (Figure 4.2.3-12). *Clidemia hirta*, a highly invasive Neotropical shrub introduced to the Hawaiian Islands in 1941, was first collected in Kīpahulu Valley in 1988 and has since come to dominate large portions of the low elevation forest understory. *Psidium cattleianum*, a small tree native to Brazil and elsewhere in the Neotropics, is capable of forming dense, monotypic stands even under closed canopy forests and is estimated to occur at densities of 2,000 trees per hectare in heavily invaded areas (Anderson et al. 1992). Both species have fleshy fruits and may be dispersed by non-native frugivorous birds and feral pigs, contributing to their spread into both disturbed and intact native forests (Diong 1982, Medeiros 2004).

Another bird-dispersed weed which was found to increase in cover from 1986 to 2011 is *H. gardnerianum* (Figure 4.2.3-12). Although it's spread into these plots was modest compared to *C. hirta* and *P. cattleianum*, *H. gardnerianum* often has a highly aggregated distribution, at least initially. While it produces dispersible fruits, this ginger also spreads vegetatively, forming large clumps that almost completely exclude all other understory vegetation (Loope et al. 1992). Additional insight into the invasion pattern of *H. gardnerianum* and *C. hirta* in low elevation forest is provided by data from a pair of weed transects in Kīpahulu Valley that have also been monitored on several occasions in recent decades. The two transects, designated 1 and 3, are located on the upper shelf of the valley between 3,500 and 4,000 ft, and on the lower shelf between 2,900 and 3,200 ft, respectively (Figure 4.2.3-6). *Hedychium gardnerianum* was nearly absent from transects 1 and 3 in 1995 and 1998, but had made substantial inroads on both transects by 2013 (NPS 2017d, Figures 4.2.3-13, 4.2.3-14). By 2017, it was present throughout the length of the transect 1, and was locally highly abundant, exceeding 80% cover at the lowest station (NPS 2017d, Figures 4.2.3-13). Similarly, *C. hirta* was absent or nearly so from both transects in 1995 and 1998, but has begun to invade both transects since 2013, especially transect 3 (Figures 4.2.3-13, 4.2.3-14).

For both weeds, the invasion process therefore appears to have occurred most rapidly some time after the mid to late 1990's. The timing of these invasions may have therefore coincided to some degree with the two large koa defoliation events, in 2003 and 2008, caused by *S. paludicola* caterpillars. This has led some to hypothesize that the resulting increased light levels passing through the koa canopy may have been responsible, at least in part, for the rapid spread of these understory weeds, especially *C. hirta* (P. Welton pers. comm. 2018). Aggressive invasion in the understory may in turn now suppress recruitment of new koa seedlings, potentially creating a positive feedback. This type of disturbance, as well as that caused by the increasing presence of feral pigs in a number of low elevation locations in Kīpahulu in recent years (HALE RM staff, pers. comm. 2017), likely assists the invasions of these and other weeds. However, disturbance does not appear to be required for their continued spread, as at least some non-native plants have progressively invaded higher elevation locations even when disturbance is lacking.



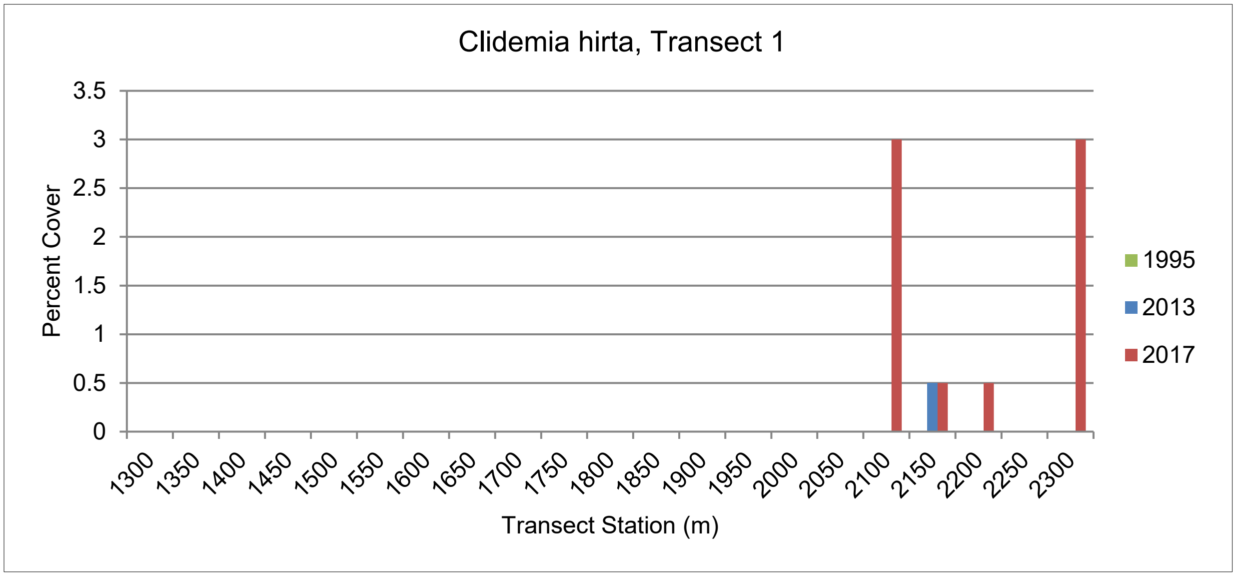
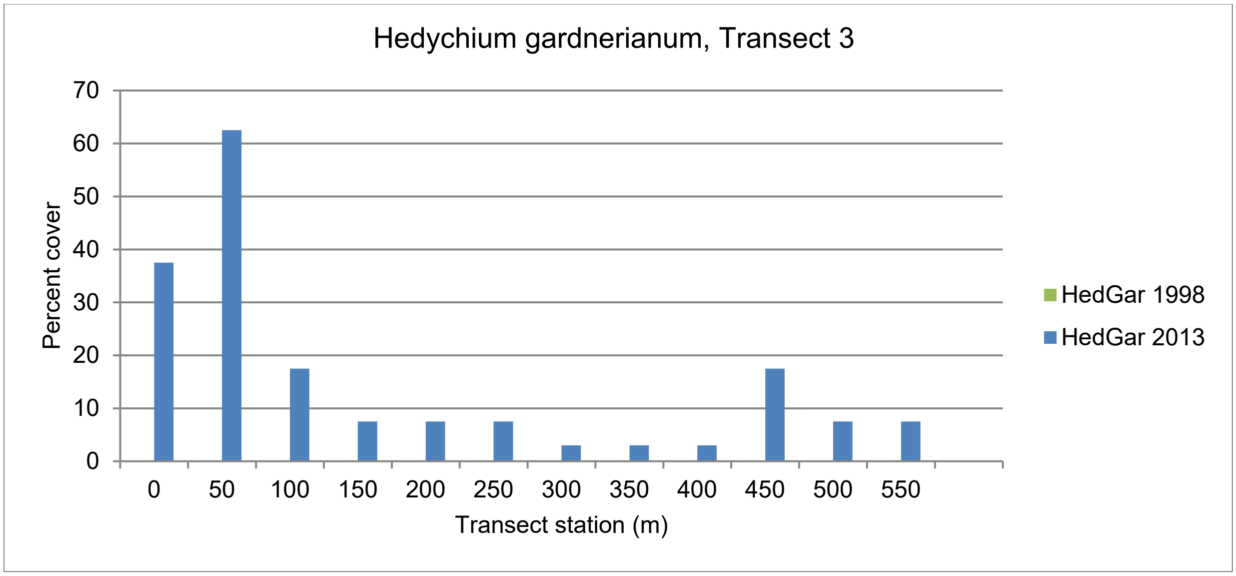


Figure 4.2.3-13. Percent cover of *Hedychium gardnerianum* and *Clidemia hirta* along weed transect 1, located on the upper shelf of Kīpahulu Valley between 4,000 ft (station 1300m) and 3,500 ft (station 2300m) elevation in 1995, 2013 and 2017. Source NPS (2017d).



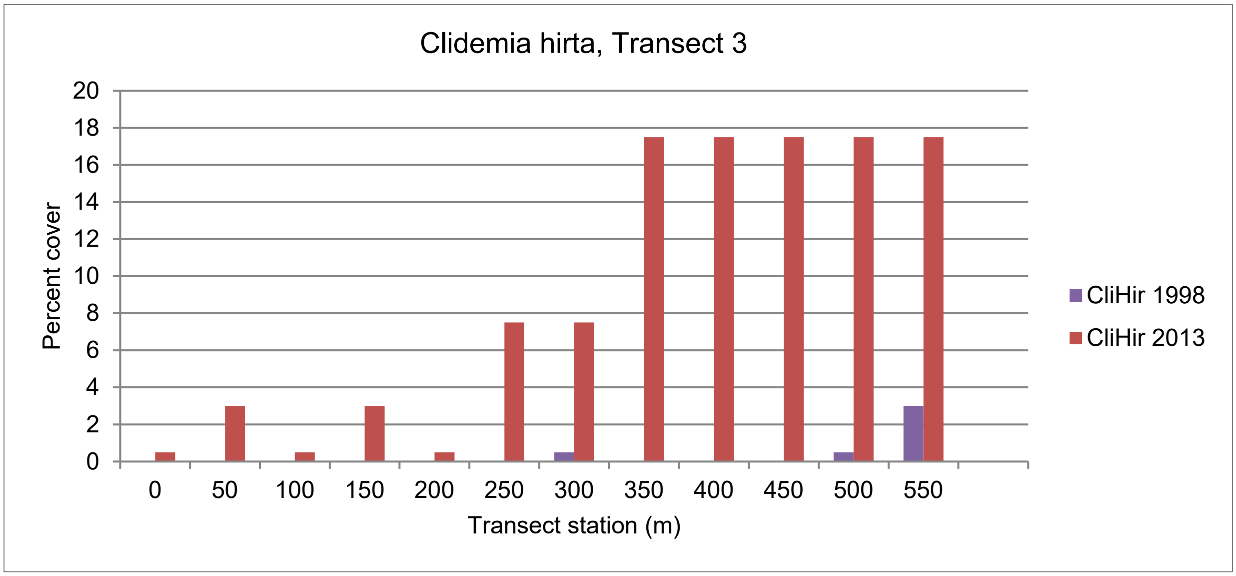


Figure 4.2.3-14. Percent cover of *Hedychium gardnerianum* and *Clidemia hirta* along weed transect 3, located on the lower shelf of Kīpahulu Valley between 3,200 ft (station 0m) and 2,900 ft (station 600m) elevation in 1998 and 2013. Source NPS (2017d).

Management of weeds in low elevation forests of HALE has historically focused most heavily on the three highly invasive species discussed above (*C. hirta, H. gardnerianum, P. cattleianum*, Table 4.2.3-2). Other serious invasive and habitat-modifying rain forest weeds include *Cyathea cooperi* (Australian tree fern), *Rhynchospora caduca*, *Spathodea campanulata* (African tulip tree) and *Tibouchina herbacea* (Loope et al. 1992, Medeiros et al. 1992). *Miconia calvescens* (velvet tree), first detected in Kīpahulu Valley in 1994, can invade relatively undisturbed Hawaiian rain forest and poses a serious threat to ecosystem functioning and diversity of native rainforests (Medeiros et al. 1997). Detections of mature *Miconia* trees in the lower elevation forest of the valley emphasize the importance of early detection of and rapid response to this habitat-modifying tree. Ongoing control efforts in collaboration with the Maui Invasive Species Committee and University of Hawaii utilizing aerial control measures hope to prevent its spread into more pristine higher elevation forests (Leary et al. 2014). Numbers of *Miconia* individuals controlled within park boundaries to date are shown in Table 4.2.3-2.

While invasive plant control efforts in low elevation forests were extensive in the 1990's and early 2000's, this effort has diminished in recent years as it became apparent that physical and chemical control methods were insufficiently effective to reverse or even halt the spread of firmly established weeds. As noted above, efforts have increasingly been directed at stopping incipient invasions of species such as *H. gardnerianum* into more pristine high elevation forests. Little management attention, for example, has been focused on low elevation forests of Kaʻāpahu, which are unfenced and harbor many feral goats and pigs (HALE RM staff pers. comm. 2017), and which are highly invaded by non-native plants. Welton and Haus (2008) recorded 123 non-native plant species, representing 42% of all vascular plants, in a survey of Kaʻāpahu forests.

Despite the serious problem with non-native plant invasion, HALE RM staff has attempted to maintain populations of a number of rare plant species in low elevation forests, including many federally listed taxa, through outplanting. Cumulative totals of individuals planted in low elevation wet forest ecosystems are shown in Table 4.2.3-3.

Condition summary for low elevation forest vegetation

In summary, the condition of low elevation forest vegetation is judged to warrant significant concern. These forests have been the most heavily modified, and over one third of original native cover has been replaced by vegetation dominated by non-native species. Although fencing and control of feral pigs appears to have initially resulted in a dramatic recovery of native understory diversity, subsequent invasion of habitat-modifying weeds like *C. hirta*, *H. gardnerianum*, and *P. cattleianum* is now threatening to reverse these gains. Cover of the dominant native canopy tree *A. koa* has also shown signs of decline, although it is as yet unclear whether this is a transient change. Because of the increasing diversity, distribution, and cover of highly invasive non-native plants detected across low elevation forest monitoring plots and transects, and because Vegetation Management staff have anecdotally observed a severe and rapid degradation of low elevation plant communities in recent years (P. Welton pers. comm. 2018), we assign a declining trend in the condition. Confidence in this assessment is medium. Additional monitoring data would help clarify spatial patterns in non-native plant invasion, and there is limited information on the population status of most rare native plant species.

High elevation bird communities (>4,200 ft)

Six species of native birds are known to reside in high elevation forests of HALE today: ʻapapane (*Himatione sanguinea*), Maui ʻamakihi (*Chlorodrepanis virens wilsoni*), Maui ʻalauahio (Maui creeper, *Paroreomyza montana newtoni*), ʻiʻiwi (*Drepanis coccinea*), ʻākohekohe (crested honeycreeper, *Palmeria dolei*), and kiwikiu (Maui parrotbill, *Pseudonestor xanthophrys*) (Judge et al. 2013). Of these six, ʻiʻiwi is currently listed as federally threatened, while ʻākohekohe and kiwikiu are federally endangered. Four non-native birds also now inhabit high elevation forests: red-billed leiothrix (*Leiothrix lutea*), Japanese white-eye (*Zosterops japonicus*), and Japanese bush-warbler

(*Cettia diphone*), and hwamei (*Garrulax canorus*) (Judge et al. 2013). In recent history, three additional native forest birds have been detected in HALE, as summarized below.

During an expedition to Kīpahulu Valley in 1967, Banko (1968) reported seeing Maui nukupuʻu (*Hemignathus lucidus affinis*) on three occasions on 24 August. The birds were seen singly at elevations of 5,909 ft, 5,860 ft, and 5,709 ft (1,801, 1,786, and 1,740 m) elevation. Banko thought all three were different individuals based on their plumage color. Banko (1968) also reported that a nukupuʻu was seen in the same area on 11 September by a Mr. Morrison, a national park ranger. A single Maui nukupuʻu was detected in Kīpahulu in the early 1980s at 5,280 ft (1,600 m) elevation (Scott et al. 1986), and the last known observations of the Maui nukupuʻu were in Hanawī Natural Area Reserve on the northern slope of Haleakalā in 1994-1996 (Reynolds and Snetsinger 2001). There have been no recent reports in HALE, although rare bird searches have also not be conducted recently. The species may no longer occur in the park and may be extinct (Camp et al. 2009, Gorresen et al. 2009, NPS 2012).

Maui ʻākepa (*Loxops coccineus ochraceus*) was reported in Kīpahulu Valley in 1995 based on an audio detection, but because of possible confusion with calls of other bird species this report is not considered confirmed. The poʻouli was discovered in 1973 in Hanawī Natural Area Reserve on the northern slope of Haleakalā (Casey and Jacobi 1974), but fossil evidence indicates they once also inhabited drier forests at lower elevation on the leeward slopes of Haleakalā (James and Olson 1991). Surveys from 1997-2000 located only three birds (Reynolds and Snetsinger 2001). The last known individual died in captivity during efforts to establish a captive breeding program (VanderWerf et al. 2006). As with Maui nukupuʻu, there have been no recent reports of Maui ʻākepa or poʻouli in HALE, but rare bird searches have not be recently conducted. Both species may no longer occur in the park and may be extinct (Camp et al. 2009, Gorresen et al. 2009, NPS 2012).

One of the earliest reported bird surveys in HALE wet forests was the 1967 Kīpahulu Valley Expedition mentioned above. In addition to the three sightings of Maui nukupuʻu, and a single sighting of kiwikiu (Banko 1968), Warner (1968b) reported that all observations of rare and endangered native forest birds were made between 5,000 and 7,350 ft elevation (1,540-2,260 m; Warner's elevation estimates, not exact conversions between English and metric systems). He also made estimates of relative abundance of species on a scale from 0-10 (least-most), as follows: ʻapapane 10, ʻiʻiwi 7, ʻamakihi 7, Maui ʻalauahio 3, the non-native red-billed leiothrix (*Leiothrix lutea*) 3, the non-native Japanese white-eye (*Zosterops japonicus*) 0.5, and ʻākohekohe 0.01. Warner (1968b) described the range of the ʻākohekohe to encompass the upper forest zone from approximately 6,000 ft (1,840 m) to the upper forest edge.

The following decade, Stemmerman (1976) conducted forest bird surveys on five transects in the Manawainui area in 1976. Six species of native forest birds and seven species of non-native forest birds, in addition to the nēnē (Hawaiian goose, *Branta sandivicensis*) and several species of non-native gamebirds, were detected. Native species included the pueo (Hawaiian short-eared owl, *Asio flammeus sandwichensis*) and five species of Hawaiian honeycreepers: Maui ʻamakihi, Maui

ʻalauahio, ʻapapane, ʻiʻiwi, and ʻākohekohe. Stemmerman also speculated that other rarer species of Hawaiian honeycreepers might occur in the higher elevation areas of Manawainui adjacent to Kīpahulu Valley, which were not included in the surveys. Most native forest bird species were reported to be more numerous at higher elevations, particularly above 5,000 ft (1,524 m). ʻĀkohekohe were observed only above 5,000 ft elevation, and numbers of ʻiʻiwi were highest at and above 5,300 ft (1,616 m), with only four detections at lower elevations, though the actual elevations of those detections were not reported. Maui ʻalauahio were reported only on the upper portions of several transects, above 4,300 ft (1,311 m), with the highest density between 5,300 ft and 6,900 ft (1,616-2,104 m). In contrast, the Maui ʻamakihi was reported to be more numerous in the open koa parkland from 3,800 ft to 4,300 ft (1,158-1,311 m) than in denser ʻōhiʻa forest above.

Conant and Stemmerman (1979) observed five species of native birds and four species of non-native birds in forested sections of Haleakalā Crater (Palikū and Kaupō) during surveys in 1976-1978. Densities of the species varied greatly (from <1 to 275 birds per 40 ha area), but no means or measures of variance were reported.

The Hawaiʻi Forest Bird Survey (HFBS) was a large-scale quantitative assessment of the status of Hawaiian forest birds, with a series of extensive surveys throughout the main islands conducted from 1976 to 1983 (Scott et al. 1986). This effort was replicated in 1992, and since that time, forest bird surveys were conducted in high elevation forests in HALE until 2008 (HALE unpub. data). Forest bird survey data collected since the HFBS have been analyzed and summarized by Camp et al. (2009) and Gorresen et al. (2009), and provide valuable comparisons among different time periods. Analysis of those data required pooling across geographic areas because of small sample sizes for most native bird species, making it difficult to draw conclusions about the status of forest birds in HALE alone. However, since most surveys took place predominantly in forests above 4,200 ft, (1,280 m) we discuss the findings here in relation to more recent data collected at HALE.

Camp et al. (2009) found that abundance of most native forest birds in east Maui, including HALE, was stable to increasing from 1980 to 2001, but much of the data on which those analyses were based came from outside HALE. The kiwikiu showed a decreasing trend in abundance over this time period, with a possible contraction of the lower range limit from 3,608 ft to 3,936 ft (1100 m to 1200 m), but overall the analysis of trend data was inconclusive. Camp et al. (2009) found that abundance of several native forest birds, including Maui ʻamakihi, Maui ʻalauahio, ʻiʻiwi, and ʻākohekohe, showed increasing population trends from 1980-2001. However, some of those apparently increasing trends may have been caused by differences in the season when surveys were conducted in different time periods, with surveys during the 1980 HFBS conducted later in the year after the peak in vocalizations, resulting in lower initial abundance estimates and potentially erroneous increasing trends.

The most current published information about forest birds in HALE comes from the Pacific Island landbird monitoring annual report (Judge et al. 2013), which includes forest bird survey data collected in Kīpahulu Valley, Hāna Rainforest (referred to as Northeast Rift), and Manawainui from March to July of 2012. The data were collected at 160 points along 11 transects (Figure 4.2.3-15), using point-transect distance sampling, which were used to estimate density and abundance of native and non-native species in the survey area. Points were located 150 m apart, and an eight-minute survey was made at each point. A total of 12 forest bird species was detected in high elevation forests during surveys, including all six extant native species, and six non-native species. The two endangered species, ‘ākohekohe and kiwikiu, were detected at elevations above 4,200 ft but not below 4,200 ft. The richness of native bird species was thus 50% higher (6 vs. 4) in high elevation vs. low elevation forests. Abundance of two native bird species, Maui ‘alauahio and ‘apapane, also was higher in high elevation forest than in low elevation forest (Figure 4.2.3-16). Abundance of one non-native species, the Japanese white-eye, was higher in low elevation forest than in high elevation forest. Population estimates in the park for native species were made and are shown in Table 4.2.3-8, but these include birds in high elevation and low elevation forests combined.

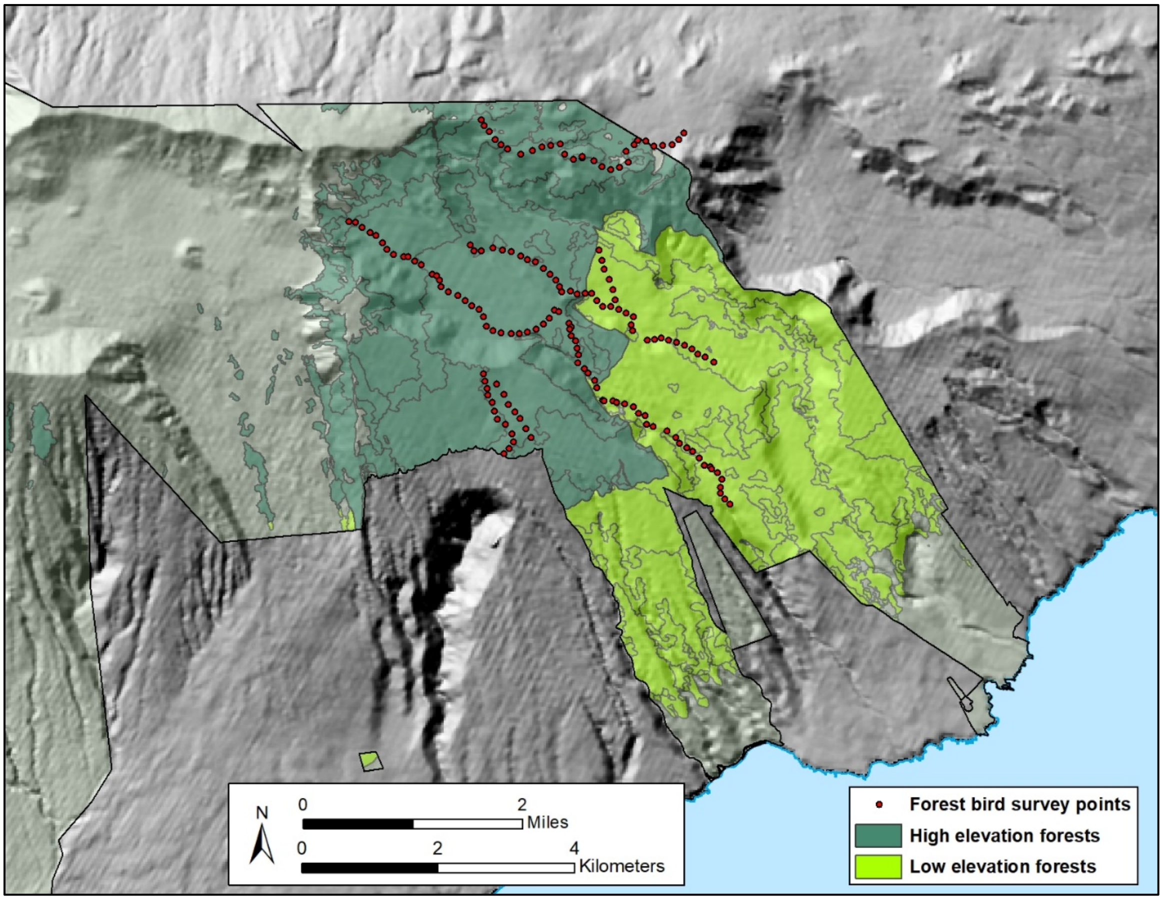


Figure 4.2.3-15. Locations of bird survey point transects used at HALE in the 2012 Pacific Island Landbird Survey. Source: Judge et al. (2013).

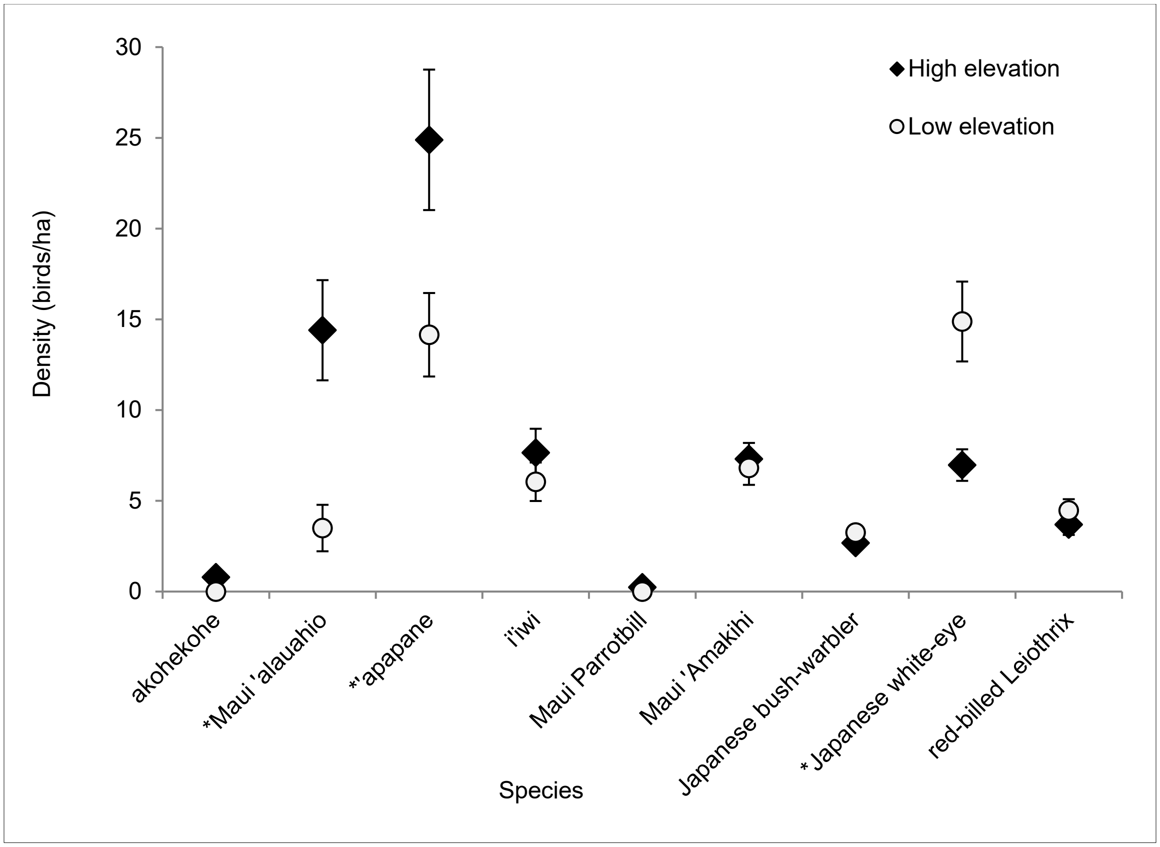


Figure 4.2.3-16. Comparison of abundance of native and non-native bird species in high elevation (>4,200 ft) and low elevation (1,000 - 4,200 ft) forests in HALE. Species marked with an asterisk differed in abundance between the elevation strata. Source: Judge et al. (2013).

Table 4.2.3-8. Estimated total abundance of the six extant native forest bird species in HALE in 2012. Abundance estimates include both high and low elevation forest areas surveyed in the park (1,458 ha). Source: Judge et al. (2013).

Taxon	**Abundance (± SE)**
Maui ʻalauahio *(Paroreomyza montana newtoni)*	31,109 ± 6,004
ʻākohekohe *(Palmeria dolei)*	1,150 ± 389
ʻapapane *(Himatione sanguinea sanguinea)*	64,055 ± 9,531
ʻiʻiwi *(Vestiaria coccinea)*	21,850 ± 3,452
kiwikiu *(Pseudonestor xanthophrys)*	495 ± 261
Maui ʻamakihi *(Chlorodrepanis virens wilsoni)*	22,172 ± 2,370

There are several actual or potential threats to high elevation bird communities at HALE. The clearest current threat is predation by non-native rats, which are known to be significant nest predators of birds in Hawaiʻi (Atkinson 1977, van Riper and Scott 2001, VanderWerf 2009). Rats are prolific in high elevation forests of HALE (HALE unpub. data). The large numbers of non-native forest birds may result in competition with native species for food resources, at least to some degree. Some researchers have posited that competition from the Japanese white-eye is responsible for declines in native bird populations at a high-elevation wet forest on Hawaiʻi Island (Freed et al. 2008, Freed and Cann 2009), but others have questioned this claim and pointed out that negative impacts from competition are difficult to demonstrate conclusively (Kingsford 2010, Camp et al. 2014, Banko et al. 2015). Non-native birds may also serve as reservoirs of diseases that negatively impact native forest birds (Ahumada et al. 2009, Samuel et al. 2011). The most important of these diseases are avian malaria and pox, which are mostly restricted to low elevation forests (see below). However, recent studies have documented declines in Hawaiian forest bird populations, even at higher elevations, as mosquitos and the diseases that they vector move upslope in response to warming temperatures and drying conditions (Paxton et al. 2016, and papers cited therein); moreover, projections of further upward spread of mosquitos and disease over the coming century due to climate change lead to predictions of substantially diminished suitable habitat area and bird richness in high elevation forests, including those within HALE (Fortini et al. 2015). Similarly, continued spread of habitat-altering non-native plants into high elevation forests has the potential to reduce habitat quality for forest birds (see below).

Condition summary for high elevation bird communities
In summary, the available evidence suggests that high elevation bird communities at HALE are currently in relatively good condition, and appear to be relatively stable, at least across East Maui as a whole and over the past several decades. Comparison with earlier time periods, however, would result in a less favorable picture, with several known or presumed extinctions, and probable population declines among at least several remaining species. Confidence in this assessment is medium, because while recent information is comprehensive, a shortage of standardized repeated surveys specific to HALE results in lower confidence in estimates of recent trends. Continued monitoring under the PACN landbird protocol will help address this deficiency; monitoring conducted in 2017 will be used in 2018 for an NPS Focused Condition Assessment for HALE. Additional high elevation forest survey data in the park between 1993 and 2008 (C. Bailey unpub. data) will also be incorporated in this assessment. Bird communities in mesic forests in Kaupō, however, do not appear to be addressed in these monitoring efforts, and little information on birds in this area currently exists.

Low elevation bird communities (1,000 - 4,200 ft)
During the 1967 Kīpahulu Valley Expedition, Warner (1968b) reported that no native forest birds were observed below 2,800 ft (860 m) elevation, and that several introduced species were seen regularly, including the "Pekin nightingale" (red-billed leiothrix) and Japanese white-eye. He noted that this absence of native birds below 2,800 ft was already observed and remarked upon in 1945 by the park assistant to the superintendent P.H. Baldwin, who descended through Kīpahulu Valley and found the bird distribution pattern inexplicable because the forest at that elevation appeared to otherwise be in excellent condition. Warner (1968b) therefore attributed the absence of native birds to the lethal effects of avian malaria and pox below approximately 2,800 ft (860 m) elevation. From 2,800-3,300 ft (860-1050 m) elevation, Warner (1968b) reported a steady increase in the abundance

of native honeycreepers, primarily ʻapapane and Maui ʻamakihi, with smaller numbers of Maui ʻalauahio and fewer ʻiʻiwi, and no appreciable decrease in abundance of introduced bird species. At mid-elevations, from 3,300-5,000 ft (1,050-1,540 m), he stated that bird numbers were uniform, with relative abundance of species estimated on a scale from 0-10 as follows: ʻapapane 10, ʻiʻiwi 6, Maui ʻamakihi 5, Maui ʻalauahio 4, red-billed leiothrix 3, and Japanese white-eye 1.

In 1980, Stemmerman (1980) observed two species of native forest birds and eight species of non-native forest birds in Kīpahulu Valley between 1,000 and 2,000 ft (305-610 m) elevation. The native species were ʻapapane and Maui ʻamakihi, which were found to occur above 1,800 ft (549 m) and 1,600 ft (488 m), respectively.

In Kaʻāpahu, Natividad Bailey (2007) detected four species of native forest birds and five species of non-native forest birds on 11 stations along a single transect extending from 1,800 to 3,880 ft (549-1,183 m) in elevation. Native forest birds included the Maui ʻamakihi, Maui ʻalauahio, ʻiʻiwi, and ʻapapane. Non-native forest birds included the Japanese bush-warbler, hwamei, red-billed leiothrix, Japanese white-eye, and northern cardinal (*Cardinalis cardinalis*). The transect was surveyed twice, in 2002 and 2005, with species composition being the same in both years. Abundance of three of the four native forest birds was lower in 2005 than in 2002, and abundance of four of five non-native species was higher in 2005 than in 2002 (Figure 4.2.3-17), suggesting a worsening status for native forest birds as a whole. However, no measures of variance were reported for the estimates of relative abundance, so it is not possible to compare these two time periods statistically. The differences in abundance also could have been caused by a difference in timing of the surveys (October in 2002 vs. May in 2005), or to a koa defoliation event that occurred in 2003-2004. Maui ʻalauahio and ʻiʻiwi were relatively rare, but their occurrence at all is somewhat unusual in this lower elevation area. The steep terrain and prevailing winds may result in lower abundance of mosquitoes and the diseases they transmit.

The 2012 Pacific Island landbird monitoring (Judge et al. 2013) included low elevation forest survey points in Kīpahulu Valley (Figure 4.2.5-17). Only four of the six extant native bird species, the same four detected by Natividad Bailey (2007) in Kaʻāpahu, were detected below 4,200 ft (1,280 m) elevation in Kīpahulu. The two endangered species, ʻākohekohe and kiwikiu, appeared to be absent, and two species that were present, ʻapapane and Maui ʻalauahio, were estimated to occur at significantly lower densities compared to forests above 4,200 ft (1,280 m) elevation (Figure 4.2.3-16). Six non-native species were detected: Japanese white-eye, Japanese bush warbler, red-billed leiothrix, hwamei, northern cardinal, and house finch (*Carpodacus mexicanus*).

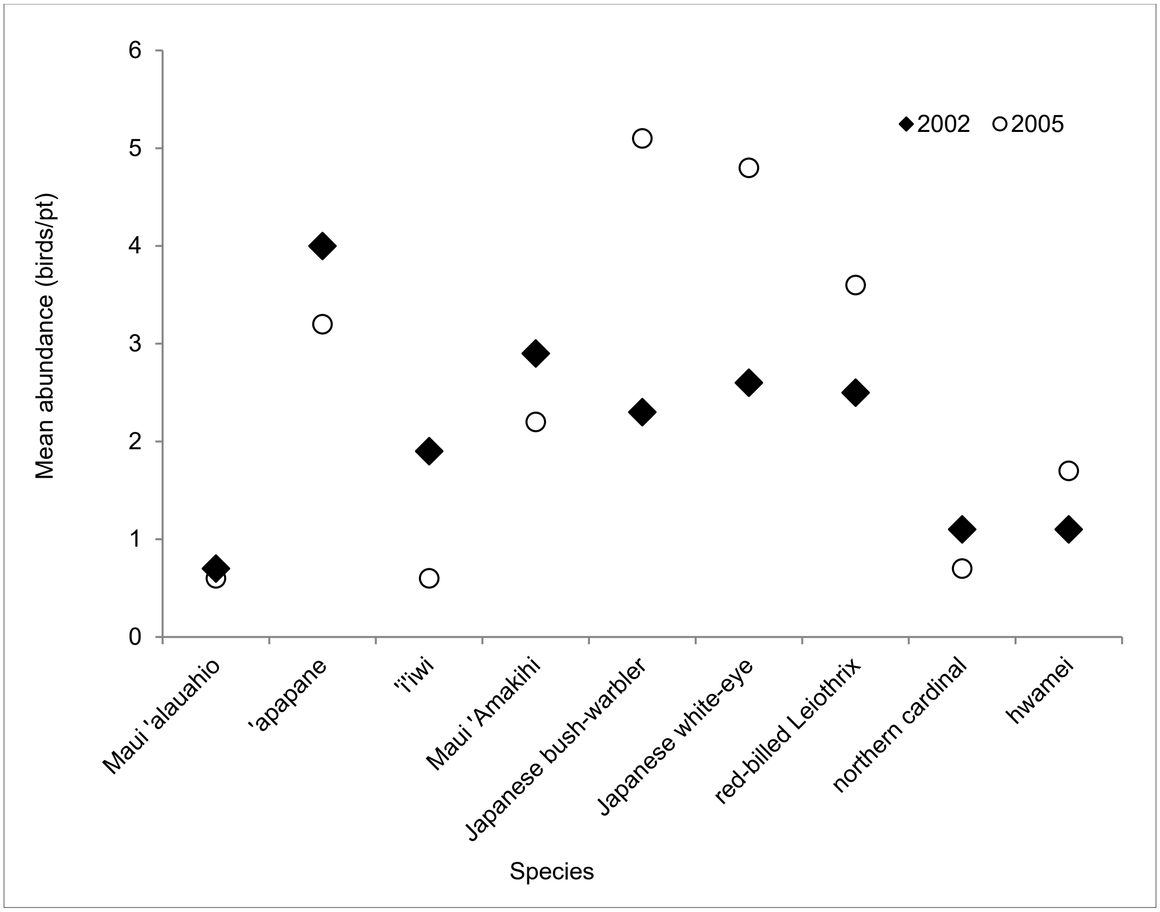


Figure 4.2.3-17. Relative abundance of native and non-native forest bird species in the Kaʻāpahu area in 2002 and 2005. Source: Natividad Bailey (2007).

All sources of information indicate a greatly diminished native bird community, both in species richness and abundance, in lower elevation forests, especially below approximately 3,000 ft (915 m) elevation. Although native birds in low elevation forests are impacted by predation from rodents and potentially by competition with non-native bird species, as described above for high elevation communities, the strongest limiting factor at lower elevations in Hawaiʻi is the prevalence of mosquitos, avian malaria and avian pox (Scott et al. 1986, Judge et al. 2013). Evidence for resistance to malaria has so far been found only for Hawaiʻi ʻamakihi (*Chlorodrepanis virens*) and Oʻahu ʻamakihi (*Chlorodrepanis flava*) (Paxton et al. 2016). Aruch et al. (2007) detected mosquitos and malarial infections in native birds at elevations of up to 3,000 ft (915 m), in Kīpahulu Valley during a 2002 study, but also measured malarial prevalence rates much lower than in comparable forests elsewhere in Hawaiʻi. It is unknown if these rates are typical or perhaps unusually low for this area. Conversion of low elevation forests to communities dominated by non-native plants may also impact habitat quality for native birds. Although data on this topic are limited, Foster (2005) found arthropod-based bird diets to differ strongly between individuals captured in introduced pine forest and those captured in native montane forest adjacent to HALE, possibly because the prey bases differed between the two forest types, and this was suggested to have a potential causal role for the much lower bird densities in pine forest. It is possible that habitat-modifying invasive plants in low elevation forests, such as *P. cattleianum*, *H. gardnerianum*, *C. hirta* and *M. calvescens*, could have similar effects on arthropod prey densities.

Condition summary of low elevation bird communities

In summary, we judge the current condition of low elevation native bird communities to warrant significant concern, with a declining trend. Confidence in this assessment is high. Several extant species no longer occur in low elevation forests, and half of the four remaining native species occur at lower densities compared to high elevation forests. In comparison, six non-native bird species occur, several at high densities. Moreover, most remaining native forest birds appear to now be restricted to a relatively narrow band near the upper end of our definition of low elevation forest, i.e. mainly between 3,000 and 4,200 ft (915-1280 m). As mosquitos and the diseases they vector move upslope, this band of suitable habitat can be expected to disappear. Other stressors, listed above, contribute to the current condition.

Invertebrate communities

Information on forest invertebrates at HALE is limited. Although a number of taxonomic studies have reported on species occurring within HALE forests (e.g. Schmitz and Rubinoff 2011, Liebherr 2015), and a variety of informal collections have been made, there are very few reported studies that have surveyed forest invertebrates in a way that allows for quantification of metrics such as native diversity, invertebrate habitat quality, or degree of non-native invertebrate invasion. This is undoubtedly due to the difficulty of completing such surveys, especially in diverse forest ecosystems. For example, the NPS PACN I&M monitoring plan postponed development of inventory and monitoring protocols for invertebrates until phase two of the program (HaySmith et al. 2006). Below, we summarize chronologically the invertebrate surveys that have been conducted in HALE forests, and what their findings suggest regarding the condition of forest invertebrate communities in the park.

The 1967 Kīpahulu Valley Expedition (Warner 1968a) conducted general but methodologically unspecified invertebrate surveys in the vicinity of three basecamps located at 3,100 ft (945 m), 4,100 ft (1,250 m), and 6,500 ft (1,982 m) elevation (Wilson 1968). Although identification of the collected material was preliminary at the time the report was written, approximately 130 insect and 22 mite species were collected, of which approximately 62% and 65%, respectively, were judged to be native (Wilson 1968). It was further stated that “If only those species collected in the native forest or from native animals are considered, then the percentages are considerably higher.” Otherwise, the reported results were not associated with the three basecamps, other elevation or locality information, or other descriptions of habitat quality. A separate report on native *Drosophila* pomace flies, however, reported the collection of 60 species, and specifically remarked on the excellent quality of the koa forest habitat around basecamp 1 (3,100 ft, 945 m), stating that it was “undoubtedly one of the best *Drosophila* collecting spots in the Islands. The forest of giant koa trees, completely uncut and unspoiled, shelters the delicate and complex native plants which in turn support the fly population” (Carson 1968). These findings may be interpreted to indicate relatively good condition of the invertebrate community in a very general way, but the lack of information on methodology and the mainly qualitative nature of the results makes further interpretation difficult.

An extensive insect survey effort from 1975-77, as part of the Resources Basic Inventory project (Berger et al. 1975), focused mainly on subalpine shrubland and aeolian ecosystems of the Crater District of the park (Beardsley 1980). However, several sampling sites in Kaupō Gap and around Palikū were located in forest or woodland habitat. A total of 204 species were collected in these habitats, of which 139 (68%) were native. Notably, collecting sites around Palikū yielded 182 species, of which 127 (70%) were native, while sites in the forested areas along Kaupō trail yielded only 56 species, of which 33 (59%) were native. This suggests that the wetter, slightly higher elevation forests around Palikū supported both a higher number and a higher proportion of native species, compared to the forested areas of Kaupō. Because it is unclear whether sampling effort was similar between these two areas, it is difficult to be certain of this conclusion. However, Beardsley did note that "Overall the Kaupō Gap had a higher percentage of introduced insects which appear to be correlated to the greater percentage of introduced vegetation" (Berger et al. 1975).

The 1976 Manawainui Expedition (Peterson 1976) conducted insect and spider sampling along a variety of transects installed across the Manawainui area of East Maui (Villegas 1976). Two of these transects were located partly to mostly within HALE boundaries, on the upper Manawainui Planeze: Healani transect running vertically along Healani Stream up to Kuiki summit, and Hihia transect contouring across the planeze near the park boundary at about 5,000 ft (1,524 m) elevation. A variety of collecting methods were used at stations along these transects; it is uncertain if sampling effort was completely standardized across stations, but the results probably provide reasonable estimates of relative diversities of native and non-native arthropods at each. Similarly, identification of samples was incomplete at the time of reporting, but was apparently sufficient to allow preliminary tallies of native and adventive species at each sampling station. On Healani transect, seven stations were located in forest habitat within the park (above 5,000 ft), and the percentage of species collected that were native was estimated to be 87.4% ± 4.8% (mean ± SE). Ten stations were within or relatively close to the park boundary (at about 5,000 ft) along Hihia transect, and for these forested stations the percentage of species collected that were native was estimated to be 86.1% ± 3.4% (Villegas 1976). Therefore, at this time the wet forests of Manawainui appeared to support minimally invaded, and likely reasonably intact, native arthropod communities.

The same expedition also made efforts to search for *Partulina* and *Newcombia* tree snails (Meyer 1976). None were found, but in the process a variety of other conspicuous snails were observed and collected, mainly within the relatively intact wet forests above 5,000 ft (1,524 m) elevation. A total of 130 individuals were collected, 94 of which were native. These represented approximately 25 species in 8 families or subfamilies, only three of which were non-native. The latter included the garlic snail, *Oxychilus alliarius*, an invasive predatory species that may impact native snails (Meyer and Cowie 2010), plus *Oxychilus cellarius*, and a third unidentified *Oxychilus* species. Two additional non-native snails were observed at lower elevation, more disturbed sites outside the park: the highly destructive, predatory rosy wolf snail (*Euglandina rosea*), and *Bradybaena similaris*. The native snails were collected mainly on native vegetation, with *Myrsine* spp., *Broussaisia arguta*, and *Clermontia* spp. being favored (Meyer 1976).

Several low elevation forest areas and riparian corridors below 2,000 ft (610 m) elevation in Kīpahulu Valley were surveyed for invertebrates in 1980 (Gagné 1980, Gon and Pinter 1980, Severns 1980). The poor status of invertebrates found in the coastal zone below 1,000 ft (305 m) elevation in these surveys is summarized in the coastal ecosystems section (Chapter 4.2.2). Approximately 283 species of insects and myriapods (millipedes and centipedes) were collected in forests and forested riparian corridors between 1,000 and 2,000 ft (305-610 m), of which about 174 (61%) were judged to be native (Gagné 1980). Most native insects were collected “in the primarily native vegetation situated between the Gaging Station on Palikea Stream and Palikea Summit, and the upper pasture fence line” (Gagné 1980). In these same areas, Gon and Pinter (1980) collected 13 species of spiders, of which 10 (77%) were native. They similarly remarked that native species “were associated with areas of more or less intact native vegetation” (Gon and Pinter 1980). Severns (1980) collected 7 snail species in the forests between 1,000 and 2,000 ft (305-610 m) elevation, of which 5 (71%) were native. However, the non-native species included *O. alliarius* and *E. rosea*, the two predatory species mentioned above, and *E. rosea* in particular was judged to be placing “severe stress” on the remaining native species (Severns 1980). Native tree snails such as *Partulina porcelana* and *Partulina nivea* had been collected in the area between 1919 and 1949, but were thought to be extirpated at the time of this survey (Severns 1980).

Some of the above survey results, particularly those pertaining to high elevation forests of Manawainui, suggest a reasonably intact and generally good condition of HALE forest invertebrates, while surveys from low elevations (especially below 2,000 ft) and from mesic forest areas of Kaupō indicate more degraded and more highly invaded invertebrate communities. However, the spatial coverage and intensity of the surveys are highly restricted, and any conclusions drawn from them derive from collections and observations made around 40 to 50 years ago. Although it is possible that arthropod communities in Kaupō mesic forests have subsequently increased in abundance and diversity in response to vegetation restoration activities, it is more likely that invertebrate communities in most forest locations have either persisted in a similar state, or degraded in response to a variety of forces. These include the invasion of non-native plants in low elevation forests detailed above, which reduces the cover and diversity of host plants for herbivorous invertebrate species, and also likely cascades through other parts of the invertebrate food web, as suggested by the concentration of native predatory spiders within native forest patches (Gon and Pinter 1980). In his introduction to the entomological series *Insects of Hawaii*, Zimmerman (1948) summarized the observations of many prior workers in concluding that most of the lowland forests of Hawaiʻi, and even upland forest areas, have undoubtedly suffered numerous invertebrate extinctions due to the loss or conversion of native vegetation. Other invertebrate biologists have subsequently drawn similar conclusions (Gagné and Christensen 1985, Howarth 1990).

In addition, non-native invertebrate predators threaten native invertebrate communities. As mentioned above, the predatory snail *E. rosea* is thought to have decimated the native snail fauna (Cowie 2001), and continues to threaten the species that remain; the predatory snail *O. alliarius* may be exerting additional pressure (Meyer and Cowie 2010). Similarly, predatory social insects, such as ants and yellowjacket wasps, are known to strongly impact native arthropod communities (Krushelnycky et al. 2005, Wilson et al. 2009), and may even disrupt insect-mediated pollination and impact plant reproduction (Hanna et al. 2013). Most ant species are not able to invade mesic to wet forests in Hawaii (Reimer 1994, Krushelnycky et al. 2005), and only a few ant species have been observed in the forests of Kīpahulu Valley (Fellers 1982, P. Krushelnycky unpub. data). The more open mesic forests and woodlands of Kaupō, however, may be more susceptible to ant invasion. Limited surveys of ants in the southeastern portion of Kaupō Gap in 2001 detected several species in or near forested areas (Figure 4.2.3-18, P. Krushelnycky unpub. data). These included the big-headed ant (*Pheidole megacephala*), a highly destructive species. However, *P. megacephala* appeared to be restricted to areas below approximately 4,750 ft (1,448 m) at that time, and tended to be found in shrubland or forest peripheries, rather than under forest canopy. Its current distribution in Kaupō is unknown.

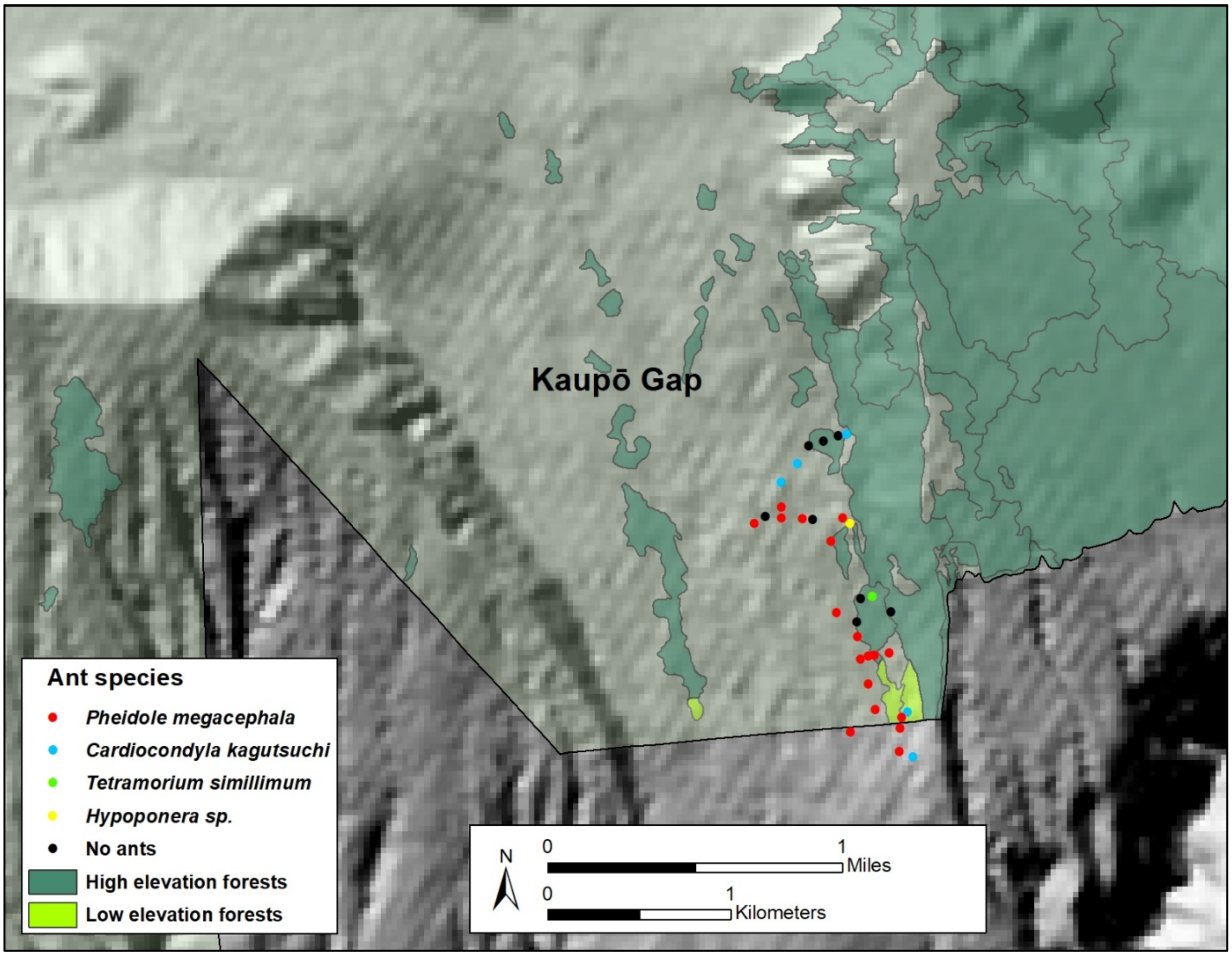


Figure 4.2.3-18. Ant species detected in and around Kaupō forest and woodland communities in 2001. Two additional species, *Plagiolepis alluaudi* and *Tapinoma melanocephalum*, were collected at locations with *C. kagutsuchi* and *P. megacephala*, respectively, and are not shown. Source: P. Krushelnycky unpub. data.

The western yellowjacket, *Vespula pensylvanica*, is found throughout much of the drier and higher elevation portions of the West Slope and Crater (Chapter 4.2.6 and 4.2.8), but its abundances in mesic and wet forests is unmonitored. Many other non-native predatory arthropod species have also become established and undoubtedly exert pressures, individually and collectively, on the native forest fauna, but these impacts are difficult to measure. Similarly, many non-native arthropods compete for resources with native species, but these dynamics are also very difficult to assess and are largely unstudied.

Finally, non-native vertebrates, including birds, mice and rats, prey on native invertebrate species. Non-native Japanese bush warblers, Japanese white-eyes, and red-billed leiothrix were all found to consume large quantities of arthropods in a high elevation wet forest adjacent to HALE (Foster 2005). Mice (*Mus musculus*) and both black rats (*Rattus rattus*) and Polynesian rats (*Rattus exulans*) have been shown to prey on native arthropods elsewhere in Hawaiʻi (Cole et al. 2000, Shiels et al. 2013), and a rat diet study in the 1980's documented their consumption of native arthropods in wet forests of HALE. Howarth and Stone (2007) examined the stomach contents of 159 *R. rattus* and 174 *R. exulans* trapped in forests from 4,100 to 6,800 ft (1,250–2,073 m) elevation in Kīpahulu Valley during 1983 and 1984, and found both rat species to consume a wide variety of arthropod prey. Araneae, Hemiptera, Coleoptera and Lepidoptera were the most common pretty items, and several rare arthropod species were found in the samples. Altogether, arthropods were present in nearly 100% of stomachs. Rats are also known to prey on native snails and are thought to have strongly impacted their populations (Hadfield 1986). However, rats also consume the predatory rosy wolf snail (*E. rosea*), potentially generating an indirect beneficial effect for native snails (Meyer and Shiels 2009).

Condition summary of invertebrate communities

In summary, the status of available information for forest invertebrate communities precludes a valid assessment of their current condition or trend. None of the existing surveys clearly state whether standardized collecting efforts were used, and may therefore be biased towards higher percentages of native species by the common tendency to maximize the collection of native species while passing over common non-native species. Furthermore, none of the surveys recorded numbers of individuals collected, precluding any estimates of the relative abundances (as opposed to relative diversities) of native and non-native species in forest communities. Most importantly, all of the information is very dated, and conditions may have changed substantially in the intervening decades. Snail surveys appear to be completely lacking in most wet forest areas.

Overall assessment, including threats and stressors

Integrating the five indicators for forest ecosystems according to SotP aggregation rules yields an overall current condition warranting moderate concern, with an unchanging trend. However, it is apparent that conditions, and trends, differ strongly between high elevation and low elevation forests. High elevation wet forests are in relatively good condition, and have retained or slightly increased native plant richness and diversity over the past few decades, potentially in response to fencing and exclusion of feral pigs. However, some highly damaging invasive plants, such as *H. gardnerianum*, show signs of making increasing inroads in high elevation forests, and the invasion of non-native plants will likely be accelerated if feral pigs continue to infiltrate the existing fence network. Mesic forests in Kaupō, located mostly in the high elevation zone, are more degraded than wet forests in Kīpahulu, Manawainui and Hāna Rainforest, having been more damaged by feral ungulates and weed invasion in the past. However, some recovery in this area has occurred, and can likely continue, with active management. Bird communities in high elevation forests are also in generally good condition, at least relative to modern historic reference points. Trends for some of the rarer species, like ʻākohekohe and kiwikiu, however, are more uncertain, and future trends for all species may become much less favorable if mosquitos and the diseases they vector spread upslope. Although the level of certainty is not high, at least some climate projections predict that the trade wind inversion may drop in elevation in the future (Lauer et al. 2013), which would lead to a downward shift in the tree line on the mountain (Loope and Giambelluca 1998, Crausbay et al. 2014a,b). This would cause a further contraction of suitable, mosquito-free forest habitat for native birds, as well as an overall reduction in high elevation forest habitat. Invertebrate communities in high elevation forests are likely in relatively good condition at present, but information to substantiate this assumption is very limited.

In contrast, over one-third of low elevation forests have non-native plants as the most common dominant or co-dominant species, and the remaining areas are under constant pressure from invasion and further modification by habitat-modifying weeds. In addition, further declines in *Acacia koa*, the most important canopy tree of low elevation forests, could result in an accelerated deterioration in native understory flora and continued expansion of non-native plant cover. Despite best management practices, invasive plant numbers have continued to steadily increase over time. Yoshinaga (1980a) reported that the first scientific survey of Kīpahulu Valley in 1945 documented only five non-native plant species, a total which increased to 22 in 1967 and 55 in 1980. Green et al. (2015) recorded 53 non-native plants in 66 monitoring plots, a small subset of the entire forest area, and Welton and Haus (2008) recorded 123 non-native plant species in the forests of Kaʻāpahu alone. Vegetation Management staff have anecdotally observed a severe and rapid degradation of low elevation forest communities in recent years (P. Welton pers. comm. 2018). Unless biological control agents can be found that reduce the ability of these weeds to reproduce, spread into and detrimentally impact native forest communities, management and control will be necessary in perpetuity to maintain their continued ecological integrity. In the absence of feral pigs (through fencing and control), non-native plant spread is likely to be slowed, but is still expected to continue without other forms of management intervention.

Low elevation bird communities are also in poor condition, with the most significant limiting factor being disease. Although some evidence for evolution of resistance to malaria has been found for Hawaiʻi ʻamakihi (*C. virens*) and Oʻahu ʻamakihi (*C. flava*) (Paxton et al. 2016), it is unclear if similar developments will emerge in other native species in time to ameliorate current patterns and trends. Although information on invertebrate communities in low elevation forests is also limited, it can be expected that they will be impacted by the invasion of non-native plants in these ecosystems. This may further stress bird communities if it affects their food resource base, and the degradation of both bird and invertebrate communities may in turn impact native plant diversity via loss of important pollinators for specialized plant species.

Perhaps the most significant imminent threat to native forests at all elevations in HALE is posed by emerging forest pathogens. In particular, fungal pathogens (*Ceratocystis lukuohia* (formerly species A) and *Ceratocystis huliohia* (formerly species B)) responsible for widespread mortality of *M. polymorpha* on Hawaiʻi Island, have not yet been detected on Maui (Keith et al. 2015, Mortenson et al. 2016, Barnes et al. 2018). The resulting disease has been termed "Rapid ʻŌhiʻa Death" because healthy trees die within weeks after first showing symptoms. If the pathogen becomes established on Maui, a substantial probability due to the wind-dispersed nature of the pathogen's spores and Maui's proximity to Hawaiʻi island, the loss of *Metrosideros* trees would completely transform high and low elevation forests and likely result in the subsequent decline of much of the remaining associated native flora and fauna. (return to Condition Summary)

Level of confidence

Overall level of confidence in this assessment is medium. Much information has been reviewed to provide the current assessment, but very few data have been repeatedly collected over time, creating more uncertainty in assessment of trends. Information is fragmentary, lacking, or insufficiently quantitative for certain resource areas, especially invertebrates, which also lowers overall level of confidence.

Information gaps and research recommendations

Previously, repeated quantitative information on forest ecosystem vegetation (i.e., cover and richness, degree of non-native plant invasion) was fairly limited. This is in the process of being remedied via the FTPC and the EIPS monitoring protocols of the NPS I&M program (Ainsworth et al. 2011, 2012). The repeated monitoring of these two sets of plots, including most recently in 2017-2018, will allow for site-specific comparisons over time that will greatly enhance future vegetation trend analyses. However, mesic forests of Kaupō are not covered in these monitoring systems. Due to time and staff limitations, consistent monitoring is not conducted for most of the many threatened, endangered, or otherwise rare plant species in forest ecosystems, making assessment of trends in these species difficult. Repeated standardized monitoring of forest birds will provide greater certainty to assessments of population trends of native birds going forward. Although challenging to conduct, recent quantitative surveys of native forest invertebrate communities are lacking, leaving a large information gap for this highly diverse and ecologically important forest component.

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4.2.4. Bog Ecosystems

Condition Summary

Overall resource condition warrants moderate concern with an unchanging or stable trend; confidence in condition determination is low. Rationale is discussed in the "*Overall assessment*" section and following.

⇔

Relevance and Context

Bogs occur in scattered locations of poor drainage in the montane wet forests of the five largest Hawaiian Islands, with notable examples found in the Alaka'i Swamp of Kaua'i, Mount Ka'ala on O'ahu, Pēpē'ōpae Bog on East Moloka'i, Pu'u Kukui and Mount 'Eke on West Maui, and smaller bogs on East Maui and the summit area of the Kohala Mountains on Hawai'i (Gustafson et al. 2014). A series of high elevation montane bogs occur within HALE in the Hāna Rainforest on the outer northeast rift at 5,380-6,150 ft (1,640- 1,875 m) elevation (Figure 4.2.4-1; Loope et al. 1991a). High annual rainfall amounts of up to 400 inches (ca. 10,000 mm) and a nearly impervious substrate have created these open areas within otherwise dense rain forest. A separate series of smaller bogs, dominated by non-native grasses and herbs, occur at lower elevation mainly along Palikea Stream on the lower shelf of Kīpahulu Valley, between elevations of 2,400 and 2,950 ft (730-900 m) (Figure 4.2.4-1).

The higher elevation bogs in Hāna Rainforest support a unique community of native grasses, sedges and herbs as well as dwarf, nearly prostrate shrubs. Approximately 30 native species of vascular plants occur in these ecosystems, about half of which also grow in surrounding wet forests, while the other half are primarily restricted to the bog environment (Loope et al. 1991a). Under pristine conditions, ground cover in these bogs is dominated by sedges. These include *Oreobolus furcatus*, a tussock- or mat-forming species, and four species of *Carex*: *C. alligata*, C. *echinata*, C. *montis-eeka*, and C. *thunbergii.* With the exception of *C. alligata*, these sedges are mostly restricted to bogs (Loope et al. 1991a). Other characteristic species include the endangered shrub *Geranium hanaense*, the herb *Plantago pachyphylla*, the woody shrub *Lobelia gloria-montis* subsp. *longibracteata*, and the greensword *(Argyroxiphium grayanum*), a striking rosette or branched prostrate shrub closely related to the silversword ('āhinahina, *Argyroxiphium sandwicense*) that grows in much drier Crater ecosystems (Loope et al. 1991a) (Figure 4.2.4-2). Another endangered species, *Clermontia samuelii* subsp. *samuelii*, grows along bog margins.

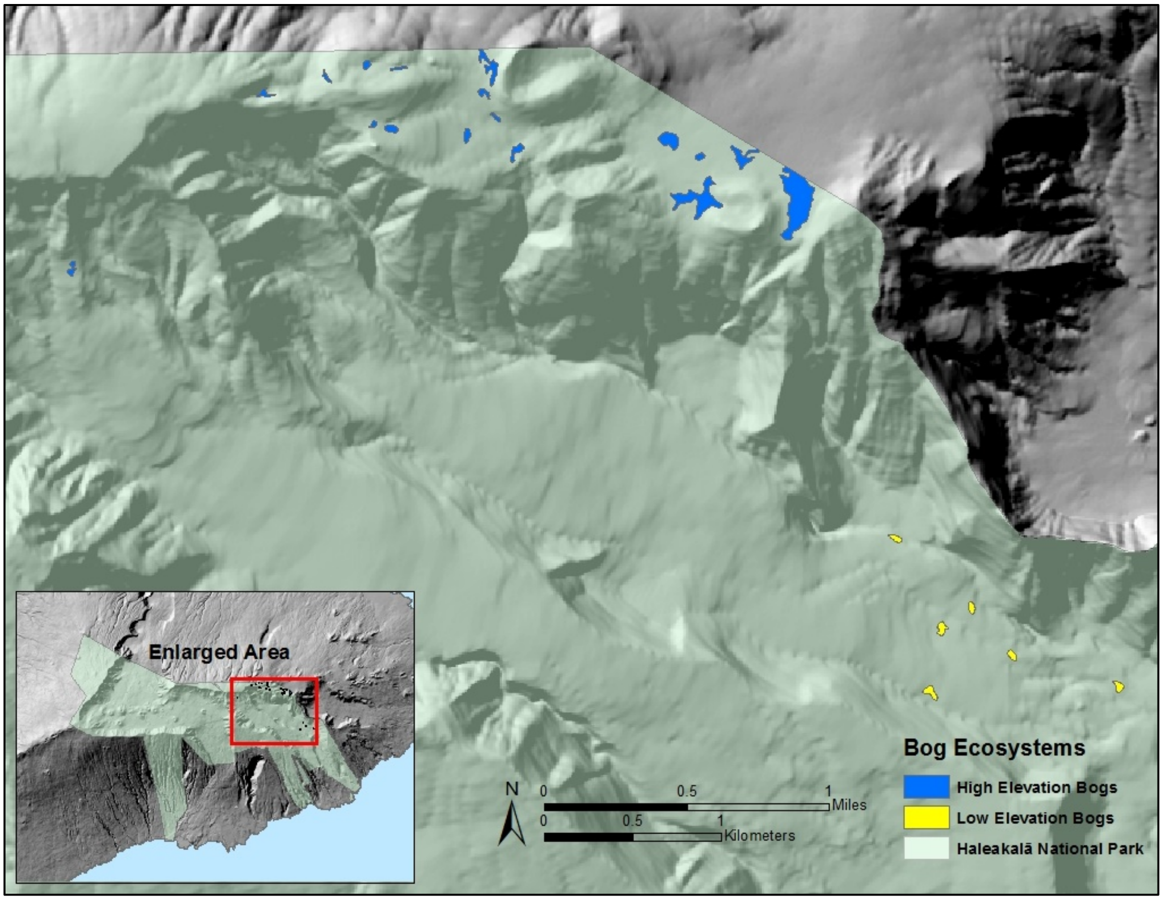

Figure 4.2.4-1. Location of high elevation and low elevation bogs in HALE. Source: Green et al. (2015).

All six of the extant native forest birds can be found in the forests surrounding at least some of the high elevation bogs (Loope et al. 1991a, Judge et al. 2013), and may forage within them to at least some extent. These are ʻapapane (*Himatione sanguinea*), Maui ʻamakihi (*Chlorodrepanis virens wilsoni*), Maui ʻalauahio (Maui creeper, *Paroreomyza montana newtoni*), ʻiʻiwi (*Drepanis coccinea*), ʻākohekohe (crested honeycreeper, *Palmeria dolei*), and kiwikiu (Maui parrotbill, *Pseudonestor xanthophrys*). The pueo (Hawaiian short-eared owl, *Asio flammeus sandwichensis*) is rarely seen in the vicinity of the bogs, and the migratory kōlea (Pacific golden plover, *Pluvialis fulva*), frequents bogs during the fall to spring non-breeding seasons (Loope et al. 1991a). A variety of non-native passerine bird species also reside in surrounding forests (Loope et al. 1991a, Judge et al. 2013). At least several species of native insects are thought to be tightly associated with bog plants, such as the fruit fly *Neotephritis paludosae* and two species of *Plagithmysus* long-horned beetles, all of which feed on greenswords (Gagné 1983, Loope et al. 1991a). Most members of the bog invertebrate communities, however, are also likely to be found in surrounding forests.

Figure 4.2.4-2. Top left: Upper elevation bog with *Oreobolus furcatus* tussocks; Top right: *Geranium hanaense*, endemic to montane bogs of East Maui; Bottom left: Greenswords, *Argyroxiphium grayanum*; Bottom right: *Plantago pachyphylla*. Photos: G.D. Carr, except Forest and Kim Starr for bottom right.

High elevation bogs at HALE began experiencing severe impacts in the 1970s from the digging and rooting of feral pigs, possibly in search of earthworms (Loope et al. 1991a). This activity not only damaged rare native plants, but also promoted the invasion by non-native grasses and herbs, including *Holcus lanatus* (velvetgrass), *Tibouchina herbacea* (cane tibouchina) and *Andropogon virginicus* (broomsedge) (Loope et al. 1991a, Medeiros et al. 1991, Loope et al. 1992). In response, NPS fenced the bogs between 1979 and 1988 to exclude pigs and encourage the recovery of native vegetation. Other non-native animals that are present and might impact native bog ecosystems include the black rat (*Rattus rattus*) and house mouse (*Mus musculus*), and less commonly mongoose (*Herpestes auropunctatus*) and feral cat (*Felis cattus*) (Loope et al. 1991a). The invasive slug, *Milax gagates* has been seen feeding on greenswords (Gagné 1983).

We assess the condition of bog ecosystems using a single indicator: bog vegetation. We focus on the high elevation bogs in Hāna Rainforest, as relatively little information is available for the low elevation bogs in Kīpahulu Valley. We do not consider native birds here, because forest birds are a peripheral component of bog ecosystems, and are addressed primarily in the forest ecosystems section (Chapter 4.2.3). Very little information is available regarding bog invertebrate communities, and their condition is likely to be similar to those of surrounding forests.

Indicators

- Bog vegetation

Data and Methods

A review of available data and information was used to assess conditions, and if possible, trends, in the indicator listed above. Some additional analyses were performed on a subset of geospatial and vegetation classification data collected by HALE and PACN staff specific to bog vegetation communities (Green et al. 2015).

Sources of information

Monitoring, outplanting, and propagule collection data for common, rare, and endangered native taxa have been recorded by HALE Vegetation Management staff for the entire park from 1993 through present (NPS 2017a).

Control data for invasive, non-native plants have been collected by HALE Vegetation Management staff for the entire park from 1986 through present (NPS 2017b).

NPS I&M vegetation classification data were also collected by HALE and PACN staff in 220 circular plots (400 m^2) and observation points in 2011, 2 of which were in bog ecosystems (Green et al. 2015).

Vegetation data for three bogs from 1982 to 1988 were provided in Loope et al. (1991b) and Medeiros et al. (1991).

Reference condition

Ideal reference conditions for bog vegetation would include diverse and fully intact native communities, uninvaded by non-native weeds, and undisturbed by pigs or other non-native ungulates. However, since the arrival of humans and the subsequent introduction of non-native plants and animals to the Hawaiian Islands, no bog ecosystem currently exists in an entirely pristine state. Nevertheless, examples of relatively high-quality Hawaiian bogs, in which native plant species richness and abundance exceeds that of non-natives, can still be found, particularly at higher elevations and remote locations throughout the islands. A desirable reference condition for HALE's bog vegetation is therefore that the richness and cover of native species exceeds that of nonnative species.

Condition and Trend

Bog vegetation

By 1988, all of the high elevation bogs in Hāna Rainforest had been fenced, but 23 non-native plants had already become established within them (Loope et al. 1991a). Loope et al. (1991a) stated that these non-native plants have difficulty invading undisturbed bogs, some of which were near-pristine as recently as 1974 (Loope et al. 1991b), but proliferate after bare ground is exposed by pig rooting.

Exclusion of pigs with perimeter fences demonstrated that, at least in one bog, native vegetation was capable of recovering to a large degree. Greensword Bog, originally dominated by the mat forming sedge *O. furcatus*, recovered from a state of almost complete (94%) pig-tilled bare ground in 1981 to one dominated by native plant cover (88%) within about three years after pig exclusion (Loope et al. 1991b). This recovery was driven by growth of dominant sedges like *O. furcatus* and *C. echinata*, and the native grass *Deschampsia nubigena*. Recovery of less common species like *A. grayanum* and *P. pachyphylla*, however, was much less extensive even after seven years (Loope et al. 1991b).

The two largest bogs, Big Bog and Mid-camp Bog, were not fenced until 1987 and 1988 (Medeiros et al. 1991). Monitoring of these bogs from 1982 to 1988 documented a decline in native cover and invasion by several non-native plants as a result of repeated pig disturbance. The largest declines among native species occurred for *O. furcatus*, *A. grayanum*, *C. thunbergii*, *P. pachyphylla*, *Selaginella deflexa* and *Trisetum glomeratum*. Invasion by non-native species was greatest for *H. lanatus*, *Juncus planifolius*, and *Sacciolepis indica*. Medeiros et al. (1991) noted that areas of the bogs dominated by *C. echinata* were much more prone to non-native plant invasion than areas dominated by *O. furcatus*: over the six year period, non-native cover increased from 6% to 30% in plots dominated by *C. echinata*, compared to an increase of only 0.2% to 2.6% in plots dominated by *O. furcatus*. They were hopeful that fencing of these large bogs would result in recovery of native vegetation, as occurred in Greensword Bog (Loope et al. 1991b).

Nearly 25 years after the last bog was fenced, Green et al. (2015) mapped and classified vegetation for 23 bogs within and along the northern rim of Kīpahulu Valley. A subset of these bogs were visited and analyzed to classify vegetation for all 23 bogs. Six of these bogs occur on the lower shelf of Kīpahulu between elevations of 2,400 and 2,950 ft (730-900 m), and the remainder occur within Hāna Rainforest along the northern rim of the valley and on the northeast rift at elevations between 5,400 and 6,800 ft (1,645-2,070 m) (Figure 4.2.4-1). The lower elevation bogs, occupying an area of 3.5 ac (ca. 1.4 ha) on gentle southeast aspects along Palikea Stream, are dominated by non-native grasses and herbs and have been classified as *Ageratina adenophora* - *Paspalum conjugatum* Semi-natural Herbaceous Bogs, although some native plants including stunted *Metrosideros polymorpha* and *Acacia koa* trees and *Carex alligata* sedges, among others, also persist (Green et al. 2015). The 17 upper elevation bogs, occupying a total area of 26.9 ac (10.9 ha), are largely native-dominated, and have been classified as *Carex* (*alligata*, *echinata*, *montis-eeka*, *thunbergii*) - (*Deschampsia nubigena*) Mixed Herbaceous Montane Bogs (Green et al. 2015). These high elevation bogs are dominated or codominated by the sedges *C. alligata*, *C. echinata*, *C, montis-eeka*, and/or *C. thunbergii*, and have lower cover of *A. grayanum* and *G. hanaense* (Green et al. 2015). Disturbed sites may be co-dominated by the invasive grass *H. lanatus*.

Two inventory and vegetation mapping plots were located within the higher elevation, native-dominated bogs, one within Big Bog and one in a nearby bog, and provide a sample of some of the species found therein (Green et al. 2015). Species identity and cover was recorded in multiple vegetation layers, or strata, including herbs, shrubs and trees (after Tart et al. 2005). A total of 22 native and four non-native species have been recorded, with a mean species richness of 13.5 ± 4.5 native and 3.5 ± 0.5 non-natives per plot (Figure 4.2.4-3; Green et al. 2015). Mean native plant cover in these plots is almost four times greater than non-native plant cover in all strata (Figures 4.2.4-4). As would be expected in bogs, tree and shrub cover is very low (<14%), whereas herbaceous vegetation accounts for almost 95% cover (Figures 4.2.4-4). Within the herb stratum, mean native plant cover is 66.55% ± 7.45% and non-native cover is 28% ± 24% (Figure 4.2.4-4). No recent data have been collected in the lower elevation *Ageratina adenophora* - *Paspalum conjugatum* Semi-natural Herbaceous Bogs.

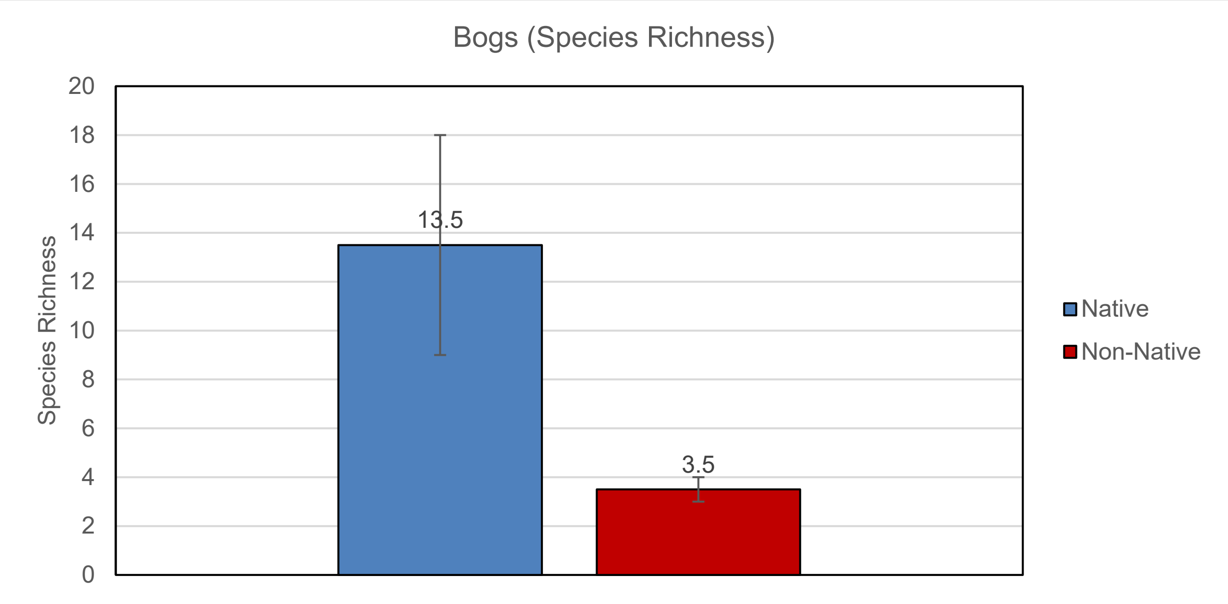

Figure 4.2.4-3. Mean native and non-native plant species richness recorded within 400 m^2 inventory and mapping plots in upper elevation bogs at HALE (n = 2). Source: Green et al. (2015).

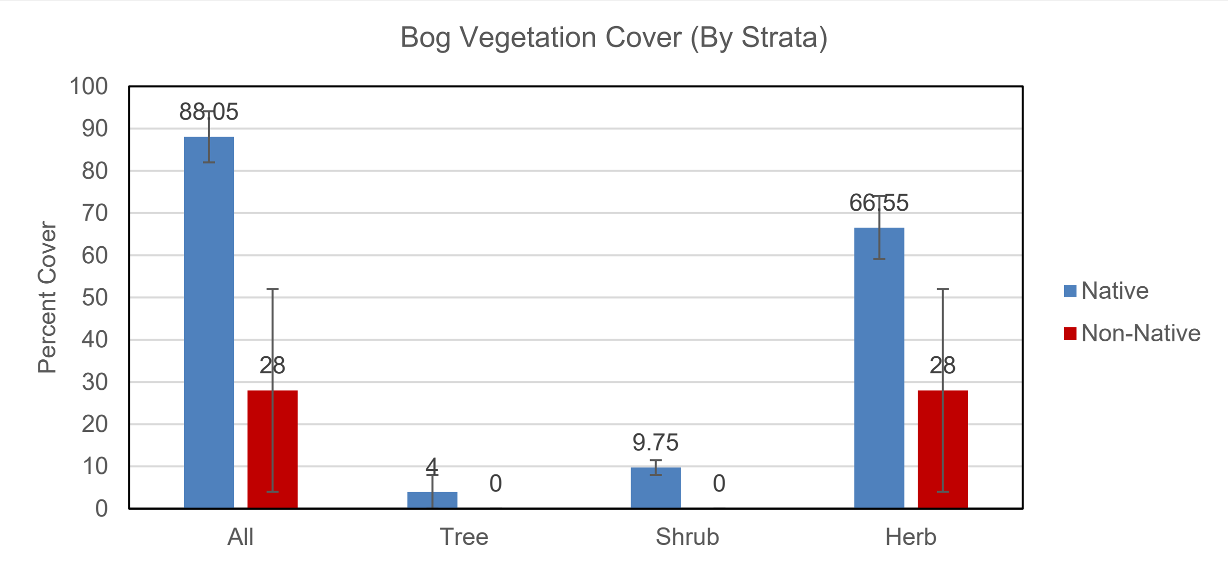

Figure 4.2.4-4. Percent cover of native and non-native plants, divided by strata, within 400 m^2 inventory and mapping plots in upper elevation bogs at HALE (n = 2). Source: Green et al. (2015).

The total of four non-native species recorded by Green et al. (2015) in 2011 is much lower than the 23 non-native species present in 1988 as reported by Loope et al. (1991a). However, these totals are not comparable because the former applies to only two 400 m^2 plots, while the latter was a total across the entirety of numerous bogs on the northeast rift. Both Medeiros et al. (1991) and Green et al. (2015) also measured vegetative cover within bogs on the northeast rift. Because the plot locations and the sampling methods between the two efforts differed, it is difficult to make any robust comparisons between the two data sets. However, the 28% cover by non-native plants measured in 2011 by Green et al. (2015) in Big Bog and a nearby bog is similar to the 30% cover of non-native plants in *C. echinata*-dominated plots in Mid-camp Bog in 1988 (Medeiros et al. 1991). This may suggest that the condition of these bogs has remained fairly similar in a general sense since fences were installed, and that native plants have not been able to displace the non-native species that established earlier. More specific or consistent monitoring would be needed to decipher finer patterns in vegetation change.

In efforts to restore the bog ecosystems to their pristine states, HALE Vegetation Management Unit staff has targeted a number of priority non-native plants for control in both low elevation (Kīpahulu Valley) and high elevation (Hāna Rainforest/northeast rift) sets of the bogs (Table 4.2.4-1). Numbers reported are not from discrete areas and are not necessarily reflective of the status and trends of individual weeds, but rather are indicators of the amount of management effort expended on weed control in these habitats. As can been seen, most of this effort has targeted the more highly invaded lower elevation bogs in Kīpahulu Valley, although several important weeds threaten the high elevation bogs as well.

Table 4.2.4-1. Numbers of notable non-native plants controlled in and around bog ecosystems in HALE between 1992 -2016. Source: NPS (2017b).

Taxon	Low elevation bogs (Kīpahulu Valley)	High elevation bogs (Hāna Rainforest)
Andropogon virginicus	40	76
Angiopteris evecta	43	0
Anthoxanthum odoratum	0	3
Clidemia hirta	18,683	0
Cyathea cooperi	1,096	0
Erharta stipoides	0	30
Hedychium gardnerianum	5,670	1
Holcus lanatus	0	3
Juncus planifolius	0	1,551
Paspalum urvilleii	0	60
Psidium cattleianum	551	0
Psidium guajava	17	0
Spathodea campanulata	4	0
Tibouchina herbacea	2,707	30

In addition to controlling invasive weeds, HALE Vegetation Management Unit staff have augmented natural recovery of bog vegetation with outplanting of both rare and native common plants, including three endangered taxa. Cumulative totals of individuals planted in high and low elevation bogs are shown in (Table 4.2.4-2).

Table 4.2.4-2. Numbers of native plant taxa outplanted in bog ecosystems in HALE. Source: NPS (2017a).

Taxon	**Low elevation bogs (Kīpahulu Valley)**	**High elevation bogs (Hāna Rainforest)**	**T&E**
Argyroxiphium grayanum	0	228	NA
Carex alligata	86	0	NA
Clermontia samuelii ssp. *samuelii*	0	38	E
Cyrtandra ferripilosa	0	12	E
Phyllostegia haliakalae	0	12	E
Isachne distichophylla	60	0	NA
Machaerina angustifolia	624	0	NA
Total	**770**	**266**	–

Overall assessment

Based on the limited available information, the condition of bog ecosystems is judged to warrant moderate concern. Protection of bog communities has resulted in the recovery of native richness and cover in some upper elevation bogs, but the continued presence of non-native weeds in others, and future invasion by non-native species remains a constant threat. Lower elevation bogs are heavily modified and are unlikely to increase in native species diversity or cover. The trends in condition of bogs at both elevations appear relatively stable at this point in time, although data supporting this inference are limited to very general patterns. With continued management, these communities will likely remain stable, but currently non-native dominated communities are unlikely to improve without additional management intervention. (return to Condition Summary).

Level of confidence

Overall level of confidence in this assessment is low. Repeated vegetation monitoring data are restricted to a few bogs during the 1980s. More recent vegetation data are limited spatially, and do not correspond exactly to earlier data, lowering confidence in the assessment of trend.

Information gaps and research recommendations

Repeated quantitative vegetation data are limited spatially and temporally. No NPS I&M Focal Terrestrial Plant Community or Established Invasive Plant Species monitoring plots are located in bog ecosystems, yet additional vegetation plot data would greatly enhance future vegetation trend analyses. Information on invertebrate communities in bog ecosystems is very limited.

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4.2.5. Cave Ecosystems

Condition Summary

Overall resource condition is good with stable or unchanging trend; confidence in condition determination is low. Rationale is discussed in the "*Overall assessment*" section and following.

↔

Relevance and Context

Three types of caves supporting cave-specialized species occur in Hawai'i: lava tubes, limestone caves, and piping caves (Howarth 2004). Limestone caves occur only in raised limestone reefs or lithified sand dunes along Hawaiian coastlines. Piping caves form when softer volcanic layers such as ash are eroded from beneath harder rock layers. By far the most common type of cave in Hawai'i is the lava tube (Howarth 1973, 2004). Lava tubes are formed predominantly in pāhoehoe lava flows, when moving lava rivers crust over and subsequently drain, leaving behind a hollow tube. These caves can range dramatically in size, from very small voids to large subterranean passages that may stretch kilometers in length (Howarth 1973, Halliday 2004; Figure 4.2.5-1). Many additional smaller cavities, cracks and fissures typically exist in lava fields that may form a vast underground network of spaces and potential connections between them (Howarth 1973). Because lava tubes have a relatively short geological lifespan, collapsing and filling as they erode, most lava tubes exist today on Hawai'i Island and Maui, the two youngest islands. On the older islands, lava tube caves have usually fully eroded, excepting a few caves in younger, secondary flows (Howarth 1973).

Francis Howarth pioneered the biological exploration of Hawaiian lava tubes, beginning with his discovery of blind, cave-adapted planthoppers and crickets in a Hawai'i Island tube in 1971 (Howarth 1972). Since then, over 75 species, comprised of insects, spiders, and related arthropods, have been discovered using Hawaiian lava tubes, including 19 on Maui (Howarth et al. 2007, Stone and Howarth 2007a). These organisms vary in their degree of dependence on the cave environment, and include both troglophiles (facultative users of caves that also utilize similar dark and humid microhabitats outside of caves) and troglobites (obligate users of caves that are fully adapted to cave microhabitats and unable to exist outside) (Howarth 1973). Endemic members of this diverse community include planthoppers (Hemiptera: Cixiidae), terrestrial water treaders (Hemiptera: Mesoveliidae), moths (Lepidoptera: Noctuidae), thread-legged bugs (Hemiptera: Reduviidae), spiders (Araneae: Linyphiidae), crickets (Orthoptera: Gryllidae), ground beetles (Coleoptera: Carabidae), isopods (Isopoda), amphipods (Amphipoda), millipedes (Diplopoda), centipedes (Chilopoda), and others (Howarth 1973, Howarth 2004).

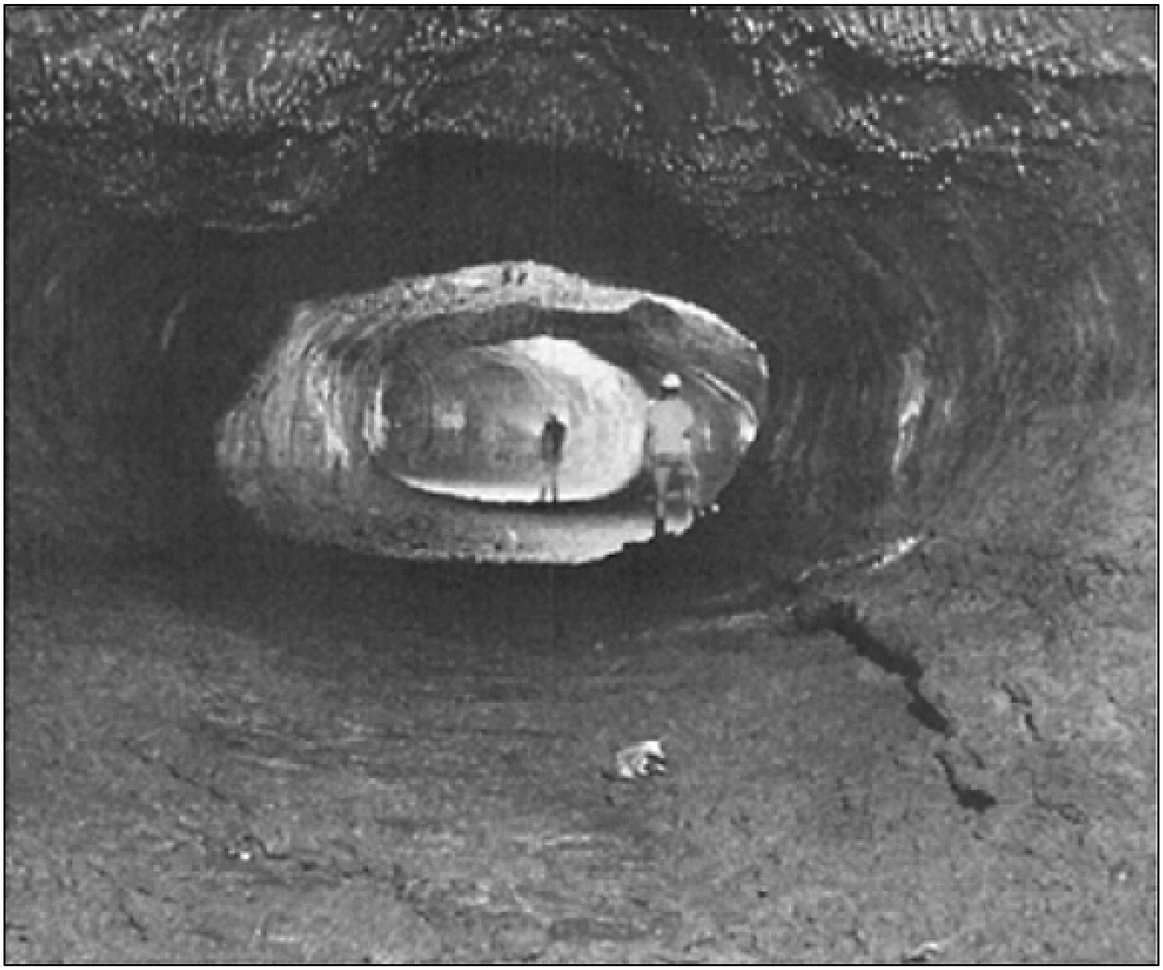

Figure 4.2.5-1. An example of a Hawaiian lava tube cave. Pictured is the main tunnel of Kazumura Cave on Hawai'i Island, which is over 30 km long (Halliday 2004). Photo: Tony Waltham, in Halliday (2004).

The principal energy source for this ecosystem is apparently plant roots that extend through the cave ceiling, especially those of the common forest and lava field pioneer tree species *Metrosideros polymorpha* (ʻōhiʻa lehua) (Howarth 1973, Stone and Howarth 2007a). Herbivores and some omnivores feed on live and dead root tissues, predators take these herbivores and omnivores as prey, and scavengers feed on dead resident arthropods and other animals that become accidently stranded and perish in caves. Slimes sometimes found on cave walls and floors, largely consisting of bacterial or fungal growth on rotting organic material, also support the food web to some degree (Howarth 1973).

Hawaiʻi's endemic bat, *Lasiurus cinereus semotus* (ʻōpeʻapeʻa), is a forest roosting species, and is generally not thought to use caves (Howarth 1973). However, one report describes observations of bats flying in and out of a lava tube pit on Hawaiʻi Island, possibly for the purposes of roosting, and relates other observations that suggest possible roosting in lava tubes (Fujioka and Gon 1988). Notwithstanding, no definitive regular roosting of bats in Hawaiian lava tubes has yet been reported.

Because plant roots provide the main source of food for cave animals, the structure and composition of the plant community directly above lava tubes may influence the condition of the cave community below (Stone and Howarth 2007a). Troglobitic members of the planthopper genus *Oliarus* are specialized herbivores, feeding on xylem sap in live root tissues, and are host specific in the plant species they utilize (Hoch and Howarth 1999). Replacement of native plant species by invasive weeds on the surface above lava tubes, therefore, threatens the ability of planthoppers to persist (Howarth et al. 2007). Other endemic herbivores, such as caterpillars of the noctuid moth *Schrankia howarthi*, and cave-adapted omnivores, such as *Thaumatogryllus* crickets, have been found to feed on non-native plant roots (Howarth et al. 2007, Medeiros et al. 2009), and may better withstand plant invasions above their caves. It is unknown, however, how reduction or elimination of specialist herbivores may disrupt the other components of the cave food web. Similarly, many non-native arthropod species have now been incorporated into cave food webs, including isopods, spiders, cockroaches, earwigs, springtails, silverfish, and others, and their impacts on the unique endemic cave fauna are unknown (Howarth 1973).

Other invasive species that have been documented in lava tubes include the black rat (*Rattus rattus*) and the mongoose (*Herpestes auropunctatus*). These species are probably relatively common in the entrance and twilight zones of caves, and may impact the native biota that also use these habitats (Howarth 1973, Medeiros et al. 1989, Stone and Howarth 2007b). For example, cave-specializing noctuid moths “once roosted in caves in huge numbers, but the group has become rare” (Howarth 2004); it is possible that predation by rats, which are known to favor lepidopteran prey (Shiels et al. 2013), has contributed to this decline. Carcasses of both rats and mongoose are occasionally observed deeper in caves, but it is not known if these species regularly forage in these dark zones (Howarth 1973, Stone and Howarth 2007b).

Although water may flow through many lava tubes during heavy rain events, even causing erosional enlargement in some situations (Howarth 1973, Halliday 2004), permanent or even frequent streams are not a regular feature of most Hawaiian caves. Flowing water usually hastens the siltation and destructive erosion of Hawaiian lava tubes, shortening their duration (Howarth 1973). Consequently, tubes that experience temporary streams and/or frequent flooding typically have poor faunas (Howarth 1973), and unlike continental caves, water quality and hydrology are therefore not among the most important indicators of cave ecosystem health in Hawaiʻi. Only one cave in HALE, in upper Kīpahulu Valley, has been reported to contain a small stream (Liebherr and Zimmerman 2000), but nothing is known about its hydrology.

In addition to supporting unique biological communities, Hawaiian caves are often valuable repositories of fossil bird bones, and have been instrumental in reconstructing the archipelago’s former, now largely extinct, bird fauna (Olson and James 1982). For example, bones of five extinct bird species have been found in two caves in HALE, in Kīpahulu Valley (Medeiros et al. 1989). However, as these represent individuals that accidentally died in these caves, rather than species that utilized the cave ecosystems, we do not comment on them further here.

Caves are often highly sensitive environments. As already mentioned, Hawaiian cave communities can be impacted by alterations to the vegetation above them, and other surface activities can also degrade them (Stone and Howarth 2007a). Moreover, many of the species inhabiting Hawaiian lava tubes are restricted to deep zones characterized by still air that is saturated with water vapor, whose conditions can be impacted by human visitation, alteration of cave morphology, and other activities (Stone and Howarth 2007a). Because unique cave faunal communities are so sensitive, and because caves may also contain important archaeological resources (like bird bones and Native Hawaiian cultural artifacts and burials), caves receive special protection under the Federal Cave Resources Protection Act of 1988 (NPS 2004). All caves in National Parks are deemed to be "significant caves" under the act (NPS 2004), therefore we only provide general location descriptions of HALE caves in this assessment, and do not disclose confidential information regarding cave contents.

There are relatively few caves known to support cave-adapted biota within HALE boundaries, and most have been only minimally explored by experienced speleologists. We focus on two main resource indicators: cave-dwelling invertebrates and other fauna, and vegetation. We assess the effects of invasive species on the condition of each of these indicators.

Indicators

- Cave-dwelling invertebrates and other fauna
- Vegetation

Data and Methods

A review of available data and information was used to assess conditions, and if possible, trends, in the indicators listed above. No additional data analysis was performed.

Sources of information

Most of the information on HALE cave fauna is contained in an unpublished, confidential report to the US Fish and Wildlife Service (Howarth 2002). In addition, a published report of preliminary investigations of two caves in Kīpahulu Valley (Medeiros et al. 1989), and several taxonomic descriptions of troglobite species found in or near the park (Christiansen and Bellinger 1992, Liebherr and Samuelson 1992, Hoch and Howarth 1999, Liebherr and Zimmerman 2000) were used in this assessment. Data collected by Anderson et al. (1992), and a comparable subset collected by Meston and Mallinson (2011) were used to analyze the status of *M. polymorpha* and invasive weeds in the wet forests of Kīpahulu Valley, as potential indicators of status of vegetation overlying cave ecosystems.

Reference condition

An ideal reference condition for cave ecosystems in HALE consists of a diverse endemic invertebrate fauna, uninvaded by non-native species, and an undisturbed native vegetation community on overlying surfaces. We consider good conditions as only minimal deviation from ideal conditions, such that the persistence of endemic cave-adapted species is not threatened. In general, areas in which cover of *M. polymorpha* or other dominant native plants is stable or increasing would indicate that root resources, the prime energy source for native subterranean biota, should not be a limiting factor in the health of any underlying cave communities.

Condition and Trend

Cave-dwelling invertebrates and other fauna

Only two troglobitic arthropod species have been described from HALE caves to date. These are two species of ground beetle (Coleoptera: Carabidae), each in one of two small caves in upper Kīpahulu Valley (Medeiros et al. 1989): *Blackburnia aaae* from Pukamoa Cave, and *B. howarthi* from Lua Manuiwi Cave (Liebherr and Samuelson 1992, Liebherr and Zimmerman 2000). Phylogenetic analyses indicate that these beetles represent two independent colonizations of, and subsequent adaptation to, the subterranean environment, as each has as its closest relative a different surface species within the large endemic radiation that is *Blackburnia* (Liebherr and Zimmerman 2000). Both beetles exhibit typical adaptations to cave life, such as reduced eyes and elongated, thin legs (Figure 4.2.5-2), and each is known to occur only in its respective cave (Liebherr and Zimmerman 2000). Both Pukamoa and Lua Manuiwi are relatively small lava tubes covered by montane wet forest. Each cave harbors at least one or two additional cave species that have yet to be described (Howarth 2002, Table 4.2.5-1). Black rats, or their droppings, were observed in both caves (Howarth 2002).

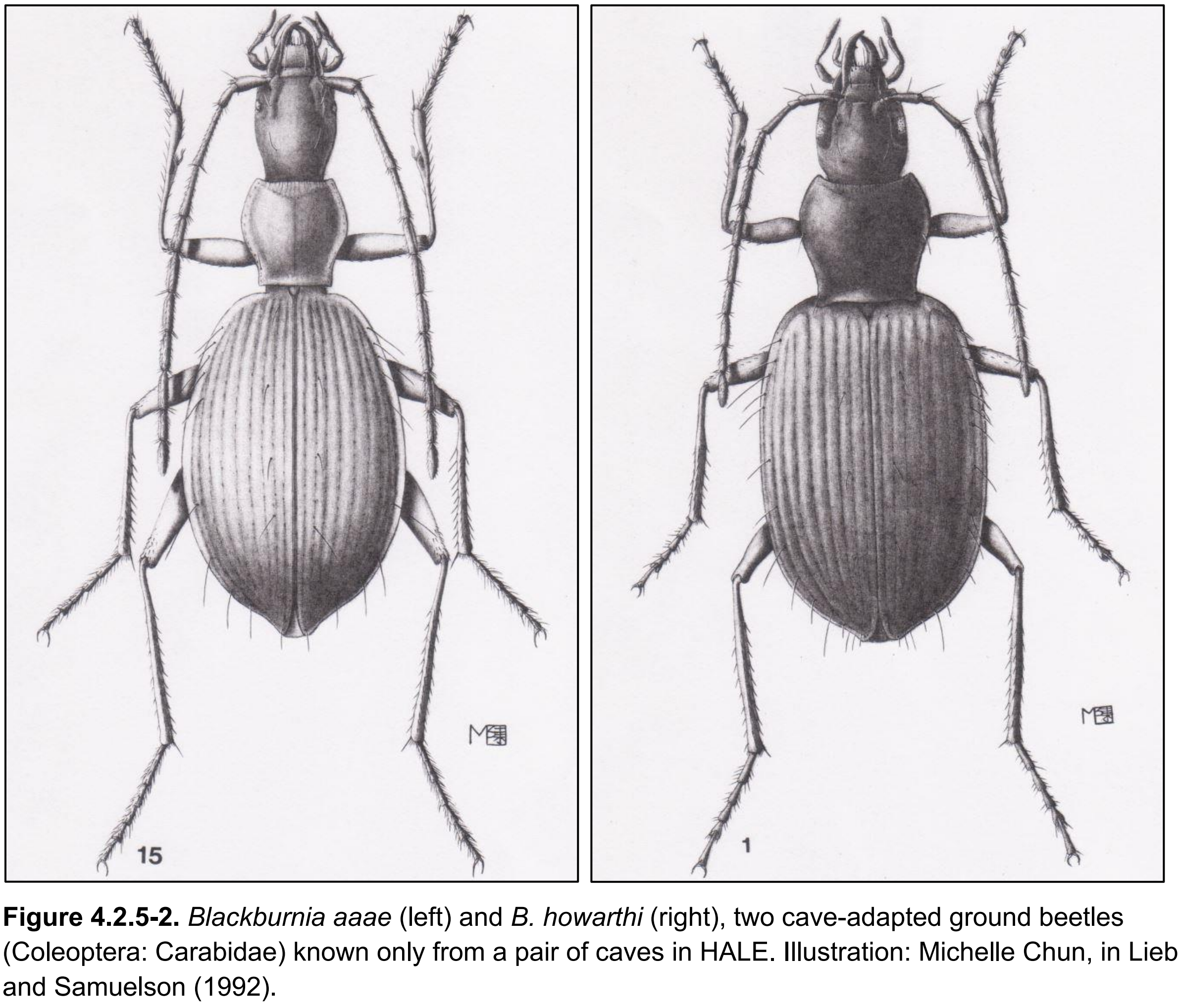

Figure 4.2.5-2. *Blackburnia aaae* (left) and *B. howarthi* (right), two cave-adapted ground beetles (Coleoptera: Carabidae) known only from a pair of caves in HALE. Illustration: Michelle Chun, in Liebherr and Samuelson (1992).

Table 4.2.5-1. Native troglobitic fauna known to occur in caves in and around HALE. Source: Howarth 2002.

Cave	Known native biota
Pukamoa Cave	• Carabid beetle (*Blackburnia aaae*) • Undescribed millipede (*Nannolene* sp.) • Possibly *Thaumatogryllus* crickets (eggs seen, no adults observed)
Lua Manuiwi Cave	• Carabid beetle (*Blackburnia howarthi*) • Undescribed springtail (Collembola)
Waikau Cave	• Cixiid planthopper (*Oliarus waikau*) • Springtail (*Sinella nupa*) • Undescribed spider (*Theridion* sp.) • Undescribed cricket (*Thaumatogryllus* sp.) • Undescribed millipede (*Nannolene* sp.) • Undescribed centipede (*Lithobius* sp.)
West slope road break cave	• Unidentified *Thaumatogryllus* crickets
Holua Long Cave	• Unidentified centipede

Two additional species have been described from Waikau Cave, a lava tube just outside the park boundary in Koolau Gap. *Oliarus waikau* is an endemic cave-adapted cixiid planthopper (Hemiptera: Cixiidae) that sucks xylem sap from live plant roots. Waikau Cave is overlain with native shrubs and small trees, and *O. waikau* was seen to feed on unidentified roots that may have belonged to *M. polymorpha*, *Leptecophylla tameiameiae*, *Dodonea viscosa*, *Coprosma* sp., or *Vaccinium* sp. (Hoch and Howarth 1999). *Sinella nupa* is a cave-adapted springtail (Collembola: Entomobryidae), presumably feeding on decomposing organic matter; Christiansen and Bellinger (1992) state that this species is “the most highly troglomorphic” cave collembolan in Hawaii and one of the most troglomorphic members of the genus *Sinella*. Only one specimen has been seen. In addition to these two described species, at least four additional cave-adapted native invertebrates have been observed in Waikau cave: undescribed species of spiders, crickets, millipedes and centipedes (Howarth 2002, Table 4.2.5-1). *Oliarus waikau, Sinella nupa*, and the undescribed *Theridion* spider are known only from this cave, and Howarth (2002) indicated that the cave probably supports additional endemic species yet to be detected.

Kalua Awa is comprised of two vent caves in the central crater area; one contains a pit that has not been biologically surveyed, and the other is relatively shallow with its entirety in the twilight zone (Howarth 2002). The latter is reported to have once been an important roosting site for millions of native noctuid moths (Howarth 2002). These are now rare or extinct, perhaps as a result of non-target impacts from biological control agents and predation by rats (Howarth 2002).

A small lava tube was discovered in 1999 in the west slope shrubland, when road-widening construction on the park road to the summit accidentally penetrated the cave (Howarth 2002). Investigation of this cave revealed the presence of unidentified, troblobitic *Thaumatogryllus* crickets (Howarth 2002). The cave was subsequently re-sealed.

Perhaps the best-known lava tube in HALE is Holua Long Cave. This approximately 300 m long tube has frequently been explored by park visitors, especially before a metal ladder was removed from the lower entrance. It is sparsely vegetated on the surface, and an unidentified cave-adapted centipede is the only animal known to inhabit it (Howarth 2002).

Numerous additional shallow caves, pits and rock overhangs have been identified in HALE (Halliday 1993), but these are not deep enough to possess dark zones. They therefore support only facultative cave-using species, both native and introduced, and are not known to support unique cave-adapted species (Howarth 2002).

Surveys for the Hawaiian bat, *L. cinereus semotus* (‘ōpe‘ape‘a), in HALE have been very limited (see Chapter 4.2.10), but there have been no bat detections in the immediate vicinity of any known significant caves (Fraser et al. 2007), and there have been no reports of bats using HALE caves for roosting or foraging. It is possible that Kalua Awa may have attracted foraging bats in the past, when many moths were reported to emerge at dusk (Howarth 2002).

The known native fauna in HALE caves appears to be relatively undisturbed, with the exception of likely predation by invasive rats in certain portions of certain caves. Invasive arthropods have not been reported from most of these caves, but it is certain that at least some occur, especially in entrance and twilight zones. Because of limited surveys in these caves, it is difficult to estimate the overall impacts of these invasive species on the native communities. There is currently little indication that plant root resources have declined or degraded in recent years. (The potential implications of vegetation are discussed further in the following section.) With the information at hand, it would appear that the condition of native cave-dwelling invertebrates and other fauna (of which there appears to be none) is generally good. Confidence in this assessment is medium, because of limited empirical information. The trend in this condition appears relatively unchanging, as there have been no obvious new or intensifying sources of degradation subsequent to the introduction of rats.

Vegetation

Very little has been published on the vegetation in or above caves within HALE. Aside from the information already noted above, several other native plants are often associated with, but not restricted to cave environs. For example, the endemic Hawaiian begonia, *Hillebrandia sandwicensis*, occurs along stream courses in Kīpahulu Valley and the Hāna Rainforest, but had been previously collected in Kaupō Gap with the note, "Found growing on very wet cave in deep shade. Kaupo trail, 3900 ft" (Medeiros et al. 1998). Likewise, the endemic, and federally listed endangered fern, *Asplenium peruvianum* var. *insulare*, is found in moist, low light environments and “at the mouths or in remote corners of lava tubes that receive very little light” (Palmer 2003). Similarly, the rare fern *Cystopteris douglasii* is found in both mesic forests and cave mouths, and on wet, protected ledges and within lava tube entrances of the upper elevation subalpine-alpine zone (Palmer 2003).

All of the caves discussed in the previous section are, in general terms, overlain with relatively undisturbed native vegetation. However, no quantitative, site-specific data exist for above-ground or subterranean flora of these caves, precluding a robust determination of the current condition or trend of native vegetation in these particular cave ecosystems. In addition, it is likely that other, undiscovered lava tube caves exist in the park. The condition of native vegetation within other ecosystems of concern may therefore provide a general indication of the status of the flora above any underlying cave ecosystems.

Within the wet forest ecosystem of Kīpahulu Valley, a subset of 400 m^2 plots analyzing changes of dominant plant cover between the early 1980s (1983-86) and 2011 indicate that overall cover of *M. polymorpha* has increased from 37.5% to 44.4% (Figure 4.2.5-3), but that this increase is restricted to upper elevation forests above 4,000 ft (1,220 m) elevation. While *Metrosideros* cover in the upper valley has increased from 32.8% to 60%, it has declined from 46% to 16.2% below 4,000 ft (1,220 m). This decline in *Metrosideros* cover may be associated with an increase in cover of three habitat-modifying invasive weeds (*Hedychium gardnerianum, Psidium cattleianum, Clidemia hirta*; Figure 4.2.5-4), although further analysis is needed to determine the extent of the relationship. Such changes in the cover of this keystone species suggest that dependent biota in any underlying cave ecosystems may be resource-limited by its decline at lower elevations, but may have stable or increasing nutrient resources in upper elevation wet forests.

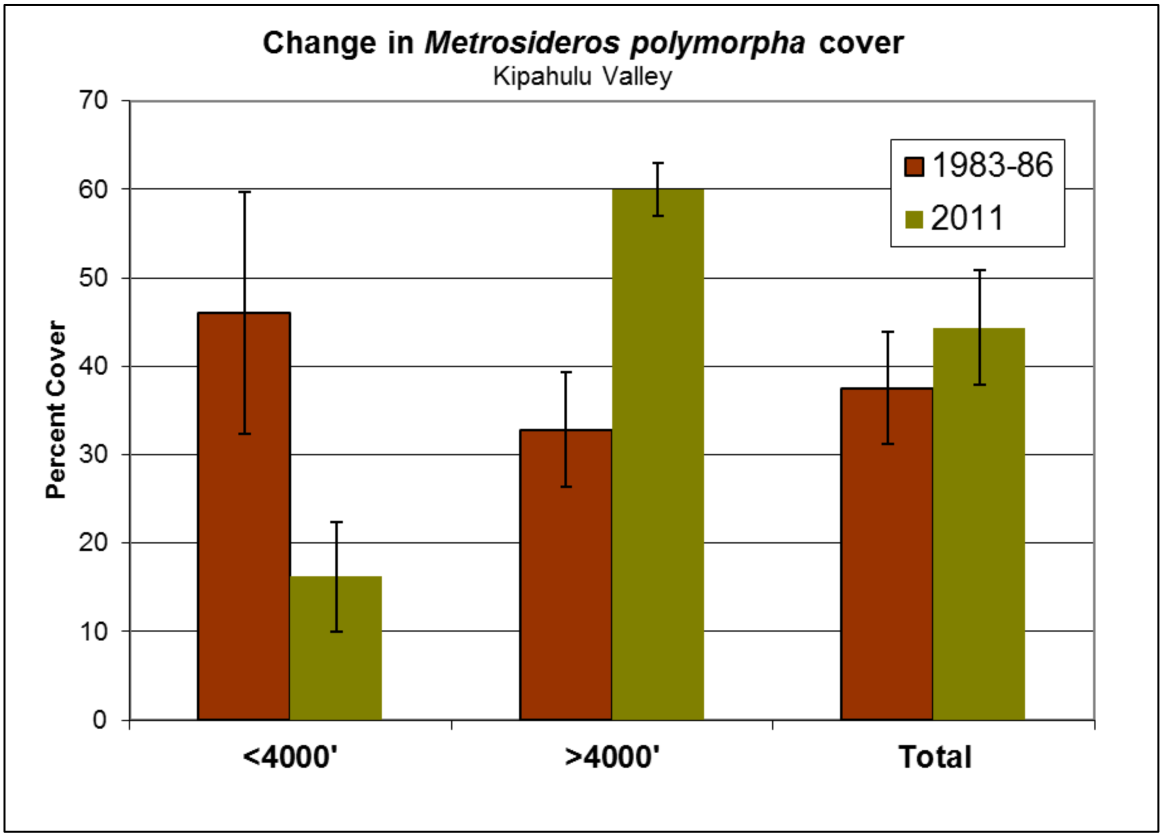

Figure 4.2.5-3. Changes in percent cover of *Metrosideros polymorpha* in 400 m^2 monitoring plots within wet forest above and below 4,000 ft elevation, Kīpahulu Valley (1983-86 to 2011). Source: Anderson et al. (1992) and Meston and Mallinson (2011).

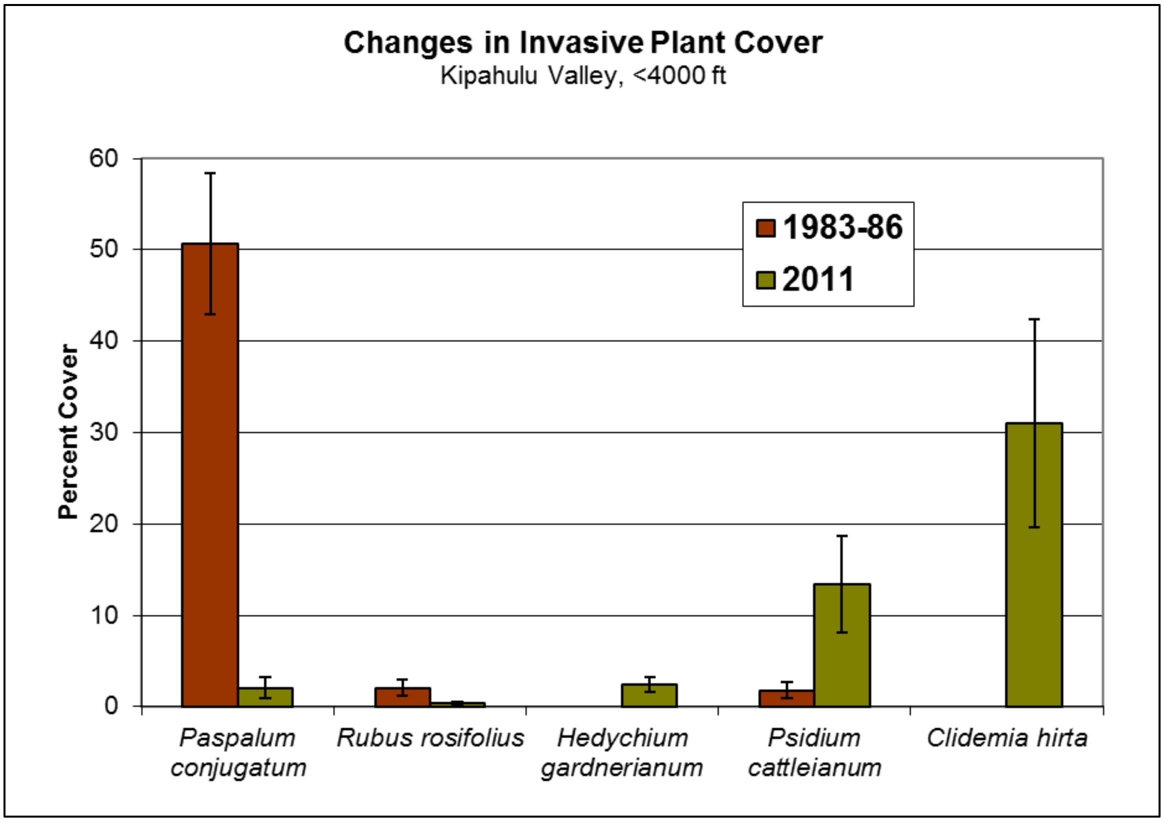

Figure 4.2.5-4. Changes in percent cover of five invasive plants in 400 m^2 monitoring plots within wet forest below 4,000 ft elevation, Kīpahulu Valley (1983-86 to 2011). Source: Anderson et al. (1992) and Meston and Mallinson (2011).

The condition of vegetation above HALE caves appears to generally be good at higher elevations, where all of the known lava tubes that support cave faunal communities occur. This appears to be true for caves located under wet forest as well as shrubland and sparsely vegetated ecosystems (see chapters 4.2.3, 4.2.6, and 4.2.8). However, the condition of any caves that might be located at lower elevations, such as in lower Kīpahulu Valley, might now be degraded as cover of overlying weeds has increased in recent decades (chapter 4.2.3). Similarly, while the trend in native vegetation cover appears to be stable or even improving at higher elevations, the opposite trend is occurring at lower elevations. Because of these conflicting trends, and because site-specific data for known cave ecosystems are lacking, confidence in the good condition of cave-associated vegetation is low.

Overall assessment

In summary, integration of the two main indicators of cave ecosystems yields an overall good condition. The trend in this condition was assessed as unchanging, because while vegetation at lower elevations has become increasingly invaded by non-native species, no known caves supporting troglobitic faunal communities have yet been discovered in these areas, and not all cave-adapted species appear to be dependent on the presence of native plant roots (Howarth et al. 2007). To the extent that vegetation structure on overlying soils remains similar and roots continue to penetrate, even caves located below areas experiencing plant invasion may continue to support portions of their invertebrate communities, provided that other forms of direct habitat disturbance are prevented.

Level of confidence

Overall confidence in this assessment is low. Although a number of unique cave-adapted species have been detected in HALE, most of the caves have been only minimally surveyed. The true diversity of both native and invasive invertebrates is likely underestimated, and no direct information exists to rigorously assess trends in condition. Site-specific information on the composition of cave-associated vegetation, which may influence the condition in the caves below, is also lacking. Judgment of both good condition and stable trend is therefore largely inferred from an absence of evidence to the contrary. (return to Condition Summary).

Information gaps and research recommendations

Currently, no monitoring plots are knowingly located directly above identified cave ecosystems. Data collected in such plots would provide site-specific information on the status and trends of native and non-native vegetation and could serve as an indicator of the status of potential energy inputs for the resident invertebrate community. Similarly, no monitoring plots are currently located around the mouths of caves or lava tubes, but would provide valuable data on the status and trends of rare and endangered terrestrial plants associated with these light-limited environments. Finally, more extensive and repeated surveys of cave faunas would provide more information on their status and trends, and identification of plant roots within caves would help inform their management (Howarth et al. 2007). However, because of the sensitive nature of these ecosystems, such intensified monitoring has the potential to degrade them, and thereby undermine the park's mission to protect the biological, archaeological and cultural resources inside caves. Consultation with experienced biospeleologists and archaeologists is strongly advised when considering such activities.

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4.2.6. Shrubland Ecosystems

Condition Summary

Overall resource condition warrants moderate concern with an unchanging or stable trend; confidence in condition determination is medium. Rationale is discussed in the "*Overall assessment*" section and following.

↔

Relevance and Context

Gagné and Cuddihy (1990) classify plant communities by elevation, precipitation, and physiognomy (i.e. overall structure or physical appearance of the community and the dominant species), and recognize five physiognomic classes in Hawaiʻi including Herblands, Grasslands, Shrublands, Forests, and Mixed Communities. Shrublands are those communities comprised of 40% or greater cover in shrubs, defined as "perennial woody plants with usually several to numerous primary stems arising from or relatively near the ground" (Wagner et al. 1999, Figure 4.2.6-1).

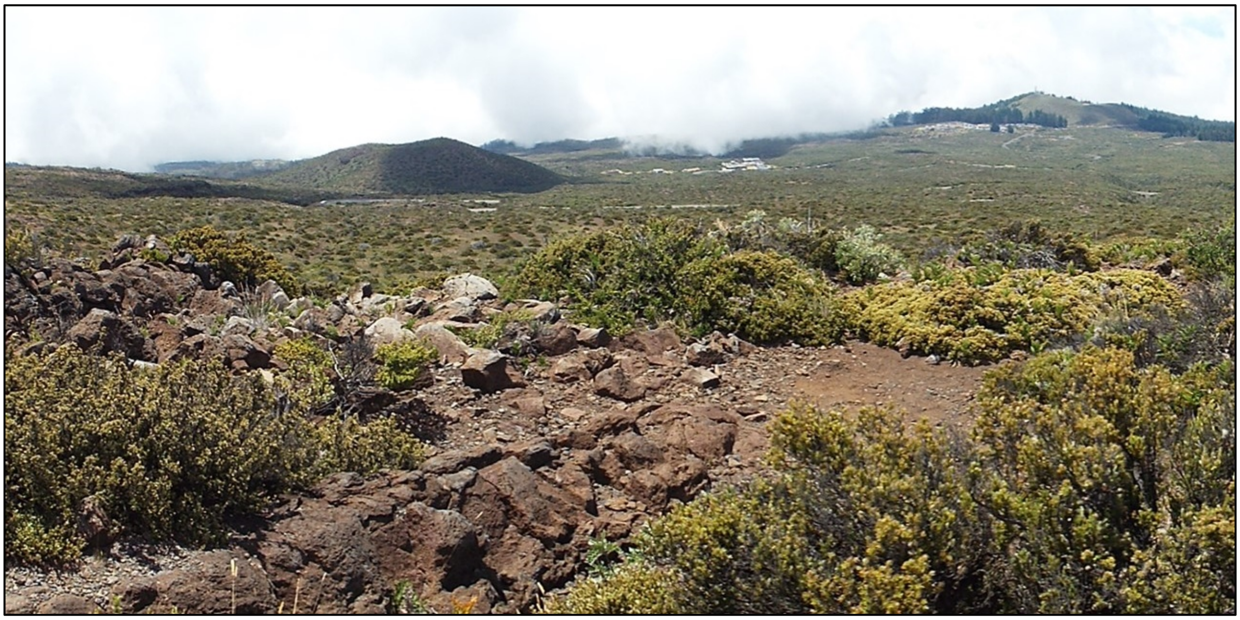

Figure 4.2.6-1. Subalpine shrubland ecosystem on the West Slope of Haleakalā National Park. (Source: Forest and Kim Starr).

Within the Hawaiian archipelago, subalpine shrublands (and high-elevation woodlands) surround the upper windward and leeward slopes of Haleakalā, Maui, and Mauna Loa, Mauna Kea, and Hualālai, Hawaiʻi island. The composition and structure of shrubland communities vary by elevation, precipitation, and substrate age within these regions (Gustafson et al. 2014).

Green et al. (2015) have classified approximately 11,575 ac (4,684 ha) as covered in shrubland vegetation communities throughout HALE (Figure 4.2.6-2). This important ecosystem type therefore comprises 34.3% of the park. Haleakalā's subalpine shrublands occur on the volcano's western and northwestern slopes from below the park boundary at 6,724 ft elevation (2,050 m) up to the alpine zone at approximately 8,530 ft (2,600 m) (Figures 4.2.6-1 and 4.2.6-2). Shrublands can also be found on the northern and southern cliffs and outer rims of the crater, on a variety of typically older substrates within the crater, and in the upper western Koʻolau Gap. In Kaupō Gap, subalpine shrublands intergrade with leeward shrublands in central Kaupō (<5,576 ft (1,700 m) elevation), with leeward forests along the eastern and western cliffs, with wet forest in upper eastern Kaupō near Palikū, and with *Acacia koa* forest in lower eastern Kaupō (Medeiros et al. 1998). Shrublands in HALE are mostly situated at or above the trade wind inversion (TWI), and therefore typically experience clear and dry atmospheric conditions (Giambelluca and Nullet 1991), with relatively low annual rainfall (approximately 1,000 to 1,400 mm on the western slope; Giambelluca et al. 2013). Although mean annual rainfall can exceed 2,500 mm in parts of Kaupō shrublands (Giambelluca et al. 2013), young soil substrates inhibit the development of forest throughout most of Kaupō Gap.

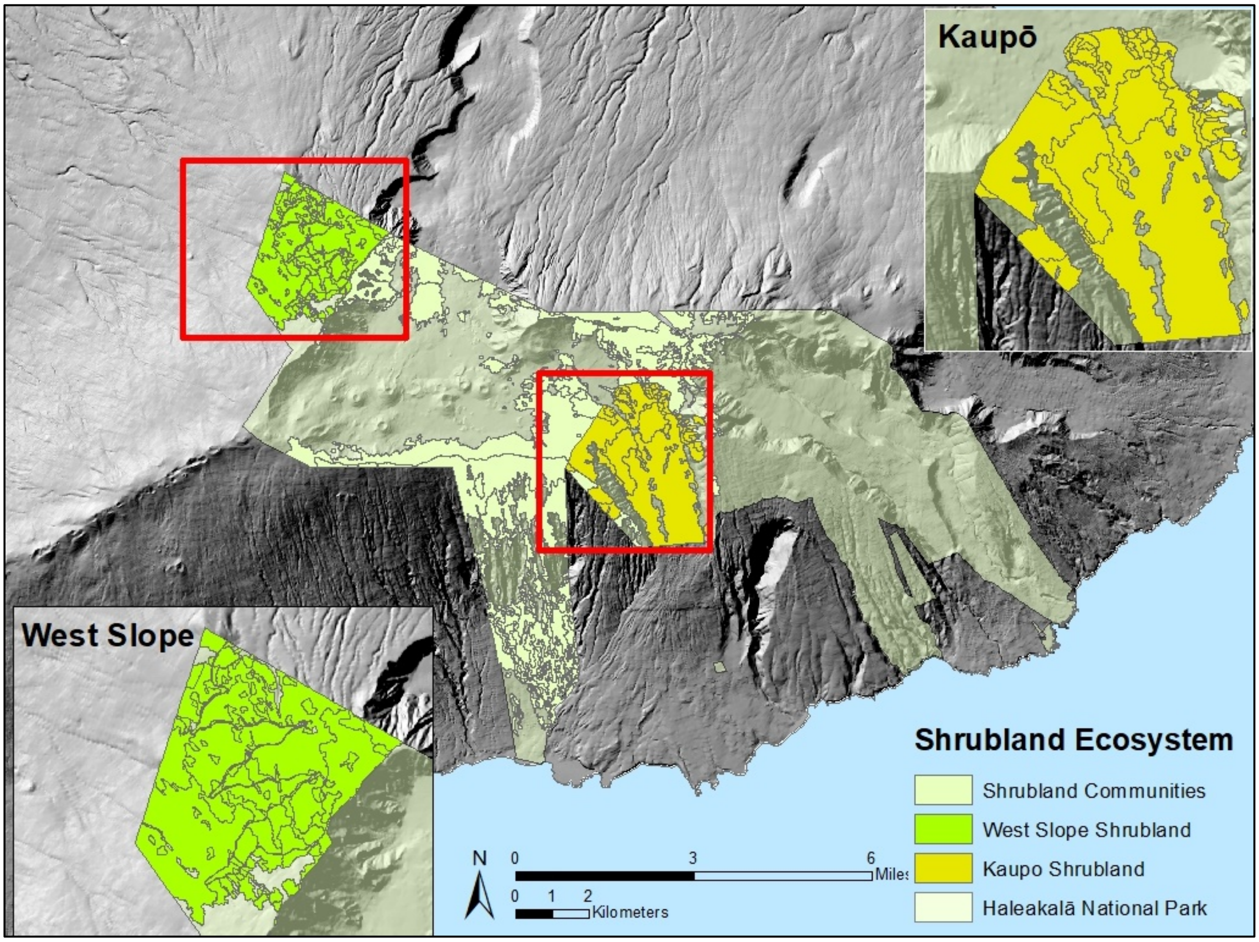

Figure 4.2.6-2. All shrubland ecosystems located within HALE. West slope and Kaupō boundaries delineated by NPS designated management units. (Source: Green et al. 2015).

Leptecophylla tameiameiae (pūkiawe), a coriaceous, small-leaved shrub, is the most common native species of the subalpine shrubland ecosystem, and *Sophora chrysophylla* (māmane), a seasonal nectar source for native honeycreepers, is the tallest native tree-shrub (Medeiros et al. 1998, Figure 4.2.6-3). The native shrubs *Vaccinium reticulatum* (ʻōhelo) and *Dubautia menziesii* (kūpaoa) are also relatively abundant in the subalpine shrublands. Other common and characteristic native subalpine species include the shrubs *Coprosma montana* (pilo), *Coprosma ernodeoides* (kūkaenēnē), *Geranium cuneatum* subsp. *tridens* (hinahina), and *Dodonaea viscosa* (ʻaʻaliʻi), and the herbs *Carex wahuensis*, *Deschampsia nubigena*, and *Morelotia gahniiformis* (ʻuki). *Santalum haleakalae* (ʻiliahi), an uncommon tree usually found in small groves, is largely confined to the subalpine shrublands of Maui. Several endangered plant taxa occur in HALE shrublands: *Bidens micrantha* subsp. *kalealaha*, *Geranium arboreum*, *Geranium multiflorum*, *Plantago princeps* var. *laxiflora*, *Sanicula sandwicensis*, and *Schiedea haleakalensis* (see also chapter 4.2.10).

Figure 4.2.6-3. *Leptecophylla tameiameiae* (left) and *Sophora chrysophylla* (right), common components of Haleakalā's subalpine shrubland ecosystem. (Source: Forest and Kim Starr).

Shrubland Birds

Forest birds, including ʻapapane (*Himatione sanguinea*), Maui ʻamakihi (*Chlorodrepanis virens wilsoni*), ʻiʻiwi (*Drepanis coccinea*), and Maui ʻalauahio (Maui creeper, *Paroreomyza montana newtoni*), forage in shrublands where they adjoin forests (Medeiros et al. 1998), but are typically uncommon or absent in shrublands located more distant from forest edges at HALE. Conant and Stemmerman (1979) found that abundance of those four species of native forest birds was lower in closed shrub habitat than in forest habitat. Shrublands are an important habitat for the endangered nēnē (Hawaiian goose, *Branta sandvicensis*), however, which commonly nests and forages in dense and open shrubland ecosystems (Conant and Stemmerman 1979, NPS 2012). The pueo (Hawaiian short-eared owl, *Asio flammeus sandwichensis*) sometimes hunts over shrublands, but is more numerous and nests primarily in grasslands (Conant and Stemmerman 1979). The migratory kōlea (Pacific golden plover, *Pluvialis fulva*), uses shrublands primarily during the fall to spring non-breeding seasons, but a few may stay through the summer (Conant and Stemmerman 1979). Two introduced gamebirds, the ring-necked pheasant (*Phasianus colchicus*) and chukar (*Alectoris chukar*), and a number of non-native passerine bird species, also reside in HALE shrublands (Conant and Stemmerman 1979, Cole et al. 1995a, b). The endangered Hawaiian hoary bat or ʻōpeʻapeʻa (*Lasiurus cinereus semotus*) has also been detected in HALE shrublands, but the nature of its use or reliance on this ecosystem is currently unknown (see chapter 4.2.10).

Shrubland Invertebrates

Shrublands support diverse invertebrate communities, predominantly arthropods (insects and their relatives, such as spiders, mites, millipedes and centipedes). Although less speciose than forest arthropod communities, at HALE they are much better studied, particularly on the more accessible western slopes, which were visited by entomologists as early as 1878 (Krushelnycky et al. 2005a). Collecting over the following century revealed that a fairly high proportion (21%) of the insect species residing in the high-elevation shrubland, alpine and crater ecosystems at HALE are endemic to Haleakalā volcano (Beardsley 1980). A number of highly unique and unusual forms have evolved in these ecosystems, including flightless species of moths (*Thyrocopa apatela*), lacewings (e.g., *Micromus cookeorum*), ground beetles (family Carabidae) and long-legged hunting flies (family Dolichopodidae) (Loope and Medeiros 1994, Medeiros and Loope 1994). Haleakalā volcano is also the center of diversity for a massive radiation (239 species) of *Mecyclothorax* ground beetles, and HALE shrublands harbor an impressive suite of these beetles (Liebherr 2015). However, non-native arthropods have invaded these communities to a considerable degree (Beardsley 1980, Krushelnycky et al. 2007), with two of the most damaging species being the Argentine ant (*Linepithema humile*) and western yellowjacket (*Vespula pensylvanica*) (Cole et al. 1992, Gambino and Loope 1992). Efforts to manage both species of social insects have been undertaken by the park.

Impacts to Shrublands

Subalpine shrublands had also been heavily modified by the impacts of feral goats, cattle, and pigs over the past few centuries (Loope et al. 1992a). Ungulate browsing, grazing and trampling created disturbance that facilitated the invasion of this ecosystem by dense mats of non-native grasses which in turn have inhibited recruitment of native plants. To mitigate this problem, fencing of the West Slope (often referred to as "frontcountry") area of the park was completed in 1991, and apart from periodic incursions, feral ungulates have been largely eliminated from the area (Gross et al. 2017). A rapidly increasing population of European rabbits (*Oryctolagus cuniculus*), originating from six cage-reared rabbits released near Hosmer Grove by a pet owner in October 1989, was recognized as a significant threat to subalpine shrublands and other vegetation communities (Loope 1992, Loope et al. 1992b). Upon their discovery in 1990, control efforts were quickly initiated and ultimately resulted in the removal of 93 rabbits from a 25-ha area of high-elevation (2,075 - 2,135 m) native shrubland between August 1990 to March 1991(Loope 1992, Loope et al. 1992b). Although eradication of this population was successful, the potential threat of future release or escape of rabbits remains due to their ready availability in the pet trade (Loope 1992). Despite these management successes, construction, military operations, vehicular traffic, experimental tree plantings, and proximity to agricultural lands have contributed to the continued invasion of this zone by non-native plants (Gross et al. 2017). Pines (*Pinus* spp.), eucalypts (*Eucalyptus* spp.), gorse (*Ulex europaeus*), and blackberry (*Rubus argutus*) have been identified as serious invaders of subalpine shrublands (Loope et al. 1992a), and are among the species being managed by park staff throughout the area.

Kaupō shrubland and associated native plant communities have also been heavily impacted by feral ungulate damage and the subsequent invasion by habitat-modifying weeds. In 1984, park crews completed the Kaupō Gap boundary fence, initiating the recovery of resilient native shrubland species (NPS 1999, Figure 4.2.6-4). Nevertheless, establishment of smothering non-native grasses such as kikuyu grass (*Cenchrus clandestinus*), flammable, fire-promoting grasses such as molasses grass (*Melinis minutiflora*) and thicket-forming shrubs and trees such as silk oak (*Grevillea robusta*), castor bean (*Ricinus communis*) and Christmas berry (*Schinus terebinthifolius*), among others, have impeded natural regeneration in areas and pose a risk to the continued health of the native communities (Loope et al. 1992a). Park vegetation management staff have therefore targeted these and other invasive weeds for continued suppression and control within two Kaupō management units (NPS 1999).

Figure 4.2.6-3. Kaupō Gap boundary fence demonstrating the stark contrast between native-dominated shrubland (right), and non-native grasslands and ungulate-damaged vegetation (left). (Source: C. Chimera).

An additional stressor on shrubland ecosystems is changing climate conditions. While this has the potential to affect all ecosystems at HALE, the clearest evidence for changes already in progress pertains to ecosystems near the top of the mountain above the TWI, including subalpine shrublands. The frequency of occurrence of the TWI appears to have increased in Hawaiʻi in recent decades (Longman et al. 2015a), and this has led to drier, sunnier conditions above its base height on Haleakalā (Longman et al. 2015b). Combined with increasing temperatures over the previous century, which have been warming fastest in recent decades and at higher elevations (Giambelluca et al. 2008), these conditions may now be leading to more frequent or severe water stress for shrubland plants.

Although shrublands occur across much of the top of Haleakalā volcano, the largest continuous blocks of shrubland habitat in the park are in the West Slope frontcountry and in Kaupō Gap (Figure 4.2.6-2). We therefore focus our assessment on these two regions, using three indicators: West Slope vegetation, Kaupō vegetation, and invertebrates. We do not assess native birds, because forest birds are a minor component of these shrublands, and nēnē are addressed primarily in the T&E Species section (Chapter 4.2.10).

Indicators

- West Slope vegetation
- Kaupō vegetation
- Invertebrates

Data and Methods

A review of available data and information was used to assess conditions, and if possible, trends, in the indicators listed above. Some additional analyses were performed on a subset of geospatial and vegetation classification and monitoring data collected by HALE and PACN staff specific to the shrubland ecosystem subcategories of the West Slope and Kaupō (Green et al. 2015, PACN and Ainsworth 2016). Ecosystem-based summaries of invertebrate communities were extracted from data on the entire Summit District presented in Beardsley (1980). Summary analyses were also performed on HALE yellowjacket monitoring data (NPS 2017c).

Sources of information

Monitoring, outplanting, and propagule collection data for common, rare, and endangered native taxa have been recorded by HALE Vegetation Management staff for the entire park from 1993 through May 2017 (NPS 2017a).

Control data for invasive, non-native plants have been collected by HALE Vegetation Management staff for the West Slope from 1982-2017 and for Kaupō from 1983 through May 2017 (NPS 2017b).

NPS I&M vegetation classification data were collected by HALE and PACN staff in 220 circular plots (400 m^2) and observation points in 2011 (Green et al. 2015). NPS I&M also monitored 30 subalpine shrubland plots (20 x 50 m) in 2012-2013 as part of the Focal Terrestrial Plant Community monitoring system (Ainsworth et al. 2011, PACN and Ainsworth 2016), and recorded non-native plant frequency and abundance data along belt transects located in the subalpine shrublands of the West Slope and Crater by HALE and PACN staff (Gross et al. 2017).

Data on invertebrate diversity and distributions, and status of non-native species, were obtained from surveys of arthropods across the Summit District in 1975-77 (Beardsley 1980), and in the West Slope shrubland in 2002-2004 (Krushelnycky et al. 2007).

Information on rates of spread and ecological effects of the Argentine ant were taken from Cole et al. (1992), Krushelnycky et al. (2005b, 2011) and Krushelnycky and Gillespie (2008, 2010). Information on ecological effects of the western yellowjacket was taken from Gambino and Loope (1992) and Wilson et al. (2009), and unpublished data on monitoring and control of yellowjackets (NPS 2017c) was provided by HALE staff.

Information on ecological effects of non-native gamebirds and rodents was taken from Cole et al. (1995a, b) and Cole et al. (2000).

Reference condition

Ideal reference conditions for shrubland ecosystems would include fully intact native vegetation, uninvaded by non-native weeds, in both West Slope and Kaupō regions. Similarly, invertebrate communities would be highly diverse in species and function, and uninvaded by non-native invertebrate species and unaffected by non-native vertebrates. Due to direct and indirect anthropogenic impacts, including the introduction of non-native plants and animals to the Hawaiian Islands, no shrubland ecosystem currently exists in an entirely pristine state. Nevertheless, examples of relatively high-quality Hawaiian shrublands, in which native plant species richness and cover exceeds that of non-natives, can still be found, particularly at higher elevations and remote locations throughout the islands (e.g. Hughes et al. 2014). A desirable reference condition for HALE's shrubland vegetation is therefore that the richness and cover of native species exceeds that of non-native species. A desirable reference condition for shrubland invertebrate communities is that native species richness exceeds non-native species richness, and that highly damaging invasive species like social insects are absent.

Condition and Trend

West Slope vegetation

Green et al. (2015) have classified six vegetation community types as native shrublands in the West Slope area of HALE, totaling 850.5 ha and almost 80% of the cover for the entire region (Figure 4.2.6-5, Table 4.2.6-1). The native shrub *Leptecophylla tameiameiae* is the most common and dominant species in five of the six shrubland communities, including the largest community by area on the West Slope, *Leptecophylla tameiameiae* - *Vaccinium reticulatum* Subalpine Mesic Shrubland (Table 4.2.6-1). The sixth shrubland community, and third largest by area on the West Slope, is *Sophora chrysophylla* - (*Coprosma montana* - *Leptecophylla tameiameiae* - *Dodonaea viscosa*) Subalpine Shrubland (Table 4.2.6-1). This community is characterized by an open to moderately dense tall-shrub (2-4 m) canopy with 10-50% cover of *S. chrysophylla* and often with the shrub *C. montana* present to codominant (Green et al. 2015). The remainder of the West Slope cover consists of native vegetation (i.e. non-shrubland), non-native dominated communities, unvegetated areas and built-up sites (Figure 4.2.6-5 and Table 4.2.6-1).

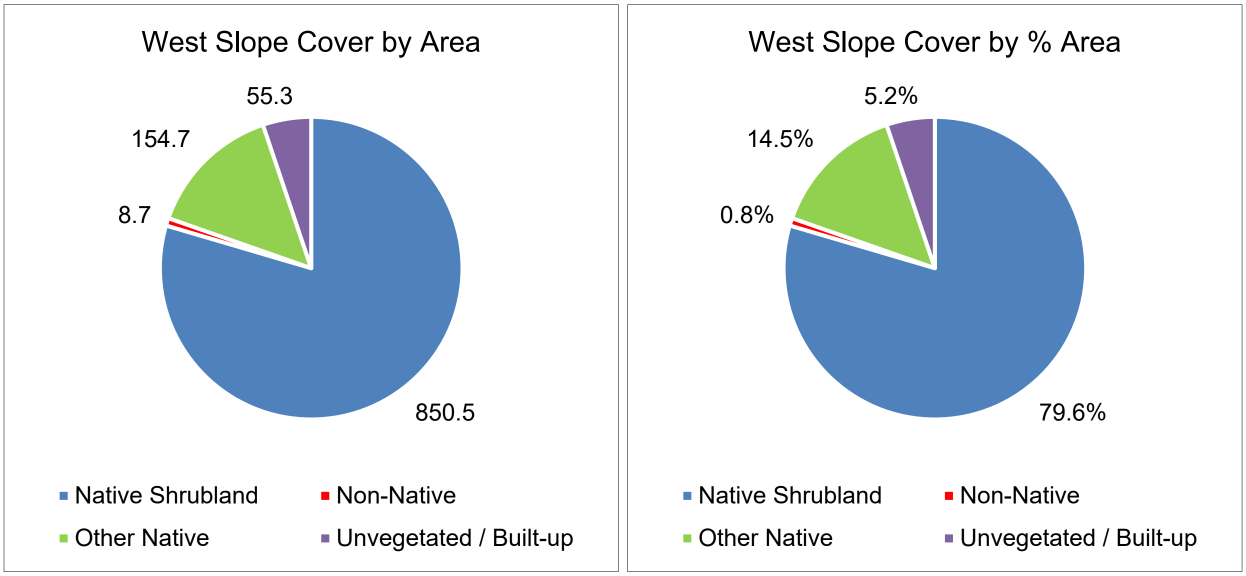

Figure 4.2.6-5. Area (hectares) and percent cover of shrubland and other vegetation within HALE West Slope (i.e. Frontcountry) management units. (Source: Green et al. 2015).

Table 4.2.6-1. Shrubland community types located within HALE West Slope (i.e. Frontcountry) management units, ranked by total area. Source: Green et al. (2015).

Community Classification	**Area (ha)**
Leptecophylla tameiameiae - *Vaccinium reticulatum* Subalpine Mesic Shrubland	317.9
L. tameiameiae Subalpine Dry Shrubland	219.4
Sophora chrysophylla - (*Coprosma montana* - *L. tameiameiae* - *Dodonaea viscosa*) Subalpine Shrubland	182.8
L. tameiameiae - *V. reticulatum* - (*Geranium cuneatum*) / *Deschampsia nubigena* Mesic Shrubland	112.6
L. tameiameiae - *Sadleria cyatheoides* Subalpine Mesic Shrubland	10.6
L. tameiameiae - (*D. viscosa*) / *Cenchrus clandestinus* Montane - Subalpine Mesic Semi-natural Shrubland	7.2

Although not a comprehensive list of all plant taxa found in the West Slope shrublands of HALE, Green et al. (2015) provide a broad sampling of species through a series of 16 vegetation inventory and mapping plots. A total of 26 native and 25 non-native plant species were recorded in these plots, with a mean species richness of 10.81 ± 0.97 natives and 4.75 ± 0.69 non-natives per plot (Figure 4.2.6-6). Native species richness was also higher than non-native richness in all inventory and mapping plots located on the West Slope, which included three non-shrubland plots (n = 19, Figure 4.2.6-6). Mean native species cover was 52.86% ± 5.30%, more than double the 22.81% ± 5.63% mean non-native species cover (Figure 4.2.6-6). Native species cover was also higher than non-native cover when considering all monitoring plots located on the West Slope (n = 19, Figure 4.2.6-6).

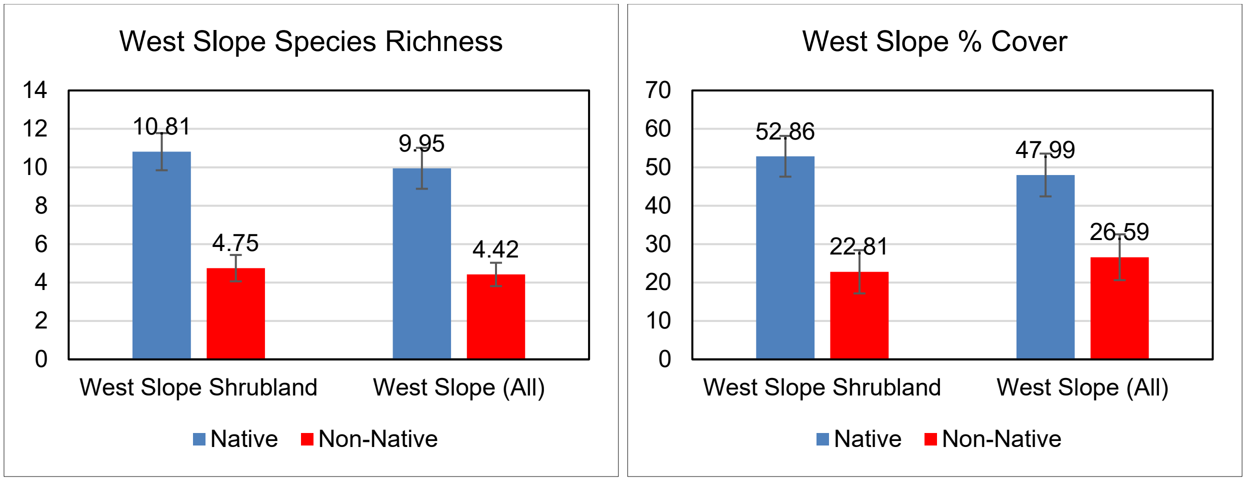

Figure 4.2.6-6. Species richness and percent cover in West Slope (n = 16) shrubland plots, and all West Slope vegetation plots (n = 19). Source: Green et al. (2015).

A similar pattern is evident from the first round of monitoring in the NPS I&M Focal Terrestrial Plant Community (FTPC) subalpine shrubland plots, eight of which were located within West Slope shrublands (PACN and Ainsworth 2016). These recorded a mean species richness of 12.12 ± 0.55 natives and 7.75 ± 1.19 non-natives per plot (Figure 4.2.6-7). Mean native cover in the FTPC plots was 56.88% ± 5.93%, compared to 30.88% ± 7.42% cover for non-natives species (Figure 4.2.6-7).

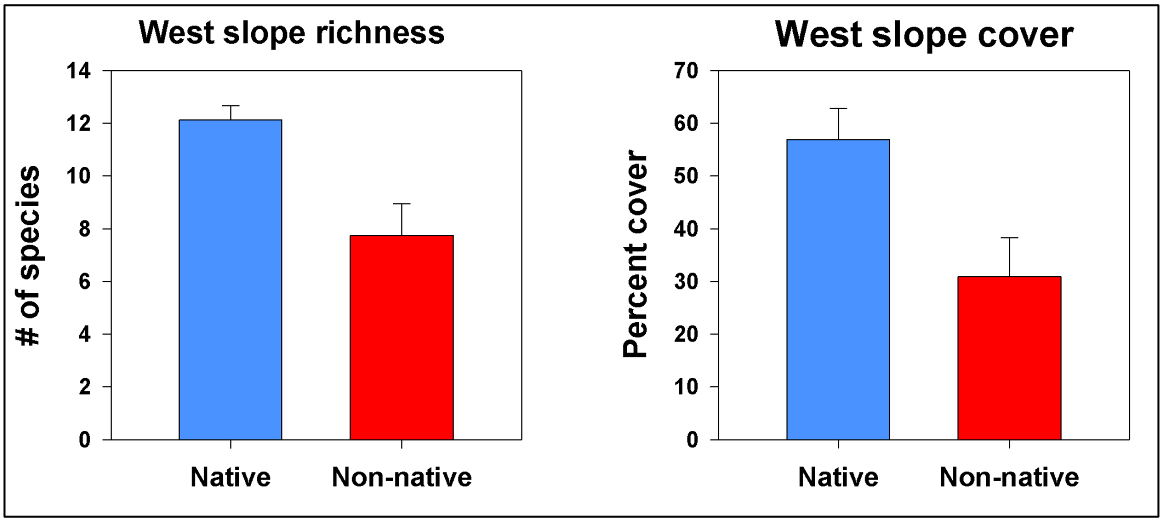

Figure 4.2.6-7. Mean (±SE) native and non-native plant species richness (left) and percent cover (right) recorded within 1,000 m^2 FTPC monitoring plots in West Slope shrublands (n = 8). Source: PACN and Ainsworth (2016).

Data collected in NPS I&M efforts to monitor the status of Established Invasive Plant Species (EIPS) at HALE suggest a slightly different pattern. Four 500 m long belt transects (consisting of 25 contiguous 5 x 20 m plots) were established within West Slope shrubland, and baseline monitoring of non-native plants along the transects was completed in 2012 (Gross et al. 2017). On two of the four transects, mean cover of non-native plants was between 1% and 5%, and on the remaining two transects was between 5% and 10%, and between 10% and 25%, respectively. If one assigns a midpoint value in each cover class range, the four transects averaged 7.75% cover of non-native plants overall, which is considerably lower than 22.81% and 30.88% cover measured in the inventory and FTPC plots, respectively (Figures 4.2.6-6 and 4.2.6-7). The total number of non-native plant species per EIPS transect ranged from 9 to 15, and averaged 12.2 species across the four transects (Gross et al. 2017). This, in contrast, is higher than the average 4.75 non-native species recorded in the inventory plots and the average 7.75 non-native species recorded in the FTPC plots, although a higher richness on the transects would be expected since each transect covers 2,500 m^2, compared to only 400 m^2 and 1,000 m^2 covered in the inventory and FTPC plots, respectively.

In spite of the differences in these values, all three sets of plots or transects suggest that the West Slope shrubland is dominated by native plants. This is particularly true of the shrub canopy layer, which is essentially composed entirely of native species. Several problematic invasive plants, however, threaten to change this situation. Medeiros et al. (1998) identified *Cortaderia jubata* (Andean pampas grass), *Eucalyptus globulus* (blue gum), *Pinus radiata* (Monterey pine), *Pinus pinaster* (maritime pine), *Pinus patula* (Mexican weeping pine), and *U. europaeus* (gorse) as the most serious weeds of the subalpine zone. Of the *Eucalyptus* species in the park, *E. globulus* is the most invasive due to its ability to establish seedlings, even at a distance from parent trees, and within otherwise undisturbed native-dominated habitat. This and other *Eucalyptus* species planted at Hosmer Grove and at 8,500 ft elevation are a source of new propagules and continued invasion into adjacent shrubland communities (Medeiros et al. 1998). Pines, particularly *P. radiata*, *P. patula* and *P. pinaster*, have become significant invaders within the subalpine shrubland and Crater ecosystems. These trees possess wind-dispersed seeds capable of spreading long distances from parents, can establish in and overtop shorter-statured native communities, and can eventually convert them into fire-prone coniferous monocultures (Burns and Honkala 1990, Loope et al. 1992a). *Ulex europaeus* (gorse) is a spiny, flammable shrub that can invade subalpine shrubland habitat and form dense, impenetrable thickets that exclude all other vegetation and impede movement. Due to the persistence of its long-lived seeds, continual control measures will be required in perpetuity within previously invaded areas (Loope et al. 1992a).

Another important non-native plant is *Cenchrus clandestinus* (kikuyu grass), an aggressive mat-forming perennial grass that was first reported as a problem species in the park in 1939, damaging highway paving on the West Slope (Medeiros et al. 1998). Although not common in undisturbed areas of the West Slope, it can smother low growing vegetation in areas where it occurs, and can suppress native seedling recruitment, possibly due to its reported allelopathic properties (Smith 1985). *Holcus lanatus* (velvetgrass) is also problematic in this region, and was recorded as high as 25-50% cover in some FTPC plots and EIPS transect subplots (Gross et al. 2017). *Rubus argutus* (Florida blackberry) is an invasive shrub capable of spreading by rhizomes, aerial runner branches, and bird-dispersed seeds, and can establish thorny thickets within native shrublands, grasslands and forest communities (Loope et al. 1992a, Medeiros et al. 1998). Finally, *Senecio madagascariensis* (fireweed) is an invasive annual or short-lived perennial herb that can spread long distances via its prolific wind-dispersed seeds and establish and form dense cover in subalpine and alpine habitat

(Ramadan et al. 2011). Large infestations outside the park boundary provide a perpetual seed source, and this weed is becoming increasingly common not only in disturbed areas like the mule pasture, but also throughout the shrublands.

Pines (*Pinus* spp.), *Eucalyptus* species, and gorse were prioritized for management and control in the park's Resource Management Plan (NPS 1999). These and other invasive plants of subalpine shrubland ecosystems, including those summarized above, have been actively targeted by park vegetation management staff since 1982 (Table 4.2.6-2). These species have demonstrated an ability to persist and continue to spread in the absence of ungulate disturbance, and will need to be managed in perpetuity to maintain the native-dominated composition and ecological integrity of this community type.

In addition to controlling invasive weeds, HALE Vegetation Management Unit staff have augmented natural recovery of West Slope shrubland vegetation with outplanting of both rare and common plants, including all of the extant threatened and endangered species known to occur (Table 4.2.6-3).

Table 4.2.6-2. Numbers of high priority invasive weeds controlled on the West Slope (i.e. Frontcountry) of HALE. Source: NPS (2017b).

Taxon	Total Controlled
Cortaderia jubata (1998-2015)	28
Eucalyptus spp. (1992-2017)	23,225
All Pine species (1982-2017)	118,788
Pinus patula (1982-2017)	109,970
Pinus radiata (2001-2017)	8,187
Rubus argutus (1996-2017)	82,316 canes
Senecio madagascariensis (2006-2017)	58,662
Ulex europaeus (1982-2016)	98,562
All Non-Native Plant Species (1982- 2017)	663,605 individuals

Table 4.2.6-3. Numbers of native plant taxa outplanted and seeds sown in West Slope (i.e. Frontcountry) shrublands in HALE. Source: NPS (2017a).

Taxon	Plants	Seeds	T&E
Acacia koa	8	0	–
Argyroxiphium grayanum	24	0	NA
Argyroxiphium sandwicense ssp. *macrocephalum*	146	0	T
A. sandwicense ssp. *macrocephalum X Dubautia menziesii*	30	0	NA
Artemesia mauiensis	10	0	NA

Table 4.2.6-3 (continued). Numbers of native plant taxa outplanted and seeds sown in West Slope (i.e. Frontcountry) shrublands in HALE. Source: NPS (2017a).

Taxon	**Plants**	**Seeds**	**T&E**
Bidens micranthra ssp.*kalealaha*	38	0	E
Carex wahuensis	4	0	NA
Coprosma ernodeoides	237	0	NA
Coprosma montana	424	2,664	NA
Cyperus javanicus	5	0	NA
Deschampsia nubigena	1,481	3,294,500	NA
Dodonaea viscosa	640	33,180	NA
Dryopteris wallichiana	33	0	NA
Dubautia menziesii	230	124,200	NA
Fragaria chiloensis subsp.sandwicensis	21	0	NA
Geranium arboreum	343	0	E
Geranium cuneatum subsp. tridens	88	0	NA
Geranium multiflorum	2	0	E
Leptocophylla tameiameiae	8	0	NA
Luzula hawaiiensis var. *hawaiiensis*	36	0	NA
Lysimachia remyi	6	0	NA
Metrosideros polymorpha	33	0	NA
Osteomeles anthyllidifolia	5	0	NA
Pittosporum confertiflorum	30	0	NA
Plantago princeps var*. laxiflora*	55	0	E
Pseudognaphalium sandwicensium	1	0	NA
Pteridium aquilinum ssp. decompositum	30	0	NA
Rubus hawaiensis	291	0	NA
Rubus macraei	7	0	NA
Santalum haleakalae	3	0	NA
Sanicula sandwicensis	86	100	E
Schiedea haleakalaensis	133	0	E
Sisyrinchium acre	4	0	NA
Sophora chrysophylla	804	4,700	NA
Stenogyne microphylla	64	0	–
Tetramolopium humile var. haleakalae	19	0	NA
Trisetum glomeratum	70	0	NA
Vaccinium reticulatum	78	0	NA
Viola chamissoniana ssp.*tracheliifolia*	495	0	NA
Total	6,022	3,459,344	–

Vertebrate Impacts

Smaller, non-ungulate vertebrates also have potential effects on shrubland vegetation. Nēnē consume a wide variety of native and non-native plants, including grasses and sedges, herbs, and fruits of shrubs (Baldwin 1947, Black et al. 1998). Seeds of *L. tameiameiae*, *V. reticulatum*, and *C. ernodeoides* were found to be common in their droppings on Hawaiʻi Island (Baldwin 1947), suggesting that they are active dispersers of these native shrubs. However, nēnē are currently present at low densities in the West Slope shrubland (NPS 2012, Figure 4.2.10-8), and therefore do not likely provide significant seed dispersal services there. In contrast, two introduced gamebird species, ring-necked pheasants and chukars, are abundant in West Slope shrublands. For example, they were estimated to occur at average densities ranging from approximately 18 to 95 pheasants, and 6 to 161 chukars, per 100 hectares in 1985-1986 (Cole et al. 1995a). Like nēnē, both birds consume a wide variety of vegetation, including fruits of the native shrubs *L. tameiameiae*, *V. reticulatum*, *C. ernodeoides*, and *C. montana* (Cole et al. 1995a). These authors also found that six native and seven non-native plant species germinated from droppings of pheasants and chukars, with native germinants outnumbering non-native germinants by a ratio of nearly 5 to 1. The most common native plants germinating from droppings were *L. tameiameiae*, *V. reticulatum*, and *C. ernodeoides*, and Cole et al. (1995a, b) concluded that these introduced gamebirds may now be filling the beneficial role of native seed dispersers that was formerly occupied by rare or extinct species, such as flightless ducks and geese. Non-native rodents in West Slope shrublands (mice, *Mus musculus*, and black rats, *Rattus rattus*) were also found to feed extensively on plant material, at least during certain seasons, with dicot fruits being especially important for rats (Cole et al. 2000). Both rodents fed heavily on fruits of *Vaccinium* and *Coprosma*. Although no seeds in rodent droppings germinated, suggesting little effective seed dispersal, Cole et al. (2000) felt that probably only several relatively rare and highly preferred plant species, like *Santalum haleakalae* and *Pittosporum confertiflorum*, are likely experiencing population-level impacts from rodent seed predation in these shrublands. Predation of native pollinators by invasive Argentine ants and western yellowjackets in West Slope shrublands has the potential to impact reproduction of certain insect-pollinated plant species, such as *D. menziesii, G. cuneatum, and L. tameiameiae*, among others. We discuss this further in the invertebrate section below.

Climate Change

Finally, as stated earlier, climate conditions have become warmer, sunnier, and drier in areas above the TWI over the past few decades, possibly caused by an upward shift in TWI prevalence around 1990 (Longman et al. 2015a,b). These drying conditions have been linked to substantial population declines of at least one species, the Haleakalā ʻāhinahina or silversword (*Argyroxiphium sandwicense macrocephalum*) in the crater ecosystem (Krushelnycky et al. 2013, 2016), but there has been no systematic long-term monitoring of plant populations in the West Slope area that would indicate whether any shrubland plant species have been similarly affected. However, significant mortality of *L. tameiameiae* shrubs was observed on the West Slope during the summer to fall of 2010, most obviously in an elevational band around 8,500 ft (2,591 m) elevation. This dieback event appeared to be associated with unusually warm and dry atmospheric conditions throughout the 2010 dry season, and exhibited similarities to a mortality event observed in high-elevation *L. tameiameiae*, *V. reticulatum*, and *C. ernodeoides* shrubs during an El Niño associated drought in 1992 on Hawaiʻi

Island (Lohse et al. 1995). Although this 2010 event at HALE was not otherwise documented or quantified, it may represent an anecdotal example of how shrubland and other high-elevation plant communities may become vulnerable to conditions that further intensify water deficit. Such conditions also increase the risk of fire, which pre-historically was infrequent on upper Haleakalā but more common during periods of drought (Burney et al. 1995, Crausbay et al. 2014). Many native Hawaiian plants are believed to be ill-adapted to frequent fire (Smith and Tunison 1992), and fire is likely to inhibit regeneration of native shrubland species, especially in the presence of fast-growing, fire-tolerant non-native plants (Loope et al. 1992a). A small 0.6 ac (0.2 ha) fire in the West Slope shrubland in 1992 resulted in a 400% increase in cover of non-native grasses relative to adjacent unburned areas six years after the fire; the native shrubs *L. tameiameiae* and *V. reticulatum* were slow to recover (LaRosa et al. 2008).

Summary of West Slope Shrublands

Despite the threats from invasive plants, animals and changing climate, the West Slope shrubland vegetation is judged to currently be in good condition. Although no repeated vegetation data sets have been collected within HALE shrubland ecosystems, Hughes et al. (2014) documented changes in comparable subalpine shrubland communities of the adjacent Waikamoi Preserve between 1994 and 2008 (14 years after ungulate removal), and reported dramatic increases in native cover, and decreases in non-native grass cover. Similar responses to ungulate exclusion from the West Slope shrubland communities are thought to have occurred, and are reflected in the much higher native than non-native species richness and cover in this area of the park. Recovery of these West Slope vegetation communities can also be inferred by the substantially higher plant diversity apparent when visually comparing vegetation inside the park with that immediately outside the boundary fence. The native-dominated status in these areas has further been maintained and augmented through a combination of native outplanting and restoration activities, and targeted control of a select group of potentially habitat-modifying invasive plants. For these reasons, we assign an improving trend to the condition of West Slope vegetation. Confidence in this assessment is medium, owing to the lack of repeated data to confirm these apparent trends.

Kaupō vegetation

Green et al. (2015) have classified eleven vegetation community types as native shrublands in the Kaupō region of HALE, totaling 950.9 ha and approximately 74% of the cover for the entire region (Figure 4.2.6-8 and Table 4.2.6-4). The native shrub *Leptecophylla tameiameiae* is the most common and dominant species in seven of the shrubland communities, including the largest community by area in Kaupō, *Leptecophylla tameiameiae* - *Dodonaea viscosa* Montane Shrubland (Table 4.2.6-4). One native shrubland community (*Leptecophylla tameiameiae* - (*Dodonaea viscosa*) / *Cenchrus clandestinus* Montane - Subalpine Mesic Semi-natural Shrubland) is characterized by an open to moderately dense cover of native shrubs, but with an herbaceous layer strongly dominated by the invasive non-native grass, *Cenchrus clandestinus* (Green et al. 2015). A twelfth shrubland community (Unclassified Polynesian Semi-natural Montane and Subalpine Shrubland and Grassland), occurs on the western side of Kaupō Gap, where the steep terrain has prevented a more accurate community classification (Green et al. 2015).

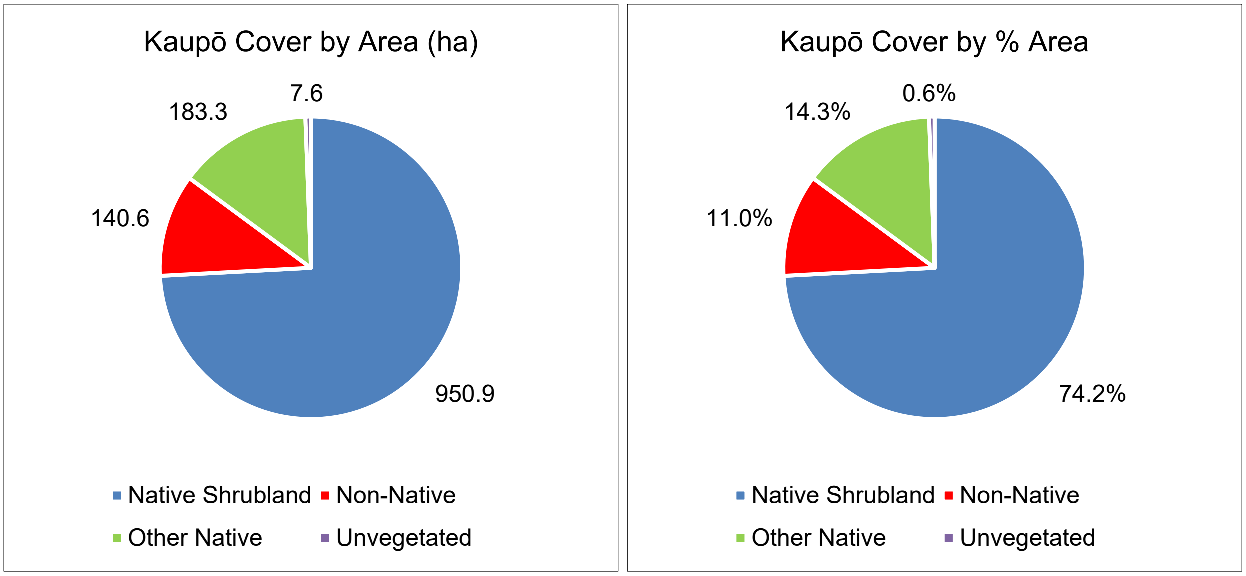

Figure 4.2.6-8. Area (hectares) and percent cover of shrubland and other vegetation within HALE Kaupō management units. Small unvegetated pie slices not visible, but represented numerically. Source: Green et al. (2015).

Table 4.2.6-1. Shrubland community types located within HALE Kaupō management units, ranked by total area. Source: Green et al. (2015).

Community Classification	**Area (ha)**
Leptecophylla tameiameiae - Dodonaea viscosa Montane Shrubland	560.6
L. tameiameiae Subalpine Dry Shrubland	153.1
L. tameiameiae Montane Mesic Shrubland	113.6
L. tameiameiae - Vaccinium reticulatum Subalpine Mesic Shrubland	86.8
Unclassified Polynesian Semi-natural Montane and Subalpine Shrubland and Grassland	36.1
Sophora chrysophylla - (*Coprosma montana - L. tameiameiae - D. viscosa*) Subalpine Shrubland	13.6
D. viscosa Montane Mesic Shrubland	9.0
L. tameiameiae - Sadleria cyatheoides Subalpine Mesic Shrubland	5.6
Rubus hawaiensis Montane Wet Shrubland	4.8
L. tameiameiae - (*D. viscosa*) / *Cenchrus clandestinus* Montane - Subalpine Mesic Semi-natural Shrubland	2.8
L. tameiameiae - V. reticulatum - (*Geranium cuneatum*) / *Deschampsia nubigena* Mesic Shrubland	1.0

Although not a comprehensive list of all plant taxa found in the Kaupō shrublands, Green et al. (2015) provide a broad sampling of species through a series of 15 vegetation inventory and mapping plots. A total of 23 native and 17 non-native plant species were recorded in these plots, with a mean species richness of 11.33 ± 1.21 natives and 4.67 ± 0.47 non-natives per plot (Figure 4.2.6-9). Native species richness was also higher than non-native richness in all inventory and mapping plots located in the Kaupō management units, which included 15 non-shrubland plots (n = 30, Figure 4.2.6-9). Mean native species cover was 47.34% ± 9.11%, almost five times greater than the 9.65% ± 6.15%

mean non-native species cover (Figure 4.2.6-9). Native species cover was also higher than non-native cover when considering all plots located in the Kaupō management units (n = 30, Figure 4.2.6-9).

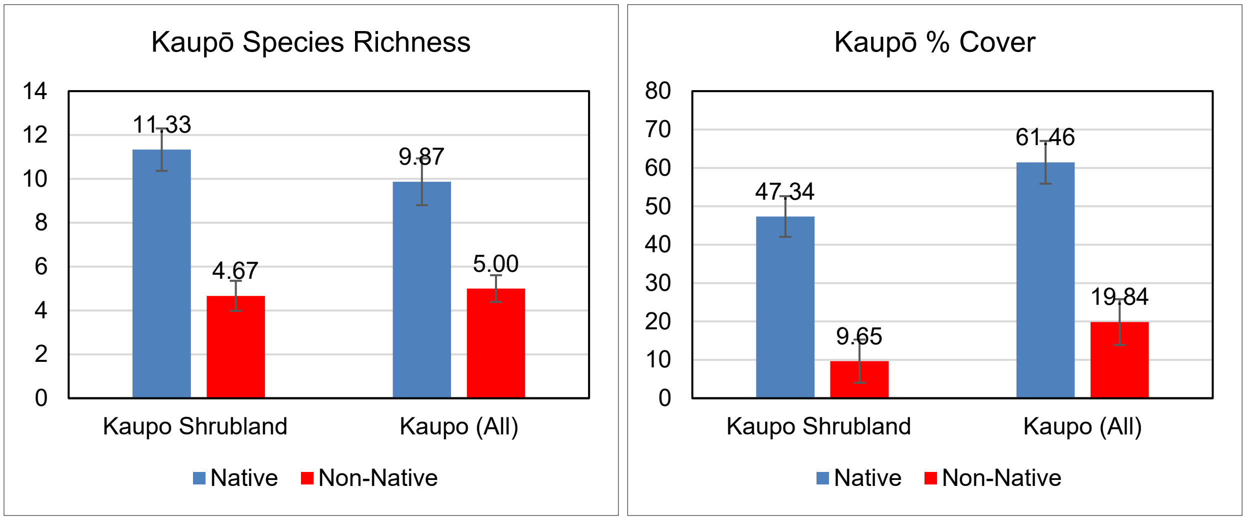

Figure 4.2.6-9. Species richness and percent cover in Kaupō (n = 15) shrubland inventory and mapping plots, and all Kaupō inventory and mapping plots (n = 30). Source: Green et al. (2015).

A similar pattern is evident from the first round of monitoring in the NPS I&M FTPC plots, two of which were located within Kaupō shrublands (PACN and Ainsworth 2016). These recorded a mean species richness of 12.00 ± 0.00 natives and 5.00 ± 0.00 non-natives per plot (Figure 4.2.6-10). Mean native cover in the FTPC plots was 58.00% ± 17.00%, compared to 16.50% ± 12.50% cover for non-natives species (Figure 4.2.6-10).

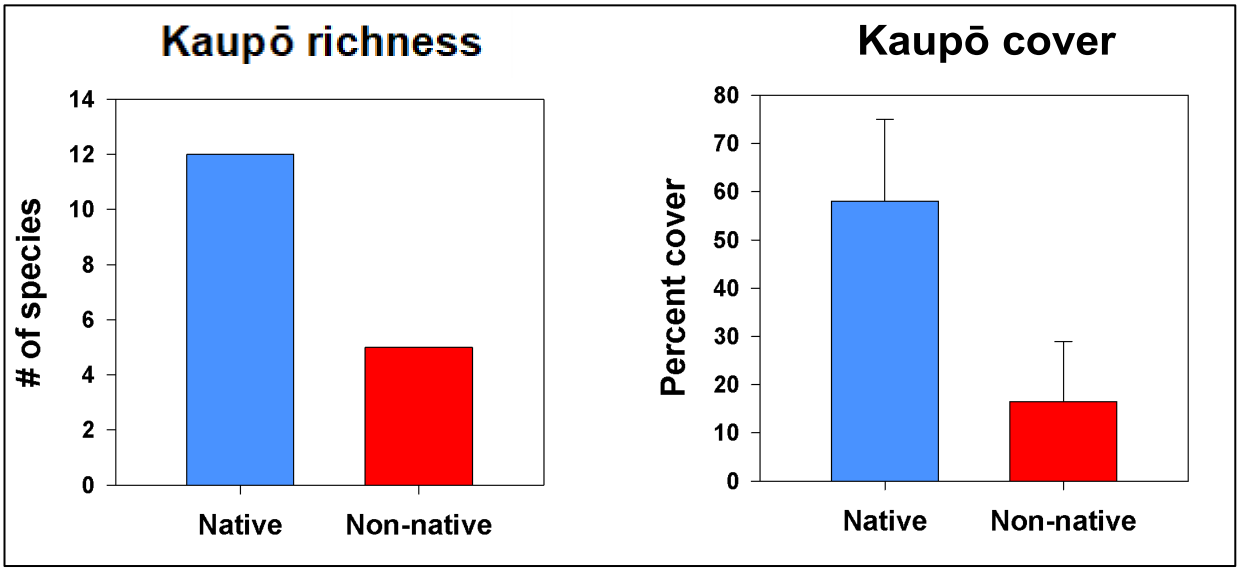

Figure 4.2.6-10. Mean (±SE) native and non-native plant species richness (left) and percent cover (right) recorded within 1,000 m^2 FTPC monitoring plots in Kaupō shrublands (n = 2). Source: PACN and Ainsworth (2016).

One 500 m belt transect (consisting of 25 contiguous 5 x 20 m plots) used by the NPS I&M EIPS monitoring program was located in shrubland habitat within the Kaupō management units (Gross et al. 2017). This transect recorded an average cover by non-native plants of between 1% and 5%, and a total of 16 non-native plant species along its length (Gross et al. 2017).

Kaupō shrubland vegetation, like West Slope shrubland vegetation, is therefore currently dominated by native plants, especially in the shrub canopy layer. Also like the West Slope, Kaupō shrublands are threatened by several habitat-modifying weeds (NPS 1999, Table 4.2.6-5), although these weeds differ considerably in identity between the two areas. Among the most serious problems in Kaupō was the dramatic increase of the fire-promoting molasses grass (*Melinus minutiflora*) following the exclusion of feral ungulates (Loope et al. 1992a, NPS 1999). In response, funding was secured to control this grass in the western side of Kaupō Gap using helicopter-deployed herbicide from 1992-1995. It continues to be invasive in other parts of Kaupō, up to 5,000 ft (1,524 m) elevation, and has proved difficult to control (P. Welton pers. comm. 2018). In contrast, *C. clandestinus* (kikuyu grass), an aggressive mat-forming perennial grass, was formerly common in the eastern side of the gap, but is susceptible to ground-based herbicide applications and is now mainly restricted to areas along trails (P. Welton pers. comm.).

Table 4.2.6-5. Numbers of high priority invasive weeds west of Kaupō trail controlled in Kaupō Shrubland Ecosystems in HALE. Source: NPS (2017b).

Taxon	Total Controlled
Cenchrus clandestinus (1996-2010)	6.4 acres treated
Cirsium vulgare (1983-2015)	10,030
Grevillea robusta (1998-2016)	27
Melinus minutiflora (1991-2008)	93.2 acres treated
Schinus terebinthifolius (1990-2016)	2,243
Senna septemtrionalis (1997-2014)	10,384
All Non-Native Plant Species (1983-2017)	64,047 individuals

Other woody shrubs and trees have been targeted for their abilities to invade recovering or intact native communities and to modify the structure and function of these ecosystems (Medeiros et al. 1998). *Schinus terebinthifolius* (Christmas berry) is currently the most threatening of these. This bird-dispersed shrub to small tree was first recorded in western Kaupō Gap in 1981 (Stemmermann et al. 1981), and Loope et al. (1992) reported that it was common at lower elevations outside the Park but was rarely found above 4,000 ft (1,220 m). However, this species is becoming much more common in Kaupō Gap, and has now been seen above 5,000 ft (1,524 m) elevation (NPS 2017b, P. Welton pers. comm. 2018). Without additional control effort, it could eventually form dense stands and transform the lower stature shrublands into a closed-canopy non-native monoculture forest (Smith 1985). *Grevillea robusta* (silk oak) is also a significant threat to Kaupō shrublands. This wind-dispersed tree is widely planted, and now naturalized, on the main Hawaiian Islands (Wagner et al. 1999). First reported in Kaupō Gap by Bill Haus in 1998 at 4,000-4,100 ft elevation, this tree is invading ranchlands adjacent to the park (Medeiros et al. 1998). With its tall stature and ability to form dense monocultures, it could completely alter and transform native shrublands if uncontrolled (Santos et al. 1992, NPS 1999). *Senna septemtrionalis* (kolomona) is a shrub to small tree that was first reported in lower east Kaupō Gap and has been targeted for control after park staff noticed it spreading rapidly inside park boundaries (Medeiros et al. 1998).

In addition to controlling invasive weeds, HALE Vegetation Management Unit staff have augmented natural recovery of Kaupō vegetation by outplanting both rare and common native plants. Most of this has occurred beneath or among trees and in shrublands adjacent to forest and woodland habitats in the eastern portion of Kaupō, and is reported in the forest ecosystems section (Table 4.2.3-6 in Chapter 4.2.3).

Because they are much more remote than West Slope shrublands, much less research has been conducted on ecological interactions in Kaupō shrublands. However, it is likely that the effects of nēnē, introduced gamebirds, and non-native rodents on vegetation are generally similar to those discussed with respect to West Slope shrublands. Although they extend to lower elevations than West Slope shrublands, and climate instrumentation there is less extensive, Kaupō shrublands may also be experiencing the effects of changing climate conditions.

Summary of Kaupō vegetation

Despite the fairly large number of habitat-altering non-native plants that are threatening to invade, and the increased risk of fire that some of them promote, the condition of Kaupō shrubland vegetation is judged to currently be in good condition, with much higher diversity and especially cover of native plants compared to non-native plants. As with West Slope shrublands, there are no published repeated vegetation data sets for Kaupō shrublands with which to rigorously assess trends. However, Loope et al. (1992) reported that in the absence of fire, ʻaʻaliʻi (*Dodonaea viscosa*), pūkiawe (*Leptecophylla tameiameiae*), and ʻūlei (*Osteomeles anthyllidifolia*) recovered rapidly in a 10-year-old exclosure at 4,000 ft (1,220 m) in western Kaupō Gap that was previously almost barren. Moreover, the obvious recovery of shrubland vegetation throughout the area after fencing and removal of ungulates is especially striking along the Kaupō fence line (Figure 4.2.6-4). The native-dominated status in these areas has further been maintained and augmented through a combination of native outplanting and restoration activities, and targeted control of a select group of potentially habitat-modifying invasive plants. For these reasons, we assign an improving trend to the condition of Kaupō vegetation. Confidence in this assessment is medium, owing to the lack of repeated data to confirm these apparent trends.

Invertebrates

Invertebrate collecting in high-elevation shrublands of HALE began with expeditions through the West Slope area of the park to the summit by Thomas Blackburn in 1878, followed shortly thereafter by the naturalist R.C.L. Perkins in 1894 and 1896 (Beardsley 1980, Krushelnycky et al. 2005a). A number of subsequent collections were made in these and other parts of the Summit District, some of which were included in Beardsley's (1980) report and checklist that resulted from his extensive insect survey that was part of the Resources Basic Inventory for the park, conducted from 1975-1977.

This report was the first comprehensive summary of the high-elevation insect fauna, as assessed through 55 survey sites across the West Slope, summit, Crater and its periphery, and Kaupō Gap.

Analysis of Beardsley's (1980) sampling results from sites within shrubland habitats, including both West Slope and Kaupō shrublands, indicates that 207 species were collected, of which 109 (53%) were native. Separating these results between the two shrubland regions yields a total of 164 species collected on the West Slope, of which 91 (56%) were native, and a total of 57 species collected in Kaupō, of which 24 (42%) were native. This suggests that the Kaupō region is more highly invaded, an observation that was made by Beardsley (in Berger et al. 1975).

Referring to the entire Summit District (called Crater District in his report), Beardsley concluded that "The current status of the insect fauna of the area is not very satisfactory." This assessment was due in part to the damaging effects of invasive goats and pigs on native vegetation that is the foundation for the native insect fauna, as well as to the numerous non-native insect species established. Goats and pigs have been removed from these ecosystems, resulting in recovery of native vegetation (see above sections), but the problem of non-native insect invasion has only worsened.

Approximately 25 years later, from 2001 to 2004, Krushelnycky et al. (2007) surveyed arthropods in two elevation zones in the West Slope shrublands. Sampling was conducted using standardized methods (pitfall traps, leaf litter extraction, vegetation beating) in 48 5 x 5 m plots, supplemented with limited opportunistic collecting. Vegetation beating samples were conducted on four shrub species: *L. tameiameiae*, *V. reticulatum*, *S. chrysophylla* and *D. menziesii*. This survey detected a total of 257 species, of which 101 (39%) were judged to be native (15 species were of unknown status). Notably, 56 of the 141 (40%) non-native species collected were new records for the park. While many of these had likely been present for many years but simply not previously identified or reported, some portion likely represented relatively recent introductions.

Neither the Beardsley (1980) nor the Krushelnycky et al. (2007) survey should be taken as an estimate of the total richness of the insect or arthropod faunas at their respective times. Invertebrate sampling inevitably fails to detect all species present. For example, species accumulation curves constructed from the Krushelnycky et al. (2007) sampling suggested that the survey was incomplete, and that additional sampling using the same methods in the same area would increase the total number of native and non-native species by 38% and 34%, respectively. Sampling on additional plant species, and employing additional sampling methods, would increase the totals even further. For this reason, and because the methods used by Beardsley (1980) and Krushelnycky et al. (2007) differed substantially, it is difficult to compare the results of the two surveys in a rigorous way. However, the change from an estimated 56% native faunal composition in 1975-77 to an estimated 39% native faunal composition in 2001-2004 may provide a very rough indication of the continued accumulation of non-native arthropod species in West Slope shrublands. More recently, a series of surveys of shrublands begun in 2009 around the park entrance station and a pair of gulches adjacent to the park road at roughly 7,500-7,600 ft (2,286-2,316 m) elevation has found similar results: composition of the communities sampled has ranged from 23% to 41% native species, averaging 30% overall, and has ranged from 31% to 55% non-native species, averaging 42% overall (Brenner 2009, 2010a, 2010b, 2011a, 2011b, 2012a, 2012b, 2013).

Little is known about the ecological effects of most individual non-native species, but for some, as in the case of the recently established predatory ground beetle, *Trechus obtusus*, impacts can be surmised from general life history. First detected at the park service area near the main entrance in 1999, this beetle has quickly spread across much of the top of Haleakalā volcano, reaching high densities within the leaf litter in some areas (Liebherr and Takumi 2002, Liebherr and Krushelnycky 2007). Because of its similarity in size and habits to native *Mecyclothorax* ground beetles (Figure 4.2.6-11), it is quite possible that *T. obtusus* represents a significant new competitor for food and possibly shelter resources for the native beetles, and may also prey upon them (Liebherr and Krushelnycky 2007). This species represents just one example. Collectively, the great diversity of new competitors and predators now established in shrubland habitats may be exerting substantial effects on native arthropod community composition, and more broadly on food web structure and function.

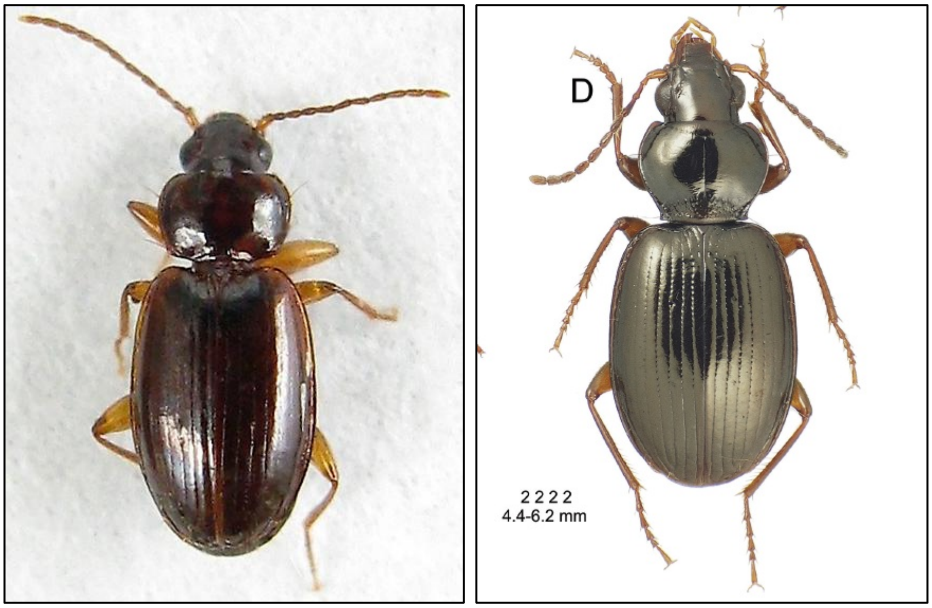

Figure 4.2.6-11. The invasive ground beetle *Trechus obtusus* (left), and the common native denizen of West Slope shrublands *Mecyclothorax cordithorax* (right). Photos: Jim Moore left, James Liebherr right.

Argentine Ants

A few non-native species invading shrubland ecosystems are known to individually have severe and potentially far-reaching ecological effects. These are social insects, including ants and yellowjacket wasps. The Argentine ant (*L. humile*), was first detected near Hosmer Grove in 1967, and has subsequently spread dramatically, with two large populations now covering over 1,606 ac (650 ha) of the park, mainly within the West Slope shrubland but now increasingly extending further into the crater (Krushelnycky et al. 2011). The boundaries of these populations, which were last mapped comprehensively in 2004, are shown in Figure 4.2.6-12, but these are known to be a substantial under-representation of the ant's current distribution. Rates of outward spread vary across different regions of the two populations, but were measured to exceed 100 m/yr in the most rapidly expanding areas (Krushelnycky et al. 2005b). Range limits and rates of spread appear to be governed largely by temperature constraints (Krushelnycky et al. 2005b, Hartley et al. 2010), which has allowed estimates to be made of the total potential distribution of this species in the park. Estimated amount of suitable habitat, which may total approximately 75% of subalpine habitats in the park (Krushelnycky et al. 2005b), varies according to fluctuations in soil temperatures (Hartley et al. 2010), suggesting that more areas will become suitable as the climate continues to warm.

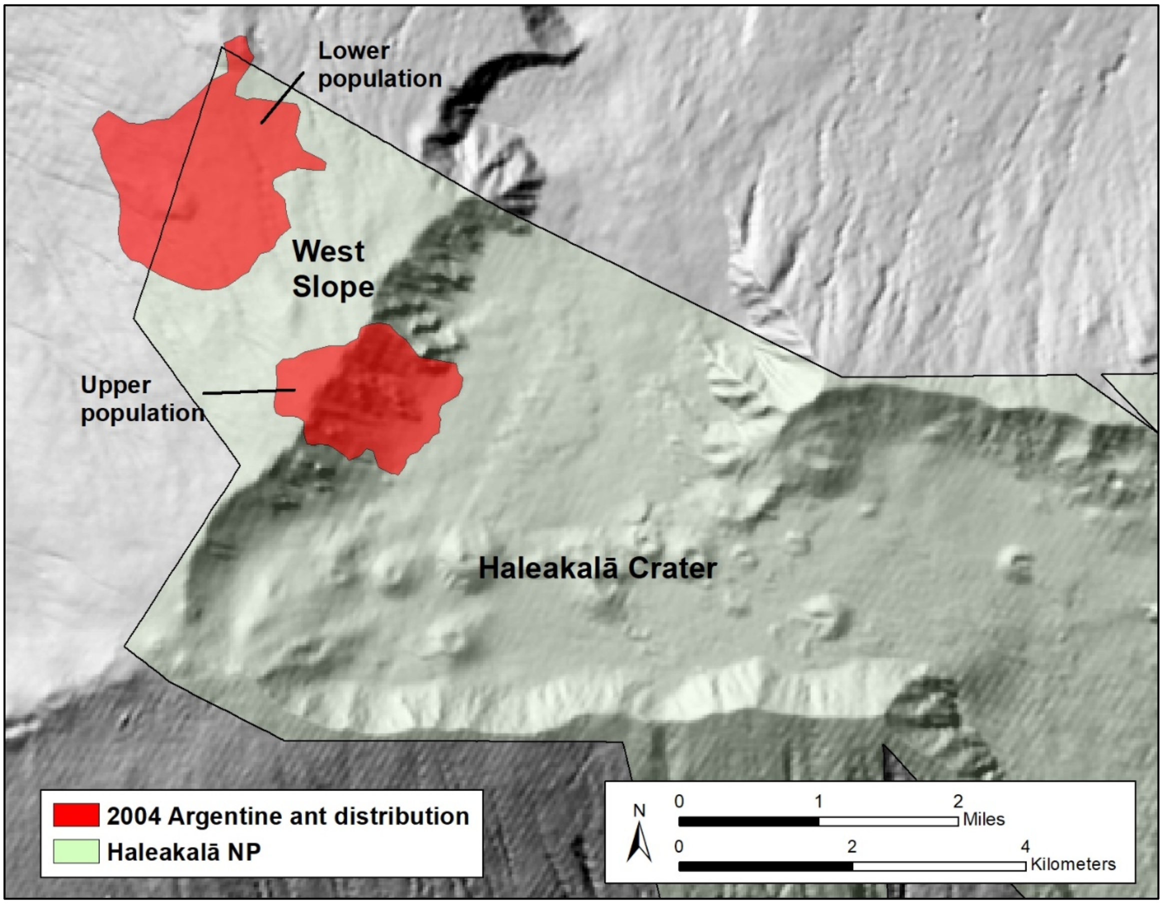

Figure 4.2.6-12. Estimated distribution of the two Argentine ant populations in HALE in 2004, totaling 1,606 acres (650 ha). Source: Krushelnycky et al. (2011), and unpublished USGS data.

In West Slope shrubland areas already invaded by the Argentine ant, impacts on native arthropods are extensive and dramatic (Cole et al. 1992, Krushelnycky and Gillespie 2008). For example, Krushelnycky and Gillespie (2008) sampled arthropod communities on shrubs and on the ground within each of the two large ant populations (Figure 4.2.6-12), and determined that native arthropod diversity was reduced by over 50% relative to adjacent un-invaded areas (Figure 4.2.6-13). A wide variety of arthropod taxa are affected, but endemic beetles, predators and rare species of all types appear to be especially vulnerable, while non-native species are more resistant (Krushelnycky and Gillespie 2010). As an example, several very rare species of native *Mecyclothorax* and *Blackburnia* ground beetles were only collected outside ant-invaded areas, and are very likely being excluded by the ants, while the non-native ground beetle mentioned above (*T. obtusus*) had similar densities in invaded and un-invaded habitat (Krushelnycky et al. 2005a, Liebherr and Krushelnycky 2007).

Overall, the invasion substantially increases the fraction of the community composed of non-native arthropod species. The trophic structure of the arthropod community is also strongly altered in ant-invaded areas, suggesting changes in energy flow through the food web, but these changes are idiosyncratic between invasion sites (Krushelnycky and Gillespie 2008).

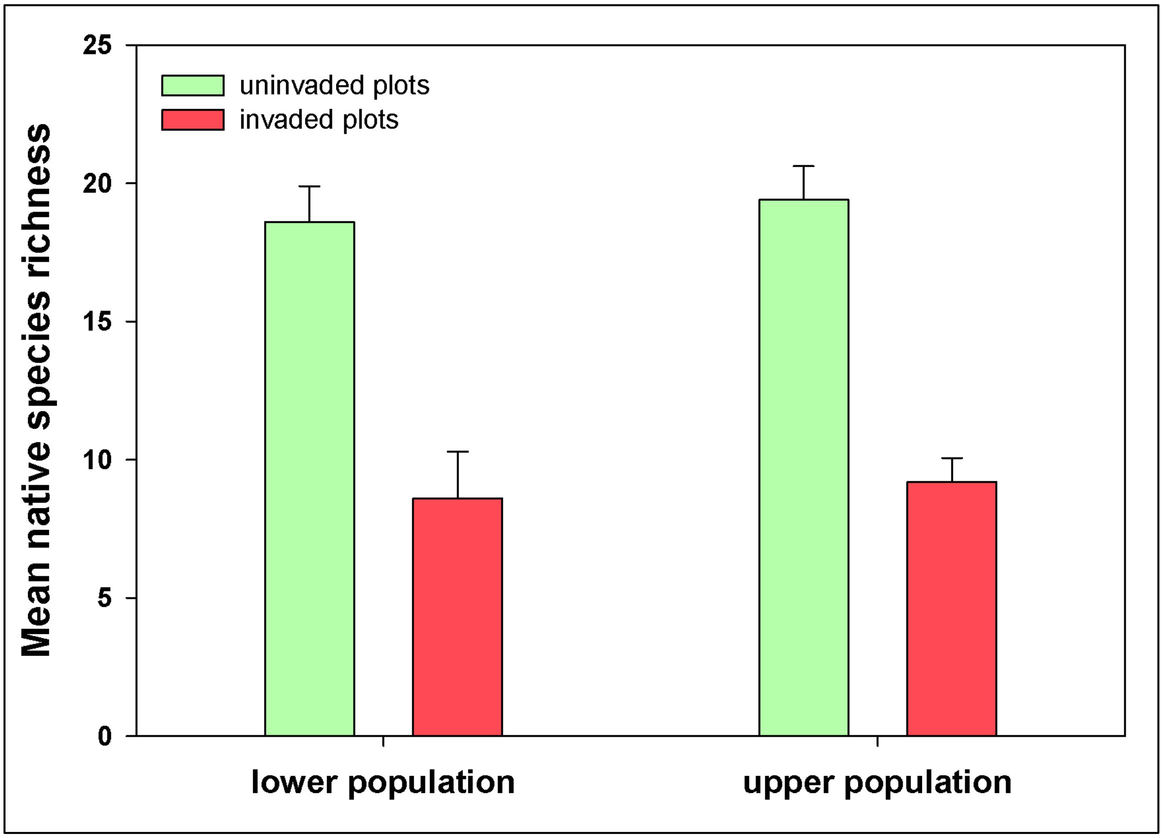

Figure 4.2.6-13. Mean native arthropod richness per plot within the lower and upper Argentine ant populations, compared to adjacent uninvaded plots in the West Slope shrubland. Native richness was reduced by 53.8% and 52.6% within the lower and upper populations, respectively. (n = 5 plots for each site x invaded status combination, n = 20 plots total). Source: Krushelnycky and Gillespie (2008).

Other ecological processes may be impacted by Argentine ant invasion, the most obvious being insect-mediated pollination. This effect was first hypothesized by Cole et al. (1992), who found dramatically reduced densities of native yellow-faced bees (*Hylaeus* spp.) in ant-invaded areas. Yellow-faced bees are important pollinators of many native plants (Magnacca 2007), and high-elevation areas of Haleakalā support the highest remaining densities of *Hylaeus* in the Hawaiian Islands (Daly and Magnacca 2003; Figure 4.2.6-14). Ant-induced reductions in their populations or displacement from flowers (Hanna et al. 2015, Sahli et al. 2016) may be especially detrimental to plants that are self-incompatible and completely reliant on insect cross-pollination. This is known to be the case for the Haleakalā ʻāhinahina (*A. sandwicense macrocephalum*), and is suspected of also being true for the related shrubland species *D. menziesii* (Carr et al. 1986). A study on floral visitation determined that yellow-faced bees and other insects were also likely to be important pollinators for other native shrubs or small trees on the West Slope, including *G. cuneatum*, *L. tameiameiae*, and *S. haleakalae* (Sahli et al. 2016). This study found that pollinator visitation rates at flowers of *G. cuneatum*, and *L. tameiameiae* in one of two years, were lower at ant-invaded study sites. Numbers of *Hylaeus* captured in pan traps were also lower at ant-invaded sites. However, other confounding environmental variables prevented conclusive determination of Argentine ants as the causal factor behind these patterns (Sahli et al. 2016). Also complicating the situation, a study on Hawai‘i Island found that Argentine ants reduced rates of floral herbivory by non-native caterpillars on *V. reticulatum*, thus increasing rates of fruit set for this plant (Bleil et al. 2011). The effects of Argentine ants on plant reproduction in shrublands therefore appear to be somewhat complex and context-dependent.

Figure 4.2.6-14. *Hylaeus difficilis*, a common native yellow-faced bee in HALE shrublands. Photo: Karl Magnacca.

Because of the various ecological impacts summarized above, NPS and collaborating USGS staff initiated an extensive research and management program aimed at developing methods to control Argentine ants in the park around 1994 (Krushelnycky et al. 2011). This program continued though about 2009, but has since waned because effective methods for eradicating this species were not discovered. Similarly, a containment strategy initiated in 1997, in which the expanding peripheries of both ant populations were treated annually with pesticidal ant bait, was discontinued after 2004 because the method was insufficiently effective in slowing outward spread (Krushelnycky et al. 2011). Currently, both populations of Argentine ants are expanding unchecked, representing one of the most significant threats to biological resources in the park.

Big-Headed Ants

Another highly damaging invasive ant species, the big-headed ant (*Pheidole megacephala*), occurs in shrubland in the lower eastern portion of Kaupō Gap (see Figure 4.2.3-20). When this area was surveyed in 2001, the big-headed ant appeared to be restricted to elevations below about 4,750 ft (1,448 m), and occurred at relatively low densities (P. Krushelnycky, unpub. data). This ant species is typically more successful at lower elevations, and is uncommon above 4,000-5,000 ft (Reimer 1994), so the upper boundary in Kaupō may represent a fairly stable climatic limit. However, the area has not been surveyed since 2001, and the ant's distribution in the western portion of Kaupō is unknown; range limits may be shifting upslope with warming temperatures, or could do so in the future. Based on what is known about the ecological effects of big-headed ants in Hawai'i (Krushelnycky et al. 2005c), this ant may be capable of exerting impacts on native arthropods in Kaupō similar to those seen for the Argentine ant on the West Slope, especially if its populations increase in abundance.

Yellowjackets

Yet another invasive social insect, the western yellowjacket (*V. pensylvanica*), also has been shown to prey on large numbers of a wide range of arthropods in the shrublands of the West Slope, including native yellow-faced bees (Gambino and Loope 1992, Wilson et al. 2009). When colonies of this wasp were experimentally removed at both HALE and HAVO, spider and caterpillar densities strongly rebounded, indicating that yellowjackets depress populations of these arthropods by 36% and 86%, respectively (Wilson et al. 2009). These wasps also displace native yellow-faced bees and non-native honeybees from flowers of shrubland plants at HALE (Wilson and Holway 2010), and may thereby impact plant reproduction (e.g. Hanna et al. 2013).

Because of these effects, and because aggressive yellowjackets also represent a hazard to park visitors, HALE staff monitor yellowjacket abundances and control individual nests. Yellowjacket relative abundances have been monitored since 1987 using two types of traps baited with heptyl butyrate attractant: "inn" traps, which were used initially through about 2007, and "water" traps, which have become the main monitoring method in more recent years owing to higher capture rates (NPS 2017c). Annual estimates of relative abundance in West Slope shrubland areas using numerous traps of the two types are shown for the period 1987-2015 in Figure 4.2.6-15. These data, which show the annual average number of yellowjackets caught per trap on a relativized scale ranging from 0 to 1, indicate that abundances of yellowjackets vary strongly from year to year, with no clear long-term trend. The data also show that, averaged over the entire time period, there is strong seasonality to yellowjacket abundances, with worker numbers increasing in June and July as colonies grow, peaking in August and September, and declining through the fall and winter (Figure 4.2.6-16). Endangered Wildlife Management program staff control individual nests in the summer and fall seasons when numbers get high, typically focusing on nests near trails, campgrounds and other facilities. Between 2007 and 2013, 64 yellowjacket nests were exterminated, 38 of which were located in the West Slope shrublands (NPS 2017c). These efforts reduce risk of visitor stinging around heavily used areas, but likely have relatively small, localized effects in terms of protecting native arthropod communities.

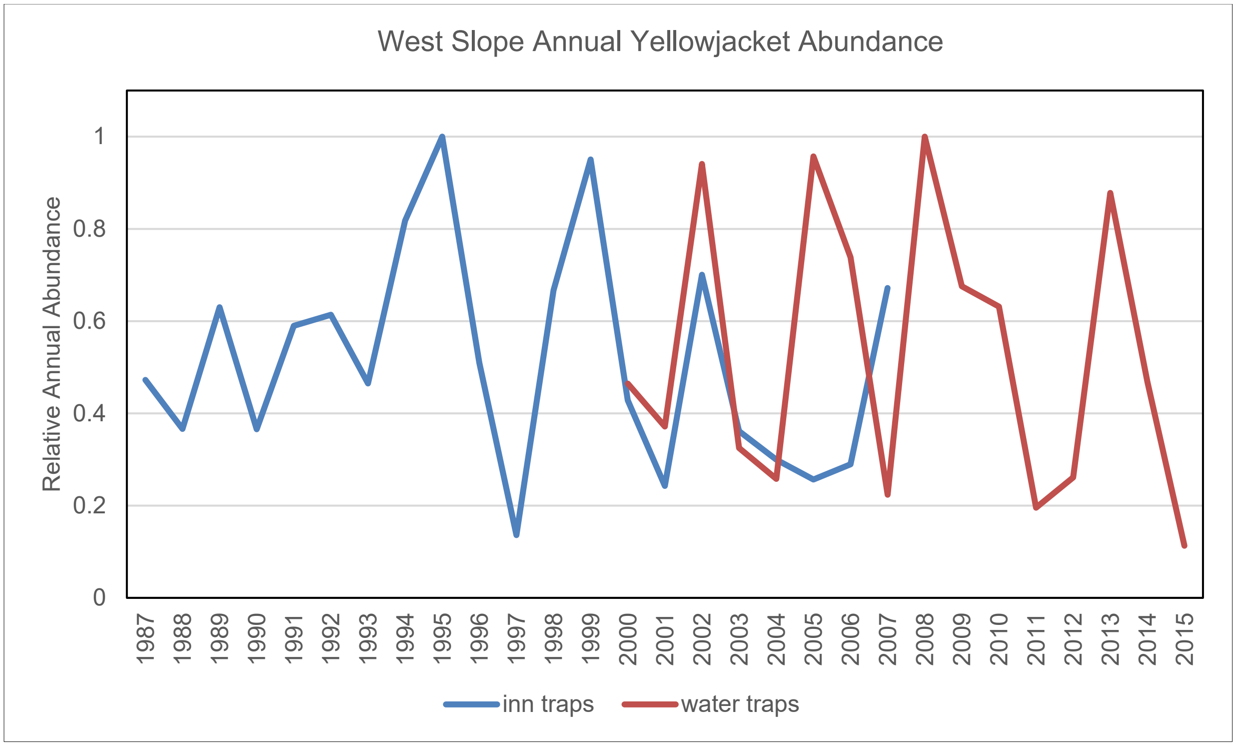

Figure 4.2.6-15. Estimated annual relative abundance of *Vespula* yellowjackets in West Slope shrubland areas from 1987 to 2015. Shown are mean number of workers captured per trap, averaged over all traps deployed per year, and relativized by the maximum annual average for each trap type. Trends for the two trap types, inn and water, are plotted. Inn trap data for 2008 are excluded due to low numbers of traps used. Source: NPS (2017c).

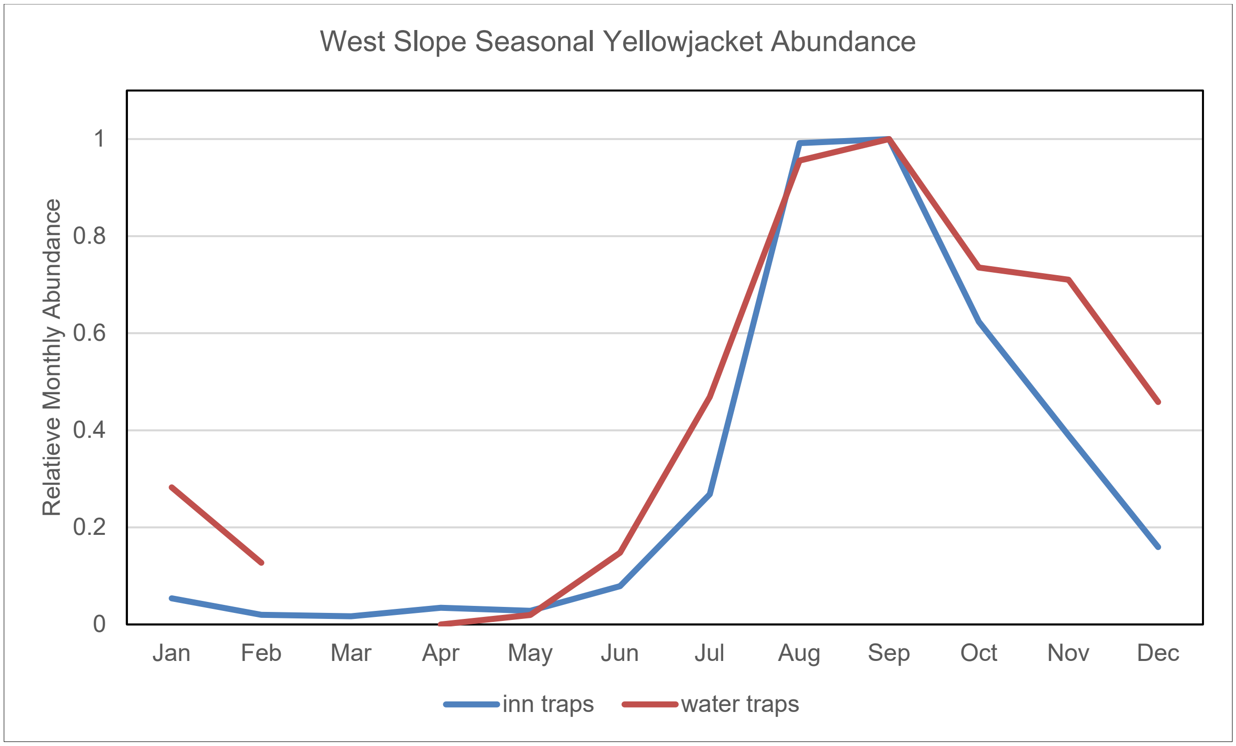

Figure 4.2.6-16. Estimated seasonal relative abundance of *Vespula* yellowjackets in West Slope shrubland areas. Shown are mean number of workers captured per trap, averaged over all traps deployed in each month over the period 1987-2015, and relativized by the maximum monthly average for each trap type. Trends for the two trap types, inn and water, are plotted. Source: NPS (2017c).

Additional Impacts of Non-native Species

Non-native gamebirds (pheasant and chukar) feed on invertebrates, but they were found to comprise a minor part of these birds' diets in West Slope shrublands, and were overwhelmingly non-native species like isopods, ladybird beetles, garlic snails and earthworms (Cole et al. 1995a, b). Non-native rodents may pose a greater threat to native shrubland arthropods. Arthropods were found to comprise 32.5 ± 2.6% and 16.4 ± 3.5% of the diets of mice and black rats, respectively, trapped in West Slope shrublands in 1984-85 (Cole et al. 2000). These rodents appear to rely on arthropod prey especially heavily during the winter to spring, when alternative plant resources are scarce. Prey items were diverse, with the most common being Araneae (spiders), Hemiptera (bugs), Diptera (flies), Coleoptera (beetles) and Lepidoptera (moths). Many endemic species belonged to this prey base, including *Agrotis* caterpillars, lycosid wolf spiders, weevils and carabid beetles. In certain areas and times, arthropods made up approximately 75% of mice diets. Consequently, Cole et al. (2000) felt that both rodents may be exerting population-level impacts, but that mice especially may have "dramatic negative effects on locally endemic arthropods, particularly during periods of high density." Endangered Wildlife Management program staff at HALE operate several small mammal predator traplines, employing about 300 traps, in the West Slope shrubland area (NPS 2012). These traps target rats, mongoose and cats to protect endangered nēnē and ʻuaʻu, but also likely provide some benefit to native arthropods in the vicinity of the traplines.

Because most of the shrubland areas in HALE are relatively dry, native mollusc diversity and abundance is likely to be fairly low. However, some native snails are known to occur in dry high-elevation shrublands (Cowie 1995), and the native shrubland snail fauna appears to be unsurveyed at HALE. The only information found concerned the presence of non-native slugs and the non-native predatory garlic snail (*Oxychilus alliarius*) in West Slope shrublands, as detected in pitfall sampling and in the diet of pheasants and chukars (Cole et al. 1992, 1995a, b).

Summary of Native Shrubland Invertebrates

Because of the fairly large fraction of shrubland arthropod communities composed of non-native species, and the widespread presence of several highly damaging species of invasive ants and wasps, we judge the current condition of shrubland invertebrates to warrant significant concern. We assign a deteriorating trend in this condition, owing to 1) the continuous introduction and establishment of new non-native species, with no practical means of controlling the vast majority of them, and 2) the continuing presence of the western yellowjacket and the expanding distribution of the Argentine ant in shrubland ecosystems. Confidence in this assessment is medium, because although much information exists for West Slope shrublands, information on invertebrate condition in Kaupō is much more limited. Furthermore, invertebrate communities in other shrubland areas of the park, such as along the north and south rims of the crater, are thought to be free of damaging species of invasive ants, and may be in relatively better condition but also are largely unstudied.

Overall assessment

Integrating the three indicators for shrubland ecosystems according to SotP aggregation rules yields a current overall condition that is borderline between good and warranting moderate concern. Because only one of the three indicators represent animal communities, as opposed to plants, we have decided to give additional weight to the invertebrate indicator, and thus assign an overall condition warranting moderate concern (rather than good condition). The ongoing invasion of the Argentine ant in shrubland ecosystems is one of the largest single threats to a large portion of native biodiversity in the park, and merits added influence. Our condition assignment also recognizes that the poorer condition of arthropod communities has the potential to negatively impact vegetation communities. Not only are ecological interactions such as insect-mediated pollination possibly threatened, but the continued accumulation of new non-native invertebrates risks the establishment of serious pests of native plants, like the naio thrips (*Klambothrips myopori*) that is now causing extensive damage to naio woodland areas on Hawaiʻi Island (Conant et al. 2009, K. Magnacca pers. comm. 2017). Moreover, while shrubland vegetation has improved as a result of fencing and removal of ungulates, weed control and native species outplanting, and is consequently in relatively good condition, these plant communities still face significant existing and emergent threats. Compared to shrubland communities of HAVO, Gross et al. (2017) found that non-native plant frequency and cover were greater within HALE subalpine shrubland. The authors speculate that this may be due to the greater accessibility of HALE's shrublands through its network of roads, trails and other infrastructure which may serve as pathways for the introduction of non-native plants. Warming and drying conditions may increase the frequency or severity of water stress on shrubland vegetation, and also increases the risk of fire. We assign an unchanging trend to the overall condition of shrubland ecosystems, due to the contrasting trends between plant and invertebrate indicators. (return to Condition Summary).

Level of confidence

Overall level of confidence in this assessment is medium. Much information has been reviewed to provide the current assessment, but very few data have been repeatedly collected over time, creating more uncertainty in assessment of trends. Information is fragmentary, lacking, or insufficiently quantitative for certain resource areas, which also lowers level of confidence.

Information gaps and research recommendations

Previously, repeated quantitative information on shrubland ecosystem vegetation (i.e., cover and richness, degree of non-native plant invasion) was lacking. This is in the process of being remedied via the FTPC and the EIPS monitoring protocols of the NPS I&M program (Ainsworth et al. 2011, 2012). The repeated monitoring of these two sets of plots will allow for site specific comparisons over time that will greatly enhance future vegetation trend analyses. Due to time and staff limitations, consistent monitoring is not conducted for most of the threatened, endangered, or otherwise rare plant species in shrubland ecosystems, making assessment of trends in these species difficult. While shrubland arthropod communities are fairly well characterized on the West Slope, less is known about communities in other shrubland areas, and other invertebrates like molluscs are even less poorly known. No monitoring protocol exists for shrubland invertebrate communities, which severely hampers assessment of changing conditions.

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4.2.7. Subalpine Grassland Ecosystems

Condition Summary

Overall resource condition is good with an unchanging or stable trend; confidence in condition determination is low. Rationale is discussed in the "*Overall assessment*" section and following.

⬄

Relevance and Context

Large tussocks of the endemic bunchgrass *Deschampsia nubigena* dominate high elevation grasslands found only on Maui and Hawai'i island (Gagné and Cuddihy 1990, Gustafson et al. 2014). Within HALE, the most notable Deschampsia grassland occurs in Kalapawili, with smaller representatives found at Kuiki, in small areas within the crater and in mostly cinder substrates on the outer northwestern slopes (Medeiros et al. 1998, Figures 4.2.7-1 and 4.2.7-2).

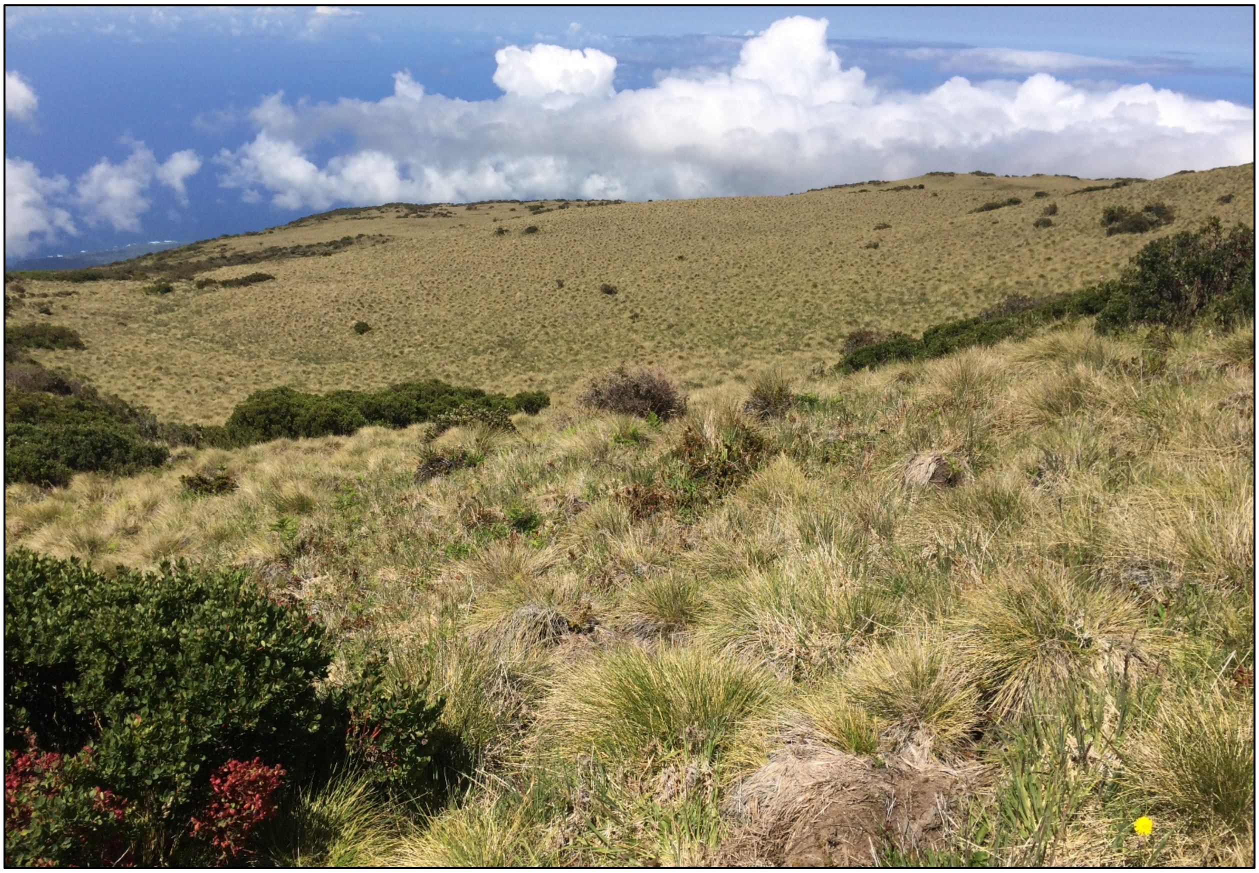

Figure 4.2.7-1. *Deschampsia nubigena* grassland at Kalapawili, Haleakalā National Park. Photo: P. Krushelnycky.

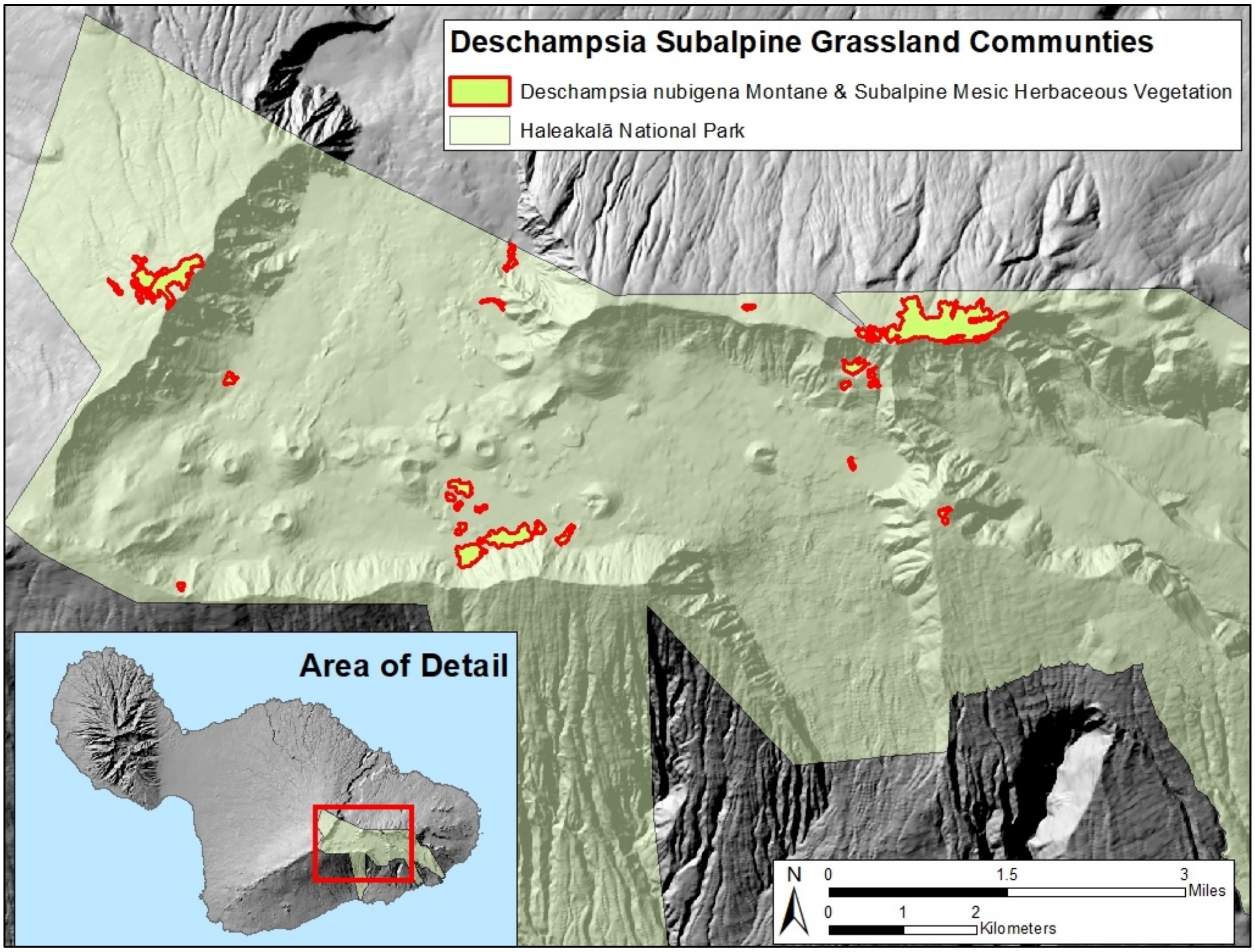

Figure 4.2.7-2. *Deschampsia nubigena* Montane and Subalpine Mesic Herbaceous Vegetation communities, Haleakalā National Park (134.5 ha). Source: Green et al. (2015).

On the northeastern outside flank of the Crater, the Kalapawili grassland is the highest elevation native grassland, and largest by area in the state, and is recognized as a unique Hawaiian plant community (Forehand 1970, Henrickson 1971, Vogl 1971, Vogl and Henrickson 1971, Gagné and Cuddihy 1990). Associated native species include *Luzula hawaiiensis*, *Carex brevicaulis*, *Carex macloviana*, *Vaccinium reticulatum*, and *Pteridium aquilinum* ssp. *decompositum* (Gagné and Cuddihy 1990, Medeiros et al. 1998). A dense zone of the small tree fern *Sadleria cyatheoides* (‘ama‘u) occurs on the northern and northeastern edges of the *Deschampsia* grassland immediately above the treeline of *Metrosideros polymorpha*-dominated rain forest. The substrate towards the western side of Kalapawili ridge becomes rockier with less cinder, and *Deschampsia* becomes less common (Medeiros et al. 1998).

In a survey of the northeast outer slope of Haleakalā, Vogl (1971) provides a descriptive overview of the terrain which contributed to the formation of the Kalapawili grassland, writing:

> *“The grassland consists of a series of inclined benches of lava and ash that descend or slope in a step-like fashion to the north and northeast. Erosion has cut shallow*

> *valleys and small ravines or gulches, and left behind a few resistant rocky knolls and ridges. The gulches contain vertical lava tubes, inactive steam vents or fumaroles, and water-enlarged and excavated fissures, all of which take considerable run-off during heavy rains. These man-sized or larger orifices are often concealed by grass, ferns, or other vegetation, or are covered by water.”*

The Kalapawili grassland is used for both nesting and flocking by the endangered nēnē (Hawaiian goose, *Branta sandvicensis*) (Vogl 1971, NPS 2012, Chapter 4.2.10). Similarly, the pueo (Hawaiian short-eared owl, *Asio flammeus sandwichensis*) uses subalpine grasslands for both nesting and hunting (Vogl 1971, Conant and Stemmerman 1979). The Kalapawili grassland was also identified as a key wintering ground for the kōlea (Pacific golden plover, *Pluvialis fulva*), and this migratory species may have contributed to the dispersal of *Deschampsia*, *Luzula*, *Rumex*, *Carex*, and *Trisetum* from its mainland breeding grounds and migratory routes to its island wintering grounds (Proctor 1968, Stoner 1969).

Little information has been reported on invertebrate diversity in subalpine grasslands of HALE. The Resources Basic Inventory for HALE, conducted in 1975-1977, indicated two insect sampling sites in grassland habitats, but only two species were reported from these locations: a non-native aphid from the grassland crater of Kaluanui, and the native damselfly *Megalagrion blackburni* from Kalapawili grassland (Beardsley 1980). The latter likely represented a visitor from stream channels below, as this species typically breeds in swift stream waters (Polhemus and Asquith 1996). Although information is limited, the remote grasslands of Kalapawili may serve as important habitat refuges for species that are threatened by non-native invertebrates in similar but more accessible areas of the West Slope or Crater. For example, the rare endemic ground beetle *Blackburnia lenta* was thought to be restricted to a total area of 358 ac (145 ha) of shrubland and grassland habitat on the West Slope, with this area being increasingly squeezed between the two expanding populations of Argentine ants (*Linepithema humile*) that occur there (Krushelnycky et al. 2005). Discovery of individuals of *B. lenta* at the edge of a small *Deschampsia* grassland on Kalapawili Ridge in 2005, however, revealed that at least one population of this species is currently safe from extirpation by invasive ants (Liebherr and Krushelnycky 2009).

As with other native ecosystems of Haleakalā, subalpine grasslands have been degraded by the impacts of feral ungulates and invasive weeds. von Tempski (1940) reported that domestic cattle were brought into the Crater in the late 1800’s and sometimes taken up to graze the Kalapawili grassland. Starting about 1970, extensive rooting by feral pigs within the Kalapawili grassland promoted an increase in non-native species, especially the grass *Holcus lanatus*, and two herbs, *Hvpochoeris radicata* and *Rumex acetosella* (Medeiros et al. 1998). After about 10 years, bare ground exposed by pig rooting averaged 10-40% of total cover (Jacobi 1981). Following exclusion of pigs in the mid-1980’s, the grassland has dramatically recovered in terms of *Deschampsia* ground cover, although *Holcus* still dominates in periodic stream courses. *Rubus argutu*s (blackberry), capable of invading grasslands without pig-induced disturbance, is a very serious long-term weed threat (Loope et al. 1992, NPS 1999). *Senecio madagascariensis* (fireweed), an invasive annual or short-lived perennial herb, can spread long distances via its prolific wind-dispersed seeds and form dense cover in subalpine and alpine habitat (Ramadan et al. 2011). Large fireweed infestations outside the park provide a perpetual seed source, and it is becoming increasingly common in disturbed areas throughout the shrublands. Although current precipitation amounts in the subalpine grasslands may preclude invasion, a changing climate and drier conditions could facilitate future establishment and spread of this aggressive, disturbance-adapted weed.

The condition of the subalpine grassland ecosystem was assessed using a single indicator: grassland vegetation. Focus was placed on the largest of the park's subalpine grasslands in the Kalapawili management unit on the northeast rift, and some inferences were drawn from small grasslands scattered throughout the park. Native birds are not considered here because most are a minor component of grassland ecosystems, and nēnē, which use grasslands more extensively, are addressed primarily in the T&E Species section (Chapter 4.2.10). No information on grassland invertebrate communities was found beyond what is stated above; the condition of these invertebrate communities, if not necessarily their composition, is likely to be similar to those in surrounding shrublands.

Indicator

- Grassland vegetation

Data and Methods

A review of available data and information was used to assess conditions, and if possible, trends, in the indicator listed above. Some additional analyses were performed on a subset of geospatial and vegetation classification data collected by HALE and PACN staff specific to subalpine grassland ecosystems (Green et al. 2015).

Sources of information

Control data for invasive, non-native plants have been collected by HALE Vegetation Management staff for the entire park from 1986 through present (NPS 2017).

NPS I&M vegetation classification data were also collected by HALE and PACN staff in 220 circular plots (400 m^2) and observation points in 2011 (Green et al. 2015). NPS I&M also recorded non-native plant frequency and abundance data along a belt transect located in the Kalapawili subalpine grassland (Gross et al. 2017).

Reference condition

An ideal reference condition for grassland ecosystems would include fully intact native vegetation, uninvaded by non-native weeds. Due to anthropogenic impacts, including the introduction of non-native plants and animals to the Hawaiian Islands, no subalpine grassland community currently exists in an entirely pristine state. Nevertheless, examples of relatively high-quality grasslands, in which native plant species richness and cover exceeds that of non-natives, can still be found, particularly after non-native ungulates were excluded from the park. A desirable reference condition for HALE's subalpine grassland communities is therefore that the richness and cover of native species exceeds that of nonnative species, and that habitat-modifying invasive weeds are absent.

Condition and Trend

Green et al. (2015) have classified twenty-six areas throughout the park as *Deschampsia nubigena* Montane and Subalpine Mesic Herbaceous Vegetation, totaling 332.6 ac (134.6 ha) (Figure 4.2.7-2). This subalpine grassland community type is characterized by open to dense herbaceous vegetation dominated by *D. nubigena* with the native grass *Trisetum glomeratum* and/or indigenous fern *P. aquilinum* ssp. *decompositum* sometimes present to codominant. Scattered shrubs, including *Leptecophylla tameiameiae* and *V. reticulatum*, may also be present. Non-native grasses, including *Anthoxanthum odoratum*, *H. lanatus*, and *Poa pratensis*. are common associates, with *H. lanatus* codominant in some disturbed stands (Green et al. 2015).

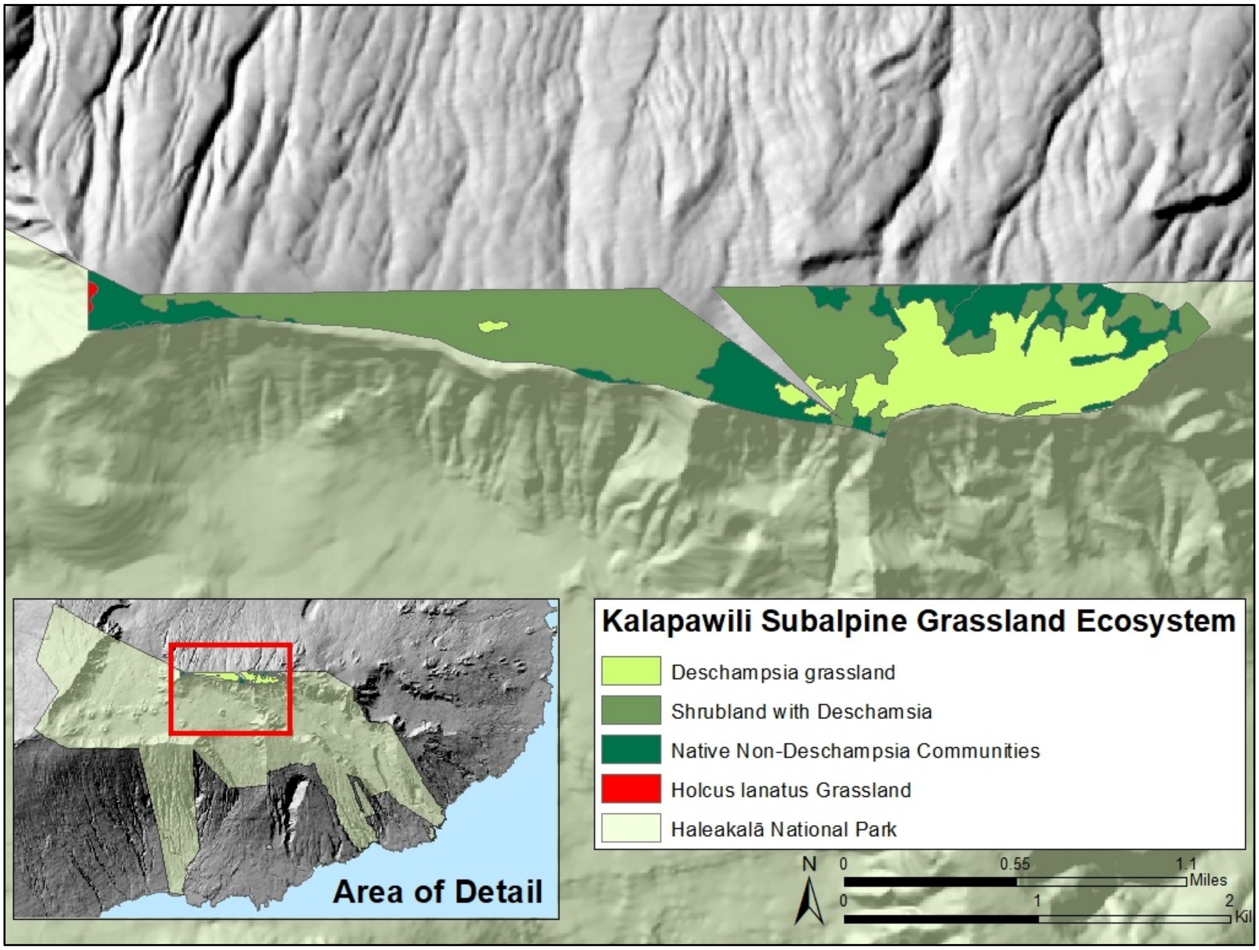

Figure 4.2.7-3. Subalpine grassland ecosystem located on Kalapawili Ridge, HALE. Boundary delineated by NPS designated management units. Source: Green et al. (2015).

The Kalapawili management unit contains the single largest example of *Deschampsia nubigena* Montane and Subalpine Mesic Herbaceous Vegetation in the park, and combined with smaller representatives within the unit, accounts for 152.5 ac (61.7 ha), or 45.8% of the park's subalpine *Deschampsia* grasslands (Figures 4.2.7-3 and 4.2.7-4, Table 4.2.7-1). The majority of the Kalapawili management unit is covered by *Leptecophylla tameiameiae* - *Vaccinium reticulatum* - (*Geranium* *cuneatum*) / *Deschampsia nubigena* Mesic Shrubland (labeled "Shrubland with Deschampsia" in figures and tables), consisting of a native shrubland cover of 20-50% and an understory dominated by *D. nubigena* (Green et al. 2015, Figures 4.2.7-3 and 4.2.7-4, Table 4.2.7-1). In total, *Deschampsia* is present or co-dominant in 469 ac (189.8 ha), or almost 79% of the entire Kalapawili management unit (Figures 4.2.7-3 and 4.2.7-4, Table 4.2.7-1).

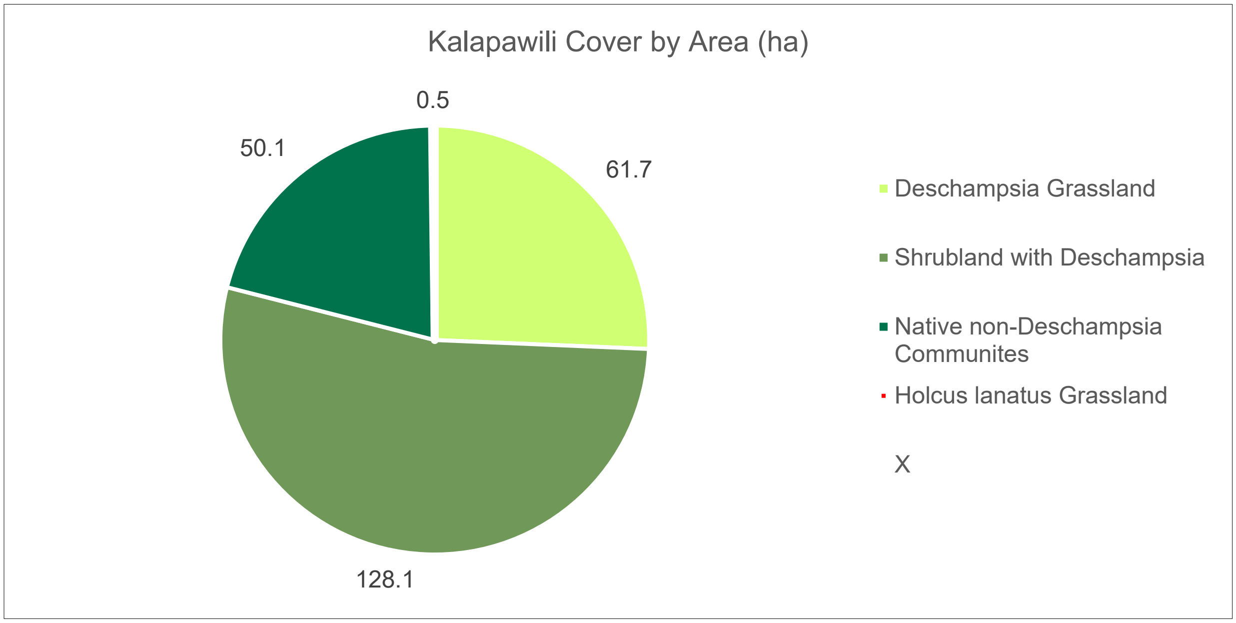

Figure 4.2.7-4. Area (hectares) of *Deschampsia* grassland, and other general vegetation communities within HALE Kalapawili management unit. *Holcus lanatus* grassland (0.5 ha) not visible on chart. Source: Green et al. (2015).

Table 4.2.7-2. Vegetation community types located within the Kalapawili management unit. Source: Green et al. (2015).

Community Classification	**Area (ha)**
Leptecophylla tameiameiae - *Vaccinium reticulatum* - (*Geranium cuneatum*) / *Deschampsia nubigena* Mesic Shrubland	128.1
Deschampsia nubigena Montane and Subalpine Mesic Herbaceous Vegetation	61.7
Leptecophylla tameiameiae - *Vaccinium reticulatum* Subalpine Mesic Shrubland	24.5
Leptecophylla tameiameiae - *Sadleria cyatheoides* Subalpine Mesic Shrubland	20.8
Metrosideros polymorpha / *Sadleria cyatheoides* Forest	3.9
Pteridium aquilinum ssp. *decompositum* Herbaceous Vegetation	0.6
Holcus lanatus Semi-natural Grassland	0.5
Leptecophylla tameiameiae - (*Vaccinium reticulatum*) Montane and Subalpine Sparse Vegetation	0.1
Leptecophylla tameiameiae Subalpine Dry Shrubland	0.1
Total	**240.3**

Green et al. (2015) provide a sampling of plant taxa in 6 inventory and mapping plots located within the *Deschampsia nubigena* Montane and Subalpine Mesic Herbaceous Vegetation community in the park. A total of 14 native and 12 non-native plant species were recorded in these plots, with a mean species richness of 5 ± 1.7 natives and 4 ± 0.6 non-natives per plot (Figure 4.2.7-5). Mean native species cover was 58.8% ± 10%, more than four times the 13.7% ± 3.7% mean non-native species cover (Figure 4.2.7-5).

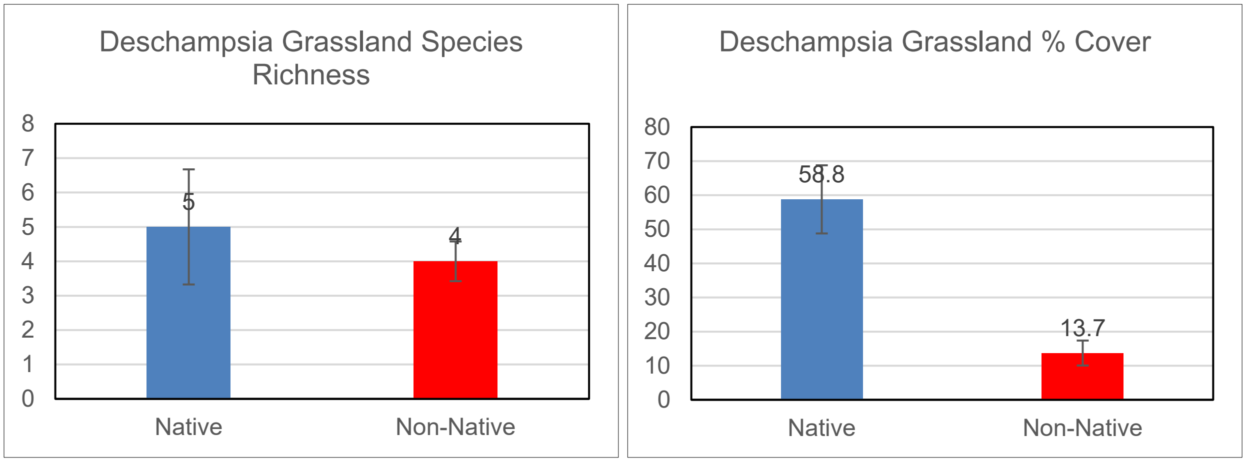

Figure 4.2.7-5. Species richness and cover in all *Deschampsia* grassland inventory and mapping plots (n = 6). Source: Green et al. (2015).

Data collected in NPS I&M efforts to monitor the status of established invasive plants at HALE come to a similar conclusion. One 500 m long belt transect (consisting of 25 contiguous 5 x 20 m plots) was established within the Kalapawili management unit, and baseline monitoring of non-native plants along the transects was completed in 2012 (Gross et al. 2017). On this transect, mean non-native species richness was 3, comparable to values reported by Green et al. (2015), and mean cover of non-native plants was between 10-25%. The midpoint cover range for this transect is 17.5%, which is not excessively higher than the 13.7% non-native cover reported in other grasslands by Green et al. (2015).

These data support the earlier findings of Jacobi (1981), and demonstrate that subalpine grassland vegetation, while relatively low in native diversity, is nevertheless fairly resilient once protected from feral ungulates and continues to be dominated by native cover. When Henrickson (1971) conducted a botanical survey of the northeast outer slope of Haleakalā in 1969, he recorded 13 non-native species in the region, and reported that only three of these were common. These included the composite herb cat's-ear (*H. radicata*), velvet grass (*H. lanatus*), and sheep sorrel (*R. acetosella*), which only occurred in areas disturbed by pigs in the tussock grasslands. Upon completion of his survey, Henrickson (1971) summarized the condition of the grassland as follows:

> "*This area, therefore, is surprisingly free of exotic species and three exotic species are common only because of their ability to thrive in disturbed habitats formed by feral pigs. When the native Deschampsia grass grows back in these areas, however,*

> *the exotic species are crowded out. It therefore appears that in spite of past and present pressures of herbivore grazing and feral pig rooting, this area has retained a highly natural physiognomy, perhaps very much as it was prior to settlement by Hawaiians.*"

Despite the native dominance of vegetation in grassland ecosystems throughout the park, however, a small set of invasive weeds threaten to change the situation in the Kalapawili grassland. Three species, in particular, not reported in previous botanical surveys of the region, have been actively targeted by park vegetation management staff since 1983 due to their ability to persist in the absence of disturbance, their potential to invade a much wider area than currently occupied, their dispersibility, and their potential to modify the structure and function of the Kalapawili subalpine grassland ecosystem (Table 4.2.7-2). *Pinus radiata* (Monterey pine), aggressively targeted for control within the subalpine shrubland and Crater ecosystems, has more recently been invading the Kalapawili grassland. This tree possesses wind-dispersed seeds capable of spreading long distances from parents, can establish in and overtop the shorter-statured grassland community, and could eventually convert it into a fire-prone coniferous monoculture (Burns and Honkala 1990, Loope et al. 1992). *Rubus argutus* (Florida blackberry) is an invasive shrub capable of spreading by rhizomes, aerial runner branches, and bird-dispersed seeds, and can establish thorny thickets within native shrublands, grasslands and forest communities, even in the absence of pig-induced disturbance (Loope et al. 1992, Medeiros et al. 1998). *Parentucellia viscosa* (yellow glandweed) is a hemiparasitic member of the Orobanchaceae, deriving nutrients both by photosynthesis as well as from the host plants it parasitizes. Previously only documented as naturalized on Hawaiʻi island (Wagner et al. 1999), this plant parasitizes many other species and could pose a threat to native taxa, a concern within other areas of its invaded range (Pate and Bell 2000).

Table 4.2.7-3. Numbers of high priority invasive weeds controlled in the Grassland Ecosystems of HALE. Source: NPS (2017).

Taxon	Total Controlled
Pinus radiata (2010-2016)	170
Rubus argutus (1983-2010)	9,405 canes
Parentucellia viscosa (1995-2015)	157
All Non-Native Plant Species (1992 – 2017)	9,831

Overall assessment

Based on the limited available information, the condition of subalpine grassland ecosystems is judged to be good, with an unchanging trend. Despite past disturbance by feral pigs, and establishment of invasive weeds, the grassland ecosystems have largely recovered following ungulate exclusion. Currently, both cover and richness of native plants exceeds that of non-native vegetation. The native-dominated status of subalpine grasslands has been maintained and augmented through a combination of upkeep of boundary fences, and targeted control of a select group of potentially habitat-modifying invasive plants. With continued management, these communities will likely remain stable. However, similar to other HALE ecosystems located above the trade wind inversion, changing climate conditions may exert significant influences on subalpine grasslands in the future. Plant species assemblages in the Kalapawili grassland vary from east to west with the prevailing moisture gradient, and future decreases in precipitation could lead to higher densities of shrubs in wetter portions of the grassland (Crausbay and Hotchkiss 2010). Past fluctuations in the moisture regime have also been suggested to significantly affect the position of the treeline in this area, with trees encroaching into areas currently consisting of grassland during periods of higher precipitation (Burney et al. 1995). Future changes in the moisture regime may therefore lead to substantial changes in vegetation community distributions in the vicinity of the Kalapawili grassland/shrubland ecotone. (return to Condition Summary).

Level of confidence

Overall level of confidence in this assessment is low. Much information has been reviewed to provide the current assessment, but very few data have been repeatedly collected over time, creating more uncertainty in assessment of trends. Repeated vegetation monitoring data are restricted to transects in the Kalapawili grassland surveyed in the mid-1970s, and published in 1981. More recent vegetation data are limited spatially, and do not correspond exactly to earlier data, lowering confidence in the assessment of trend.

Information gaps and research recommendations

Repeated quantitative vegetation data are limited spatially and temporally. No NPS I&M Focal Terrestrial Plant Community monitoring plots or Established Invasive Plant Species fixed transects are located in subalpine grassland ecosystems, yet additional vegetation plot data would greatly enhance future vegetation trend analyses. Information on invertebrate communities or other biota in grassland ecosystems is limited or lacking.

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4.2.8. Crater

Condition Summary

Overall resource condition is good with an unchanging or stable trend; confidence in condition determination is low. Rationale is discussed in the "*Overall assessment*" section and following.

Relevance and Context

Haleakalā "Crater" is not a true volcanic crater, but rather a massive cinder cone-studded, multi-hued depression encompassing the upper regions of the volcano, approximately 7 miles (11.3 km) long and 2 miles (3.2 km) wide (NPS 1999). The depression was created by stream erosion and the coalescence of two amphitheater-headed valleys, Ke‘anae and Kaupō, from opposite sides of the mountain, and was later partially filled by subsequent volcanic activity (Stearns 1942, Figure 4.2.8-1). Haleakalā Crater was both the main feature inspiring the addition of the Haleakalā section to Hawai‘i National Park in 1916, and the original section's main geographic constituent (Jackson 1972, Nakamura 2010). Today, although various other parcels have been added to HALE, the Crater's landscapes remain highly popular with visitors and are those most readily identified with the park.

Figure 4.2.8-1. View from the western rim of Haleakalā Crater. Photo: Forest and Kim Starr.

For the purposes of this report, the Crater has been delineated by the HALE RM vegetation management units that fall mostly or completely within the depression described above, and that are not addressed in other sections (such as shrubland areas extending up from Kaupō Gap, which are included in the shrubland ecosystems section, Chapter 4.2.6) (Figure 4.2.8-2). Also included is a small management unit encompassing the sparsely vegetated upper western Crater rim and summit, because of the ecological similarity of this area with much of the western Crater. The total area treated in the present section therefore covers 11,431 ac (4,628 ha), or 40.5% of the park, and ranges in elevation from approximately 6,200 to 10,023 ft (1,890-3,056 m).

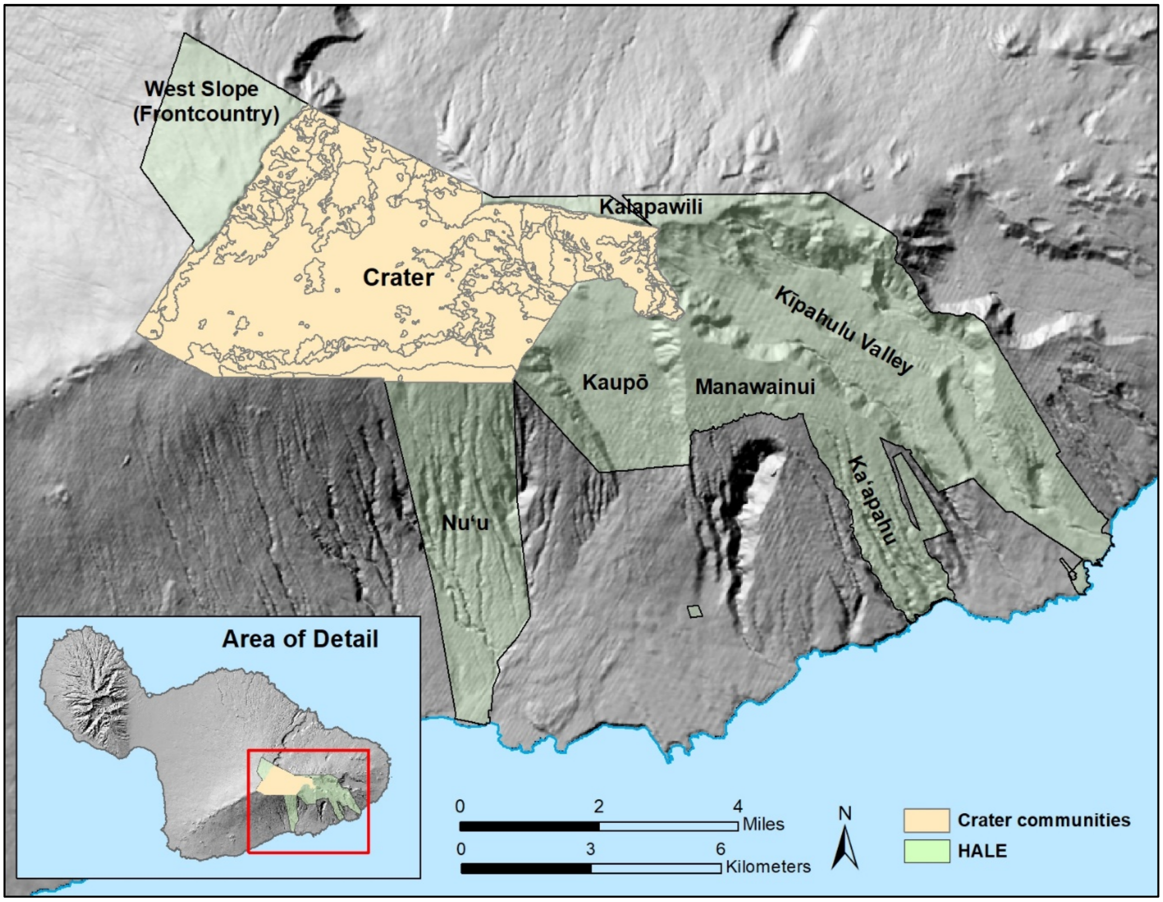

Figure 4.2.8-4. Haleakalā Crater in relation to other geographic units in the park. Polygons within Crater area indicate vegetation communities mapped by Green et al. (2015).

Plant Communities

Because the Crater is a geographic demarcation, rather than an ecological one, and because it encompasses a broad range of climatic conditions, it includes at least limited representations of several of the major ecosystem types assessed in prior sections. Mean annual rainfall, for example varies from approximately 1,040 mm (41 in) near the summit to over 3,600 mm (142 in) at the eastern end of the Crater (Giambelluca et al. 2013), and resulting biomes or major vegetation community types include grassland, shrubland, and pockets of forest and woodland (Figure 4.2.8-3, Green et al. 2015).

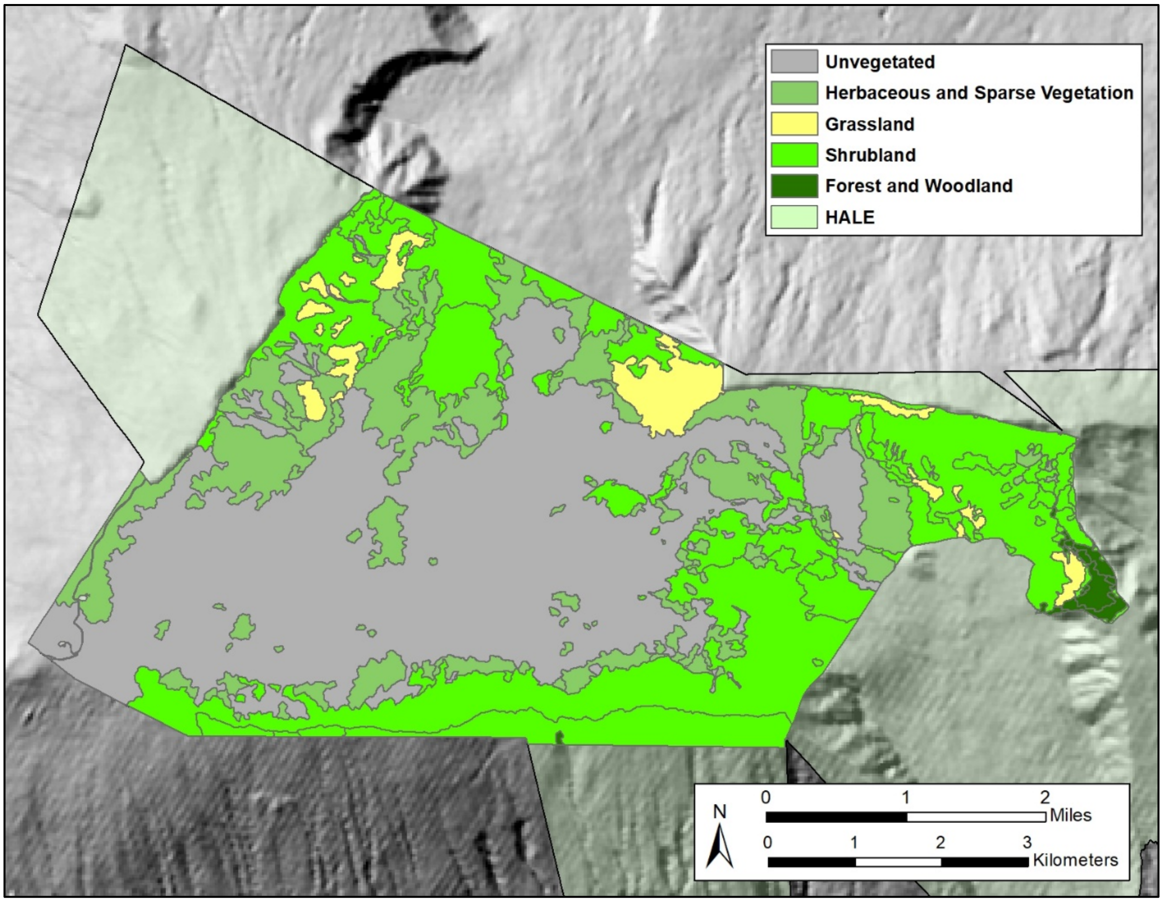

Figure 4.2.8-3. Major vegetation community types or biomes within the Crater region of HALE. Source: Green et al. (2015).

However, most of the Crater is positioned at or above the base height of the trade wind inversion (TWI), and therefore typically experiences clear and dry atmospheric conditions (Giambelluca and Nullet 1991). In addition, most soil substrates on the Crater floor are relatively young in age, ranging from approximately 750 to 30,000 years old, with large areas <5,000 years old (Sherrod et al. 2007, see Figure 4.1.2-1), and are consequently poorly developed and porous. The resulting environment is harsh, with the biota experiencing strong water and nutrient limitation, high solar radiation, and strong diurnal fluctuations in temperature (Leuschner and Schulte 1991; Pérez 2001, 2003, 2009). Much of the area supports only sparse vegetation, with the largest area classified as "unvegetated" (Figure 4.2.8-3), defined as having cover of vascular plant species less than 5% (Green et al. 2015). This cover class accounts for 4,918.9 ac (1,991.4 ha), or approximately 43% of the entire Crater area (Green et al. 2015). Despite this unvegetated designation, one of the most unique plant communities in the Hawaiian Islands, ʻĀhinahina/Naʻenaʻe (*Argyroxiphium*/*Dubautia*) Shrubland, occurs within this region (Gagné and Cuddihy 1999, Figure 4.2.8-4). This very exposed alpine dry community occurs predominantly on seemingly barren gravel and cinders, giving a nearly plant-free impression from a distance, but is characterized by one of Haleakalā's most conspicuous and charismatic endemic plant species, the federally threatened ʻāhinahina (Haleakalā silversword, *Argyroxiphium sandwicense* subsp. *macrocephalum*). Other plants occurring in these barren areas of the Crater include kūpaoa (or naʻenaʻe, *Dubautia menziesii*), a close relative of ʻāhinahina, pūkiawe (*Leptecophylla tameiameiae*), *Silene struthioloides*, the diminutive mountain daisy (*Tetramolopium humile* subsp. *haleakalae*), and the bunchgrass *(Deschampsia nubigena*).

Figure 4.2.8-4. ʻĀhinahina/Naʻenaʻe (*Argyroxiphium*/*Dubautia*) Shrubland. Photo: Forest and Kim Starr.

In addition to the threatened ʻāhinahina, several federally endangered plant taxa occur throughout the Crater (chapter 4.2.10). These are *Bidens micrantha* subsp. *kalealaha*, *Geranium multiflorum*, *Plantago princeps* var. *laxiflora*, *Sanicula sandwicensis*, and *Schiedea haleakalensis*. Another endangered species, *Ranunculus hawaiiensis*, was recorded in the Crater in 1945, but has not been documented recently.

Birds

Most native birds are relatively scarce throughout much of the Crater, owing to the low level of vegetative cover. In the eastern end of the Crater, near Palikū, denser shrubland and mesic forest habitat supports forest birds, including ʻapapane (*Himatione sanguinea*), Maui ʻamakihi (*Chlorodrepanis virens wilsoni*), ʻiʻiwi (*Drepanis coccinea*), and Maui ʻalauahio (Maui creeper, *Paroreomyza montana newtoni*). This area is also used extensively for nesting and foraging by the endangered nēnē (Hawaiian goose, *Branta sandvicensis*); nēnē densities are relatively low in most other parts of the Crater (NPS 2012, Figure 4.2.10-8). The pueo (Hawaiian short-eared owl, *Asio flammeus sandwichensis*) can be seen hunting over shrubland areas in the Crater, but is more numerous and nests primarily in grasslands (Conant and Stemmerman 1979). The migratory kōlea (Pacific golden plover, *Pluvialis fulva*), is sparingly present in the Crater during the fall to spring non-breeding seasons, although a few may stay through the summer (Conant and Stemmerman 1979). Perhaps the most widespread native bird in the Crater is the endangered ʻuaʻu (Hawaiian petrel, *Pterodroma sandwichensis*), which is found throughout much of the Crater during its spring to fall nesting season. Nests are placed in rock crevices, lava tubes, and in burrows excavated by the birds both on the Crater walls and on parts of the Crater floor (NPS 2012, Figure 4.2.10-11). Because it forages at sea during the day, this bird is rarely seen in the park by casual observers but is commonly heard calling at night. Two introduced gamebirds, the ring-necked pheasant (*Phasianus colchicus*) and chukar (*Alectoris chukar*), and a number of non-native passerine bird species, also reside in the Crater, primarily within shrubland habitats (Conant and Stemmerman 1979). The endangered Hawaiian hoary bat or ʻōpeʻapeʻa (*Lasiurus cinereus semotus*) has also been detected in the Crater, but the nature of its use or reliance on this ecosystem is currently unknown (see chapter 4.2.10).

Invertebrates

The Crater is home to a diverse invertebrate community. Shrubland and forested areas support invertebrates typical of those ecosystem types, but the sparsely vegetated central and western Crater areas, up to the summit, also support a highly unique, if less abundant, community of arthropods (Medeiros and Loope 1994). Some herbivorous species are tightly associated with several of the plant species found in this region. Examples include the planthopper *Nesosydne argyroxiphii*, the long-horned beetle *Plagithmysus terryi*, and the moth *Rhynchephestia rhabdotis*, all found on Haleakalā ʻāhinahina plants; the planthopper *Nesosydne tetramolopii* on *T. humile* plants; and the fruit fly *Trupanea cratericola* on both ʻāhinahina and kūpaoa. Other species are predators and scavengers of this aeolian environment, so named because outside, wind-borne resources like arthropods are thought to be important supplements to locally-derived resources (Howarth 1987). These species include *Mecyclothorax* and *Blackburnia* ground beetles, the large wolf spider *Lycosa hawaiiensis*, and flighted and flightless *Micromus* brown lacewings. As with shrubland invertebrate communities, the largest current threat is the combined pressures of the many established non-native invertebrates (Beardsley 1980), especially the highly invasive Argentine ant (*Linepithema humile*) and western yellowjacket (*Vespula pensylvanica*) (Cole et al. 1992, Wilson et al. 2009).

Impacts of Vertebrates

Like other areas of Haleakalā, the Crater has suffered from a long history of anthropogenic disturbance, such as the introduction of feral ungulates including cattle (*Bos taurus*), feral goats (*Capra hircus*), feral pigs (*Sus scrofa*) and axis deer (*Axis axis*). von Tempski (1940) reported that domestic cattle were first brought into the Crater in the late 1800s and were sometimes taken up to graze the Kalapawili grasslands. After his 1927 trip into Koʻolau Gap, Degener (1930) noted significant damage to native vegetation caused by the trampling and grazing of cattle. After park designation, cattle grazing finally ended in the 1930s (Loope et al. 1992).

Feral pigs were first reported in the Crater in the 1930s, presumably originating from coastal piggeries and accessing the park through Koʻolau Gap (Diong 1982). Pigs eventually became widespread in the Crater and surrounding forests. Browsing on native vegetation, digging and rooting, and dispersal of invasive weeds caused significant damage to native communities and biota (Loope et al. 1992). Because feral pigs are more elusive and solitary than goats, removal from the Crater has been difficult. Through a combination of aerial and ground shooting, and snaring, however, all known pigs were removed from within Crater fences by the end of 1993 (NPS 1999).

Feral goats were first introduced to the Hawaiian Islands by Captain Cook in 1778, and were later released on the island of Maui in the 1790s (Yocom 1967). Damage to the vegetation of the Crater and other park ecosystems was extensive, and by the time of his study, Yocom (1967) commented "I could find no place within the boundary of the park where goats had not grazed recently or in the past." Conspicuous damage to the Haleakalā silversword was reported by Ruhle (1959), as well as to the leguminous māmane tree (*Sophora chrysophylla*), in which goats had been observed eating twigs and leaves as high as they can reach (Yocom 1967). Prior to fencing and ungulate control, at least eight plant species had been extirpated from the park, while other vulnerable taxa persisted on ledges and other inaccessible sites (NPS 1999).

To mitigate the long history of ungulate-induced damage, fencing of the Crater district began along the north slope in 1976 (NPS 1999). In 1984, the Kaupō Gap boundary fence was completed. By the end of 1985, 10 miles of south boundary fence was constructed and, with the Kaupō Gap fence, provided a barrier to goats from Haleakalā's southern slopes. In 1985-86, the Park's north slope fence was completed and in 1986, the ridge separating the crater from Kīpahulu Valley was completed.

From 1984 to 1987, large-scale goat removal efforts were conducted in the crater, with a focus on western Kaupō Gap and Hana Mountain (NPS 1999). Over 5,300 goats were removed during that period. In October 1987, radio-collared "Judas" goats were released to assist resource management crews in locating the remnant herds. In 1988 an additional 1,050 goats were removed with help from the "Judas" goats. To facilitate the eradication of the remnant goat population, aerial shooting from helicopters was initiated in 1989, and by the end of 1993, the Crater was deemed free of non-collared goats (NPS 1999).

Impacts of Non-native plants

Ungulate disturbance and grazing have also resulted in the invasion of the Crater by many invasive, and potentially habitat-modifying non-native plants. Alien grasses, likely originating from grazing during the early 1900s, have come to dominate pasture-like areas below Holua and at Palikū, and many alien plant species were undoubtedly introduced to the Crater with horses (*Equus caballus*) and cattle (Loope et al. 1992). Significant weeds including blackberry (*Rubus argutus*), pampas grass (*Cortaderia jubata*), bull thistle (*Cirsium vulgare*), and pines *(Pinus* spp.), among others, pose a considerable threat to the ecological integrity of the Crater and its associated communities, and are the ongoing focus of control efforts by resources management staff (Loope et al. 1992, NPS 1999).

Like other high-elevation areas of HALE, the Crater has been experiencing warmer, sunnier and drier climatic conditions over the past few decades (Longman et al. 2015b). This may be related to an increase in the frequency of occurrence of the TWI in Hawaiʻi (Longman et al. 2015a), which has the effect of reducing the number of wet weather events in areas above its base height, such as the Crater. These conditions appear to be leading to more frequent or severe water stress and increased rates of mortality for at least ʻāhinahina plants (Krushelnycky et al. 2013, 2016), and possibly other associated plants in drier portions of the Crater.

The condition of natural resources within the Crater was assessed using two indicators: vegetation and invertebrates. Native birds are not considered here, because forest birds are a minor component in most of the Crater, and nēnē and ʻuaʻu are addressed primarily in the T&E Species section (Chapter 4.2.10).

Indicators

- Native Vegetation
- Native Invertebrates

Data and Methods

A review of available data and information was used to assess conditions, and if possible, trends, in the indicators listed above. Some additional analyses were performed on a subset of geospatial and vegetation classification and monitoring data collected by HALE and PACN staff specific to Crater vegetation communities (Green et al. 2015, PACN and Ainsworth 2016). Ecosystem-based summaries of invertebrate communities were extracted from data on the entire Summit District presented in Beardsley (1980). Summary analyses were also performed on HALE yellowjacket monitoring data (NPS 2017c).

Sources of information

Monitoring, outplanting, and propagule collection data for common, rare, and endangered native taxa have been recorded by HALE Vegetation Management staff for the entire park from 1993 through January 2016 (NPS 2017a).

Control data for invasive, non-native plants have been collected by HALE Vegetation Management staff for the entire park from 1983 through present (NPS 2017b).

NPS I&M vegetation classification data were collected by HALE and PACN staff in 220 circular plots (400 m^2) and observation points in 2011 (Green et al. 2015). NPS I&M also monitored 30 subalpine shrubland plots (20 x 50 m) in 2012-2013 as part of the Focal Terrestrial Plant Community monitoring system (Ainsworth et al. 2011, PACN and Ainsworth 2016), and recorded non-native plant frequency and abundance data along belt transects located in the subalpine shrublands of the West Slope and Crater by HALE and PACN staff (Gross et al. 2017).

Data on invertebrate diversity and distributions, and status of non-native species, were obtained from surveys of arthropods across the Summit District in 1975-77 (Beardsley 1980).

Information on rates of spread and ecological effects of the Argentine ant were taken from Cole et al. (1992), Krushelnycky et al. (2005b, 2011) and Krushelnycky and Gillespie (2008, 2010). Information on ecological effects of the western yellowjacket was taken from Gambino and Loope (1992) and Wilson et al. (2009), and unpublished data on monitoring and control of yellowjackets (NPS 2017c) was provided by HALE staff.

Information on ecological effects of non-native gamebirds and rodents was taken from Cole et al. (1995) and Cole et al. (2000).

Reference condition

Ideal reference conditions for the Crater would include fully intact native vegetation, uninvaded by non-native weeds. Similarly, invertebrate communities would be highly diverse in species and function, and uninvaded by non-native invertebrate species and unaffected by non-native vertebrates. Due to direct and indirect anthropogenic impacts, including the introduction of non-native plants and animals to the Hawaiian Islands, no high-elevation ecosystems currently exist in an entirely pristine state. A desirable reference condition for Crater vegetation is therefore that the richness and cover of native species exceeds that of non-native species. A desirable reference condition for Crater invertebrate communities is that native species richness exceeds non-native species richness, and that highly damaging invasive species like social insects are absent.

Condition and Trend

Native Vegetation

Green et al. (2015) have classified twenty-two vegetation community types or cover classes within the Crater, as defined in this report (Table 4.2.8-1). Although areas classified as unvegetated (i.e. <5% vascular plant cover) account for the single largest cover class within the Crater, seventeen vegetation communities have native species as the most common or abundant plant within that community type, and account for 51.6% of all cover within the Crater (Figures 4.2.8-5 and 4.2.8-6, Table 4.2.8-1). In addition, the unvegetated areas also include some sparse cover classified as *Dubautia menziesii* - (*Argyroxiphium sandwicense*) Subalpine Sparse Vegetation, but barely reach the threshold for that designation (Green et al. 2015). In contrast, communities with non-native plants as the most common or dominant species account for 517.2 ac (209.3 ha) of the Crater, only 4.5% cover for the entire area (Figures 4.2.8-5 and 4.2.8-6, Table 4.2.8-1).

Table 4.2.8-1. Crater vegetation communities, including unvegetated cover class. Source: Green et al. (2015).

Community Classification	Area (ha)
Unvegetated	1,991.4
Leptecophylla tameiameiae Subalpine Dry Shrubland	824.0
Leptecophylla tameiameiae - Vaccinium reticulatum Subalpine Mesic Shrubland	493.8
Pteridium aquilinum ssp. decompositum Herbaceous Vegetation	297.0
Dubautia menziesii - (Argyroxiphium sandwicense) Subalpine Sparse Vegetation	261.9
Leptecophylla tameiameiae - (Vaccinium reticulatum) Montane and Subalpine Sparse Vegetation	220.0
Holcus lanatus Semi-natural Grassland	172.6

Table 4.2.8-1 (continued). Crater vegetation communities, including unvegetated cover class. Source: Green et al. (2015).

Community Classification	Area (ha)
Sophora chrysophylla - (Coprosma montana - Leptecophylla tameiameiae - Dodonaea viscosa) Subalpine Shrubland	131.6
Leptecophylla tameiameiae - Dodonaea viscosa Montane Shrubland	59.5
Deschampsia nubigena Montane and Subalpine Mesic Herbaceous Vegetation	42.6
Leptecophylla tameiameiae - Sadleria cyatheoides Subalpine Mesic Shrubland	41.8
Ageratina adenophora - (Sporobolus africanus - Anthoxanthum odoratum) Semi-natural Herbaceous Vegetation	35.5
Metrosideros polymorpha / Sadleria cyatheoides Forest	16.6
Metrosideros polymorpha - Cheirodendron trigynum / (Cibotium spp.) Montane Wet Forest	13.9
Leptecophylla tameiameiae - Vaccinium reticulatum - (Geranium cuneatum) / Deschampsia nubigena Mesic Shrubland	11.3
Metrosideros polymorpha / Leptecophylla tameiameiae - Dodonaea viscosa Montane Woodland	4.7
Built-up Commercial and Services	3.2
Rubus hawaiensis Montane Wet Shrubland	3.1
Metrosideros polymorpha / Rubus hawaiensis Montane Wet Forest	1.3
Cenchrus clandestinus Semi-natural Herbaceous Vegetation	1.2
Metrosideros polymorpha / Dryopteris wallichiana Mesic Forest	0.9
Myrsine lessertiana - (Metrosideros polymorpha) / Coprosma foliosa - Dodonaea viscosa Montane Mesic Forest	0.4

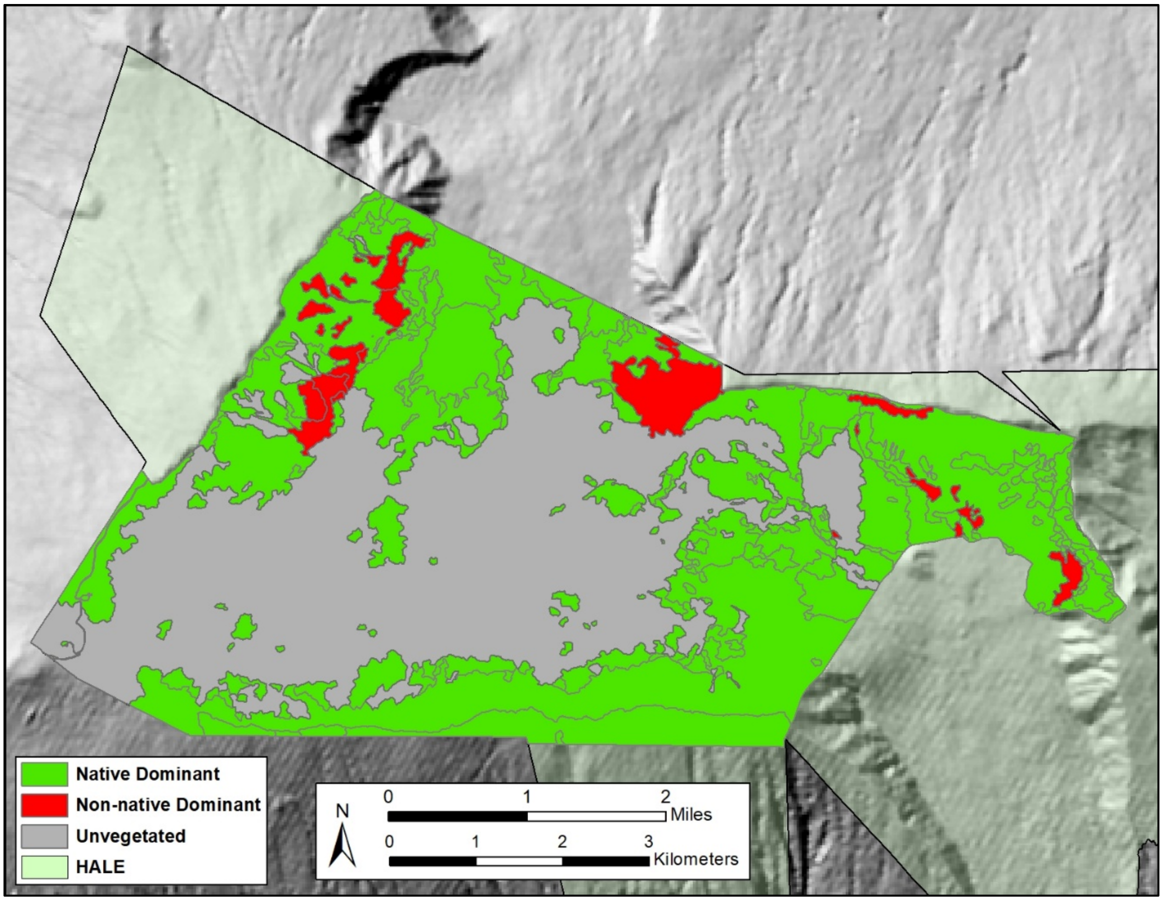

Figure 4.2.8-5. Native and non-native dominant vegetation and unvegetated areas within HALE Crater. Source: Green et al. (2015).

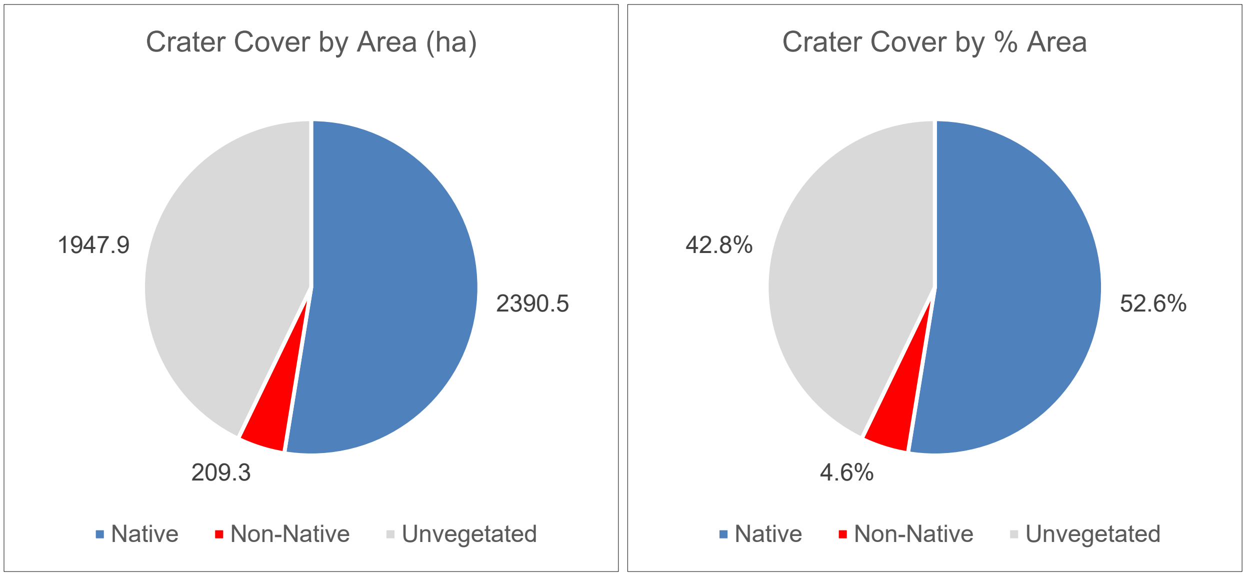

Figure 4.2.8-6. Area (hectares) and percent cover of native dominant and non-native dominant vegetation within HALE Crater. Source: Green et al. (2015). Native dominant communities contain varying degrees of non-native plant cover, and non-native dominant communities may contain native species.

Excluding the unvegetated cover class, of the 22 vegetation community types in the Crater, the five largest by area have native plants as the most common or dominant species, and three include the native shrub *Leptecophylla tameiameiae* (Table 4.2.8-1). The largest single vegetation community in the Crater is *Leptecophylla tameiameiae* Subalpine Dry Shrubland, characterized by a moderately open *Leptecophylla tameiameiae* dominated shrub layer (20-30% cover) with a depauperate understory (Green et al. 2015). Other abundant species in this community type include the native grasses *Agrostis sandwicensis* and *Deschampsia nubigena*, the native shrub *Geranium cuneatum* var. *tridens*, the native herb *Tetramolopium humile* ssp. *haleakalae*, and the non-native herb *Hypochaeris glabra* (Green et al. 2015). This community type is also found on the West Slope and in Kaupō Gap.

As previously mentioned, *Dubautia menziesii* - (*Argyroxiphium sandwicense*) Subalpine Sparse Vegetation, classified as ʻĀhinahina/Naʻenaʻe (*Argyroxiphium/Dubautia*) Shrubland by Gagné and Cuddihy (1999), makes up a relatively small, but conspicuous portion of the cover in the otherwise unvegetated western portion of the Crater (Figure 4.2.8-3, Table 4.2.8-1). In addition to *D. menziesii* and *A. sandwicense,* other common taxa include the native herbs *T. humile* ssp. *humile* and *T. humile* ssp. *haleakalae*, and native grass *Trisetum glomeratum* (Green et al. 2015).

At the far eastern end of the Crater, higher annual precipitation totals and cooler temperatures create an environment more similar to the adjacent wet forests of Kīpahulu Valley than to the rest of the Crater. This is reflected in the relatively small, but otherwise distinct community types found nowhere else in the Crater. Three communities, including *Metrosideros polymorpha* - *Cheirodendron trigynum* / (*Cibotium* spp.) Montane Wet Forest, *Rubus hawaiensis* Montane Wet Shrubland, and *Metrosideros polymorpha* / *Rubus hawaiensis* Montane Wet Forest, account for only 0.4% of the total Crater cover, but also occur in higher rainfall areas on the outer slopes and adjoining valleys of the park (Green et al. 2015; note that species denoted with parentheses within community type names indicates that those species may or may not actually be present in a given area, for example *Cibotium* is not present in the Crater).

Although not a comprehensive list of all plant taxa found in the Crater, Green et al. (2015) provide a broad sampling of species through a series of 38 vegetation inventory and mapping plots. A total of 34 native and 25 non-native plant species were recorded in these plots, with a mean species richness of 8.3 ± 0.7 natives and 4.2 ± 0.4 non-natives per plot (Figure 4.2.8-7). Mean native species cover was 46.2% ± 4.3%, more than double the 20.0% ± 3.9% mean non-native species cover (Figure 4.2.8-7).

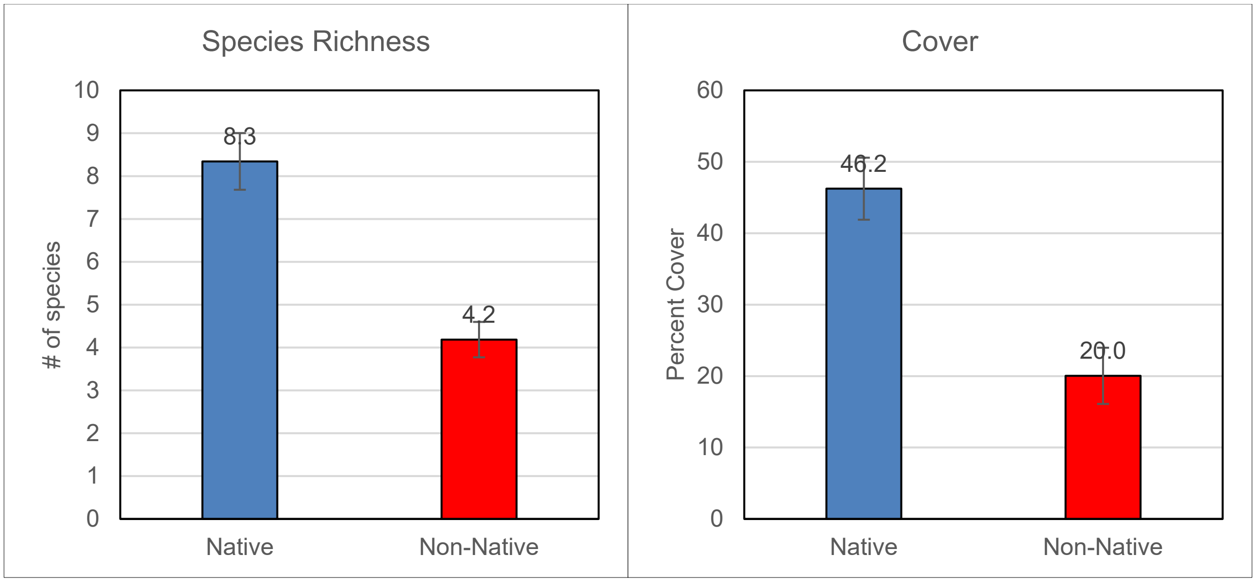

Figure 4.2.8-5. Species richness and cover in Crater inventory and mapping plots (n = 38). Source: Green et al. (2015).

A similar pattern is evident from the first round of monitoring in the NPS I&M Focal Terrestrial Plant Community (FTPC) subalpine shrubland plots, 17 of which (out of 30) were located within the Crater (PACN and Ainsworth 2016). These recorded a mean species richness of 8.94 ± 1.05 natives and 5.12 ± 0.64 non-natives per plot (Figure 4.2.8-8). Mean native cover in the FTPC plots was 32.88% ± 7.28%, compared to 16.71% ± 6.37% cover for non-natives species (Figure 4.2.8-8).

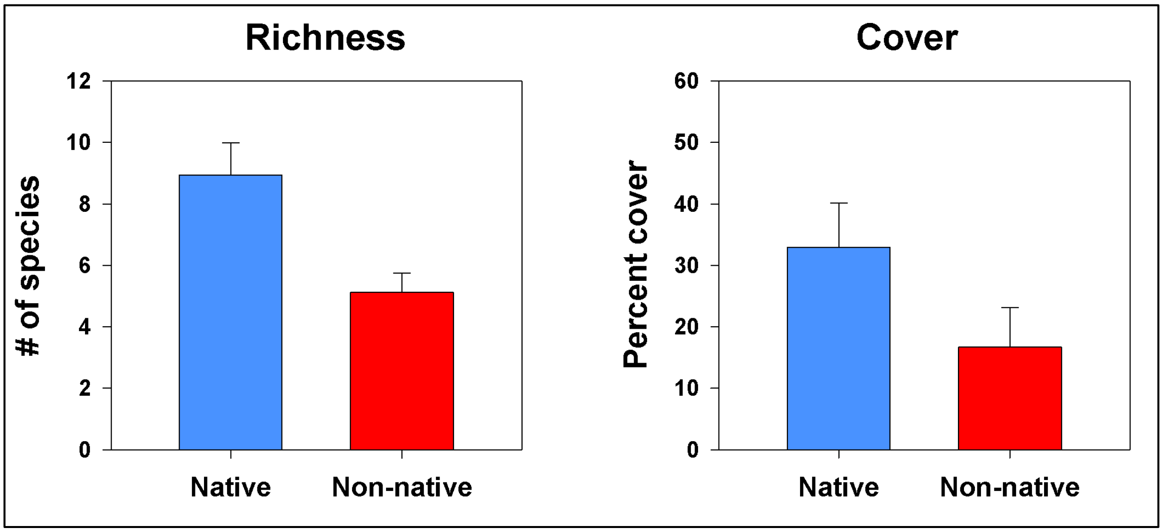

Figure 4.2.8-8. Mean (±SE) native and non-native plant species richness (left) and percent cover (right) recorded within 1,000 m^2 FTPC monitoring plots in the Crater (n = 17). Source: PACN and Ainsworth (2016).

Data collected in NPS I&M efforts to monitor the status of Established Invasive Plant Species (EIPS) at HALE suggest a somewhat different pattern. Thirteen 500 m long belt transects (consisting of 25 contiguous 5 x 20 m plots) were established within the Crater (as defined in this report), and baseline monitoring of non-native plants along the transects was completed in 2012 (Gross et al. 2017). On six of the transects, mean cover of non-native plants was <1%, was between 1% and 5% on an additional five transects, and was between 5% and 10% on the remaining two transects. If one assigns a midpoint value in each cover class range, the 13 transects averaged 2.5% cover of non-native plants overall, which is considerably lower than the 20.0% and 16.7% cover measured in the inventory and FTPC plots, respectively (Figures 4.2.8-7 and 4.2.8-8). The total number of non-native plant species per EIPS transect ranged from 1 to 12, and averaged 6.5 species across the 13 transects (Gross et al. 2017). This is slightly higher than the average 4.2 non-native species recorded in the inventory plots and the average 5.12 non-native species recorded in the FTPC plots, although a higher richness on the transects would be expected since each transect covers 2,500 m^2, compared to only 400 m^2 and 1,000 m^2 covered in the inventory and FTPC plots, respectively.

Invasive Plants

In spite of the differences in these values, all three sets of plots or transects suggest that the Crater is dominated by native plants, with the exception of several areas dominated by alien grasses and sometimes also the alien herb *Ageratina adenophora* (red areas in Figure 4.2.8-5; Table 4.2.8-1). Several problematic invasive plants, however, threaten to change this situation. These plants have been actively targeted by park vegetation management staff since 1983 because of their ability to persist in the absence of disturbance, their potential to invade a much wider area than currently occupied, their dispersibility, and their potential to modify the structure and function of the Crater's unique communities (Table 4.2.8-2). Due to the widespread nature of these and other invasive non-native plants on Maui, none of these species are targeted for eradication, with the possible exception of pampas grass (*C. jubata*). Management will therefore be necessary in perpetuity to contain their spread and to prevent them from causing irrevocable damage to the Crater ecosystems and biota.

Table 4.2.8-4. Numbers of high priority invasive weeds controlled in the Crater. Source: NPS (2017b).

Taxon	Number Controlled
Cirsium vulgare (1983-2017)	53,298
Cortaderia jubata (1990-2015)	145
Heterotheca grandiflora (1992 – 2017)	603,265
Pinus radiata (includes *Pinus* spp.) (1984-2017)	4,577
Rubus argutus (1983-2017)	238,244
Senecio madagascariensis (2010 -2017)	1,019
Trifolium arvense (1997 – 2017)	247,234
All Non-Native Plant Species (1983 – July 2017)	1,263,089 individuals

Pines, particularly *Pinus radiata*, *P. patula* and *P. pinaster*, are aggressively targeted for control within the subalpine shrubland and Crater ecosystems. These invasive trees possess wind-dispersed seeds capable of spreading long distances from parents, can establish in and overtop shorter-statured native communities, and can eventually convert them into fire-prone coniferous monocultures (Burns and Honkala 1990, Loope et al. 1992). Monterey pine (*P. radiata*) is currently the most common pine invading the Crater. This species was uncommon before 2007, when a large fire southwest of the crater burned a ~600-ac (243 ha) plantation, sending many seeds upslope and into the Crater with the wind; it is hypothesized that this event may be at least partly responsible for the recent pine invasion in the park (P. Welton pers. comm. 2018). Aerial herbicide applications using helicopters with a ball sprayer attached to a long-line have been employed since 2014 to control pines invading the steep, inaccessible walls on the western end of the Crater. Gross et al. (2017) documented *Pinus radiata* scattered across the northwest corner of the subalpine shrubland, and *Pinus patula* in one Plant Community plot in the northwest corner of the subalpine shrubland, but not within the Crater, likely because the monitoring transects were located on relatively flat terrain where aggressive management by the Vegetation Management Program has removed most of the invading pines.

Rubus argutus (Florida blackberry) is an invasive shrub capable of spreading by rhizomes, aerial runner branches, and bird-dispersed seeds, and can establish thorny thickets within native shrublands, grasslands and forest communities (Loope et al. 1992, Medeiros et al. 1998). In the Crater, it is being controlled in the Palikū pasture and adjacent areas. *Senecio madagascariensis* (fireweed) is an invasive annual or short-lived perennial herb that can spread long distances via its prolific wind-dispersed seeds and establish and form dense cover in subalpine and alpine habitat (Ramadan et al. 2011). Large infestations outside the park boundary provide a perpetual seed source, and this weed is becoming increasingly common not only in disturbed areas like the mule pasture, but also throughout the West Slope shrublands. Gross et al. (2017) recorded fireweed at low cover in the northwest portion of the subalpine shrubland. *Heterotheca grandiflora* (telegraph plant) is an aromatic annual or biennial herb with wind-dispersed seeds that can become quite dominant in level areas where cinder has been disturbed by winter flooding (Medeiros et al. 1998). It is actively being controlled by volunteer groups in the Crater. Gross et al. (2017) recorded low cover of telegraph plant scattered throughout the Crater shrubland.

Trifolium arvense (rabbit-foot clover) is an annual, mat-forming herb naturalized in disturbed, arid to wet areas such as along roadsides, on lava, and in open meadows on Maui and Hawaiʻi (Wagner et al. 1999). In the Crater, it can form dense ground cover that may inhibit seedling recruitment of desirable native vegetation. Gross et al. (2017) did not comment on the status of rabbit-foot clover in their invasive plant summaries. *Cirsium vulgare* (bull thistle) is a spiny, biennial herb with wind-dispersed seeds that readily establishes in dry to mesic areas on all the main islands (Wagner et al. 1999). In open Crater habitat, it formerly formed dense thickets (Haselwood et al. 1983), but now is outcompeted by recovering vegetation and non-native grasses, and is not currently a high priority target weed (P. Welton pers. comm. 2018). Gross et al. (2017) recorded bull thistle at two locations within the subalpine shrubland, both along transects within the Crater.

To accelerate the natural recovery of native vegetation following removal and exclusion of feral ungulates, NPS staff have conducted outplanting and restoration efforts of both rare and common plant taxa throughout the natural communities of the Crater, including all six of the extant threatened and endangered species known to occur (Table 4.2.8-3). Welton (2016) describes a significant effort at restoration of the subalpine woodland ecosystem in the Central Crater Region near Honokahua during the period from 2003-2014 (Figure 4.2.8-9); restoration at this site continues to the present. Although not an exact approximation of habitat condition, Berger et al. (1975) described a site near ʻŌʻili Puʻu, just under 2 km from the Honokahua restoration area, as having considerable disturbance, and remarked that "browsing by goats may have removed all the young seedlings so that all that is left is the mature remnant of the potential vegetation." They also failed to find any seedlings of *Sophora chrysophylla*, *Dodonaea viscosa* or *L. tameiameiae*, three otherwise common plants in the region.

Table 4.2.8-3. Numbers of native plant taxa outplanted and seeds sown in the Crater. Source: NPS (2017a).

Taxon	Plants	Seeds	T&E
Argyroxiphium grayanum	103	0	NA
Argyroxiphium sandwicense ssp. macrocephalum	3,370	222,215	T
Argyroxiphium sandwicense ssp. macrocephalum X Dubautia menziesii	54	0	NA
Bidens micanthra ssp.kalealaha	672	0	E
Chenopodium oahuense	52	1,200	NA
Coprosma ernodeoides	987	0	NA
Coprosma montana	1,450	0	NA
Deschampsia nubigena	538	1,553,600	NA
Dianella sandwicensis	4	0	NA
Dodonaea viscosa	2,763	0	NA
Dubautia menziesii	197	0	NA
Geranium multiflorum	125	0	E
Leptocophylla tameiameiae	1	0	NA
Osteomeles anthyllidifolia	613	0	NA
Pittosporum confertiflorum	34	0	NA
Plantago princeps var. *laxiflora*	134	0	E
Rumex giganteus	103	0	NA
Santalum haleakalae	30	0	NA
Sanicula sandwicensis	43	0	E
Schiedea haleakalaensis	118	10	E
Sophora chrysophylla	4,098	0	NA

Table 4.2.8-3 (continued). Numbers of native plant taxa outplanted and seeds sown in the Crater. Source: NPS (2017a).

Taxon	Plants	Seeds	T&E
Stenogyne microphylla	189	0	NA
Trisetum glomeratum	43	0	NA
Vaccinium reticulatum	440	0	NA
Viola chamissoniana ssp.tracheliifolia	159	223	NA
Total	**16,326**	**1,777,248**	–

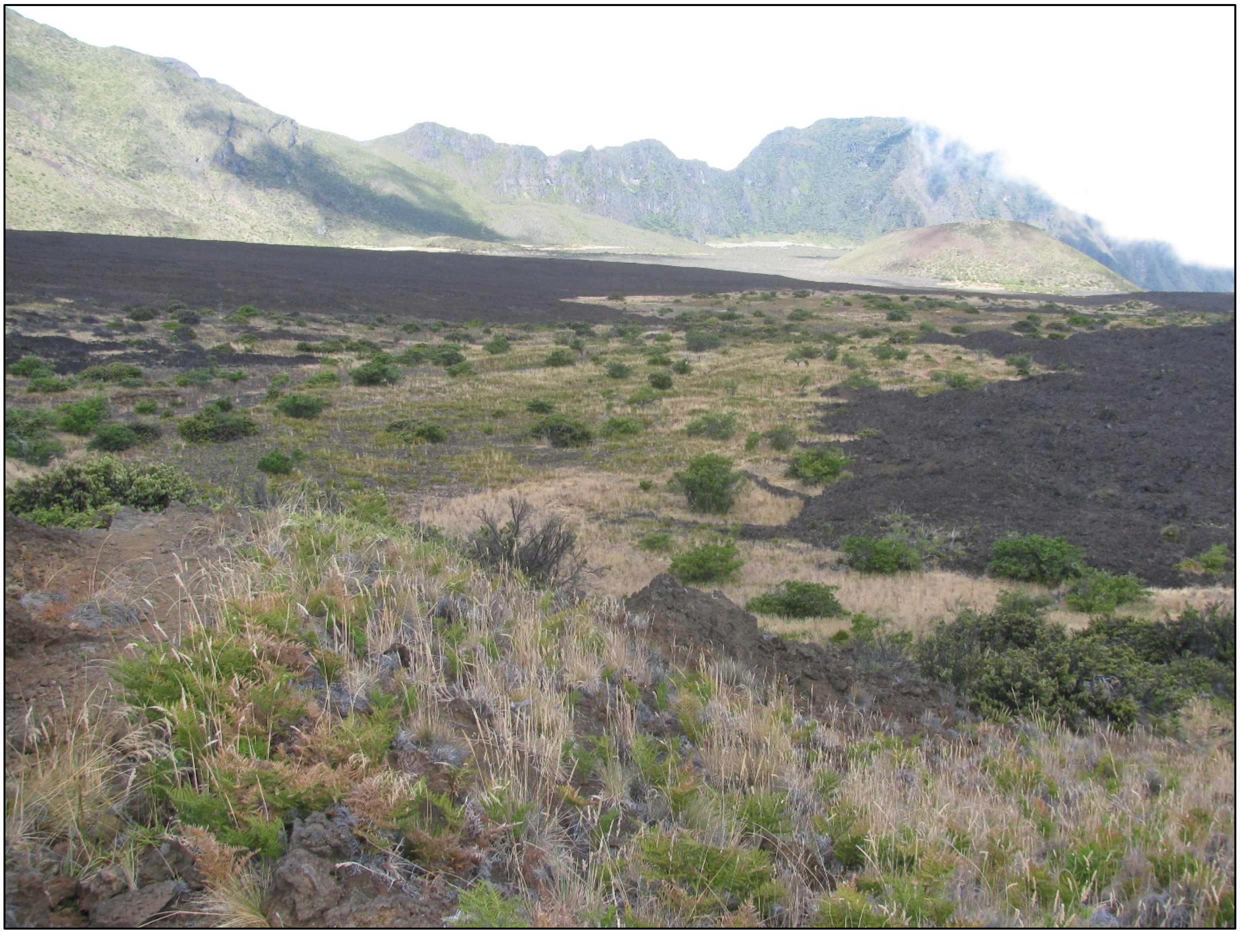

Figure 4.2.8-9. Sparse remnant vegetation in the Central Crater between Honokahua and ʻŌʻili Puʻu. Photo: Forest and Kim Starr.

The Honokahua site was selected for its biological importance to native flora and fauna, its proximity to two small exclosures in the area, and because natural recruitment by common native taxa was lacking despite absence of browsing pressure since 1986. Of the 9,152 individuals of 17 native species outplanted into the 13-ac restoration site between 2003 and 2014, only 1,669 (18.2%) were still alive in 2014, emphasizing the harsh ecological conditions to which plants are subjected in the higher elevations of the park. Since then, 2,776 more were planted and have not been remonitored (P. Welton pers.comm. 2018). Despite the low overall survival rate, almost 16% of the surviving outplants were reproductive, and natural recruitment was also observed. In addition, the distribution of two endangered plant species (*B. micrantha* subsp. *kaleahala* and *G. multiflorum*) was expanded and local species diversity has been increased (Welton 2016). Similar efforts throughout the Crater can be expected to slowly augment natural recruitment and expand native cover and diversity in previously modified native communities.

Non-native Vertebrates

Smaller, non-ungulate vertebrates also have potential effects on Crater vegetation. Although direct data from the Crater are not available, dynamics can be expected to be similar to those outlined for shrubland ecosystems at HALE (chapter 4.2.6). Namely, the introduced gamebirds (ring-necked pheasants and chukars), as well as nēnē where they occur, are likely to be important seed dispersers for native plants like *L. tameiameiae*, *Vaccinium reticulatum*, *Coprosma ernodeoides*, and *Coprosma montana* (Cole et al. 1995). Non-native rodents (mice, *Mus musculus*, and black rats, *Rattus rattus*), on the other hand, feed heavily on fruits of *Vaccinium* and *Coprosma*, but likely accomplish little or no seed dispersal (Cole et al. 2000). However, population-level impacts are most likely limited to relatively rare and highly preferred plant species like *Santalum haleakalae* (Cole et al. 2000). An additional potential dynamic involves the importation of marine nutrients by ʻuaʻu. Seabirds are known to significantly augment terrestrial nutrient budgets, especially nitrogen and phosphorus, by foraging for animal prey at sea and depositing waste at their breeding sites on land (Fukami et al. 2006, VanderWerf et al. 2014). The increasing numbers of ʻuaʻu in the Crater (chapter 4.2.10) may therefore be benefitting Crater plants, both native and non-native, but this potential effect is unstudied at HALE.

Invertebrates

Other potential effects on Crater vegetation include those caused by invertebrates. Ironically, some of the earliest entomological attention in the Crater concerned the hypothesized detrimental effects of native insect species like *R. rhabdotis* and *T. cratericola* on reproduction of ʻāhinahina plants (Swezey and Degener 1928, Degener 1930). Both insects are seed predators of ʻāhinahina, and were thought to represent a major impediment to the recovery of the ʻāhinahina population, which had reached a critical low due to ungulate browsing and human collection (Loope and Crivellone 1986). However, Kobayashi (1974) argued that these endemic insects should not severely threaten their natural host plants if other pressures were removed, and Krushelnycky (2014) confirmed that their impacts on ʻāhinahina reproduction appear to be relatively minor. However, the invasive Argentine ant may pose a more legitimate threat to reproduction of ʻāhinahina, and possibly other species reliant on insect pollination, through its impacts on native insects like *Hylaeus* yellow-faced bees (Cole et al. 1992). Krushelnycky (2014) also examined this question, and found that while *Hylaeus* floral visitation rates on ʻāhinahina were reduced by roughly 50% in an Argentine ant-invaded area, this did not translate into a reduction in the plant's seed set rate. However, Krushelnycky (2014) cautioned that Argentine ants have so far only invaded a small portion of the total ʻāhinahina range, and that further invasion and more widespread suppression of bee or other insect pollinators could impact plant reproduction. The invasive western yellowjacket, which is known to prey on *Hylaeus* bees, Lepidoptera, and a wide variety of other insects at HALE (Gambino and Loope 1992, Wilson et al. 2009), and to strongly reduce floral visitation by *Hylaeus* bees (Wilson and Holway 2010), may have similar effects on pollination (e.g. Hanna et al. 2013).

Climate Change

An additional threat to ʻāhinahina, and potentially other Crater plants, may be arising from changing climate. As noted above, ʻāhinahina reached a population low in the early 20th century, with total numbers estimated between about 11,000 and 18,000 in 1935 (Krushelnycky et al. 2016, Figure 4.2.8-10). Following the initiation of protective measures with National Park designation, including ungulate management and the prohibition of plant collecting (Loope and Crivellone 1986), the ʻāhinahina population made a dramatic recovery through the 1980s (Figure 4.2.8-10). Beginning around 1990, however, the population trajectory reversed and has subsequently declined by about 60%. This reversal was synchronous with an upward shift in the prevalence of the TWI around the early 1990s, which was followed by warmer, drier and sunnier atmospheric conditions (Longman et al. 2015a, b; Krushelnycky et al. 2016). The population decline has resulted from a combination of low seedling recruitment and mortality of immature plants, with rates of mortality higher at the lower elevational end of the population range (Krushelnycky et al. 2013). Population models indicate that if similar demographic trends continue, there is a very high probability that ʻāhinahina will become extinct throughout most of its current range within the next 100 years (Krushelnycky 2016).

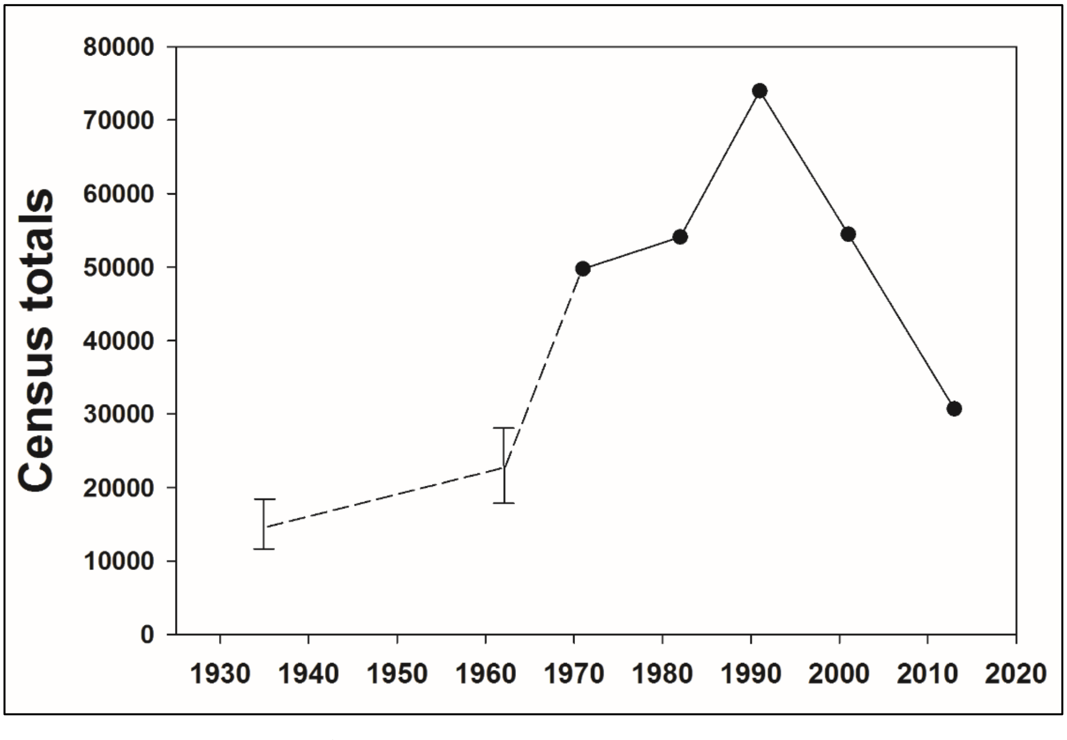

Figure 4.2.8-10. Estimated ʻāhinahina population trend over time, as indicated by total population censuses roughly every decade from 1971 to 2013. Population estimates for 1935 and 1962, with lower and upper bounds, were projected from counts on a single cinder cone. Source: Krushelnycky et al. (2016).

Comparable longitudinal data do not exist for other Crater plants, making it unclear whether changing climate conditions are also affecting other species, or whether ʻāhinahina are especially sensitive to the changes that have occurred thus far. However, anecdotal observations of a recent mortality event involving *L. tameiameiae* in HALE shrublands (see chapter 4.2.6), and other anecdotal observations of apparently increased mortality of other crater plants, raise concern that this is, or may soon become, a wider phenomenon. Moreover, warmer and drier conditions increase the risk of fire in the Crater. As noted in the shrubland ecosystems section, fire was pre-historically infrequent on upper Haleakalā (Burney et al. 1995, Crausbay et al. 2014), and many native Hawaiian plants are believed to be ill-adapted to frequent fire (Smith and Tunison 1992). Fires in the Crater are likely to inhibit regeneration of native species and promote the spread of fast-growing, fire-tolerant non-native plants (Loope et al. 1992, Smith and Tunison 1992).

Summary of Native Vegetation

Despite the threats from invasive plants, animals and changing climate, Crater vegetation is judged to currently be in good condition. With the exception of long-term demographic data for ʻāhinahina, no repeated vegetation data sets have been collected within HALE Crater. However, Hughes et al. (2014) documented changes in subalpine shrubland communities in the adjacent Waikamoi Preserve between 1994 and 2008 (14 years after ungulate removal), and reported dramatic increases in native cover, and decreases in non-native grass cover. Similar responses to ungulate exclusion from the Crater are thought to have occurred, and are reflected in the much higher native than non-native species richness and cover in this area of the park. The native-dominated status in these areas has further been maintained and augmented through a combination of native outplanting and restoration activities, and targeted control of a select group of potentially habitat-modifying invasive plants. Despite these gains, there is now clear evidence for recent population declines for ʻāhinahina, a dominant species in one important and widespread vegetation community type in the Crater. Because of these contrasting trends, we assign an unchanging trend to the condition of Crater vegetation. Confidence in this assessment is low, owing to the lack of repeated data to confirm the presumed overall recovery of native vegetation after ungulate removal, and because the wider relevance of recent trends for ʻāhinahina are as yet unclear.

Native Invertebrates

There were several early entomological collecting trips to the summit region, first by Thomas Blackburn in 1878, followed by R.C.L. Perkins in 1894 and 1896, with Perkins also collecting in at least several locations in the western Crater (Beardsley 1980, Krushelnycky et al. 2005a, Liebherr 2015). These efforts discovered a variety of unique and rare high-elevation arthropod species, such as a suite of about ten predatory ground beetle species (Carabidae) restricted to the upper portion of Haleakalā volcano (Lieberr 2015). While other focused collecting efforts followed, no comprehensive surveys of Crater invertebrates were made until Beardsley's 1975-1977 insect survey (Beardsley 1980), that was part of the Resources Basic Inventory for the park (Berger et al. 1975). This was also the last such survey within the Crater, and therefore is the primary source of information on the status of Crater invertebrates.

Beardsley (1980) surveyed 55 sites across the West Slope, summit, Crater and its periphery, and Kaupō Gap. As reported in the forest ecosystem section (Chapter 4.2.3), Beardsley collected 182 insect species, 127 (70%) of which were native, in forested areas around Palikū, which fall within the boundaries of the Crater as defined in the present NRCA report. Excluding these Palikū sampling sites, Beardsley collected 101 insect species, 63 (62%) of which were native, throughout the remainder of the Crater. This suggests that the non-forested areas of the Crater supported lower diversities of insects compared to the wetter Palikū area, as expected, and that insect communities in non-forested areas were slightly more invaded by non-native species than were communities in the small forested section around Palikū. Both areas, however, were less invaded than the West Slope and Kaupō shrubland areas surveyed (53% of species native overall, Chapter 4.2.6).

A series of arthropod surveys conducted since 2003 around the astronomy observatory facilities near the summit, just outside the park boundaries, provide some additional information (Brenner 2003, 2007, 2009, 2010a, 2010b, 2010c, 2011a, 2011b, 2012a, 2012b, 2013). The most productive of these surveys, in 2003 and 2009, collected 58 and 59 species, respectively; 29 (50%) and 27 (46%) of the species were thought to be native in the two surveys (Brenner 2003, 2009). Overall, these surveys found 47% of the sampled species to be native, on average. The slightly higher proportion of non-native species at this summit site may reflect the closer proximity to sources of introduction, namely the fairly abundant traffic and activity in the summit region, both within the park and the adjacent observatories.

Referring to the entire Summit District (called Crater District in his report), Beardsley concluded that the status of the region's insect fauna was "not very satisfactory." This assessment was due in part to the damaging effects of invasive goats and pigs on native vegetation that is the foundation for the native insect fauna, as well as to the numerous non-native insect species established. Goats and pigs have been removed from these ecosystems, resulting in presumed recovery of native vegetation (see above section, and chapter 4.2.6), but the problem of non-native arthropod invasion remains. Unfortunately, more recent comprehensive assessments do not exist for Crater invertebrate communities, so it is difficult to be certain of the current scale of the problem. However, it is likely that the situation is similar to that in adjacent shrublands of the West Slope and Kaupō, which appear to be accumulating more non-native invertebrate species over time (chapter 4.2.6).

Argentine ant

As in West Slope shrublands, perhaps the most significant threat to native Crater arthropod communities is the expanding invasion of the Argentine ant. There are two large populations of Argentine ants in the park (Figure 4.2.6-11). The upper population was apparently initiated at Kalahaku Overlook on the west rim of the Crater, and subsequently spread down the Crater walls and has begun spreading across the western Crater floor (Krushelnycky et al. 2005b, Figure 4.2.8-11). Before management efforts aimed at containing this spread were initiated in 1997, rates of outward expansion on the Crater floor were some of the fastest measured among all regions of the ant population boundaries, averaging about 100 m/year, and were most likely driven by warm relative temperatures at this location (Krushelnycky et al. 2005b, Hartley et al. 2010). As discussed in the shrubland ecosystems section (chapter 4.2.6), the containment efforts were curtailed in 2004, owing to insufficient effectiveness of the method (Krushelnycky et al. 2011), and this was the last year that the complete perimeters of the two populations were mapped (Figures 4.2.6-12, 4.2.8-11). However, one section of the upper population boundary near Hōlua in the western Crater was mapped in September of 2008 (P. Krushelnycky unpub. data), and indicated that ants were once again rapidly spreading in this region of the Crater (Figure 4.2.8-11). The population is known to have spread even further since 2008 (P. Krushelnycky pers. obs.), although the boundary has not been mapped more recently. Of particular concern is that the ant population has now overtaken the Hōlua campground and visitor cabin, increasing the risk of human-mediated dispersal to other parts of the Crater. It has been estimated that approximately 75% of subalpine habitats in the park, including most of the Crater, is likely suitable habitat for the ant (Krushelnycky et al. 2005b, Hartley et al. 2010).

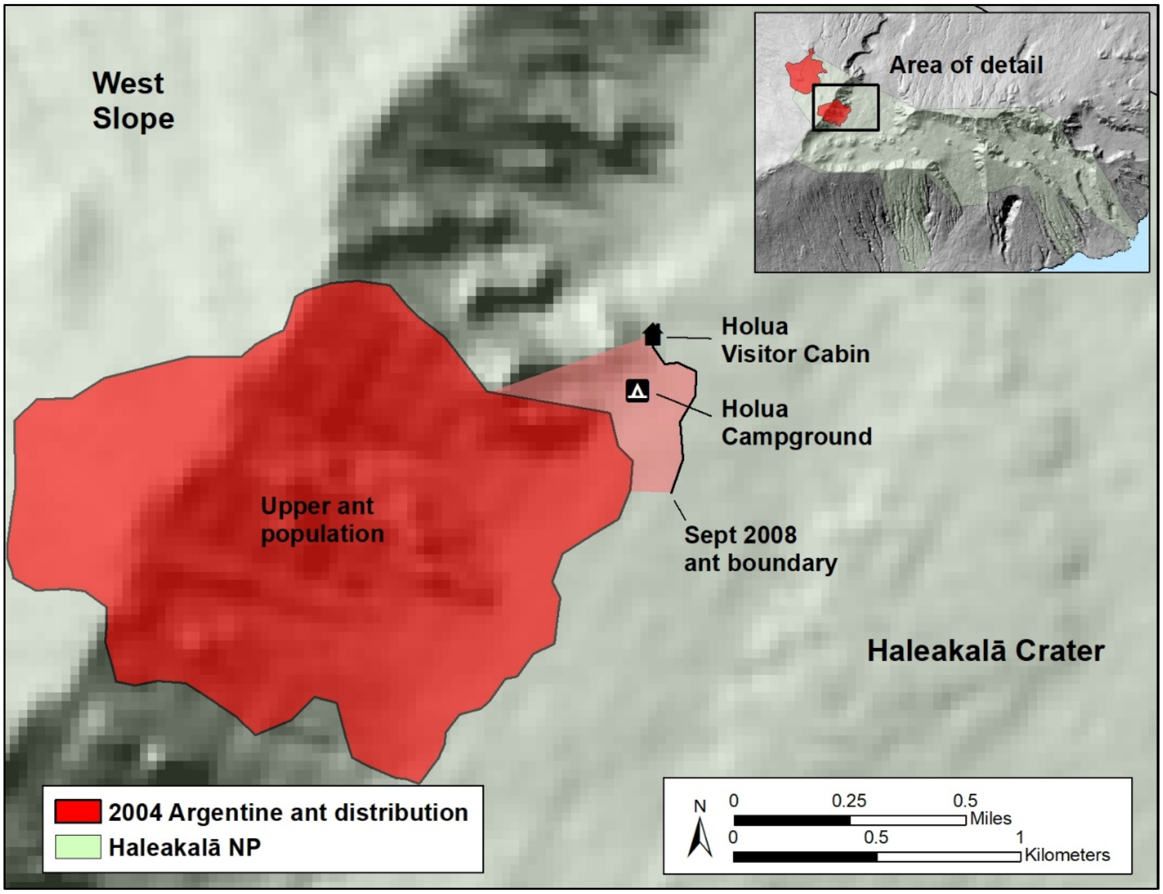

Figure 4.2.8-11. Estimated distribution of the upper Argentine ant population in HALE in 2004, and population boundary in the vicinity of Holua mapped in September of 2008. Pink area indicates territory gained between 2004 and 2008; other sections of the population boundary have not been mapped since 2004. Source: Krushelnycky et al. (2011), and unpublished USGS data.

This ongoing invasion threatens a wide variety of native arthropod species, as summarized in the shrubland ecosystems section (chapter 4.2.6). Not only is native arthropod diversity reduced by over 50% in invaded areas (Krushelnycky and Gillespie 2008), but species with important ecological functions, like *Hylaeus* bee pollinators, are threatened (Cole et al. 1992). As mentioned earlier, Krushelnycky (2014) measured *Hylaeus* floral visitation on ʻāhinahina plants (Figure 4.2.8-12) to be reduced by 50% in an ant invaded area on the Crater rim; although alternative environmental factors could not be completely ruled out as a cause, this putative impact from ants is consistent with earlier findings in the park (Cole et al. 1992) and anecdotal observations statewide (Magnacca 2007, Magnacca and King 2013). *Hylaeus* bees represent the only group of native bees in Hawaiʻi, and are today highly imperiled (Magnacca 2007); seven species were recently listed as federally endangered (USFWS 2016). High-elevation areas of Haleakalā support the highest remaining densities of *Hylaeus* in the Hawaiian Islands (Daly and Magnacca 2003), making the Crater and other parts of HALE an important refuge for these native pollinators. Continued spread of Argentine ants into the Crater will be similarly devastating for many other rare and locally endemic invertebrates.

Figure 4.2.8-12. Native *Hylaeus* yellow-faced bee visiting ʻāhinahina flower at HALE. Photo: Paul Krushelnycky.

Yellowjackets

As in the West Slope shrubland ecosystem, invasive western yellowjackets are present in the Crater, where they can be expected to strongly impact native arthropod communities through their consumption of large numbers of a wide range of arthropods, including *Hylaeus* bees (Gambino and Loope 1992, Wilson et al. 2009, Wilson and Holway 2010, see also chapter 4.2.6). Because of these effects, and because aggressive yellowjackets also represent a hazard to park visitors, HALE staff monitor yellowjacket abundances and control individual nests. Yellowjacket relative abundances have been monitored at four locations in the Crater: at the three cabins (Hōlua, Kapalaoa, Palikū) and at the mule hitching post at the base of Keonehe‘ehe‘e (Sliding Sands) Trail. Coverage of these locations varied between years, but began in 1996 and has used two types of traps baited with heptyl butyrate attractant: “inn” traps, which were used initially through about 2007, and “water” traps, which have become the main monitoring method in more recent years owing to higher capture rates (NPS 2017c). Annual estimates of relative abundance averaged over the four Crater locations are shown for the period 1996-2015 for each trap type in Figure 4.2.8-13. These data, which show the annual average number of yellowjackets caught per trap on a relativized scale ranging from 0 to 1, indicate that abundances of yellowjackets vary strongly from year to year, with no clear long-term trend. The data also show that, averaged over the entire time period, there is strong seasonality to yellowjacket abundances, with worker numbers increasing in June and July as colonies grow, peaking in August and September, and generally declining through the fall and winter (Figure 4.2.8-14). Endangered Wildlife Management program staff control individual nests in the summer and fall seasons when numbers get high, typically focusing on nests near trails, campgrounds and other facilities. Between 2007 and 2013, 64 yellowjacket nests were exterminated, 26 of which were located in the Crater (NPS 2017c). These efforts reduce risk of visitor stinging around heavily used areas, but likely have relatively small, localized effects in terms of protecting native arthropod communities.

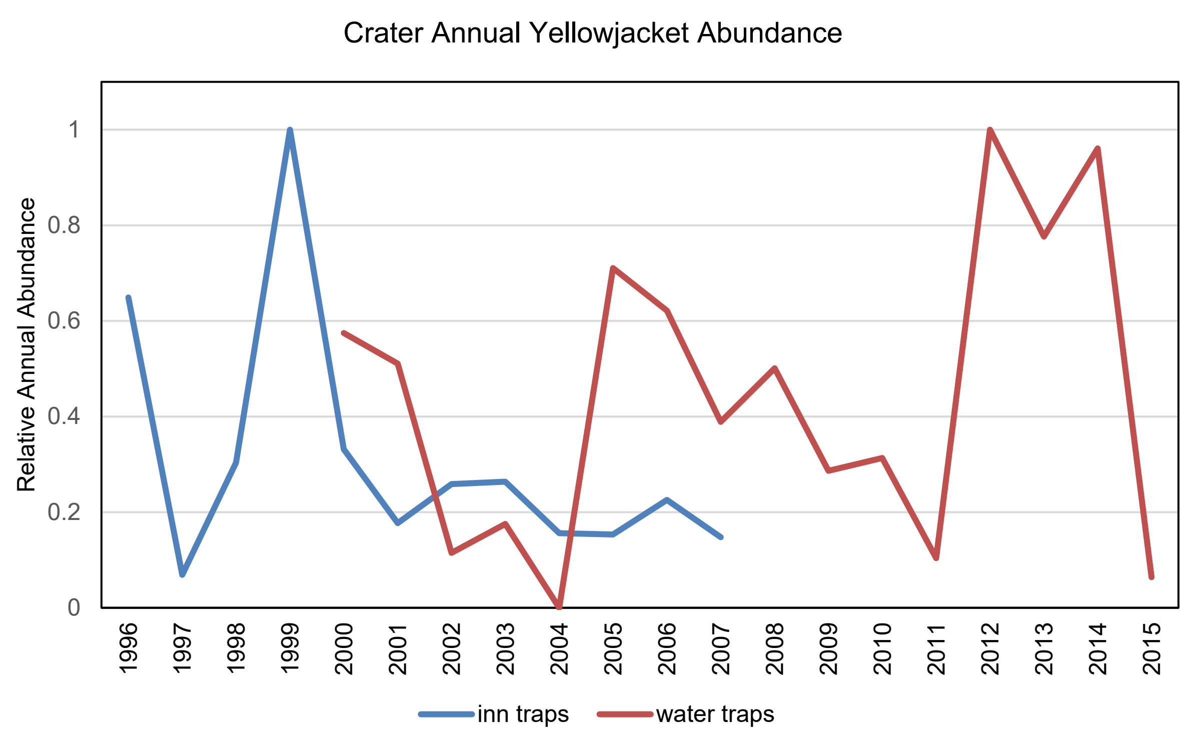

Figure 4.2.8-13. Estimated annual relative abundance of *Vespula* yellowjackets in the Crater from 1996 to 2015. Shown are mean number of workers captured per trap, averaged over all traps deployed per year at four sites, and relativized by the maximum annual average for each trap type. Trends for the two trap types, inn and water, are plotted. Inn trap data for 2008 and 2010 are excluded due to low numbers of traps and unusually high averages. Source: NPS (2017c).

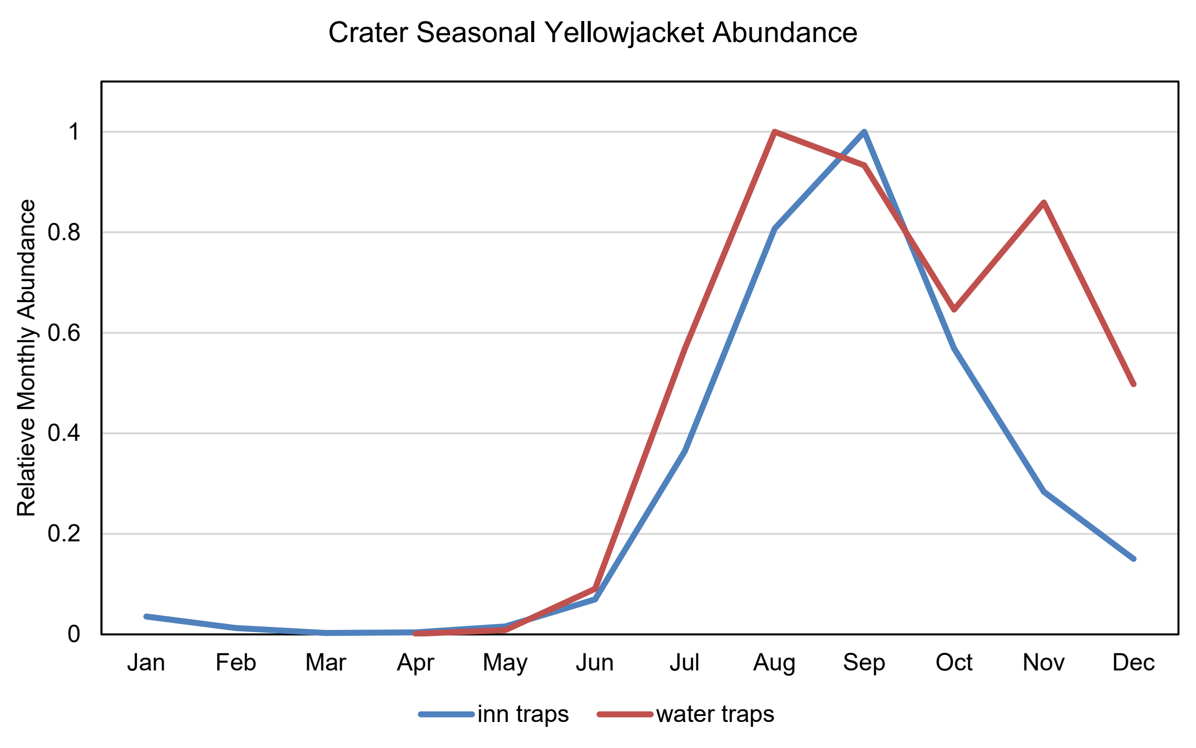

Figure 4.2.8-14. Estimated seasonal relative abundance of *Vespula* yellowjackets in the Crater. Shown are mean number of workers captured per trap, averaged over all traps deployed in each month over the period 1996-2015, and relativized by the maximum monthly average for each trap type. Trends for the two trap types, inn and water, are plotted. Water trap data for December of 2012 at Kapalaoa cabin are excluded due to unusually high numbers, likely due to a nearby nest. Source: NPS (2017c).

Other Stressors

Non-native mice and black rats can be expected to exert pressures on native arthropods similar to those assessed for West Slope shrublands (see chapter 4.2.6). Arthropods were found to comprise 32.5 ± 2.6% and 16.4 ± 3.5% of the diets of mice and black rats, respectively, trapped in West Slope shrublands in 1984-85 (Cole et al. 2000). These rodents appear to rely on arthropod prey especially heavily during the winter to spring, when alternative plant resources are scarce. In certain areas and times, arthropods made up approximately 75% of mice diets. Consequently, Cole et al. (2000) felt that both rodents may be exerting population-level impacts, but that mice especially may have "dramatic negative effects on locally endemic arthropods, particularly during periods of high density." Endangered Wildlife Management program staff at HALE operate several small mammal predator traplines, currently employing about 90 traps across five traplines in the Crater (NPS 2012, J. Tamayose pers. comm. 2018). These traps target rats, mongoose and cats to protect endangered nēnē and ʻuaʻu, but also likely provide some benefit to native arthropods in the vicinity of the traplines.

Finally, the threats to Crater plants from changing climate conditions discussed in the vegetation section above also have implications for native invertebrates, most obviously for host-specific species. As an example, the planthopper *N. argyroxiphii* and the moth *R. rhabdotis* both feed exclusively on Haleakalā ʻāhinahina plants. If the ʻāhinahina population continues to decline, populations of these associated insects will also be impacted. ʻĀhinahina also appear to provide important floral resources to *Hylaeus* bees and other flower-visiting species (Krushelnycky 2014). In addition, invertebrates may need to shift their distributions to track changing temperature or precipitation patterns. Because these species already occur near the top of the mountain, there will be limited ability to move upslope, a response that has been documented for other insects (e.g. Chen et al. 2009).

Because most of the Crater is relatively dry, and much of it supports only sparse vegetation, native mollusc diversity and abundance is likely to be fairly low. However, some native snails are known to occur in dry high-elevation shrublands (Cowie 1995), and invasive garlic snails (*Oxychilus alliarus*) and leopard slugs (*Limax maximus*) have been observed in wetter areas in the eastern Crater (R. Kaholoaʻa pers. comm. 2018). The native snail fauna, however, appears to be unsurveyed.

Summary of Condition and Trend of Native Crater Invertebrates

Owing to the moderate proportion of Crater arthropod communities composed of non-native species, at least as measured 40 years ago, and to the currently restricted distribution of the highly damaging Argentine ant but widespread presence of the western yellowjacket, we judge the current condition of shrubland invertebrates to warrant moderate concern. We assign a deteriorating trend in this condition, due to 1) the continuous introduction and establishment of new non-native species, with no practical means of controlling the vast majority of them, and 2) the rapidly expanding distribution of the Argentine ant into the Crater. Confidence in this assessment is low, because no comprehensive survey of Crater arthropods has been conducted since the 1970s, and the status of Crater molluscs is unknown.

Overall assessment

Integrating the two indicators for the Crater according to SotP aggregation rules yields a current overall good condition, with an unchanging trend. Crater vegetation appears to be in good condition, and is likely recovering from past ungulate damage, although longitudinal data to confirm this are lacking. The native-dominated status across most of the Crater has been maintained and augmented through a combination of upkeep of boundary fences, native outplanting and restoration activities, and targeted control of a select group of potentially habitat-modifying invasive plants. Changing climate, however, threatens to reverse some of these trends, but again information on this subject is limited. Crater invertebrate communities have not been assessed in over 40 years, but available data suggest many non-native species have established, and likely continue to do so. Especially damaging species like the Argentine ant and western yellowjacket are either persistent or spreading, threatening to further degrade the current condition. Collectively, these dynamics have the potential to downgrade the current good overall condition in the future. (return to Condition Summary).

Level of confidence

Overall level of confidence in this assessment is low. Much information has been reviewed to provide the current assessment, but very few data have been repeatedly collected over time, creating more uncertainty in assessment of trends. It is unknown whether recent patterns seen in ʻāhinahina plants will continue, and the extent to which they reflect trends in other Crater plants is also unclear.

Much information is fragmentary, lacking, or insufficiently recent for certain resource areas, such as invertebrates, which also lowers level of confidence.

Information gaps and research recommendations

Previously, repeated quantitative information on Crater vegetation (i.e., cover and richness, degree of non-native plant invasion) was lacking. This is in the process of being remedied via the FTPC and the EIPS monitoring protocols of the NPS I&M program (Ainsworth et al. 2011, 2012). The repeated monitoring of these two sets of plots will allow for site specific comparisons over time that will greatly enhance future vegetation trend analyses. Due to time and staff limitations, consistent monitoring is not conducted for most of the threatened, endangered, or otherwise rare plant species in the Crater, making assessment of trends in these species difficult. While Crater arthropod communities were fairly well characterized in the 1970s, this information is now very dated, and other invertebrates like molluscs are even less poorly studied. No monitoring protocol exists for Crater invertebrate communities, which severely hampers assessment of changing conditions.

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4.2.9. Nuʻu Parcel

Condition Summary

Overall resource condition warrants significant concern with an unknown trend; confidence in condition determination is medium. Rationale is discussed in the "*Overall assessment*" section and following.

●

Relevance and Context

In 2008, the National Park Service acquired and added the 4,178 ac (1,691 ha) Nuʻu parcel to HALE, extending the park boundary from the crater rim at ca. 7,700 ft (2,350 m) elevation to sea level on the southern slope of East Maui (Figure 4.2.9-1). Due to its leeward location, precipitation is sparse across this area, with rainfall averaging just over 500 mm (19.7 in) per year near the coast to approximately 1,524 mm (60 in) per year in the middle to upper elevations (Giambelluca et al. 2013).

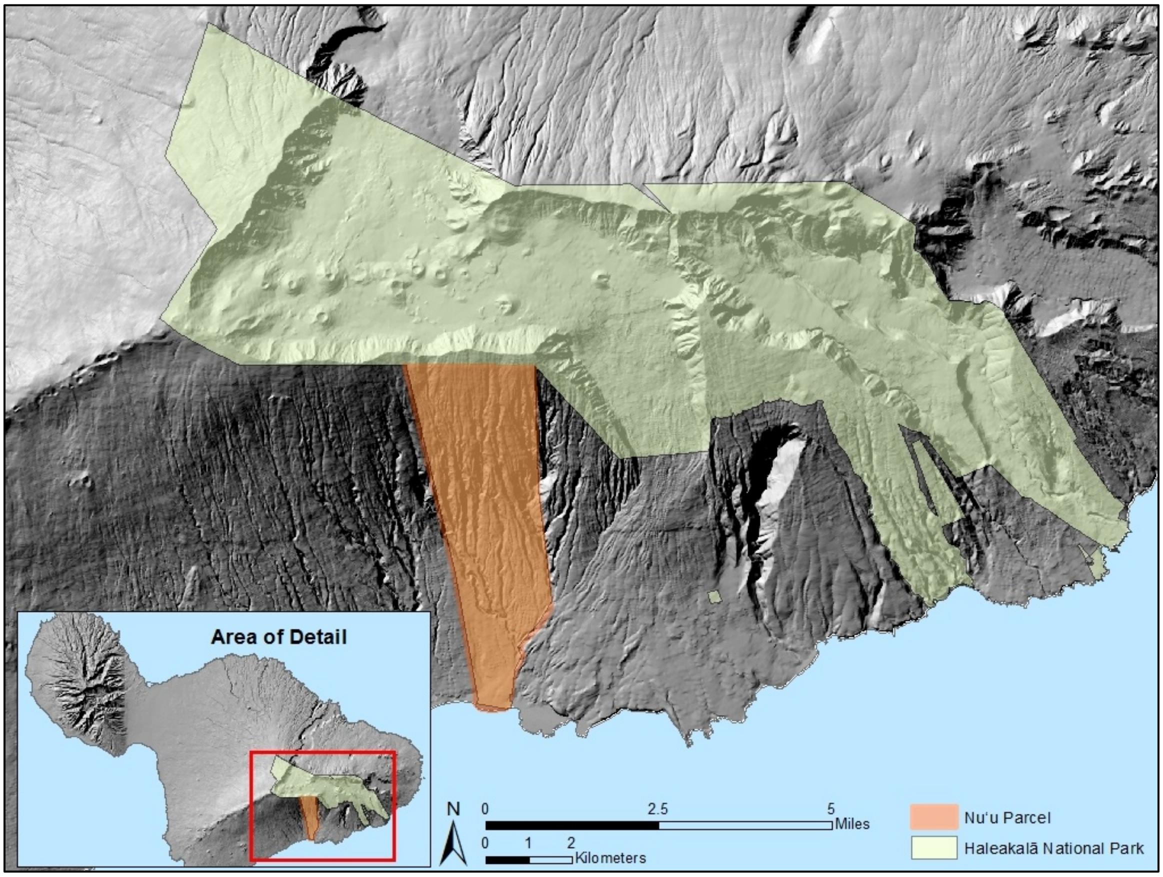

Figure 4.2.9-6. The 4,178 acre (1,691 ha) Nuʻu Parcel, added to HALE in 2008.

Nuʻu is blocked from the northeasterly trade winds so precipitation comes mainly from storms associated with south and west winds from November through March. The terrain is rough and steep, and is dissected by numerous deeply-incised gulches containing intermittent drainage channels (NPS 2016). Much of the area is inaccessible to vehicles, and is also difficult to access on foot.

Impacts to Nuʻu Systems

Before the arrival of humans to the Hawaiian Islands, native ecosystems extended from sea level to the highest elevations, and according to Pratt and Gon (1998) the Nuʻu area would have supported a diversity of major biome types including lowland dry forest and shrubland, montane dry forest and shrubland, dry cliff communities, montane mesic forest and shrubland, and subalpine dry shrubland and grassland (Figure 4.2.9-2). However, 1500+ years of human-induced changes via direct land use, and the effects of multitudes of introduced plants and animals, have in many places completely replaced the original communities with anthropogenic landscapes and alien-dominated ecosystems. Today, although large areas of relatively intact montane, subalpine, and alpine habitat remain, almost all the native coastal, lowland and middle elevation dry and mesic ecosystems of the type that formerly existed in Nuʻu have been lost (Figure 4.2.9-2).

The entire region has suffered from a long history of land degradation caused by the grazing and trampling of ungulates including cattle (*Bos taurus*), feral goats (*Capra hircus*), feral pigs (*Sus scrofa*) and axis deer (*Axis axis*). Ungulate damage results in loss of vegetation which contributes to soil disturbance, erosion and sediment run-off. Soils on the south slope of Haleakalā are generally poorly developed due to the young age of the substrates and the relatively low rainfall in the region (Medeiros et al. 1986). This is especially true of the rocky lands on the southwestern flank of the mountain; districts further east that are situated on older Kula series lava substrates, including the Nuʻu Parcel, have somewhat deeper soils (Medeiros et al. 1986). However, the combination of steep slopes on the middle to upper portions, denuded state of the vegetation, and continued activity of feral ungulates makes the soils in these areas highly vulnerable to erosion. Natural and anthropogenic fires, and invasion by non-native plants have also contributed to the region's decline (Medeiros et al. 1986).

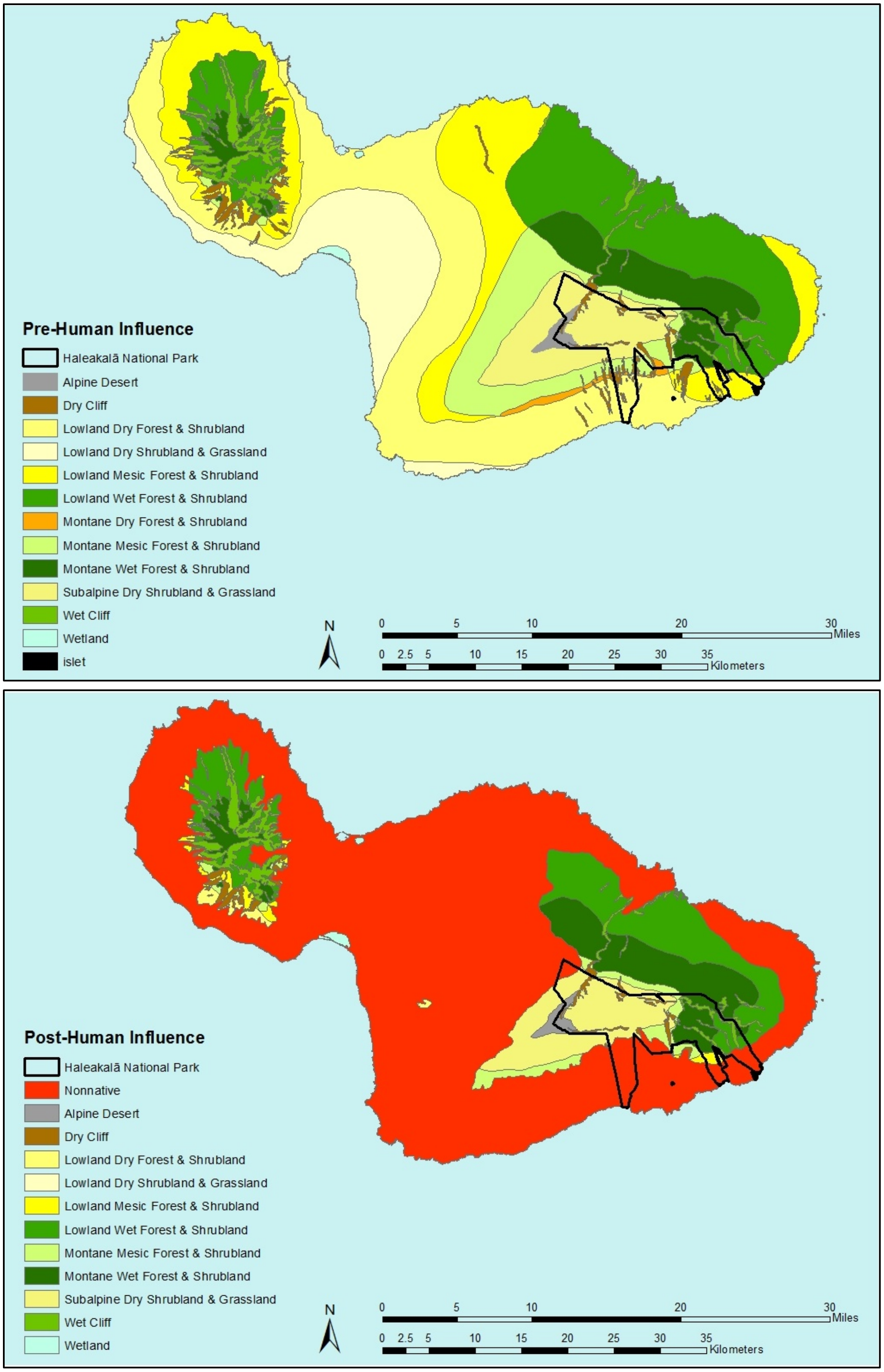

Figure 4.2.9-2. Pre- and post-human influence on vegetation communities on the island of Maui. Source: After Pratt and Gon (1998).

Such impacts continue to damage these mesic and dry forests and shrublands recognized to have been the most diverse of the Hawaiian floristic communities, with many species endemic to this region of East Maui (Medeiros et al. 1986). The rich remnant leeward flora in turn serves as the foundation for a potentially diverse community of animals that, given ecological restoration, could thrive in this remote and undeveloped portion of the mountain. For example, the endangered ʻuaʻu, or Hawaiian petrel (*Pterodroma sandwichensis*), was formerly common and widespread throughout the islands, but its range and abundance have been reduced by habitat loss, predation by alien animals, groundings, and collision with man-made objects (USFWS 1983, 2005). Haleakalā Crater currently supports the largest known breeding colony of this species. ʻUaʻu breeding pairs are also known to occur in the upper elevations of the State Kahikinui Forest Reserve, and the adjacent Nuʻu parcel contains additional habitat suitable for breeding ʻuaʻu (NPS 2014, 2016). Many ʻuaʻu nesting burrows are susceptible to trampling by feral ungulates and erosion associated with loss of vegetation. Additionally, the recently-listed endangered ʻakēʻakē (band-rumped storm petrel, *Oceanodroma castro*) was detected in the upper elevations and may also nest in the area. Several species of native passerine forest birds, including ʻapapane (*Himatione sanguinea*), ʻiʻiwi (*Drepanis coccinea*) and Maui ʻamakihi (*Chlorodrepanis virens wilsoni*) currently utilize habitat in the upper elevation of Nuʻu, especially areas with remnant native forest. These species, along with Maui ʻalauahio (Maui creeper, *Paroreomyza montana newtoni*), also inhabit mesic forest remnants in the Nakula Natural Area Reserve (NAR) and Kahikinui Forest Reserve, approximately 2.2 km (1.4 mi) to the west of Nuʻu, and re-introduction of the endangered kiwikiu or Maui parrotbill (*Pseudonestor xanthophrys*) is planned in the Nakula area (Peck et al. 2015, Figure 4.2.9-3). While only about 5% of original native forest cover remains on the leeward slopes of Haleakalā (Medeiros et al. 1986), these habitat fragments are valuable given that they are often biologically distinct from other ecosystem types on the island, and may therefore harbor unique species, or provide supplemental refuges for rare species.

Land use on neighboring properties includes livestock grazing below 4,000 ft and conservation in the upper elevations. Since acquiring the parcel, the park has conducted minimal management activities in the area. A recently completed Environmental Assessment outlines plans to fence and control feral ungulates to support ecosystem recovery and long-term resource protection (NPS 2016). The ungulate control fence, currently under construction on the HALE Nuʻu parcel, will extend from approximately 1,250 to 7,650 ft (381 to 2,332 m) elevation and enclose a total of 2,115 ac (856 ha), 1,885 ac (763 ha) within the HALE Nuʻu parcel and 230 ac (93 ha) within the adjacent Kahikinui Forest Reserve (NPS 2016, Figure 4.2.9-3). Upon completion of the fence and removal of ungulates within its boundaries, vegetation is expected to recover rapidly. Further intervention will be required to prevent the proliferation of invasive non-native plants, particularly fire-promoting grasses, that can inhibit natural regeneration of native species and interfere with facilitated restoration activities (Loope and Scowcroft 1985, Scowcroft and Hobdy 1987).

The status of native vegetation, birds, and invertebrates were identified as important indicators of the condition of the highly-altered Nuʻu Parcel. Focus is placed mainly on vegetation, because of very limited information on other resources. The status of birds and invertebrates in Nuʻu is commented on to the extent possible, in some cases using information from nearby areas or the broader region.

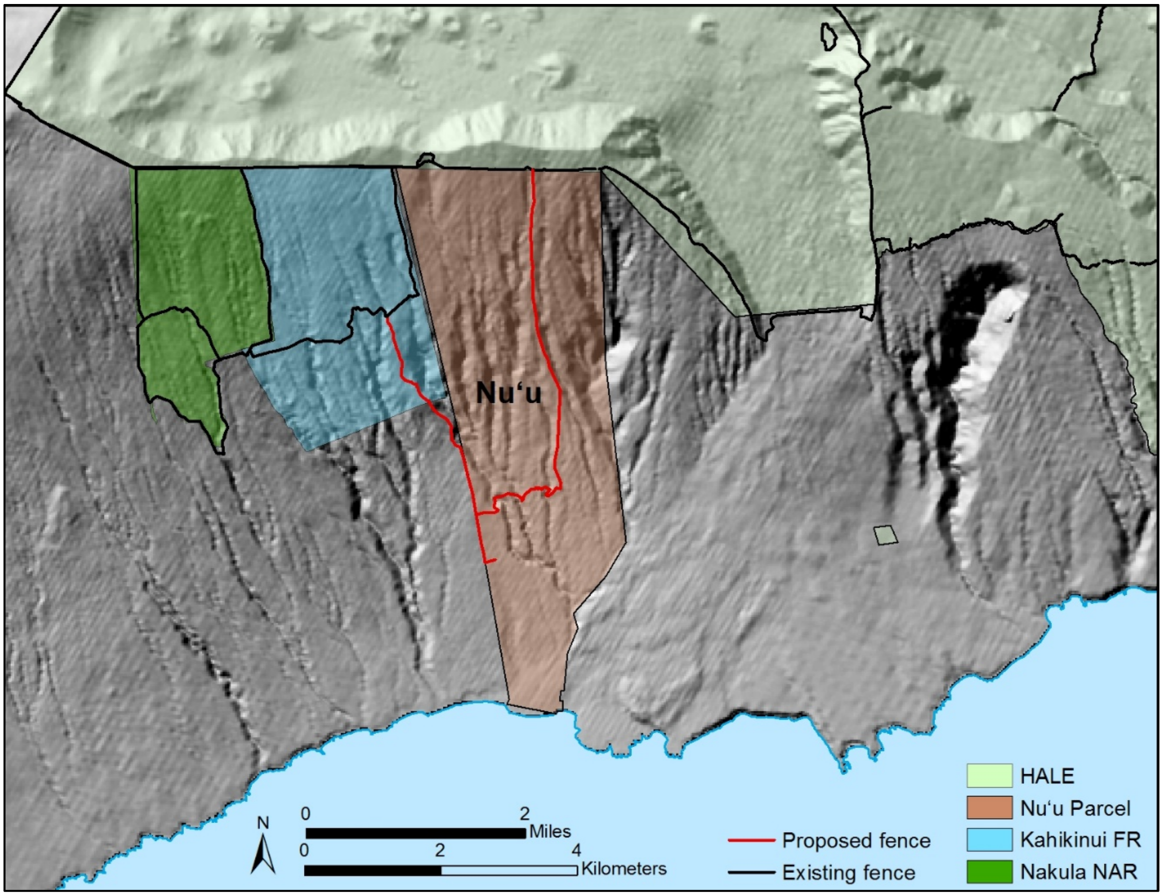

Figure 4.2.9-3. Location of State-owned conservation units adjacent to the Nu‘u Parcel (Kahikinui Forest Reserve and Nakula Natural Area Reserve), and location of planned Nu‘u fence and ungulate exclusion area. Source: NPS 2016, Hawaii Office of State Planning.

Indicators

- Vegetation
- Birds
- Invertebrates

Data and Methods

A review of available data and information was used to assess conditions in the indicators listed above. Some additional analyses were performed on a subset of geospatial and vegetation classification data collected by HALE and Pacific Island Network (PACN) staff specific to the Nu‘u parcel (Green et al. 2015).

Sources of information

Vegetation classification data were obtained from the HALE and PACN Inventory and Monitoring project (Green et al. 2015). Limited information on bird and invertebrate resources was derived from the Nu‘u Environmental Assessment for feral animal management (NPS 2016) and from Krushelnycky (2010) and Liebherr (2015).

Reference condition

Ideal reference conditions for the Nuʻu Parcel would consist of completely unaltered vegetation and associated animal communities from the coast to the upper boundary near the Crater rim. These communities would promote retention of topsoil and inhibit erosional runoff into marine environments, and would otherwise experience unimpaired ecosystem processes and function. Given the extensive alteration to the entire region surrounding Nuʻu summarized above, a desirable reference condition for Nuʻu is therefore that the richness and cover of native species exceeds that of nonnative species.

Condition and Trend

Vegetation

Vegetation in Nuʻu was recently classified by HALE and PACN staff, using 22 circular plots (400 m^2) and 4 observation points within the parcel, all below 4,000 ft (1,220 m) elevation (Green et al. 2015). Data recorded at observation points reflect the vegetation of an area of variable spatial extent around the point rather than a measured plot, and are less detailed (Green et al. 2015). This survey effort, conducted in 2011, resulted in 15 vegetation communities classified within the Nuʻu Parcel. Of these, three have native plant species as among the most abundant cover represented in that community type, ten are dominated by non-native species, and two are unclassified (Table 4.2.9-1).

Table 4.2.9-1. Vegetation communities classified within the Nuʻu Parcel of HALE. Source: Green et al. (2015).

Vegetation Community	Acres	Hectares	% of Nuʻu
Pityrogramma austroamericana - Heterotheca grandiflora Semi-natural Herbaceous Vegetation	717.45	290.34	16.71%
Unclassified Polynesian Semi-natural Montane and Subalpine Shrubland and Grassland	684.80	277.13	15.95%
Lantana camara Semi-natural Shrubland	604.25	244.53	14.07%
Unclassified Polynesian Semi-natural Lowland Shrubland, Grassland, and Savanna	505.52	204.58	11.77%
***Leptecophylla tameiameiae - Vaccinium reticulatum* Subalpine Mesic Shrubland[1]**	**474.99**	**192.22**	**11.06%**
Cenchrus ciliaris - Bothriochloa pertusa Semi-natural Herbaceous Vegetation	437.28	176.96	10.19%
Leucaena leucocephala Semi-natural Shrubland	274.95	111.27	6.40%
Prosopis pallida Coastal Dry Semi-natural Woodland	228.40	92.43	5.32%
Melinis minutiflora - (*Schizachyrium condensatum*[2]) Semi-natural Herbaceous Vegetation	168.45	68.17	3.92%
***Metrosideros polymorpha / Leptecophylla tameiameiae - Dodonaea viscosa* Montane Woodland[1]**	**152.60**	**61.75**	**3.55%**
Mixed Non-native Disturbed Herbaceous Vegetation	13.67	5.53	0.32%

[1] Native communitites, also highlighted in bold

[2] *Schizachyrium condensatum* included in community name, but not documented as present on Maui.

Table 4.2.9-1 (continued). Vegetation communities classified within the Nu‘u Parcel of HALE. Source: Green et al. (2015).

Vegetation Community	Acres	Hectares	% of Nu‘u
Spermacoce assurgens - Zinnia peruviana - Digitaria insularis Semi-natural Herbaceous Vegetation	11.89	4.81	0.28%
Unvegetated	8.25	3.34	0.19%
Thevetia peruviana Semi-natural Woodland	3.28	1.33	0.08%
***Erythrina sandwicensis* Lowland Dry Woodland**[1]	**2.92**	**1.18**	**0.07%**
Grevillea robusta Lowland Mesic Semi-natural Forest	2.85	1.15	0.07%

[1] Native communitites, also highlighted in bold

[2] *Schizachyrium condensatum* included in community name, but not documented as present on Maui.

Of the native communities, only one, *Erythrina sandwicensis* Lowland Dry Woodland, occurs in the lower elevation (between 120-200 m) of the Nu‘u Parcel. This community is a 2.92-ac (1.18 ha) patch of remnant dry forest, characterized by an open tree canopy of the native *Erythrina sandwicensis* mixed with the non-native tree *Prosopis pallida*, and a subcanopy and understory dominated by the non-native tree *Leucaena leucocephala* and non-native grasses, respectively (Green et al. 2015). *Leptecophylla tameiameiae* - *Vaccinium reticulatum* Subalpine Mesic Shrubland is a remnant native community occurring in the upper elevations of the Nu‘u Parcel (ca. 1970-2300 m), and at 474.99 ac (192 ha), is the fifth largest vegetation community within the region (Table 4.2.9-1). As its name implies, it is co-dominated by the native shrubs *Leptecophylla tameiameiae* and *Vaccinium reticulatum*, with an herbaceous layer of several native species and non-native grasses (Green et al. 2015). The third native community, *Metrosideros polymorpha* / *Leptecophylla tameiameiae* - *Dodonaea viscosa* Montane Woodland also occurs in the upper elevations of Nu‘u (ca. 1500-2000 m), and covers 152.60 ac (61.75 ha) of the region. This community is characterized by an open canopy of *Metrosideros polymorpha* trees and an understory of *Leptecophylla tameiameiae* and *Dodonaea viscosa* shrubs (Green et al. 2015). On the higher part of the middle elevations, *Acacia koa* is also present (Figure 4.2.9-4). Although pili grass (*Heteropogon contortus*) is reported to occur in small patches at all elevations of Nu‘u (NPS 2016), park staff have only observed it at lower elevations and in close proximity to the road (J. Mallinson and P. Welton pers. comm. 2018).

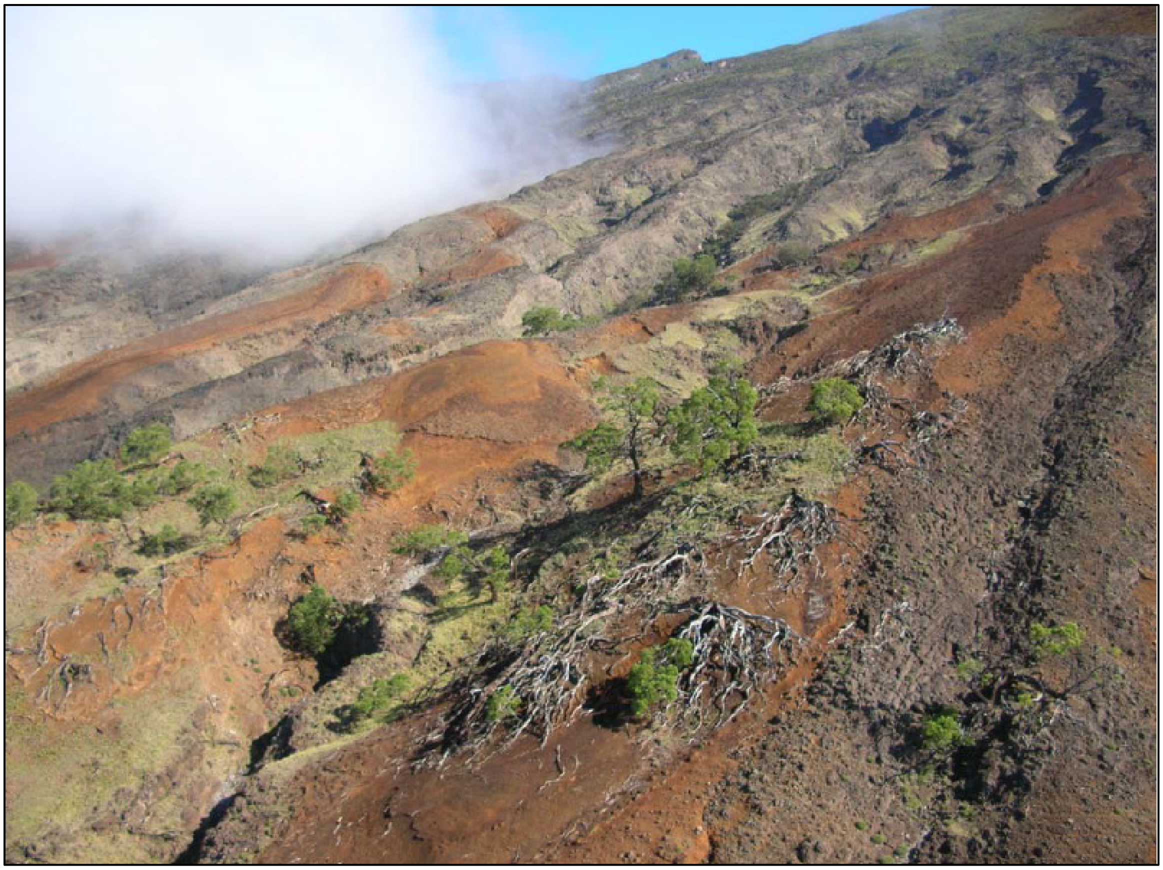

Figure 4.2.9-4. Remnant patch of *Acacia koa* on the upper slopes of the Nuʻu Parcel. Photo: A. C. Medeiros.

The remaining vegetation communities in Nuʻu are dominated by non-native species, or are unclassified community types lacking dominant species but otherwise consisting of non-native plant taxa (Table 4.2.9-1, Green et al. 2015). The largest vegetation community within Nuʻu is classified as *Pityrogramma austroamericana* - *Heterotheca grandiflora* Semi-natural Herbaceous Vegetation, accounting for 717.45 ac (290 ha) of the parcel. This community, on mesic steep slopes and ridges from 700-1400 m elevation, is characterized by low cover of the relatively unpalatable non-native fern *Pityrogramma austroamericana* and forb *Heterotheca grandiflora* (Figure 4.2.9-5, Green et al. 2015). The Unclassified Polynesian Semi-natural Montane and Subalpine Shrubland and Grassland community, in upper elevations on very steep slopes between 900 and 2100 m, accounts for the second largest area in Nuʻu (Table 4.2.9-1). Heavy browsing by feral ungulates, and the resulting erosion, have eliminated most vegetation on these steep slopes (Green et al. 2015). The remaining communities consist of a mix of invasive non-native grasses, and unpalatable or browse-tolerant shrubs and trees, at sparse to moderate cover amounts (Table 4.2.9-1).

Figure 4.2.9-5. *Pityrogramma austroamericana* (left) and *Heterotheca grandiflora* (right) Semi-natural Herbaceous Vegetation dominates in upper elevation of the Nuʻu Parcel. Photos: Forest and Kim Starr.

Overall, non-native and unclassified (largely barren or non-native) plant communities, as well as bare ground, make up 85.3% of the entire cover of the Nuʻu parcel (Figures 4.2.9-6 and 4.2.9-7). The native communities, currently accounting for 14.7% of the region, are largely restricted to the upper elevations of Nuʻu and are in relatively poor condition compared to similar community types elsewhere within HALE (Green et al. 2015). Comparable cover composition was reported in the Environmental Assessment to be 60% grassland cover, around 25% bare ground, 10% scrub cover and from 1 to 3% other categories (USGS 2011, NPS 2016).

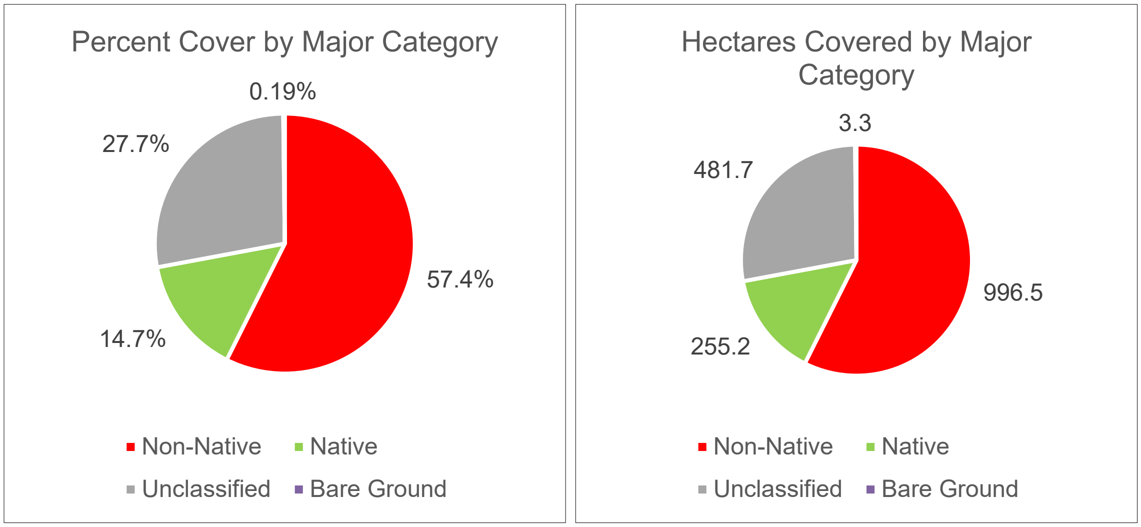

Figure 4.2.9-6. Dominant vegetation cover categories by percentage and area within the Nuʻu Parcel of HALE. Small bare ground pie slices not visible, but represented numerically. Source: Green et al. (2015).

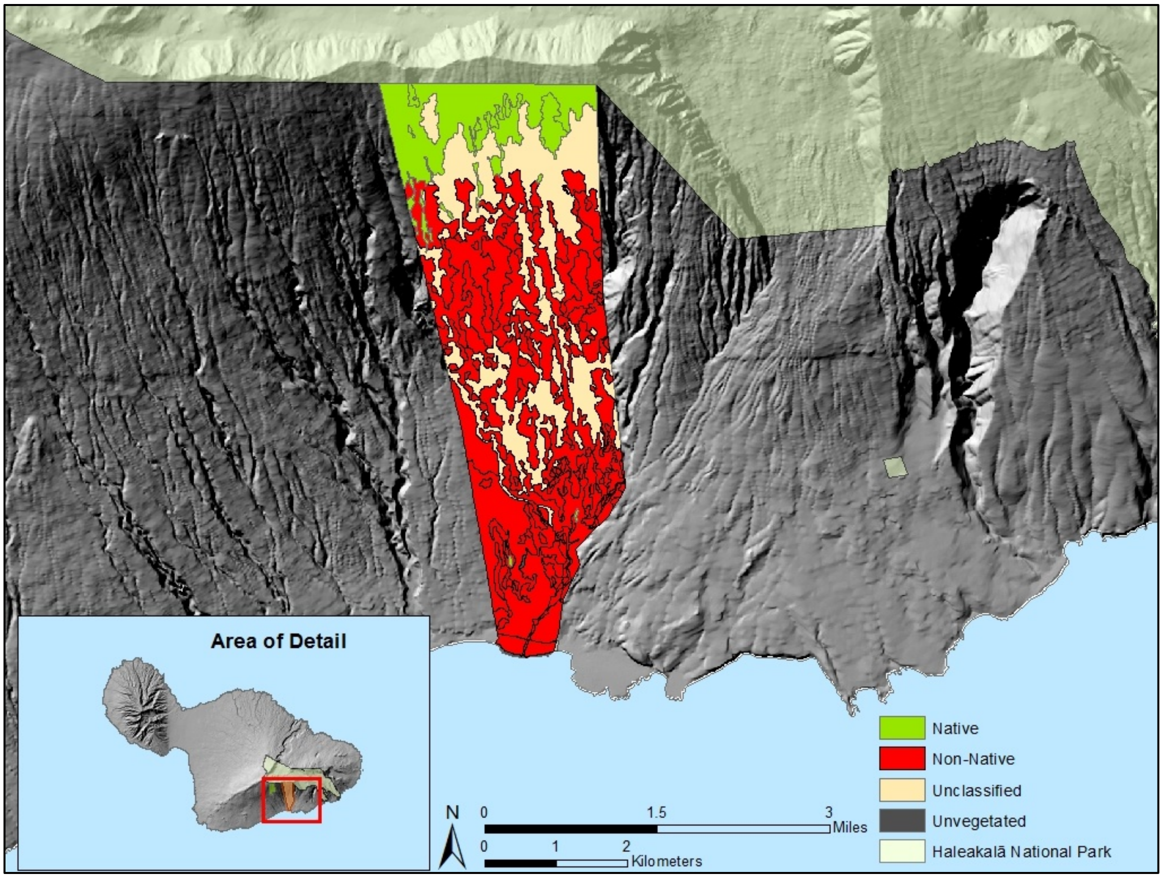

Figure 4.2.9-7. Dominant vegetation cover categories within the Nu‘u Parcel of HALE. Source: Green et al. (2015).

Green et al. (2015) also provide a limited sampling of some of the plant species occurring in the Nu‘u Parcel, through the series of circular plots and observation points surveyed. A total of 11 native and 67 non-native plant species were recorded in these plots and observation points, with a mean species richness of 1.5 ± 0.23 natives and 10.62 ± 0.82 non-natives per plot (Figure 4.2.9-8).

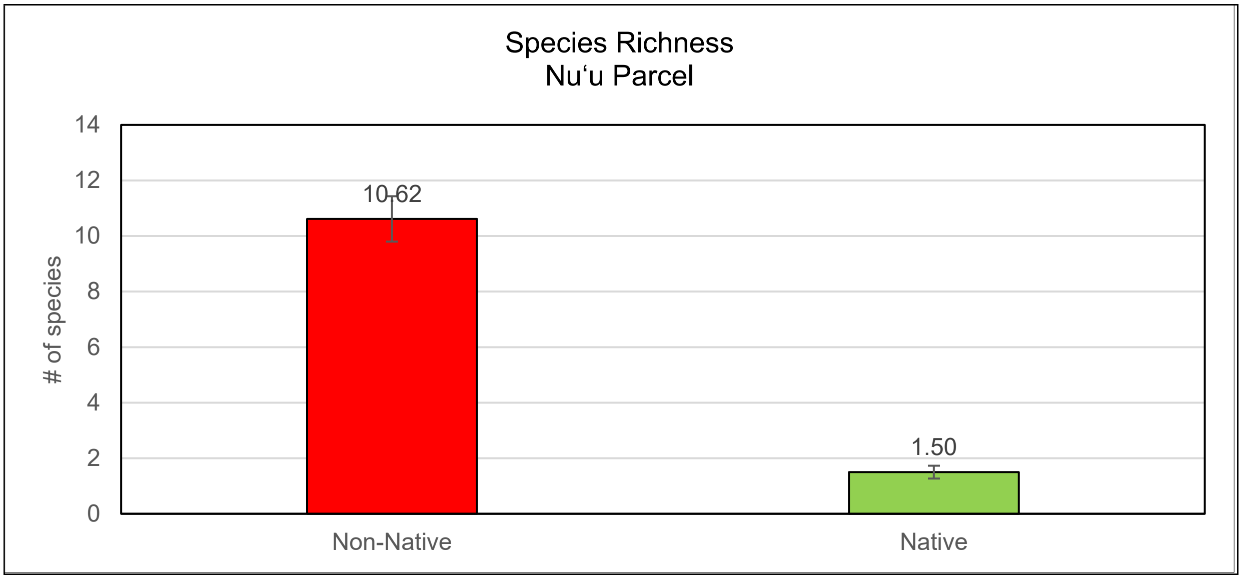

Figure 4.2.9-8. Mean species richness in vegetation monitoring and observation plots (n = 26) within Nu‘u Parcel. Source: Green et al. (2015).

Because of the relatively recent acquisition of the Nu‘u parcel, and steep, rugged aspect of much of the area, thorough vegetation surveys and searches for threatened and endangered plants have been limited. Based on known individuals or populations in areas adjacent to the Nu‘u parcel within both HALE and the Kahikinui Forest Reserve, it is possible that some rare plants occur in the area. Former critical habitat designations included critical habitat in Nu‘u for three federally listed plant species (NPS 2016): māhoe (*Alectryon macrococcus*), the Haleakalā subspecies of ‘āhinahina (Haleakalā silversword, *Argyroxiphium sandwicense* ssp. *macrocephalum*), and ko‘oko‘olau ssp. kalealaha (Kalealaha beggartick, *Bidens micrantha* ssp. *kalealaha*). None of these species are currently known to occur within the boundaries of Nu‘u, but individuals of both *A. sandwicense* ssp. *macrocephalum* and *B. micrantha* ssp. *kalealaha* are found to the north within the fenced Haleakalā crater. ‘Āhinahina plants also occur to the west in the upper elevations of the Kahikinui Forest Reserve, inside an ungulate control fence (NPS 2016). Following fencing of the upper portion of the parcel, it may be possible to reintroduce these and other species into the area as part of native restoration and rehabilitation activities. For example, under the most recent critical habitat designation scheme, the Nu‘u Parcel intersects four different critical habitat ecosystem units, which collectively provide critical habitat for many listed threatened and endangered plant taxa (USFWS 2016).

Although nearly all of Haleakalā’s dry to mesic ecosystems from the coast through montane forest zones have been lost, the most intact remnants that do remain are generally located either west of Nu‘u, or to the east within the Kaupō Gap portion of HALE. For example, Medeiros et al. (1986) identified some of the best remaining areas on the mountain as potential targets for conservation of six different vegetation zones that they used to classify the leeward Haleakalā flora. These were mainly in the districts of Auwahi, Kanaio and the western portions of Kahikinui, well west of Nu‘u. Also to the west, Nakula NAR contains relatively decent patches of koa (*Acacia koa*) and ‘ōhi‘a (*Metrosideros polymorpha*) trees, with moderate additional tree and shrub diversity in steep gulches

(Peck et al. 2015). In comparison, forest remnants at middle to upper elevations of Nuʻu are visibly smaller and less extensive, although precise extents have not been quantified. Large numbers of goats, pigs and axis deer roam these areas of Nuʻu (NPS 2016), and the resulting soil disturbance and damage to vegetation has led to massive erosion, creating deep gulches that likely transport large quantities of sediment into the marine environment after heavy rains (Figure 4.2.9-9). Medeiros et al. (1986) commented that "The middle and upper slopes of Nu'u appear to present as spectacular an example of accelerated erosion due to goat browsing as exists anywhere."

Figure 4.2.9-9. Ungulate-damaged vegetation and rugged terrain on the leeward slopes of the Nuʻu Parcel, resulting in soil erosion and sediment run-off. Photos: C. Chimera.

Summary of condition and trend of native vegetation

Owing to all of the measures and factors discussed above, the current condition of native vegetation in Nuʻu warrants significant concern. Confidence in this assessment is high. Although only areas below 4,000 ft were surveyed by PACN, these areas are so degraded that the overall condition of vegetation in the parcel is not in doubt. There is currently insufficient information with which to assess a trend in Nuʻu vegetation condition.

Birds

Nine species of native birds are thought to occur in the Nuʻu Parcel (Table 4.2.9-2, NPS 2016), although little is known about habitat use or nesting status for most of these species. The Nuʻu area is estimated to contain at least 50 to 100 nesting burrows of the endangered ʻuaʻu (Hawaiian petrel, *P. sandwichensis*) (NPS 2016). These birds are impacted by the fragmented nature of remaining native-dominated communities. They are also likely threatened by ground disturbance from ungulates that could collapse burrows and by feral dogs (*Canis lupis familiaris*) that occur in the area and prey on feral ungulates and possibly ʻuaʻu. The presence of feral dogs poses a safety risk to both humans and to adults and chicks of ʻuaʻu (USFWS 1983). With protection from these animals following fencing, and with suppression of predatory rats, cats and mongoose, the Nuʻu Parcel may support larger numbers of ʻuaʻu (NPS 2016). Endangered nēnē (Hawaiian goose, *Branta sandvicensis*) are occasionally seen in the area and likely utilize grassy areas for foraging. Nēnē nesting in the area is not likely at this time, but may occur as habitat recovers (NPS 2016, C. Bailey pers. comm. 2018). Recent vocalization detections of ʻakēʻakē (band-rumped storm petrel, *O. castro*) indicate that these birds fly over the area, and may also nest in crevices (NPS 2016, C. Bailey pers. comm.). There are no estimates of current numbers of these species, but both may increase use with predator control and ecosystem restoration. Similarly, abundances of the three native forest bird species documented at Nuʻu (Maui ʻamakihi, *C. virens wilsoni*; ʻapapane, *H. sanguinea sanguinea*; ʻiʻiwi, *D. coccinea*) are unknown, but they are regularly observed in upper elevations, especially around the cloud line (C. Bailey pers. comm.). Because of the more degraded condition of forest habitat in the Nuʻu parcel, native forest bird numbers are likely to be lower than in the relatively larger habitat fragments to the west in Kahikinui Forest Reserve and Nakula NAR. The remaining native birds are relatively common seabirds that presumably frequent the coastal region of Nuʻu, but nesting within the parcel has not been documented.

Table 4.2.9-2. Native birds thought to occur in the Nuʻu Parcel. Source: NPS (2016).

Scientific Name	Hawaiian Name	English Name	Status
Branta sandvicensis	nēnē	Hawaiian goose	Endangered
Fregata minor	ʻiwa	great frigatebird	Common
Chlorodrepanis virens wilsoni	Maui ʻamakihi	–	Common
Himatione sanguinea	ʻapapane	–	Common
Oceanodroma castro	ʻakēʻakē	band-rumped storm petrel	Endangered
Phaethon lepturus dorotheae	koaʻe kea	white-tailed tropicbird	Common
Phaethon rubricauda melanorhynchos	koaʻe ʻula	red-tailed tropicbird	Common
Pterodroma sandwichensis	ʻuaʻu	Hawaiian petrel	Endangered
Drepanis coccinea	ʻiʻiwi	–	Threatened

Seventeen species of non-native birds have been documented in Nuʻu (NPS 2016), and are observed mostly at lower elevations. The ecological effects of many of these species are largely unstudied.

However, some, such as the ring-necked pheasant (*Phasianus colchicus torquatus*) and chukar (*Alectoris chukar*), are primarily plant feeders and may act as minor food competitors to native birds such as nēnē (Cole et al. 1995a,b). Other species, like the Japanese white-eye (*Zosterops japonicus*), feed on both plants and insects, and may compete for food with native forest birds (Freed et al. 2008, Freed and Cann 2009), though the validity of this claim has been questioned (Kingsford 2010, Camp et al. 2014, Banko et al. 2015). Finally, non-native birds may serve as reservoirs of diseases that negatively impact native forest birds (Ahumada et al. 2009, Samuel et al. 2011).

Summary of condition and trend of native birds

The current condition of native birds in Nuʻu is judged to warrant significant concern. While many of the native species expected to occur in Nuʻu are observed regularly, numbers of these species are likely to be very low based on the poor quality of the remaining habitat and the diversity of current threats, particularly habitat degradation by non-native ungulates and predation by non-native animals. Confidence in this assessment is low because almost no empirical data have been collected for the birds of Nuʻu. Similarly, insufficient information exists with which to assess a trend in this condition.

Invertebrates

No invertebrate surveys have been conducted within Nuʻu. However, limited work in adjacent areas may provide some indications of potential diversity. In general, the nearly complete loss of native vegetation, and alteration of original vegetation structure, in all but the highest shrubland zones almost certainly denotes a commensurate loss in native insect diversity and abundance. For example, an expedition in the Manawainui district east of Kaupō Gap in the 1970's found that the percentage of insects sampled that were native ranged from 10% in the lowest elevation, highly altered portions of Manawainui Valley, to 70-90% in most montane sites supporting significant stands of native forest (Villegas 1976). It is likely that within the former lowland and montane forest zones of Nuʻu, only remnant patches of native trees support substantial populations of native insects. Somewhat surprisingly, however, Peck et al. (2015) found that arthropod abundances, at least when measured per unit vegetation biomass, on koa and ʻōhiʻa trees in forest fragments at Nakula NAR were similar to or higher than those at two windward Haleakalā sites (Waikamoi Preserve and Hanawī NAR) that are embedded within large continuous blocks of intact native wet forest. These arthropod communities were also fairly diverse. Although it is unknown what proportion of the arthropods at Nakula NAR were native, many belonged to groups known to be composed largely of native species, and it is likely that at least a sizeable percentage of the sampled arthropods were native. Patches of koa and other native tree species in middle to upper zones of Nuʻu may therefore still harbor remnants of native invertebrate communities. As a possible exception to this general prediction, the endangered Blackburn's sphinx moth (*Manduca blackburni*), which today commonly utilizes tree tobacco (*Nicotiana glauca*) as a secondary host plant, may occur in the Nuʻu Parcel on and around this non-native plant, although the moth's presence has not been confirmed (NPS 2016).

A fortuitous circumstance is the apparent absence of highly invasive ants in the upper southern slopes of Haleakalā, likely owing to the extremely remote nature of these areas. Surveys of ants along three transects located just west of Nuʻu failed to detect any of the most damaging invasive ant species, such as the Argentine ant and big-headed ant, above approximately 4,000 ft (1,220 m) elevation

(Figure 4.2.9-10, Krushelnycky 2010). This contrasts with the upper western slopes of Haleakalā, which are extensively invaded by the aggressive Argentine ant from approximately 7,000 to 9,200 ft (2,134 to 2,805 m) elevation (Krushelnycky et al. 2005b). Although several other non-native ant species, including *Cardiocondyla kagutsuchi, Hypoponera opaciceps*, and *Tetramorium caldarium*, were detected on these transects, these ants generally do not form large, continuous colonies, and their impacts on native arthropod species are therefore thought to be of lesser magnitude. If highly damaging invasive ants are absent from Nuʻu, as seems probable, the situation should facilitate the persistence of native arthropods.

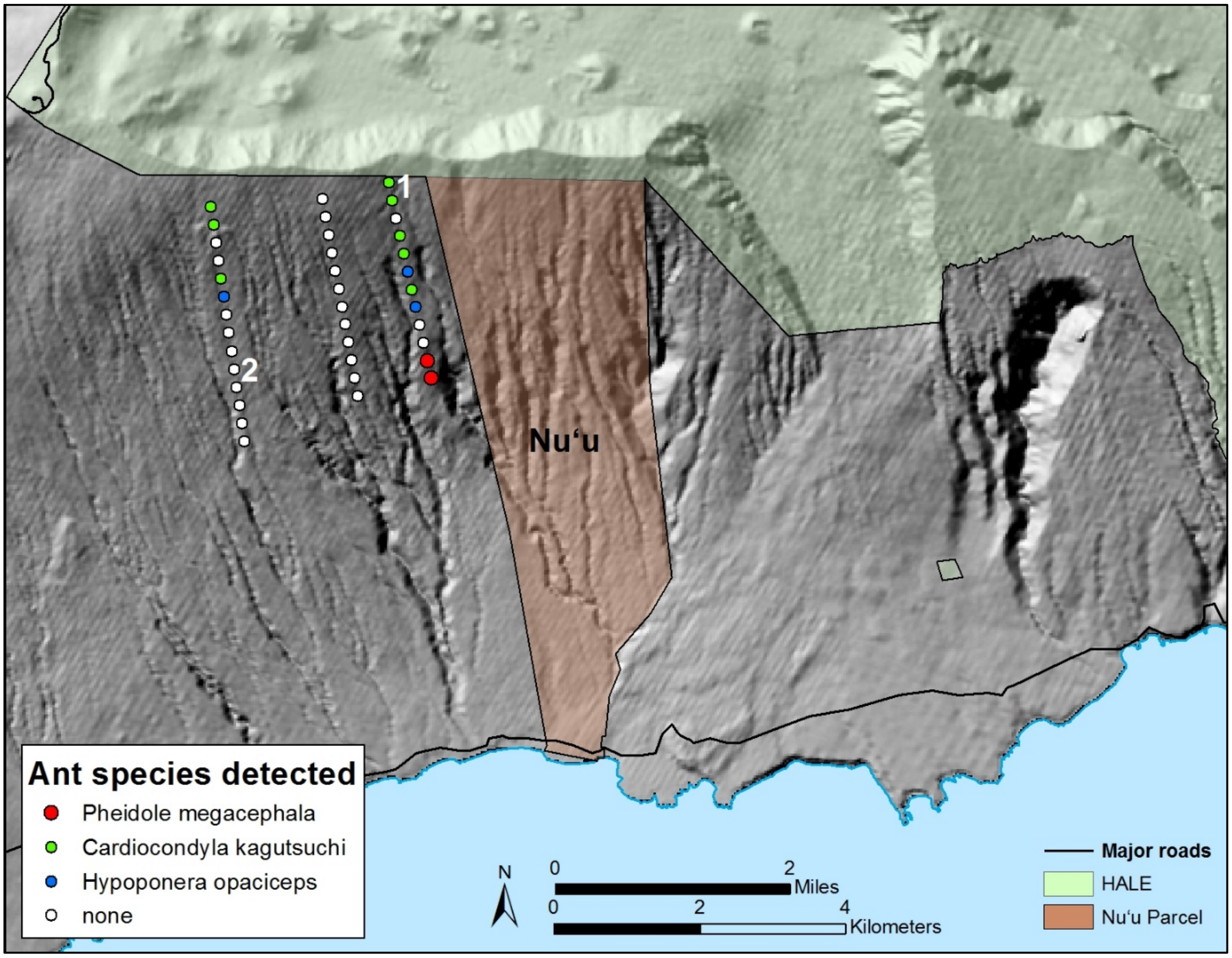

Figure 4.2.9-10. Non-native ant and native carabid ground beetles collected near the Nuʻu Parcel. One additional ant species (*Tetramorium caldarium*) was collected at a station with *Pheidole megacephala* and is not shown. The beetle species *Mecyclothorax krushelnyckyi* was collected at the location denoted with a "1", and three beetle species (*M. cordithorax, M. iteratus, M. giffardi*) were collected at the location denoted with a "2". Source: Krushelnycky (2010), Liebherr (2015).

Four species of native *Mecyclothorax* ground beetles (Coleoptera: Carabidae), which are known to be strongly impacted by aggressive ants such as *L. humile* (Krushelnycky et al. 2005a, Liebherr and Krushelnycky 2007), were found along the three ant survey transects (Figure 4.2.9-10). The presence of these beetles illustrates the potentially highly unique quality of the remaining invertebrate fauna on southern slopes of Haleakalā: while two of the species (*Mecyclothorax cordithorax* and *M. iteratus*) are widely distributed around the mountain, the other two are not (Liebherr 2015). One of the latter species, *M. giffardi*, is known only from two specimens taken west of Nuʻu, the first near the southwestern rift around Polipoli State Park in 1913, and the second within a patch of open forest (*M. polymorpha*, *A. koa*, *Ilex anomala*) in Nakula NAR in 2008 (Liebherr 2005, 2015). The other distributionally-restricted species, *M. krushelnyckyi*, is known from a single specimen collected in 2008 in subalpine shrubland just outside the boundary of Nu'u, within Kahikinui Forest Reserve (Liebherr 2015). The latter two species thus appear to be limited to the south-facing slopes of the mountain, and it is likely that at least *M. krushelnyckyi* also occurs within the boundaries of the Nuʻu Parcel. Due to the highly precinctive nature of the Hawaiian arthropod fauna (Zimmerman 1948), other locally endemic species may exist in the Nuʻu region.

Summary of condition and trend of native invertebrates

The current condition of native invertebrates in Nuʻu is judged to warrant significant concern, based on the severely degraded condition of the native vegetation. While arthropod communities in the very highest, subalpine shrubland portions of the parcel may be somewhat more intact, those in lower elevation portions are likely to be restricted to very small fragments of native vegetation, and therefore are also likely to be severely diminished in diversity and abundance. Confidence in this assessment is medium. Although there are no direct invertebrate data reported from within Nuʻu Parcel, the scale of ecosystem conversion and degradation is such that inferences regarding impacts on the invertebrate fauna can be made with some certainty. There is insufficient information with which to assess a trend in condition.

Overall assessment

Integration of the three indicators yields an overall current condition that warrants significant concern. This assessment reflects the high level of contemporary divergence from ideal reference conditions, particularly at elevations below 4,000 ft. Non-native plant richness is much higher than native richness in the lower elevation areas that were sampled. Similarly, non-native plant cover and bare ground vastly exceed that of remnant native communities, which currently persist only at upper elevations, and in a relatively degraded state. The severe erosion that has resulted from the direct and indirect effects of feral ungulates impacts both vegetation regeneration in terrestrial ecosystems and the health of nearshore marine ecosystems outside the park. Despite the persistence of remnants of the native animal fauna, the probable near absence of major bird and insect components throughout most areas of the Nuʻu Parcel implies that many important ecosystem processes are likely to be highly impaired. For example, Medeiros et al. (1986), in discussing various forces that may be hindering natural seedling recruitment in leeward Haleakalā forests and shrublands, highlight the absence of critical native birds and insects to pollinate native plants, and the same or potentially other native birds to disperse native plant seeds. Moreover, while some non-native birds may now act as effective seed dispersers of native plants (Cole et al. 1995a, b), other non-native insects, birds and rodents that have taken the place of native animals may not only fail to adequately compensate for missing interactions, but may also attack native plant tissues, depredate native seeds, and disperse invasive weeds (Medeiros et al. 1986, Chimera and Drake 2010, 2011). Finally, the conversion of closed-canopy forests and shrublands to predominantly open grasslands or bare ground is likely to have changed microclimatic conditions to the detriment of native plant seeds and seedlings (Medeiros et al. 1986, Perkins et al. 2012).

The trend in this condition was assessed as unknown: the relatively recent acquisition of the parcel by the National Park Service, and lack of any repeated quantitative data or other observations precludes an evaluation of trend. (return to Condition Summary).

Level of confidence

Overall level of confidence in this assessment is medium. Although data collected within the Nuʻu Parcel are very limited, the readily observable degree of ecosystem alteration from ideal conditions is such that uncertainty in the assessment is lessened.

Information gaps and research recommendations

Due to the rugged and steep terrain, and inaccessibility of much of the higher elevations of the Nuʻu Parcel, no NPS I&M vegetation plots or observation points were located above 4,000 ft elevation. Species richness and counts are therefore under-representative of the entire region, and particularly of the upper elevation remnant native communities. There is a complete lack of repeated quantitative vegetation information (i.e., cover, density, frequency) within the Nuʻu Parcel owing to its recent acquisition. The establishment of a network of such monitoring plots would allow for site specific comparisons over time that will greatly enhance future vegetation trend analyses. Information on vertebrate and invertebrate species is currently very limited or in most cases completely lacking.

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4.2.10. Threatened and Endangered Species

Condition Summary

Overall resource condition warrants significant concern with an unknown trend; confidence in condition determination is low. Rationale is discussed in the "*Overall assessment*" section and following.

●

Relevance and Context

Hawai'i currently has 543 federally listed threatened or endangered (T&E) species, more than any other state (USFWS 2017b). Because HALE encompasses some of the best remaining habitat for native plant and animals in the islands, it also contains many T&E species. Currently, there are 56 listed taxa known or thought to occur in the park, including species that may now be extinct or extirpated from the park (NPS 2012l, NPSpecies 2017, USFWS 2017a). This includes 43 plant taxa, 10 birds, two mammals, and one insect. Although the date of federal listing does not always coincide with the timing of a species' initial decline and endangerment, and not all taxonomic groups receive the same listing attention, the number of T&E species overall continues to grow both in HALE and statewide (Figure 4.2.10-1).

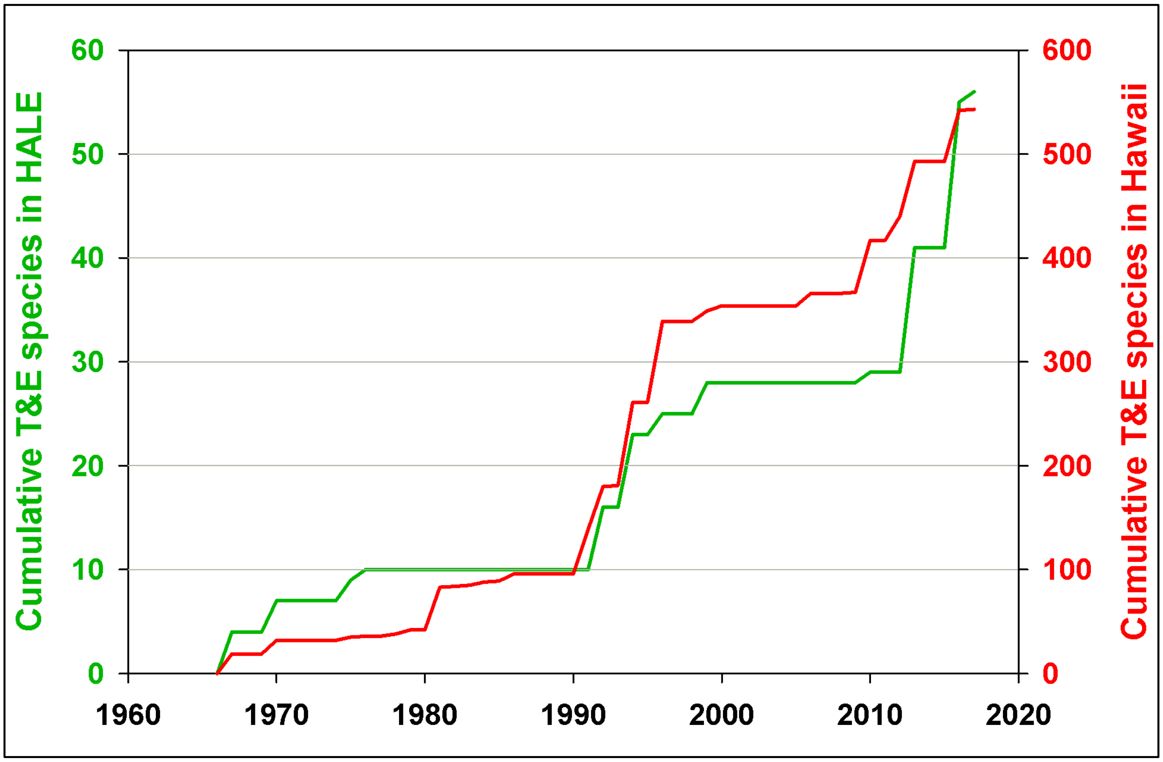

Figure 4.2.10-1. Accumulation of listed T&E species in HALE (green line and axis) and Hawai'i (red line and axis) from 1967 through 2017. Source: USFWS (2017a, b).

This suggests that the pressures responsible for species endangerment in Hawaiʻi are not abating, despite concerted management efforts on the part of private, state and federal agencies. These pressures include the various negative impacts caused by non-native species, such as browsing and trampling by ungulates (Loope and Scowcroft 1985, Cole and Litton 2014); predation and herbivory by small mammals (VanderWerf 2009, Pender et al. 2013); and predation, herbivory, and spread of disease by invertebrates (Atkinson and LaPointe 2009, Joe and Daehler 2007, Krushelnycky et al. 2017), to name a few. Past habitat destruction, especially outside the park, has decreased population sizes of several native plants and animals, exacerbating present threats. Emerging pressures from changing climate now create additional stresses (Fortini et al. 2013, 2015; Paxton et al. 2016).

In 2016, the US Fish and Wildlife Service (USFWS) designated critical habitat for 125 listed species in Maui County using an ecosystem-based approach (Figure 4.2.10-2, USFWS 2016a). This resulted in designated critical habitat covering 31,472 ac (12,736 ha) in the park, or 93% of its total area. The amount of critical habitat protected in the park also clearly indicates that HALE is an important refuge for rare plants and animals on the island of Maui (Figure 4.2.10-2).

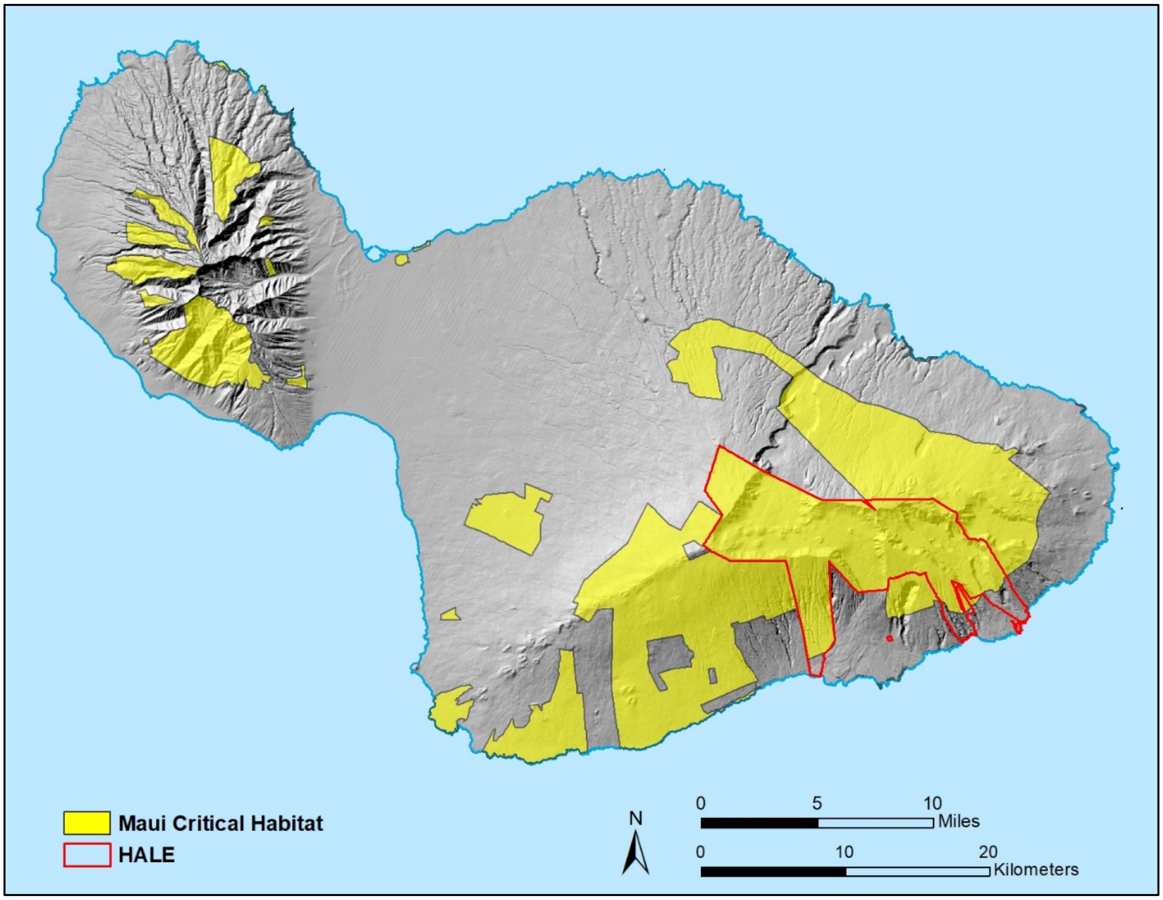

Figure 4.2.10-2. Designated critical habitat for T&E species on Maui. Source: State of Hawaiʻi Office of Planning.

Listing of species by USFWS in Hawaiʻi has focused most heavily on plants and birds, and this is reflected in the proportional makeup of T&E species in HALE. Designated critical habitat for 28 listed plant taxa together encompasses all of the critical habitat within the park, which again comprises 93% of the park area (Figure 4.2.10-3). Critical habitat has not been designated for the remaining 15 listed plant taxa thought to occur in HALE (USFWS 2016a). Critical habitat has now been designated for two of the 10 listed bird species presently or historically occurring in HALE (Figure 4.2.10-3). The designated critical habitat for these two birds (ʻākohekohe, or crested honeycreeper, *Palmeria dolei*, and kiwikiu, or Maui parrotbill, *Pseudonestor xanthophrys*) covers 26,608 ac (10,768 ha) in the park, or 79% of its total area.

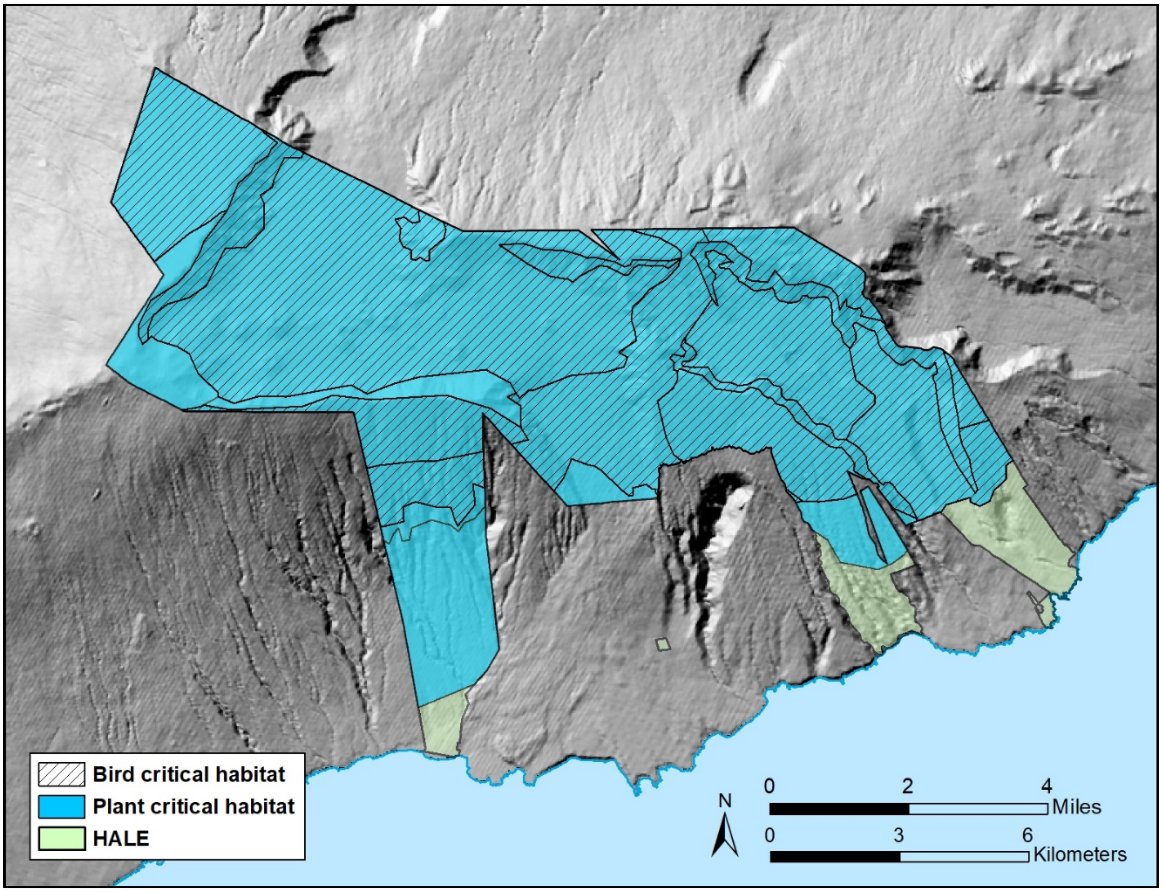

Figure 4.2.10-3. Designated critical habitat for T&E birds and plants in HALE. Polygons within critical habitat types delineate ecosystem units, as defined by USFWS. Source: State of Hawaiʻi Office of Planning.

Invertebrates represent the majority of native biodiversity in Hawaiʻi (Medeiros et al. 2013), but have only recently become more common targets for T&E evaluation and listing. A total of 76 Hawaiian invertebrate species are now recognized as threatened or endangered, including 31 arthropods and 45 snails (USFWS 2017b). However, only one of these is known to occur in HALE: the endangered pinao ʻula, or damselfly, *Megalagrion pacificum*, also known as the Pacific Hawaiian damselfly (NPS 2012l). A second endangered damselfly, *Megalagrion nesiotes*, is currently only known from a single stream on the northeast slope of Haleakalā volcano (USFWS 2010), but may also occur within the wet forests of HALE (NPS 2012l), such as in Kīpahulu Valley. Critical habitat has not yet been designated for either of these species. As mentioned in Chapter 4.2.9, the endangered Blackburn's sphinx moth, *Manduca blackburni*, has not been detected in HALE but could occur in unsurveyed portions of Nuʻu, where one of its host plants (*Nicotiana glauca*) is known to occur.

Hawaiʻi has only two native terrestrial mammal species, both of which are endangered: īlio-holo-i-ka-uaua, or the Hawaiian monk seal (*Monachus schauinslandi*), and ʻōpeʻapeʻa, or the Hawaiian hoary bat (*Lasiurus cinereus semotus*). The former occasionally hauls out on rocky shorelines at HALE, but is not considered resident in the park (NPS 2012l, Chapter 4.2.1); the latter is known to forage and may also roost in the park (Fraser et al. 2007, F. & K. Starr unpub. data). Critical habitat has not been designated for the seal within the main Hawaiian Islands, or for the bat.

The condition of the four main taxonomic groups of T&E species in HALE was assessed: plants, birds, invertebrates, and the bat (mammals). The other listed mammal, the Hawaiian monk seal, was not considered because it is not resident in the park.

Indicators

- Plants
- Birds
- Invertebrates
- Bat

Data and Methods

A review of available data and information was used to assess conditions, and if possible, trends, in the indicators listed above.

Sources of information

Information on listed plants was obtained from Krushelnycky et al. (2016), species accounts in NPS (2012a-n, 2015a-r), and USFWS (2012, 2013, 2016a,b). Monitoring, outplanting, and propagule collection data for common, rare, and endangered native taxa by HALE Vegetation Management staff were also used (NPS 2017b).

Information on listed birds was obtained from NPS (2012l), USFWS (2017a), Judge et al. (2013) and the various other literature sources cited. Additional information came from monitoring data on population numbers, nesting success, and sources of mortality collected by HALE Endangered Wildlife Management staff (NPS 2017a).

Information on listed invertebrates was obtained from Gagné (1980), Polhemus (1993), USFWS (2010), and NPS (2012).

Information on the Hawaiian hoary bat was obtained from Fraser et al. (2007), Todd et al. (2016), NPS acoustic data (NPS 2017c), and unpublished acoustic data collected by Forest and Kim Starr.

Reference condition

By definition, T&E species were judged to be in poor condition, and were often assumed to be experiencing a declining population trend, when they were listed. Current conditions and trends are therefore assessed, where possible, relative to the status of these species on or around the time of listing.

Condition and Trend

Plants

The first T&E plant species occurring within HALE were not listed until 1992, but many additional rare plant taxa in HALE have subsequently been added to the federal T&E list, mainly during the 1990's and again during the past few years (Figure 4.2.10-4). There are now 43 listed plant taxa that are known to occur, or that have historically been recorded, within HALE (Table 4.2.10-1). These include 34 dicots, three monocots, and six ferns and allies. All are federally listed as endangered except for *Argyroxiphium sandwicense* ssp. *macrocephalum*, which is currently listed as threatened.

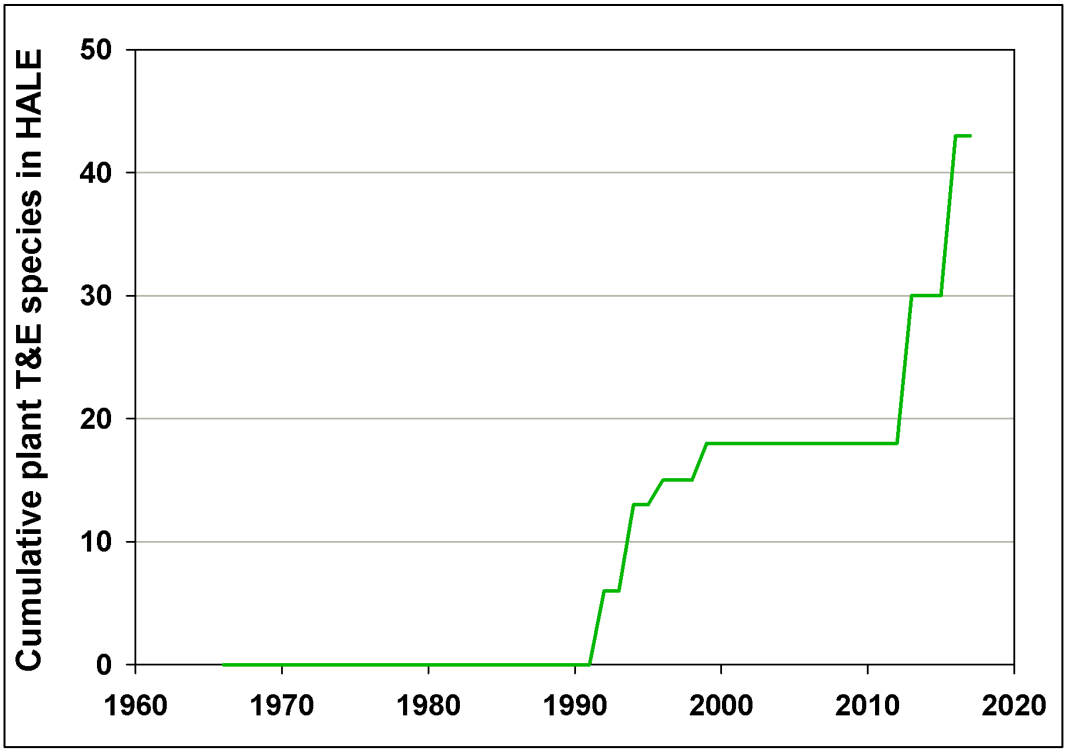

Figure 4.2.10-4. Accumulation of listed T&E plant taxa in HALE from 1967 through 2017. Source: USFWS (2017b).

Table 4.2.10-1. T&E plant taxa currently or historically occurring in HALE. All taxa are federally listed as endangered, except *A. sandwicense* ssp. *macrocephalum*, which is threatened. Sources: Except where otherwise noted, information obtained from NPS (2012a-n, 2015a-r, 2017b), USFWS (2012, 2013, 2016a, b), and unpublished expert field observations. See table footnotes for further explanations.

Scientific Name	Year listed	Ecosyst. type(s)[1]	Region(s) present[2]	Crit Hab[3]	Status in Park	Est. num. HALE[4]	Est. num. TOT.[5]
Argyroxiphium sandwicense ssp. *macrocephalum*	1992	Shrubland sparse veg	Crater West Slope	Y	Extant	39,355 (3,474)	40,000
Asplenium peruvianum var. *insulare*	1994	Shrubland	Kaupō	Y	Extant	1	100
Bidens campylotheca ssp. *pentamera*	2013	Forest	Kīpahulu	Y	Extant	50-100 (19)	200
Bidens campylotheca ssp. *waihoiensis*	2013	Forest	Kīpahulu	Y	Extant	200 (53)	200
Bidens micrantha ssp. *kalealaha*	1992	Shrubland	Crater Kaupō	Y	Extant	554 (686)	<2,000
Calamagrostis expansa	2016	Forest	ManawainuiHāna RF	N	Extant	32	<750
Clermontia samuelii ssp. *samuelii*	1999	Forest Bog	ManawainuiHāna RF	Y	Extant	24 (43)	24
Ctenitis squamigera	1994	Forest	Kīpahulu Manawainui	Y	Historic, current status unknown	0	120
Cyanea asplenifolia	2013	Forest	Ka‘apahu	Y	Extant	33 (6)	<200
Cyanea copelandii ssp. *haleakalaensis*	1999	Forest	Ka‘apahu Kīpahulu	Y	Extant	200-300 (45)	>600
Cyanea duvalliorum	2013	Forest	Kīpahulu	Y	Historic, 1919 Forbes collection	0	71

[1] Broad ecosystem type(s), as used in this report, where species occurs.

[2] General region(s) of HALE, as referenced in this report, where species occurs.

[3] Critical habitat designated within HALE by USFWS.

[4] Most recent estimated number of individuals in HALE. Numbers not in parentheses are wild individuals, numbers in parentheses are outplanted individuals (NPS 2017b). Many estimates are highly uncertain.

[5] Most recent estimated number of wild individuals in the total population for the species, including in HALE. Many estimates are highly uncertain, and approximate numbers and imprecise wording such as “a few” and “several thousand” are taken directly from listing packages or other sources.

Table 4.2.10-1 (continued). T&E plant taxa currently or historically occurring in HALE. All taxa are federally listed as endangered, except *A. sandwicense* ssp. *macrocephalum*, which is threatened. Sources: Except where otherwise noted, information obtained from NPS (2012a-n, 2015a-r, 2017b), USFWS (2012, 2013, 2016a, b), and unpublished expert field observations. See table footnotes for further explanations.

Scientific Name	Year listed	Ecosyst. type(s)[1]	Region(s) present[2]	Crit Hab[3]	Status in Park	Est. num. HALE[4]	Est. num. TOT.[5]
Cyanea hamatiflora ssp. *hamatiflora*	1999	Forest	Kaʻapahu Kīpahulu	Y	Extant	50-100 (19)	458-558
Cyanea horrida	2013	Forest	Kīpahulu ManawainuiHāna RF	Y	Extant	<5	44
Cyanea kunthiana	2013	Forest	Kaumakani Kīpahulu Manawainui	Y	Extant	100-125	165
Cyanea maritae	2013	Forest	Kaʻapahu Kīpahulu	Y	Extant	<40 (85)	23-50
Cyanea mauiensis	1996	Forest	Kīpahulu	N	Historic, 1919 Forbes collection	0	0?
Cyclosorus boydiae	2016	Forest	Kīpahulu	N	Extant	100	400
Cyrtandra ferripilosa	2013	Forest	ManawainuiHāna RF	Y	Extant	20 (40)	25
Gardenia remyi	2016	Forest	Kīpahulu	N	Extant	1-3	90
Geranium arboreum	1992	Shrubland	West Slope	Y	Extant	2 (343)	<50
Geranium hanaense	2013	Bog	Hāna RF	Y	Extant	500-700	500-700
Geranium multiflorum	1992	Shrubland Forest	Crater West Slope Kīpahulu Hāna RF	Y	Extant	400-500 (127)	500-1000

[1] Broad ecosystem type(s), as used in this report, where species occurs.

[2] General region(s) of HALE, as referenced in this report, where species occurs.

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[5] Most recent estimated number of wild individuals in the total population for the species, including in HALE. Many estimates are highly uncertain, and approximate numbers and imprecise wording such as “a few” and “several thousand” are taken directly from listing packages or other sources.

Table 4.2.10-1 (continued). T&E plant taxa currently or historically occurring in HALE. All taxa are federally listed as endangered, except *A. sandwicense* ssp. *macrocephalum*, which is threatened. Sources: Except where otherwise noted, information obtained from NPS (2012a-n, 2015a-r, 2017b), USFWS (2012, 2013, 2016a, b), and unpublished expert field observations. See table footnotes for further explanations.

Scientific Name	Year listed	Ecosyst. type(s)[1]	Region(s) present[2]	Crit Hab[3]	Status in Park	Est. num. HALE[4]	Est. num. TOT.[5]
Huperzia mannii	1992	Forest	Ka‘apahu Kīpahulu	Y	Extant	150-200 (82)	150-200
Huperzia stemmermanniae	2016	Forest	Ka‘apahu	N	Historic from early 1990s, current status unknown	0?	A few
Ischaemum byrone	1994	Coastal	‘Ohe‘o	N	Extant as outplantings	0 (822)	Several thousand
Joinvillea ascendens ssp. *ascendens*	2016	Forest	Ka‘apahu Kīpahulu	N	Extant	40 (62)	200
Melicope balloui	1994	Forest	Kīpahulu	Y	Unknown	0?	0?
Melicope ovalis	1994	Forest	Kīpahulu	Y	Extant	<50 (59)	50
Microlepia strigosa var. *mauiensis*	2016	Forest	Kīpahulu Manawainui	N	Extant	<20	<100
Nothocestrum latifolium	2016	Forest	Kaupō	N	Extant as outplantings	0 (20)	1600
Phyllostegia bracteata	2013	Forest	Kīpahulu	Y	Unknown, last seen 2010, extant as outplantings	0? (78)	?

[1] Broad ecosystem type(s), as used in this report, where species occurs.

[2] General region(s) of HALE, as referenced in this report, where species occurs.

[3] Critical habitat designated within HALE by USFWS.

[4] Most recent estimated number of individuals in HALE. Numbers not in parentheses are wild individuals, numbers in parentheses are outplanted individuals (NPS 2017b). Many estimates are highly uncertain.

[5] Most recent estimated number of wild individuals in the total population for the species, including in HALE. Many estimates are highly uncertain, and approximate numbers and imprecise wording such as “a few” and “several thousand” are taken directly from listing packages or other sources.

Table 4.2.10-1 (continued). T&E plant taxa currently or historically occurring in HALE. All taxa are federally listed as endangered, except *A. sandwicense* ssp. *macrocephalum*, which is threatened. Sources: Except where otherwise noted, information obtained from NPS (2012a-n, 2015a-r, 2017b), USFWS (2012, 2013, 2016a, b), and unpublished expert field observations. See table footnotes for further explanations.

Scientific Name	Year listed	Ecosyst. type(s)[1]	Region(s) present[2]	Crit Hab[3]	Status in Park	Est. num. HALE[4]	Est. num. TOT.[5]
Phyllostegia brevidens	2016	Forest	Kīpahulu	N	Extant as outplantings	0? (92)	5?
Phyllostegia haliakalae	2013	Forest	Kīpahulu	Y	Extant	<15 (247)	0
Plantago princeps var. *laxiflora*	1994	Shrubland Forest	Crater Kīpahulu	Y	Extant	<10 (462)	70
Platanthera holochila	1996	Forest	Hāna RF	Y	Historic, Waiʻanapanapa, Forbes 1920 collection	0	44
Portulaca villosa	2016	Coastal Grassland	Nuʻu	N	Historic, Nuʻu 1920	0 (234)	<60
Ranunculus hawaiensis	2016	Shrubland	Crater	N	Historic, 1945 Mitchell coll., east of ʻOʻili puʻu	0	14
Ranunculus mauiensis	2016	Forest	Hāna RF	N	Historic, 1945 St. John coll., Hāna RF 6300ʼ	0	165
Sanicula sandwicensis	2016	Shrubland	Crater West Slope	N	Extant	<20 (132)	(95)
Schiedea diffusa ssp. *diffusa*	2016	Forest	Kīpahulu Manawainui	N	Extant	15 (274)	<60

[1] Broad ecosystem type(s), as used in this report, where species occurs.

[2] General region(s) of HALE, as referenced in this report, where species occurs.

[3] Critical habitat designated within HALE by USFWS.

[4] Most recent estimated number of individuals in HALE. Numbers not in parentheses are wild individuals, numbers in parentheses are outplanted individuals (NPS 2017b). Many estimates are highly uncertain.

[5] Most recent estimated number of wild individuals in the total population for the species, including in HALE. Many estimates are highly uncertain, and approximate numbers and imprecise wording such as “a few” and “several thousand” are taken directly from listing packages or other sources.

Table 4.2.10-1 (continued). T&E plant taxa currently or historically occurring in HALE. All taxa are federally listed as endangered, except *A. sandwicense* ssp. *macrocephalum*, which is threatened. Sources: Except where otherwise noted, information obtained from NPS (2012a-n, 2015a-r, 2017b), USFWS (2012, 2013, 2016a, b), and unpublished expert field observations. See table footnotes for further explanations.

Scientific Name	Year listed	Ecosyst. type(s)[1]	Region(s) present[2]	Crit Hab[3]	Status in Park	Est. num. HALE[4]	Est. num. TOT.[5]
Schiedea haleakalensis	1992	Shrubland	Crater	Y	Extant	<50 (251)	<50
Solanum incompletum	1994	Forest	Kīpahulu	Y	Historic, 1919 Forbes collection	0	?
Wikstroemia villosa	2013	Forest	Manawainui	Y	Extant	<10	<10

[1] Broad ecosystem type(s), as used in this report, where species occurs.

[2] General region(s) of HALE, as referenced in this report, where species occurs.

[3] Critical habitat designated within HALE by USFWS.

[4] Most recent estimated number of individuals in HALE. Numbers not in parentheses are wild individuals, numbers in parentheses are outplanted individuals (NPS 2017b). Many estimates are highly uncertain.

[5] Most recent estimated number of wild individuals in the total population for the species, including in HALE. Many estimates are highly uncertain, and approximate numbers and imprecise wording such as “a few” and “several thousand” are taken directly from listing packages or other sources.

These species occur across all of the main ecosystem types referenced in this report except subalpine grasslands, and within all of the main park regions (Figure 4.2.10-5 and Table 4.2.10-1). However, T&E plants are much more prevalent in forests than in other ecosystem types, and are more common within Kīpahulu Valley than other park regions (Figure 4.2.10-5). These imbalances are roughly correlated with differences among ecosystems and regions in overall plant diversity.

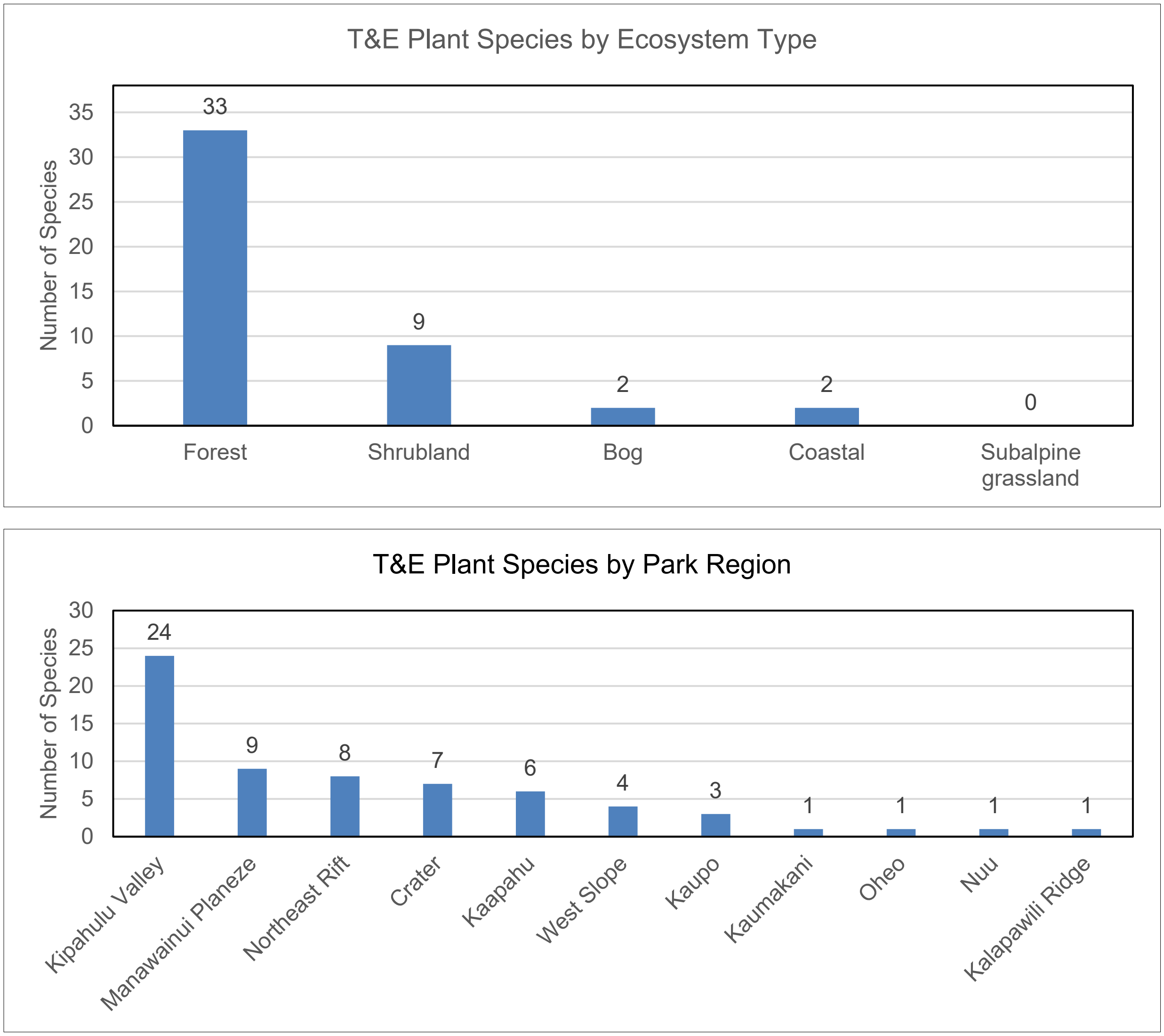

Figure 4.2.10-5. Number of T&E plant taxa currently or historically occurring in HALE, grouped by ecosystem type (top panel) and major park region (bottom panel). Source: NPS (2012a-n, 2015a-r, 2017b), USFWS (2012, 2013, 2016a, b).

USFWS shifted critical habitat designation in Hawai‘i from to species-specific habitat delineation to ecosystem unit-based habitat designation, in which ecosystem units identify critical habitat for multiple species, some of which may not currently occur within the units (USFWS 2016a). The numbers of HALE T&E plant taxa (current and historic) that are protected within designated critical habitat under this system are skewed towards mesic and wet forested areas in the eastern end of the park (Figure 4.2.10-6). It is again important to note that not all of the listed taxa currently occur in each of their respective critical habitat units, and in fact most of these units have been designated as critical habitat for many additional plant taxa that have never been recorded in the park. The latter species are indicated in Table 4.2.10-2, and represent species that could be considered for introduction into the appropriate park units in the course of ecosystem restoration.

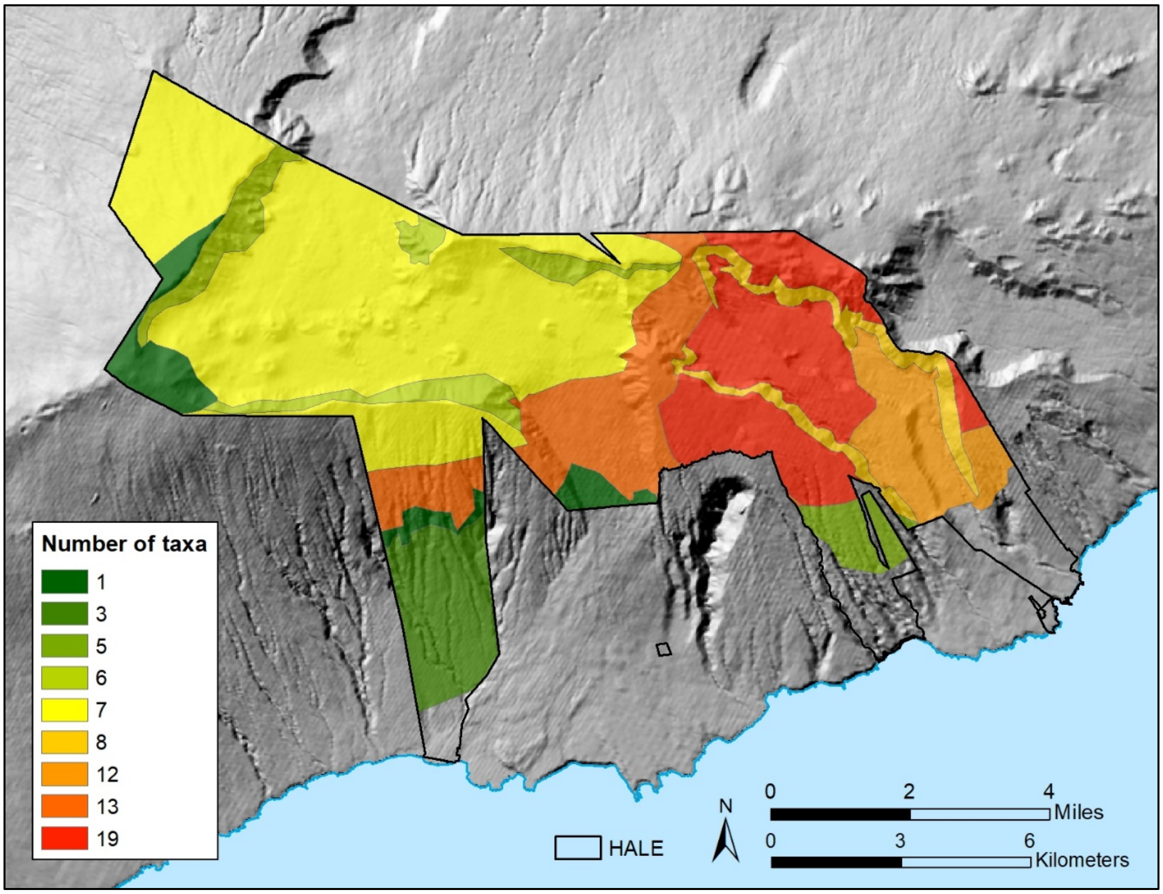

Figure 4.2.10-6. Number of T&E plant taxa currently or historically occurring in HALE that are protected within designated critical habitat units. Not all of the protected taxa currently occur in each of their respective units. Unshaded areas have not been designated as critical habitat. Source: USFWS (2016a), Hawai‘i Office of State Planning.

Table 4.2.10-2. T&E plant taxa not historically recorded in HALE, but nominally protected within the HALE critical habitat ecosystem units shown (units defined by USFWS). Source USFWS (2016a). An "X" indicates that species was detected and recorded.

Taxon	Alpine	Dry Cliff	Lowland Dry	Lowland Mesic	Lowland Wet	Montane Dry	Montane Mesic	Montane Wet	Subalpine	Wet Cliff
Adenophorus periens	–	–	–	–	–	–	–	X	–	–
Alectryon macrococcus	–	–	X	–	–	X	X	–	–	–
Asplenium dielerectum	–	–	–	–	–	–	X	–	–	–
Bonamia menziesii	–	–	X	–	–	–	–	–	–	–
Canavalia pubescens	–	–	X	–	–	–	–	–	–	–
Cenchrus agrimonioides	–	–	X	–	–	–	–	–	–	–
Clermontia lindseyana	–	–	–	–	–	–	X	–	–	–
Clermontia oblongiflora ssp. *mauiensis*	–	–	–	–	X	–	–	X	–	–
Clermontia peleana	–	–	–	–	X	–	–	–	–	–
Colubrina oppositifolia	–	–	X	–	–	–	–	–	–	–
Cyanea glabra	–	–	–	–	–	–	X	X	–	–
Cyanea mceldowneyi	–	–	–	–	X	–	X	X	–	–
Cyanea obtusa	–	–	–	–	–	–	X	–	–	–
Cyrtandra oxybapha	–	–	–	–	–	–	X	–	–	–
Diplazium molokaiense	–	–	–	–	–	–	X	X	–	–
Flueggea neowawraea	–	–	X	–	–	–	–	–	–	–
Hibiscus brackenridgei	–	–	X	–	–	–	–	–	–	–
Melanthera kamolensis	–	–	X	–	–	–	–	–	–	–
Melicope adscendens	–	–	X	–	–	–	X	–	–	–
Melicope knudsenii	–	–	–	–	–	X	–	–	–	–
Melicope mucronulata	–	–	X	–	–	X	–	–	–	–
Mucuna sloanei var. *persericea*	–	–	–	–	X	–	–	–	–	–
Neraudia sericea	–	–	X	–	–	–	X	–	–	–

Table 4.2.10-2 (continued). T&E plant taxa not historically recorded in HALE, but nominally protected within the HALE critical habitat ecosystem units shown (units defined by USFWS). Source USFWS (2016a). An “X” indicates that species was detected and recorded.

Taxon	Alpine	Dry Cliff	Lowland Dry	Lowland Mesic	Lowland Wet	Montane Dry	Montane Mesic	Montane Wet	Subalpine	Wet Cliff
Nototrichium humile	–	–	X	–	–	–	–	–	–	–
Peperomia subpetiolata	–	–	–	–	–	–	–	X	–	–
Phyllostegia mannii	–	–	–	–	–	–	X	X	–	–
Phyllostegia pilosa	–	–	–	–	–	–	–	X	–	–
Santalum haleakalae var. *lanaiense*	–	–	X	–	–	X	X	–	–	–
Schiedea jacobii	–	–	–	–	–	–	–	X	–	–
Sesbania tomentosa	–	–	X	–	–	–	–	–	–	–
Spermolepis hawaiiensis	–	–	X	–	–	–	–	–	–	–
Zanthoxylum hawaiiensis	–	–	X	–	–	X	X	–	X	–

Owing to the large number of T&E plant taxa in HALE, combined with the extremely remote nature of many of them, the Vegetation Management team is not able to regularly track population numbers of most of these species. As indicated in Table 4.2.10-1, most of the T&E species are currently estimated to exist at very low numbers in the park, and even globally in many cases, especially when considering only wild individuals. Thirteen of the 43 taxa are not currently known to have wild individuals occurring within HALE, and of these 13, wild individuals have not been seen in many decades for at least eight of them.

T&E populations have now been augmented with outplanted individuals for 58% of the taxa (25 of 43), bringing the total number of taxa currently extant in the park up to 33 (Table 4.2.10-1). Even so, many of these populations remain very small and must be considered precarious. A review of the available information suggests that only 14 of the 33 extant taxa have been observed to exhibit natural regeneration in the park (Table 4.2.10-3). For the remaining 19 taxa, natural recruitment has not been observed or observed only very rarely, and it is unknown or unclear whether any natural regeneration is occurring. Efforts to increase plant numbers via propagation and outplanting may therefore lead to self-sustaining populations for some of the taxa, but possibly not for others. For example, outplants of the coastal grass *Ischaemum byrone* have been highly successful, and are now the seed sources for further restoration efforts (P. Welton pers. comm. 2018). Conversely, outplants of *Portulaca villosa* in the same region have now all perished, without establishing local progeny. In another example, many outplants of *Geranium arboreum* have now been planted to supplement the two known wild individuals remaining in the park, but because natural seedling recruitment has only very rarely been observed (NPS 2015h), such efforts might need to be continued indefinitely to avoid extinction.

Table 4.2.10-3. Status of natural regeneration for T&E plant taxa in HALE. Sources: NPS (2012a-n, 2015a-r, 2017b), USFWS (2012, 2013, 2016a,b), and expert field observations.

Status	T&E taxa
Yes:	*Argyroxiphium sandwicense* ssp. *macrocephalum, Bidens campylotheca* ssp. *waihoiensis, Bidens micrantha* ssp. *kalealaha, Cyanea asplenifolia, Cyanea copelandii* ssp. *haleakalaensis, Cyanea hamatiflora* ssp. *hamatiflora, Cyanea kunthiana, Cyclosorus boydiae, Geranium multiflorum, Huperzia mannii, Ischaemum byrone, Melicope ovalis, Phyllostegia haliakalae, Sanicula sandwicensis*
Unknown/ Unreported:	*Asplenium peruvianum* var. *insulare, Bidens campylotheca* ssp. *pentamera, Calamagrostis expansa, Clermontia samuelii* ssp. *samuelii, Cyanea horrida, Cyanea maritae, Cyrtandra ferripilosa, Gardenia remyi, Geranium arboreum*[1]*, Geranium hanaense, Joinvillea ascendens* ssp. *ascendens*[2], *Melicope balloui, Microlepia strigosa* var. *mauiensis, Nothocestrum latifolium, Phyllostegia bracteata, Phyllostegia brevidens, Plantago princeps* var. *laxiflora, Schiedea diffusa* ssp. *diffusa, Schiedea haleakalensis, Wikstroemia villosa*[3]
Taxa not currently occurring in park:	*Ctenitis squamigera, Cyanea duvalliorum, Cyanea mauiensis*, *Huperzia stemmermanniae, Platanthera holochila, Portulaca villosa, Ranunculus hawaiensis, Ranunculus mauiensis, Solanum incompletum*

[1] Natural recruitment reported as “seldom” (NPS 2012e, 2015h)

[2] Natural recruitment reported as “rarely” observed (NPS 2015j)

[3] One seedling was reportedly observed near the single adult (USFWS 2013)

Specific causes for lack of regeneration are rarely known for certain, but most of the taxa are thought to potentially be impacted by a range of factors that may cause mortality or reduce reproduction (NPS 2012a-n, 2015a-r, 2017b; USFWS 2012, 2013, 2016a,b). Although plants in most areas of HALE are now protected from browsing and trampling by ungulates, some species like *Cyanea asplenifolia* occur outside permanently fenced areas (NPS 2015c), and recent incursions of pigs into parts of Kīpahulu Valley and Hāna Rainforest on the northeast rift area may threaten some T&E populations or individuals. Non-native rodents and slugs are thought to destroy fruits or seedlings of some species, and many species are likely to be threatened by competition from invasive non-native plants. Other common potential threats listed for HALE T&E plants include loss of effective pollinators or dispersers, fire, landslides, pathogens and changing climate (NPS 2012a-n, 2015a-r, 2017b; USFWS 2012, 2013, 2016a,b). In addition, loss of genetic diversity and deleterious demographic forces are common concerns for species with very few remaining individuals.

Sufficient data with which to reasonably assess population trends exist for only one of the T&E plant taxa: ʻāhinahina, or Haleakalā silversword (*A. sandwicense* ssp. *macrocephalum*). This plant has been studied intensively since the 1960's, with annual demographic plot data collected in most years since 1982 (Loope and Crivellone 1986, Krushelnycky et al. 2013), and full population censuses conducted roughly every decade since 1971 (Krushelnycky et al. 2016). This wealth of information has revealed that the Haleakalā ʻāhinahina made a strong population recovery from the early part of the 20th century until around 1990, most likely in response to management actions implemented by HALE, such as exclusion of feral goats and prohibition of harvesting by people (Loope and Crivellone 1986, Krushelnycky et al. 2016). Subsequently, the population entered a period of decline over the next few decades (Figure 4.2.8-11), with the total number of plants in 2013 estimated to have decreased by about 60% since 1991 (Krushelnycky et al. 2016). This population decline was driven by mortality among mostly smaller plants, combined with greatly reduced seedling recruitment, and has been tied to changing climate conditions on upper Haleakalā (Krushelnycky et al. 2013, 2016; see also chapter 4.2.8). If recent climate trends continue, most of the population can be expected to continue its decline, but with plants closer to the summit currently appearing to fare better.

Summary of condition and trend of plant species

For the remainder of HALE T&E plant taxa, very limited information precludes species-specific assessments of population trends and changes in condition since the time of listing. However, as mentioned above, the very small current population sizes of most of these taxa inherently place them at risk from both continuous pressures and stochastic events. In addition, anecdotal information suggests declining numbers of known wild individuals over the past few decades, either within HALE or for the entire global population, for some of these taxa, including *C. expansa*, *C. boydiae*, *G. arboreum*, *M. ovalis*, and *P. princeps* var. *laxiflora* (NPS 2012h,i, 2015a,f,h). For only one taxon, *B. micrantha* ssp. *kalealaha*, is there anecdotal evidence for potentially increasing population size (NPS 2012a). There is therefore little evidence to suggest that the number of T&E plant species is likely to decrease in the near future through delisting, and in fact the trend has been in the opposite direction (Figure 4.2.10-4). For all of these reasons, we assess the current condition of T&E plants to warrant significant concern, with a declining trend in this condition. Confidence in this assessment is low, because of the limited nature of data concerning most of these taxa. In particular, the effects of outplanting efforts on population trends are unclear for many of the taxa.

Birds

A total of 10 bird species with status under the Endangered Species Act (ESA) have been reported for HALE (Table 4.2.10-4). Of these, eight are listed as endangered, and two are listed as threatened. Seven of the species are known to be present in the park currently (seasonally in the case of seabirds), while three occurred in the park historically but have not been observed in recent times. One additional endangered species, the ʻio or Hawaiian hawk (*Buteo solitarius*), has been reported as a vagrant but does not occur in the park regularly (NPS 2012l).

Most of the T&E birds in HALE were listed from the late 1960's to the mid 1970's, soon after the ESA was passed (Figure 4.2.10-7, USFWS 2017b). More recently, ʻakeʻake, or band-rumped storm-petrel (*Oceanodroma castro*), was added in 2016 (USFWS 2016b), and ʻiʻiwi (*Drepanis coccinea*) was listed in 2017 (USFWS 2017a). Below, we summarize what is known about the current status and recent population trends for each of the T&E bird species in HALE.

Table 4.2.10-4. T&E bird species reported for HALE. Source: NPS (2012l), USFWS (2017a).

Scientific name	**Common name**	**Listing status**	**Year listed**	**Status in Park**
Branta sandvicensis	nēnē or Hawaiian goose	Endangered	1967	Extant
Hemignathus lucidus affinis	Maui nukupuʻu	Endangered	1970	Historical
Loxops coccineus ochraceus	Maui ʻākepa	Endangered	1970	Historical
Melamprosops phaeosoma	poʻouli	Endangered	1975	Historical
Oceanodroma castro	ʻakeʻake or band-rumped storm-petrel	Endangered	2016	Extant
Palmeria dolei	ʻākohekohe or crested honeycreeper	Endangered	1967	Extant
Pseudonestor xanthrophys	Maui parrotbill	Endangered	1967	Extant
Pterodroma sandwichensis	ʻuaʻu or Hawaiian petrel	Endangered	1967	Extant
Puffinus auricularis newelli	ʻaʻo or Newell's shearwater	Threatened	1975	Extant
Drepanis coccinea	ʻiʻiwi	Threatened	2017	Extant

Nēnē

The nēnē or Hawaiian goose (*Branta sandvicensis*) was present on all of the larger Hawaiian Islands before the arrival of humans, but was extirpated from all islands except Hawai'i by the early 1900's (Banko et al. 1999). Nēnē were reestablished on the island of Maui through a captive-breeding and release program. Nēnē were released in large numbers at HALE from 1962-1978 through a joint effort by the state and NPS. A total of 555 birds were released in the Palikū area, and 16 birds were released at Hosmer Grove (USFWS 1994, Banko et al. 1999, NPS 2017a). Despite the large number of birds released, population estimates were low (Devick 1981). Regular, intensive monitoring, research, and increased management efforts for nēnē began in the early 1990's. Much of the information about this species in the park has not been published, and is maintained in the park's unpublished data (NPS 2017a).

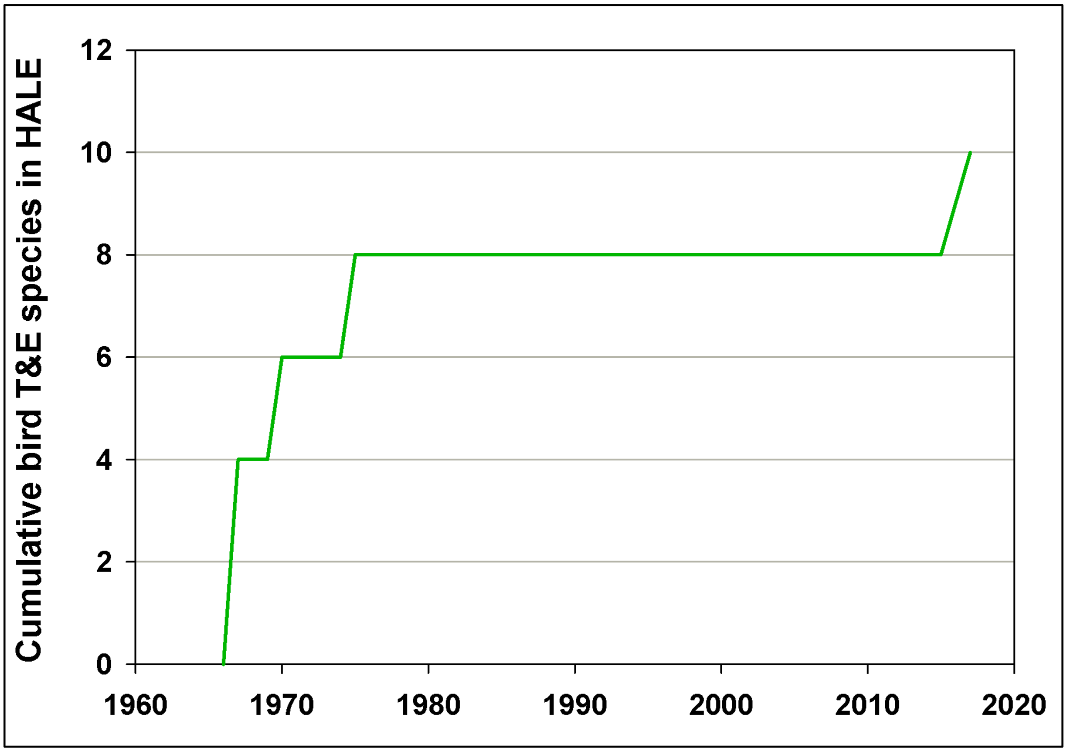

Figure 4.2.10-7. Accumulation of listed T&E bird species in HALE from 1967 through 2017. Increasing numbers result from new listings, rather than colonization of the park by previously listed species. Source: USFWS (2017b).

Nēnē are present year-round in higher elevation areas of the park, but tend to nest and form post-breeding flocks in certain areas (Figure 4.2.10-8). The estimated nēnē population in the park based on mark-resight data was reported to be 250-350 individuals in NPS (2012l). Recent unpublished data through 2016 indicate the population fluctuates from about 200-300 individuals (Figure 4.2.10-9, NPS 2017a).

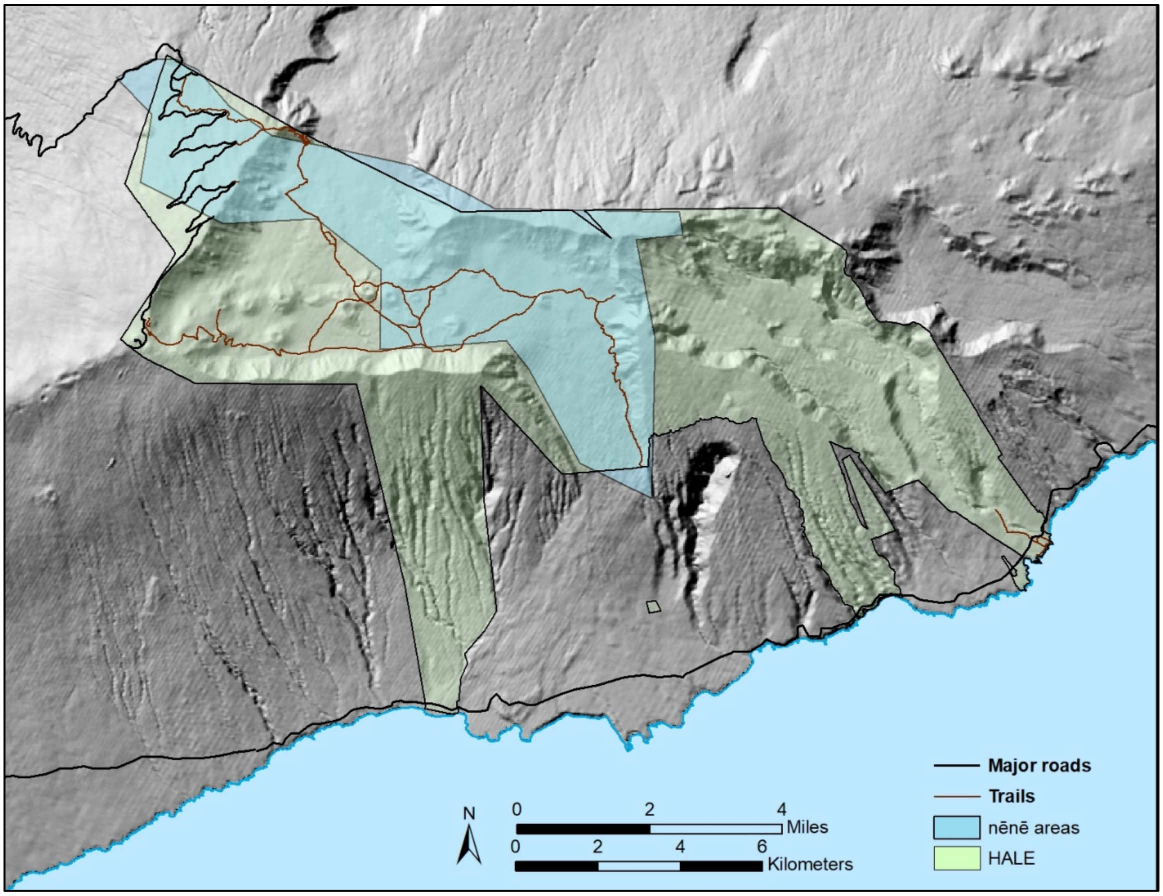

Figure 4.2.10-8. Map of nēnē general nesting and flocking areas in HALE, 2000-2017. Source: NPS (2018a).

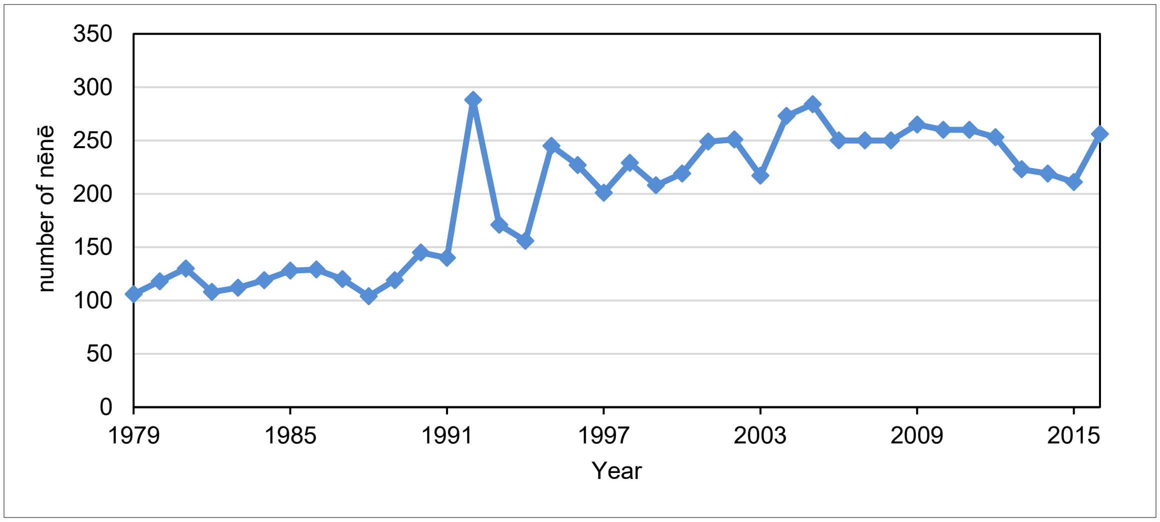

Figure 4.2.10-9. Estimated number of nēnē in HALE from 1979 to 2016. The population has remained largely stable over the last 25 years. Source: NPS (2018a).

The population size has remained fairly stable over the past few decades. It is thought that the population has not grown because the high elevation open shrubland and grassland used by nēnē in the park, and in other areas, is suboptimal habitat, owing to the cold, wet climate and the relative scarcity of adequate forage plants (Banko et al. 1999). Nevertheless, nēnē within HALE represent approximately 60% of the population for the island of Maui, based on 2017 estimates (C. Bailey pers. comm. 2018).

The average number of nēnē nests recorded in the park from 2012-2017 was 46 ± 12.7 (NPS 2018a). Nesting success of nēnē (defined as at least one egg per nest hatched) in the park from 2012 to 2017 was 76.4% ± 0.8%, and goslings fledged in 34.5% ± 12.1% of nesting attempts (nesting attempt defined as any evidence of nesting, even in absence of nests observed, including presence of brood patch or observations of goslings) (NPS 2018a). In comparison, nesting success of nēnē from 1979-1981 on Maui and Hawaiʻi Island was at least 36% and goslings fledged in 7% of nesting attempts (Banko et al. 1999). Nēnē at HALE nest during the winter months at elevations above 6,500 ft, and the cold, wet climate sometimes causes nest abandonment and gosling mortality (NPS 2012l). However, intensive habitat management including fencing and control of feral ungulates and removal of introduced predators has allowed the park's nēnē population to remain relatively stable despite the suboptimal habitat and despite the cessation of additional bird releases after 1999 (Figure 4.2.10-9). While populations currently appear to be neither increasing nor decreasing, the population is still at risk. A series of increased mortalities from vehicle collisions and dog predation in 2012-2014 (Kaholoaʻa et al. 2014, Tamayose 2015) caused a marked population decline (Figure 4.2.10-9). Such mortality events can be highly damaging, especially during low recruitment years, because nēnē nesting success at HALE is generally low compared to other species such as the related Canada goose (Tamayose 2015), owing to the suboptimal breeding habitat in the park.

Maui nukupuʻu

The endangered Maui nukupuʻu (*Hemignathus lucidus affinis*) was last observed in the park on 24 August 1967, when three individuals were reported in Kīpahulu Valley (Banko 1968). The most recent observations outside the park were in Hanawī Natural Area Reserve (NAR) in 1994-1996 (Reynolds and Snetsinger 2001). There have been no recent reports in the park, and no detections were made during the 2012 landbird survey of HALE (Judge et al. 2013), although the survey methods used were not ideal for detecting rare species. The species may no longer occur in the park and may be extinct (Camp et al. 2009, Gorresen et al. 2009, NPS 2012l). Any surviving birds would likely be on the Northeastern slopes of Haleakalā between 5,577 ft and 6,562 ft (1,700-2,000 m) and in Kīpahulu Valley (USFWS 2006).

Maui ʻākepa

The endangered Maui ʻākepa (*Loxops coccineus ochraceus*) was last seen in the park in 1980 during the Hawaiʻi Forest Bird Survey, when a total of eight individuals were observed in Kīpahulu Valley, Waikamoi, and Hanawī (Scott et al. 1986). Sightings also were reported in Hanawī NAR in 1988 (Engilis 1990). Vocalizations identified as Maui ʻākepa were reported on 24 October 1994 in Hanawī NAR and on 28 November 1995 in Kīpahulu Valley, but because of possible confusion with vocalizations of other bird species these reports are not considered confirmed (Reynolds and

Snetsinger 2001). The Maui ʻākepa was not detected during the 2012 landbird survey of HALE (Judge et al. 2013), although the survey methods used were not ideal for detecting rare species. There have been no recent reports in the park, and the species may be extinct (Camp et al. 2009, Gorresen et al. 2009, NPS 2012l). Any surviving birds would likely be on the Northeastern slopes of Haleakalā between 5,577 ft and 6,562 ft (1,700-2,000 m) and in Kīpahulu Valley (USFWS 2006).

Poʻouli

The poʻouli (*Melamprosops phaeosoma*) was discovered in 1973 in Hanawī NAR on the northern slope of Haleakalā (Casey and Jacobi 1974). Fossil records from the Ulupalakua area indicate poʻouli once inhabited mesic and drier forests at lower elevations on the leeward slopes of Haleakalā (James and Olson 1991), possibly including areas that are now part of HALE. During the Hawaiʻi Forest Bird Survey in the early 1980's only two individuals were detected, and the population size was estimated to be 140 ± 280 birds (Scott et al. 1986). Surveys from 1997-2000 located only three birds (Reynolds and Snetsinger 2001). The last known individual died in captivity during efforts to establish a captive breeding program (VanderWerf et al. 2006), and the species is thought to be extinct (Camp et al. 2009, Gorresen et al. 2009).

ʻĀkohekohe

The ʻākohekohe or crested honeycreeper (*Palmeria dolei*) occurs only in native forests in the Kīpahulu, Manawainui, and upper Hana areas of the park from 5,550-6,500 ft (1,667-1,970 m) elevation (NPS 2012l, Judge et al. 2013). The population trend of this species appears to be increasing throughout its range, which includes areas outside HALE, but is difficult to determine with certainty because of limited data and the species' rarity. Based on data from the Hawaiʻi Forest Bird Survey in the 1980's, much of which was collected outside the park, the population size of the ʻākohekohe was estimated to be 3,753 ± 373 birds (Scott et al. 1986). Subsequent surveys from 1997-2001 over the species' entire range yielded higher densities, which resulted in a population estimate of 6,745 ± 1,546 individuals (Camp et al. 2009). However, the apparently increasing trend may have been caused by differences in the season when surveys were conducted in different time periods, with surveys during the 1980 HFBS conducted later in the year after the peak in vocalizations, resulting in lower initial abundance estimates and potentially erroneous increasing trends. Judge et al. (2013) estimated the density of ʻākohekohe in the park to be 0.79 ± 0.27 birds/ha (SE), and that the population in the park was 1,150 ± 389 birds in the 1,458 ha study area. For comparison, surveys for ʻākohekohe in the core of the species' range (Hanawī NAR) indicated the density increased from 1.83 birds/ha in 1980 to 2.89 birds/ha in 1995-1997 (Scott et al. 1986, Simon et al. 2002), supporting the conclusion of range-wide increases in ʻākohekohe densities. These density estimates from Hanawī NAR are consistently higher than the density reported by Judge et al. (2013) in in the park (0.79 ± 0.27 birds/ha). Forest birds in HALE were surveyed again in 2017, and density estimates and trends for ʻākohekohe will be calculated in an NPS Focused Condition Assessment.

Kiwikiu

The kiwikiu or Maui parrotbill (*Pseudonestor xanthrophys*) is relatively rare in the park and occurs only in the higher elevation native forests in the Kīpahulu and Manawainui areas (NPS 2012l, Judge et al. 2013). The population trend of this species appears to be stable, but is difficult to determine with certainty because of limited data and the species' rarity. Based on data from the Hawaiʻi Forest Bird Survey in the 1980's, much of which was collected outside the park, the population size of kiwikiu was estimated to be 502 ± 116 birds (Scott et al. 1986). Population density (birds/ha) in 1980, 1992-1996, and 1997-2001 was estimated to be 0.17 ± 0.04, 0.17 ± 0.04, and 0.12 ± 0.03, respectively, suggesting a possible decline in abundance, but the large errors associated with the estimates caused the trend assessment to be statistically inconclusive (Camp et al. 2009, Gorresen et al. 2009). There also was some indication of a possible upward contraction of the lower range limit from 3,609 ft to 3,937 ft (1,100 to 1,200 m), but the small number of detections made the range difficult to determine accurately. However, based on surveys in 2012, Judge et al. (2013) found the density of kiwikiu to be higher in the park (0.24 ± 0.13 birds/ha) than throughout its range from 1980-2001 (0.11 to 0.17 birds/ha; Camp et al. 2009), though again the large standard errors associated with the estimates suggest they are not really different. Stein (2007) surveyed the Manawainui area above 5,200 ft (1,585 m) in January-August 2006 using spot-mapping methods in 27 10-ha sections, and estimated that 16 individual kiwikiu occupied about 100 ha of the 270 ha study area, or about 0.06 birds per hectare, suggesting this area is lower quality habitat for kiwikiu. Kiwikiu at Manawainui preferred sections with more large trees and shrubs and a denser sub-canopy, and spent the most time foraging in *Cheirodendron trigynum* (23%), *Coprosma* spp. (16%), *Ilex anomala* (15%), *Myrsine* spp. (13%), *A. koa* (10%), *Rubus hawaiensis* (10%), and *Vaccinium* spp. (9%). Kiwikiu used the canopy and sub-canopy and small trees more than expected based on the distribution of vegetation, and used *C. trigynum*, *A. koa*, and *R. hawaiensis* more than expected based on their abundances. As with ʻākohekohe, revised density estimates and trends based on a 2017 survey will be calculated for kiwikiu in an NPS Focused Condition Assessment.

ʻIʻiwi

The ʻiʻiwi (*D. coccinea*) was listed as threatened throughout its range on 20 September 2017 (USFWS 2017a). During the 2012 land bird surveys of the park, Judge et al. (2013) found that ʻiʻiwi were widespread in the forested areas of the park and were detected at 70% of stations surveyed. Judge et al. (2013) estimated their average abundance was 7.00 ± 1.11 birds/ha overall, that total population in the park was 21,850 ± 3,452 birds, and that density of ʻiʻiwi was surprisingly similar at elevations above 4,200 ft (7.65 ± 1.32 birds/ha) and below 4,200 ft (6.05 ± 1.06 birds/ha). Natividad Bailey (2007) found that relative abundance of ʻiʻiwi in the Kaʻāpahu area (between 1,800 and 3,880 ft elevation) was relatively low, and lower in 2005 (0.6 birds per point) than in 2002 (1.9 birds per point).

Based on data from the Hawaiʻi Forest Bird Survey in the 1980's, much of which was collected outside the park, the population density of the ʻiʻiwi was estimated to be 206 ± 13.2 birds/km^2 (Scott et al. 1986). Subsequent surveys over the species' entire range yielded higher density estimates in 1992-1996 (556.6 ± 22.3 birds/km^2) and 1997-2001 (520.5 ± 21.7 birds/km^2), suggesting a population increase (Camp et al. 2009). However, the apparent increasing trend may have been caused by differences in the season when surveys were conducted in different time periods, with surveys during the 1980 HFBS conducted later in the year after the peak in vocalizations, resulting in lower initial abundance estimates and potentially erroneous increasing trends. The average density of ʻiʻiwi reported by Judge et al. (2013) in the park, 7.00 ± 1.11 birds/ha (or 700 ± 111 birds/km^2), was higher than previous estimates for the species entire range, suggesting that either the density in the park is higher or that the species abundance has increased. Forest bird survey data from HALE in 2017 will be used to calculate revised density estimates and trends for ʻiʻiwi in an NPS Focused Condition Assessment.

ʻAkeʻake

The Hawaiʻi population of the ʻakeʻake or Band-rumped Storm-Petrel (*O. castro*) was listed as endangered in September 2016, after many years of being considered a candidate for listing (USFWS 2016b). As with ʻuaʻu (Hawaiian petrel, *Pterodroma sandwichensis*) and ʻaʻo (Newell's Shearwater, *Puffinus auricularis newelli*), ʻakeʻake nest in burrows or rock crevices, fly over land only at night, and can be detected by distinctive calls. ʻAkeʻake calls were heard above Kapalaoa Cabin in Haleakalā Crater on 5 August 1992, between 2000 h and 2045 h (Natividad Bailey 2009). No ʻakeʻake calls were heard during 192 hours of nocturnal surveys on 64 nights at 14 locations throughout HALE from 25 April through 2 September 2005, despite the locations being chosen based upon historical observations of this species and other seabirds, and no nests have been located in the park (Natividad Bailey 2009). However, ʻakeʻake calls were detected on acoustic meters at the Nuʻu area of the park in 2014 (NPS 2016). ʻAkeʻake calls have also been heard at Delta Camp (3,100 ft (940 m) elevation) in Kīpahulu Valley (P. Welton pers. comm. 2018). ʻAkeʻake calls are softer, higher-pitched, and more difficult to hear than ʻuaʻu calls, especially in windy conditions and in locations where many ʻuaʻu are present (Natividad Bailey 2009). ʻAkeʻake probably are present seasonally in the park each year, but their status and trend are unknown. Regular surveys of ʻakeʻake in locations away from dense ʻuaʻu colonies could be used to monitor ʻakeʻake numbers (Natividad Bailey 2009).

ʻUaʻu

The endangered ʻuaʻu or Hawaiian petrel (*P. sandwichensis*) has been regularly and intensively monitored and managed in the park since the 1970's. Much of the information about this species in the park has not been published, and is maintained in the park's unpublished data (NPS 2017a). Larson (1967) provided information about the behavior and natural history of the species, and the first quantitative data on its abundance and distribution within the park as well as the benefits of predator control.

Long-term habitat management by HALE staff, including fencing and removal of feral ungulates and removal of introduced predators, has allowed for a dramatic increase in the ʻuaʻu population in the park. There are currently over 1,850 known ʻuaʻu nesting burrows in HALE, an increase from the 14 known nests found in 1966 (Figure 4.2.10-10; NPS 2017a). Based on breeding data and GIS modeling, the ʻuaʻu population in the park is estimated to consist of 3,000-4,000 breeding pairs and a total of 8,000-9,000 individual birds. This estimate represents a 10-fold increase from the estimated population of 300-400 breeding pairs in 1966 (NPS 2012l). Although some of this increase could have resulted from increased survey effort, improved survey methods, and increases in areas surveyed, the number of active nesting burrows has undoubtedly increased substantially.

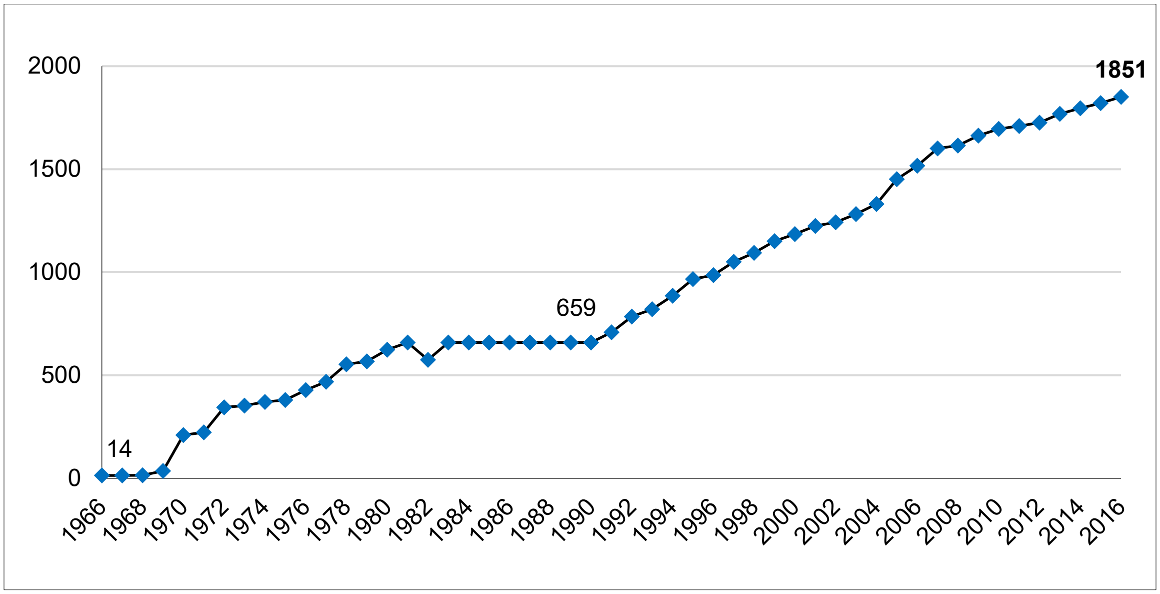

Figure 4.2.10-10. Number of known ʻuaʻu nesting burrows in HALE by year since monitoring began in 1966. Management including fencing and removal of feral ungulates and removal of introduced predators has allowed the population to increase. Source: NPS (2017a).

Larson (1967) reported that all known nesting sites were within a three square-mile area, with the greatest concentrations on the Kalahaku and Leleiwi Pali, and fewer birds on the Kapalaoa Pali. Larson (1967) estimated there were at least 300 breeding pairs on the Kalahaku and Leleiwi Pali and at least 100 pairs on the Kapalaoa Pali, but felt those estimates were conservative and conceded that the number of pairs may have been twice as high. Nests currently are distributed throughout the higher elevations of the park, mostly in rocky terrain with a slope >25 degrees (Figure 4.2.10-11).

Predation by non-native mammals and nest-trampling by feral ungulates is a serious threat to ground-nesting seabirds throughout Hawaiʻi. The predator and feral ungulate control programs in the park likely contributed greatly to the growth of the population. Predation decreased from 48% to 36% of known ʻuaʻu mortalities after an intensive predator control program began in 1982 (Natividad Hodges and Nagata 2001).

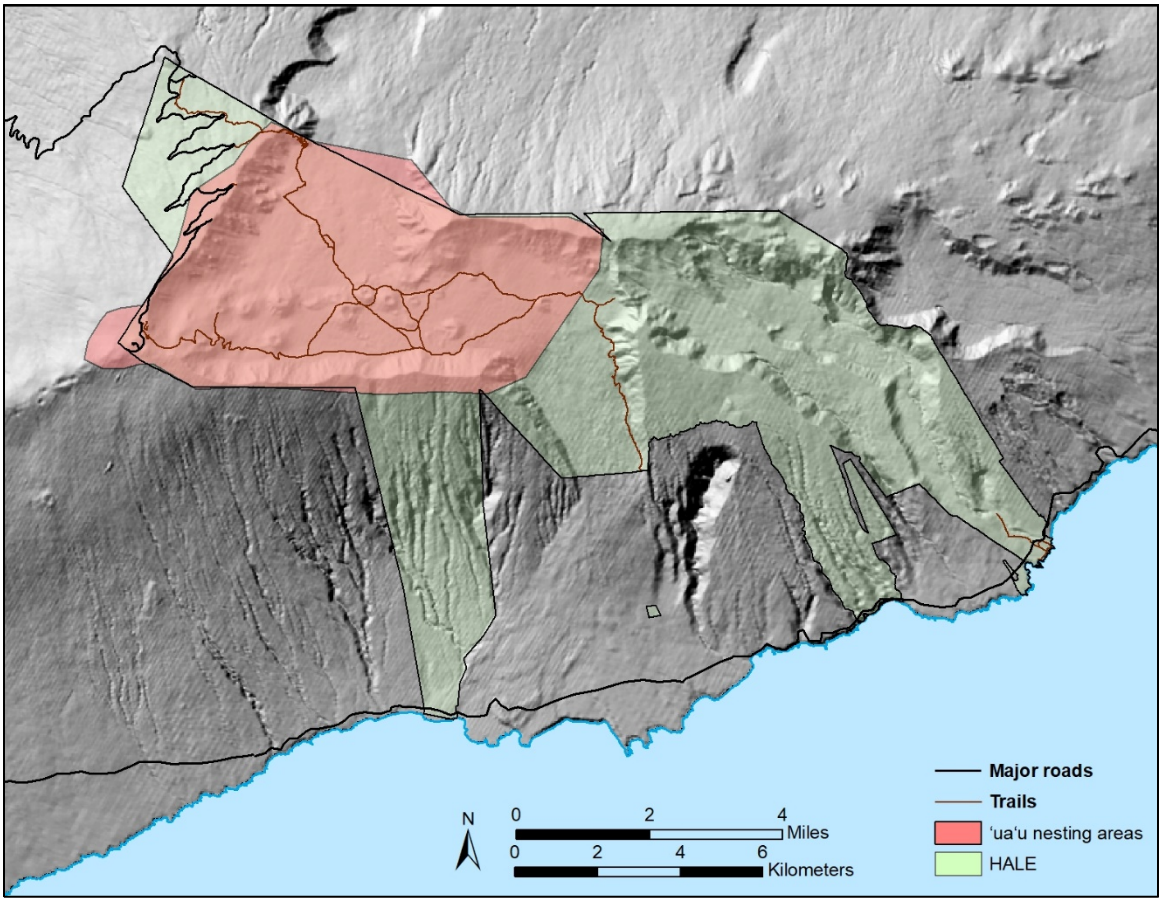

Figure 4.2.10-11. Current map of ‘ua‘u general nesting areas in HALE, 1980-2017. Source: NPS (2017a).

‘A‘o

The threatened ‘a‘o or Newell’s Shearwater (*P. auricularis newelli*) is likely to be present seasonally in the park each year, but its status and trend are unknown because of the species’ nocturnal behavior at nesting sites and the difficulty of locating nests in dense vegetation. ‘A‘o calls have been regularly reported by park field crews from various locations within Kīpahulu Valley and along the northern slope of Haleakalā near Ko‘olau Gap (Natividad Bailey 2009). One ‘a‘o was heard calling from a patch of native forest at 1,750 ft (530 m) elevation above Pua‘alu‘u Stream near Kīpahulu Valley in 1980 (Stemmerman 1980) and could have been nesting in the area or commuting to nesting areas at higher elevation in the park. ‘A‘o were detected at Delta Camp (3,100 ft (940 m) elevation) on the northeastern cliffs of Kīpahulu Valley on 24 July 1998, at around 0500 h, by Charles Chimera, who heard several loud calls from the cliffs (Natividad Bailey 2009). ‘A‘o calls also were heard at Delta Camp on 5-6 June 2012 by Judge et al. (2013). No ‘a‘o calls were heard during 192 hours of nocturnal surveys on 64 nights at 14 locations throughout HALE from 25 April through 2 September 2005, despite the locations being chosen based upon historical observations of this species and other seabirds (Natividad Bailey 2009).

A nocturnal seabird survey using a portable marine radar was conducted in the park by Cooper and Day (2003), which indicated the presence of ‘a‘o, but was not able to determine the number of individuals present. Seabirds were detected flying inland at the mouths of Kīpahulu Valley, Kaupō Gap, and Koʻolau Gap, and the timing of the detections relative to sunset was used to identify the birds as either ʻaʻo, which typically begin passing at least 60 minutes after sunset, or ʻuaʻu, which usually pass by earlier. Most of the birds detected were ʻuaʻu, but some of the detections occurred >60 minutes after sunset, suggesting they were ʻaʻo.

The radar survey conducted by Cooper and Day (2003) can serve as a baseline that can be used to monitor abundance of ʻaʻo in HALE. Similar radar surveys were important in detecting declines of ʻaʻo on Kauaʻi (Day et al. 2003, Raine et al. 2017). When conducted at sea level, radar surveys detect the movement of flying seabirds likely traveling to inland nesting sites. These surveys detected birds at sea level at Kaupō, Mokula Point, and ʻOheʻo, which were probably ʻaʻo flying inland to park lands at higher elevations (Cooper and Day 2003). Follow-up radar surveys from these locations could provide insight about ʻaʻo nesting locations and relative population trends within HALE.

ʻIo

In addition to the species discussed above, the endangered ʻio or Hawaiian hawk (*B. solitarius*) is currently resident only on Hawaiʻi Island, but it has been reported a number of times from several of the other Southeastern Hawaiian Islands. There are at least nine reports of ʻio from Maui, three of which were in HALE: in Kaupō Gap on 2 June 1971, at Hosmer Grove on 10 July 1994, and in Kīpahulu Valley on September 7 Mar 2005 (Pyle and Pyle 2017). However, none of the descriptions completely eliminated other migratory species of hawks that might occur in the Hawaiian Islands, and some reports involved observers that lacked previous experience with the species. All reports away from Hawaiʻi Island are considered unsubstantiated, including the three reports from HALE (Pyle and Pyle 2017).

Summary of condition and trend of bird species

In summary, of the seven listed bird species that are currently extant in HALE, one (ʻuaʻu) has been increasing in population size over recent decades, two (nēnē, ʻiʻiwi) appear to currently have relatively stable populations but remain at risk, and two (ʻākohekohe, kiwikiu) exist at low densities and are currently difficult to assess in terms of population trends. For the two remaining species (ʻaʻo and ʻakeʻake), no population estimates for the park currently exist, and it is not possible to assess current conditions or population trends since listing. Based on this information, we assess T&E birds in HALE to warrant moderate concern, and to possess an unknown trend. Confidence in this assessment is medium, not only because information on ʻaʻo and ʻakeʻake is very limited, but also because survey data for ʻākohekohe, kiwikiu and ʻiʻiwi are relatively sparse. We also caution that warming temperatures are likely to drive mosquitos and the diseases they vector into higher elevations in the future (Fortini et al. 2015, Paxton et al. 2016), which is likely to strongly negatively impact the future condition and trends for forest bird species such as ʻākohekohe, kiwikiu, and ʻiʻiwi.

Invertebrates

Although it once occurred on all of the main Hawaiian Islands (Polhemus and Asquith 1996), *Megalagrion pacificum* is now found in only 14 streams on Maui and at least seven streams on Molokaʻi (USFWS 2010). However, no population estimates are available for any of these localities (USFWS 2010). In HALE, information on *M. pacificum* is restricted to two sets of observations in

1980 and 1993. Gagné (1980) collected adults near a taro patch along Puaʻaluʻu Stream between 492 and 984 ft (150-300 m) elevation in July of 1980, but did not encounter them along the Palikea Stream system. A year earlier, Hardy (1979) failed to detect this species during his survey of Puaʻaluʻu Stream. During a two-day survey of the Palikea Stream system in 1993, Polhemus (1993) reported that adults were "regularly observed along the course of Palikea Stream between 450 and 1200 ft, indicating that a large and healthy breeding population is present." Polhemus (1993) observed this species at 4 of 11 collecting stations. While this species typically breeds in slow-moving, seepage-fed side pools adjacent to primary stream channels, an absence of this habitat type at Palikea Stream led to its breeding "in seepage fed pools in bedrock, some of which were nearly stagnant, and most of which lacked emergent vegetation. This indicates that the species has a somewhat wider ecological tolerance in terms of breeding sites than was previously realized" (Polhemus 1993). No subsequent estimates of its distribution or relative abundance in HALE could be found.

Because of the very limited information on *M. pacificum*, an unknown condition and trend is assigned for T&E invertebrates at HALE.

Bat

The Hawaiian hoary bat or ʻōpeʻapeʻa (*Lasiurus cinereus semotus*) is the only land mammal native to Hawaiʻi. It is considered a subspecies of the North American hoary bat. It is found on all of the larger Hawaiian Islands (USFWS 1998, Gorresen et al. 2013), and was federally listed as endangered in 1970 owing to apparent population declines, habitat loss, and lack of information on its distribution (USFWS 1998). Unlike many bats, ʻōpeʻapeʻa roost primarily in trees (USFWS 1998, Todd et al. 2016), but may also rarely use lava tubes (Fujioka and Gon 1998), cracks in rocks, or man-made structures. Owing to its solitary and cryptic roosting behavior (Bonaccorso et al. 2015), robust estimates of its population size and trend are unavailable (USFWS 1998).

Similar to other insectivorous bats, echolocation is used by ʻōpeʻapeʻa to locate a variety of native and non-native night-flying insects, including moths, beetles, crickets, mosquitoes, and termites (Whitaker and Tomich 1983, Belwood and Fullard 1984, Jacobs 1999, Bernard 2011). Detection of ecolocation calls is therefore frequently used to assess its presence and activity (e.g. Todd et al. 2016). Individuals begin foraging either just before or after sunset depending on the time of year. Elevation also may affect activity patterns; bats have been detected from sea level to at least 11,810 ft (3,600 m) elevation (Bonaccorso et al. 2015). Bats appear to be most numerous in native rain forests up to at least 1,830 m (6,000 ft), but they also use mesic and xeric habitats and disturbed areas with non-native vegetation and urban areas (Bonaccorso 2010). Water courses and edges (e.g., coastlines and forest/pasture boundaries) are important foraging areas and the species also is attracted to insects that congregate near artificial lights.

Information about ʻōpeʻapeʻa in and around HALE is currently limited to several efforts to detect bat activity using acoustic bat detectors and a limited amount of visual surveying. In May and June of 2005, Fraser et al. (2007) attempted to detect bats over the course of six days along transects in the West Slope frontcountry and ʻOheʻo areas of the park. On the West Slope, they conducted surveys for two nights and one morning around Hosmer Grove, and conducted a driving transect along the park road to the summit. At ʻOheʻo, they conducted three dawn and dusk surveys around the bunkhouse and maintenance yard, and conducted a walking transect along Pīpīwai Trail and other paths north of the road. In these efforts, they detected five feeding buzzes on 31 May 2005 at Hosmer Grove, and made a single detection along Pīpīwai Trail on 4 June 2005 (Fraser et al. 2007).

More recently, acoustic bat detectors with data loggers to record overnight activity were placed at 16 locations in HALE, plus one within the Haleakalā Observatories complex immediately outside the park near the summit (Figure 4.2.10-12, F. & K. Starr unpub. data). Between May 2014 and May 2016, a total of 135 detector-nights were recorded across the 17 locations (Table 4.2.10-5). Bats were detected at 12 of the 17 locations (Table 4.2.10-5, Figure 4.2.10-12).

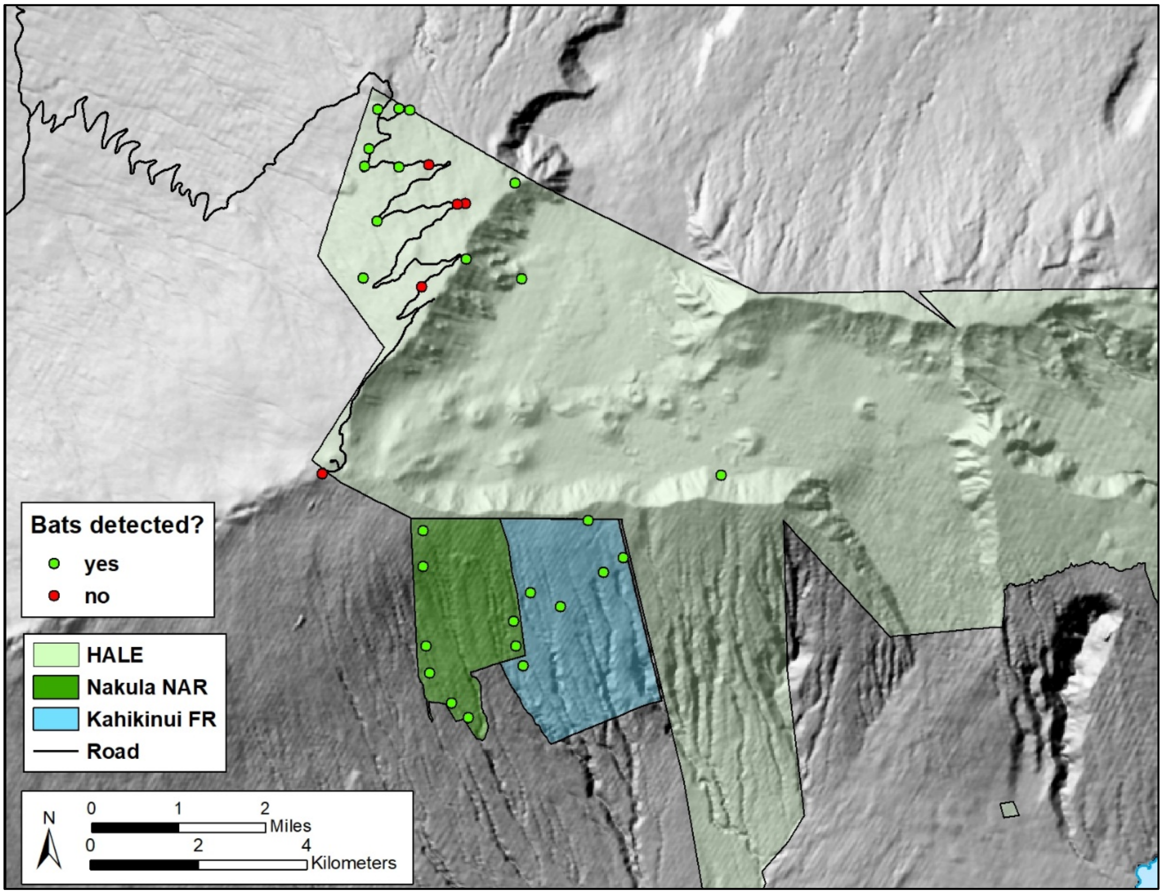

Figure 4.2.10-12. Map of bat acoustic detector sampling locations in and around HALE. Seventeen locations were sampled within HALE and near the summit by F. & K. Starr from 2014 to 2016. Fourteen locations in Kahikinui Forest Reserve and Nakula Natural Area Reserve were sampled by USGS from 2012 to 2014. One additional detection was made along Pīpīwai Trail at ʻOheʻo in 2005. Source: Fraser et al. (2007), F. & K. Starr (unpub. data), Todd et al. (2016).

The highest calling rates per night were recorded near the park entrance station and Hosmer Grove, and at a Eucalyptus Grove near the 8,500 ft elevation turn in the park road (Table 4.2.10-5). The latter location was thought to potentially represent a roosting site, because calls were detected just after sunset on multiple nights (F. Starr pers. comm. 2018). At most other locations, bats appear to be mainly passing through because they exhibited only short-duration passes and little evidence of feeding bursts. These data suggest that bats occur frequently but in relatively low numbers at most of the sampling locations (F. Starr pers. comm.). It is notable that no bats were detected over the course of 27 sampling nights at the Haleakalā Observatories near the summit.

Table 4.2.10-5. Survey effort, frequency of detection, and average acoustic activity level at 17 bat detection sites in HALE between 2014 and 2016. Source: F. & K. Starr (unpub. data).

Location	No. nights surveyed	Detection freq (% of nights)	Average no. pulses/night
RM - Housing / Fee Station	11	100	554.7
8500 ft Eucalyptus Grove	15	100	248.1
Hosmer Grove - FAA Road	28	68	146.9
Hōlua Hilton – Horse Pasture	2	100	35.0
Halemauʻu Trail – at old gate	2	100	30.0
Park Road – Green Bridge Gulch	1	100	18
HALE Headquarters	1	100	16
Hosmer Grove - TNC gate	7	100	15.0
Kapalaoa Cabin	1	100	6.0
Leleiwi Overlook	5	40	4.0
Park Road – below Headquarters	1	100	4.0
Park Road – 7700 ft turn	6	33	0.3
Park Road – between Leleiwi and Kalahaku	6	0	0.0
Halemauʻu parking lot	7	0	0.0
Park Road – near water intake pond	9	0	0.0
Park Road – Halemauʻu hitchhiker pullout	6	0	0.0
Haleakalā Observatories – gravel parking lot	27	0	0.0

Between July 2012 and November 2014, acoustic bat detectors were placed near HALE at 14 stations in Kahikinui Forest Reserve and Nakula Natural Area Reserve (Figure 4.2.10-12, Todd et al. 2016). These detected bat vocalizations at all 14 stations, and feeding pulses at 13 of the stations, during the study period. Overall, bats were detected on 60% of study nights and during every month of the study period. Activity was significantly higher in forest remnant habitats than in subalpine shrubland habitats; feeding activity in the higher elevation shrubland habitats was generally restricted to the first few hours after sunset, likely because air temperatures subsequently became too cold for bat or insect prey flight (Todd et al. 2016).

In the summer of 2017, acoustic bat detectors were placed at two locations in ʻOheʻo (ʻOheʻo Baseyard and ʻOheʻo Gulch) and at two locations in the Summit District (RM Office and Kalahaku Overlook). Preliminary data indicate that bats were regularly detected during 2,567 hours of recordings at ‘Ohe‘o (NPS 2017c). Bats were also detected during 1,294 hours of recordings in the Summit District, but less frequently than at ‘Ohe‘o (NPS 2017c).

Surveys are currently being conducted by H.T. Harvey and Associates to determine home range, seasonal movements, foraging and roosting habitat use, and diet and prey availability of ‘ōpe‘ape‘a on Maui. Information from this study will provide further insight into bat presence and activity at HALE. Preliminary data suggest that within the Maui upcountry and HALE areas, bats prefer gulches and grassland/shrubland habitats (D. Johnston, K. Jonasson, and B. Yuen (H. T. Harvey & Associates), unpubl. data).

Collectively, the bat detection data suggest that ‘ōpe‘ape‘a are commonly traversing and foraging in large parts of upper Haleakalā volcano, including within HALE. They also may frequent coastal areas of HALE, and may be roosting in certain favorable locations in the park. It is not possible at this time, however, to estimate population densities or trends in population size since the time of listing. We therefore assign an unknown current condition and trend for this endangered species.

Overall assessment

Integrating the four indicators for T&E species according to SotP aggregation rules yields a current overall condition that warrants significant concern. Most of the many listed plant taxa exist as very small populations, which inherently places them at risk from both continuous pressures and stochastic events. In addition, anecdotal information suggests declining numbers of known wild individuals over the past few decades for some of these taxa, while anecdotal evidence for potentially increasing population size exists for only one taxon. There is therefore little evidence to suggest that the number of T&E plant species is likely to decrease in the near future through delisting, and in fact the trend has been in the opposite direction. Of the seven listed bird species that are currently extant in HALE, one (‘ua‘u) has been increasing in population size over recent decades, two (nēnē, ‘i‘iwi) appear to currently have relatively stable populations but remain at risk, and two (‘ākohekohe, kiwikiu) exist at low densities and are currently difficult to assess in terms of population trends. For the two remaining species (‘a‘o and ‘ake‘ake), no population estimates for the park currently exist, and it is not possible to assess current conditions or population trends since listing. Future warming temperatures are likely to drive mosquitos and the diseases they vector into higher elevations, which is likely to strongly negatively impact the future condition and trends for T&E forest bird species. Current information is too limited to assign a condition or trend for the single listed invertebrate and bat species in HALE.

SotP aggregation rules would result in an unchanging trend in the overall condition, but we judge this to be inappropriate because the declining trend among plants involves many more species than the unknown trend among birds. This, combined with consideration of the unknown trends for the listed invertebrate and bat, led us to forego assigning an overall trend for T&E species. (return to Condition Summary)

Level of confidence

Overall level of confidence in this assessment is low. Robust data on population sizes, and especially trends, are lacking for many of the T&E plant taxa, making it difficult to assess the degree to which their precarious conditions may have changed since the time of listing. Limited information on several of the T&E bird species, as well as the single listed invertebrate and bat, also lowers confidence in the overall assessment.

Information gaps and research recommendations

For a number of the T&E species, including the invertebrate, bat, and several birds and plants, current or past population sizes are unknown. For most of the T&E species, repeated monitoring data allowing estimation of population trends are not available.

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4.3. Landscape Condition Context

4.3.1. Fuel and Fire Dynamics

Condition Summary

Overall resource condition warrants moderate concern with an unknown trend; confidence in condition determination is medium. Rationale is discussed in the "*Overall assessment*" section and following.

🟡

Relevance and Context

Hawai‘i is generally thought to have had a relatively low natural frequency of wildfire prior to human arrival (Loope et al. 1990, Smith and Tunison 1992, LaRosa et al. 2008). The principal natural ignition sources in Hawai‘i are volcanism and lightning (Trauernicht et al. 2015). However, fires ignited by active lava flows would only have occurred regularly during the early, volcano-building stages of islands (Loope et al. 1990). Similarly, lightning is relatively infrequent in Hawai‘i because the small landmasses of the islands do not commonly generate large thunderstorms, and lightning-ignited fires are thought to be fairly uncommon events (NPS 2005).

Although it is generally not possible to reconstruct the natural fire history across the many Hawaiian ecosystem types, the available evidence suggests that the natural fire regime in many areas of Hawai‘i can be characterized as "fire-independent" (Smith and Tunison 1992). This implies that "pre-settlement fires were probably irregular, infrequent, short-term ecological perturbations from which vegetation eventually recovered" (Smith and Tunison 1992), and that native Hawaiian ecosystems are generally not dependent on regular fire occurrence for their persistence. Palynological evidence from upper Haleakalā, for example, suggests that fire was infrequent pre-historically, but more common during periods of drought (Burney et al. 1995, Crausbay et al. 2014). Estimates of the mean fire return interval also indicate that most regions of east Maui, with the exception of lowland areas, probably experienced infrequent natural wildfire (Figure 4.3.1-1).

As a consequence, many native plant species are traditionally thought to be relatively fire-intolerant (Loope et al. 1990, Smith and Tunison 1992). Although a variety of species have been observed to survive and/or resprout after fire (LaRosa et al. 2008, Ainsworth and Kauffman 2009), other species appear much more vulnerable to fire, and few native species exhibit clear adaptations to fire (Smith and Tunison 1992, LaRosa et al. 2008). Perhaps more importantly, native species must now contend with a greatly altered fire regime, created by a much higher frequency of human-caused ignitions combined with widespread invasion of fire-promoting non-native plants (Trauernicht et al. 2015). These non-native plants, especially grasses, not only greatly increase the scale and intensity of fire, but also re-colonize burned areas more quickly than native woody species. This can create a positive feedback, the so-called grass/fire cycle, whereby native plant communities are converted to communities dominated by fire-adapted grasses and other weeds (Hughes et al. 1991, D'Antonio and Vitousek 1992, LaRosa et al. 2008).

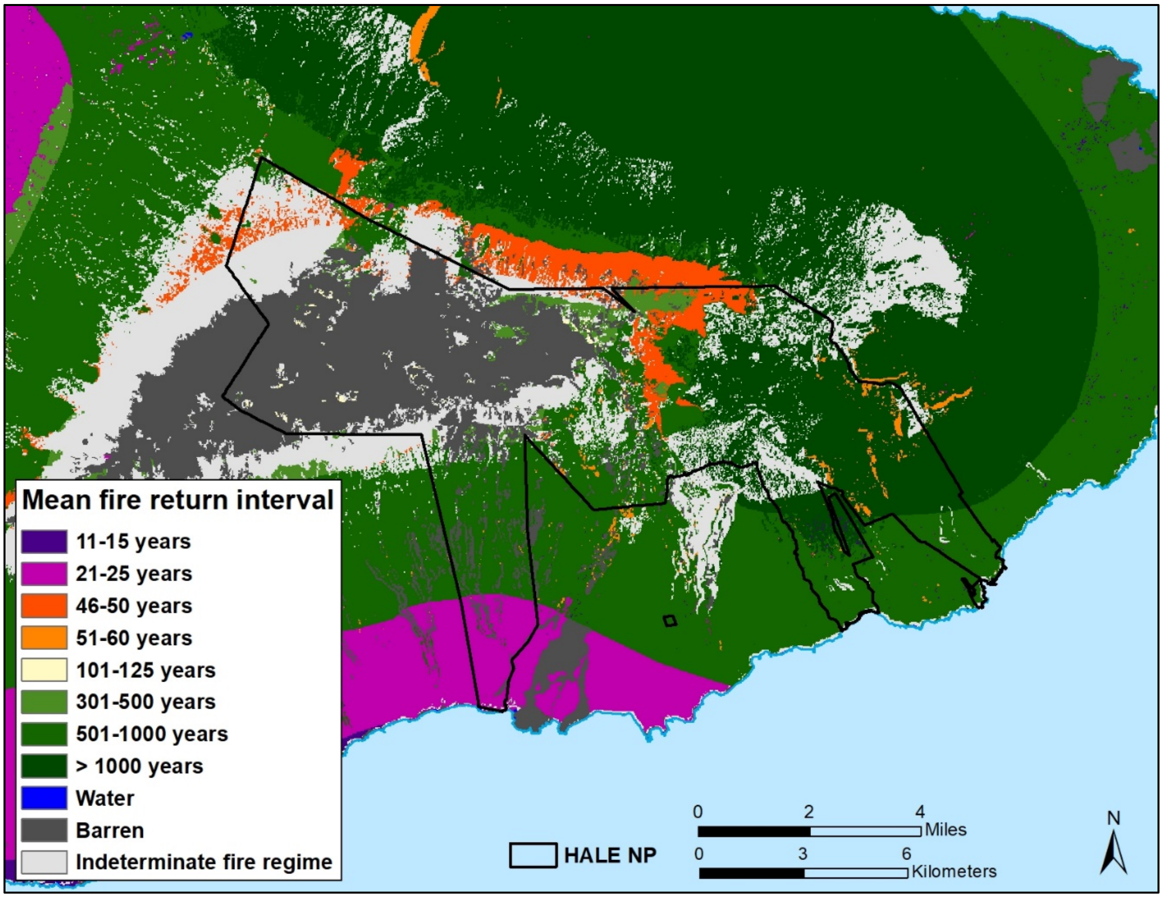

Figure 4.3.1-1. Presumed historical mean fire return interval in east Maui. Source: LANDFIRE (2017).

The responses of native plant communities in Hawai‘i to contemporary fire regimes have not been extensively studied, though available findings suggest variable and sometimes complex changes depending on local characteristics (LaRosa et al. 2008). Frequent fire is generally recognized to decrease the cover and diversity of native plants in most Hawaiian plant communities (LaRosa et al. 2008; Ainsworth and Kauffman 2010, 2013). In an example from HALE, a small 0.6 ac (0.2 ha) fire in the West Slope shrubland in 1992 resulted in a 400% increase in cover of non-native grasses relative to adjacent unburned areas six years after the fire; the native shrubs *Leptecophylla tameiameiae* and *Vaccinium reticulatum* were slow to recover (Figure 4.3.1-2, LaRosa et al. 2008). This event illustrates the concern that fire is likely to inhibit regeneration of native shrubland species in HALE, especially in the presence of fast-growing, fire-tolerant non-native plants (Loope et al. 1992). More recently, several species of pines have been invading rapidly into shrubland and sparsely vegetated Crater ecosystems. Monterey pine (*Pinus radiata*) is currently the most common pine invading the Crater, but was uncommon before 2007, when a large fire southwest of HALE burned a ~600-ac (243 ha) plantation, sending many seeds upslope and into the Crater with the wind. This and related species, such as *P. patula* and *P. pinaster,* can overtop shorter-statured native communities and eventually convert them into fire-prone coniferous monocultures (Burns and Honkala 1990, Loope et al. 1992).

Figure 4.3.1-2. 1998 photo of 0.2 ha area that burned near park headquarters in 1992. Non-native grasses dominate the burned area in the foreground, while the unburned background is comprised of native shrub canopy. Photo: S. Anderson.

For the reasons outlined above, fire management policy at HALE is to prevent and suppress all wildfires. The 1990 Fire Management Plan (FMP) for the park stated that “In natural areas of Hawai‘i, fire is therefore considered a negative influence which must be suppressed to the extent possible” (Loope et al. 1990). Similarly, the most recent HALE FMP, completed in 2005, stated the following as one of its goals: “Suppress all wildland fires, regardless of ignition source, to protect employees, the public, property and natural and cultural resources within and adjacent to the park.” (NPS 2005). Both FMPs specify that prescribed burns are not currently recognized as acceptable management tools at HALE, with the possible exception of maintaining archeological sites in certain highly specific and carefully reviewed situations (Loope et al. 1990). All of HALE is designated as a single Fire Management Unit (FMU) (NPS 2005). However, the FMU has been divided into nine FMU Segments (Figure 4.3.1-3), which allow for greater control and flexibility in managing fire in the park (NPS 2005). Presumably, the recently acquired Nu‘u Parcel will be added as a tenth FMU Segment in future versions of the FMP.

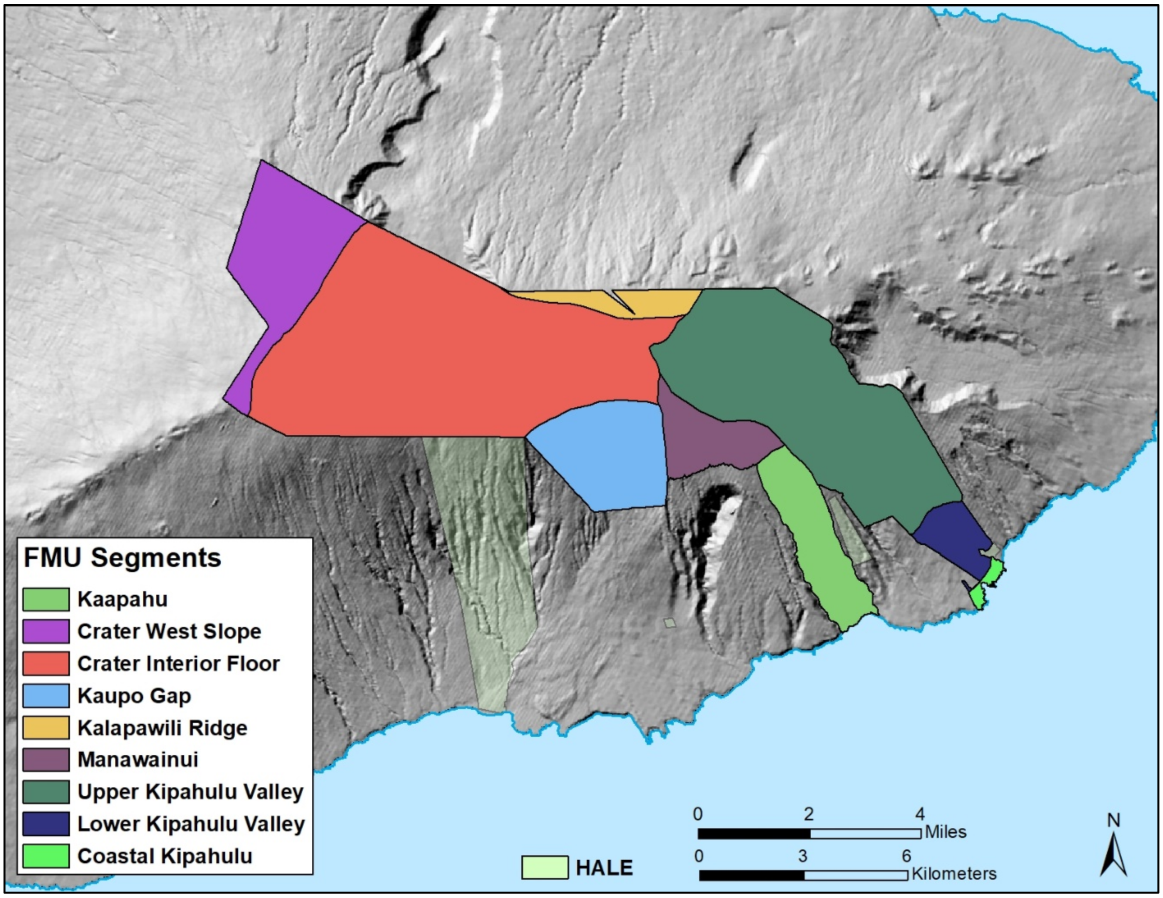

Figure 4.3.1-3. The nine Fire Management Unit (FMU) Segments designated in HALE, as of the most recent Fire Management Plan adopted prior to the acquisition of the Nu‘u Parcel. Source: NPS (2005).

The ability to achieve the park's goal of zero tolerance for wildland fires depends in large part on the condition of fuels in HALE, particularly those comprised of non-native plants that promote fire. The degree of success in achieving the goal may be assessed by the frequency and size of wildfires in the park. We therefore assess the condition of fuel and fire dynamics at HALE in terms of these two indicators: fuel characteristics and fire frequency and size.

Indicators

- Fuel characteristics
- Fire frequency and size

Data and Methods

A review of available data and information was used to assess conditions, and if possible, trends, in the indicators listed above. No additional data analysis was performed.

Sources of information

Information on fuel characteristics was obtained from the two existing FMPs for HALE (Loope et al. 1990, NPS 2005), non-native fine fuel distributions were extracted from Green et al. (2015), and fuel model information was obtained from Anderson (1982) and LANDFIRE data sources (LANDFIRE 2017).

Information on fire frequency and size was obtained from the 2005 HALE FMP (NPS 2005) and the Hawaii State Wildfire History Dataset (HWMO 2013).

Reference condition

The reference condition for fuel characteristics is fuel types and loads existing in native vegetation communities. Fuels comprised of non-native plants, especially fire-prone grasses and herbs, represent a divergence from natural fuel loads. The reference condition for fire frequency and size is the natural fire regime occurring across HALE prior to human arrival. Although this is not known with certainty, it is inferred to consist of infrequent fire, with fire return intervals likely exceeding 100 years for most regions of the park. In practical terms, this equates to little or no fire over contemporary time scales.

Condition and Trend

Fuel characteristics

Fuel models for Hawaiian vegetation types have not been developed, with the exception of a localized effort specific to three community types on Mauna Kea (Thaxton and Jacobi 2009). Prior assessments suggest that it is unclear if fire behavior fuel models (FBFM) developed on the mainland are reasonable predictors of fire behavior for fuel types found in Hawaiʻi (Smith and Tunison 1992, Trauernicht et al. 2015). However, the 2005 HALE FMP presented representative fuel models from the Anderson FBFM system (Anderson 1982) to provisionally illustrate a range of potential fire behaviors in the park (NPS 2005). Distributions of these fuel model types have now been mapped for Hawaiʻi, and are shown for areas in and around HALE in Figure 4.3.1-4. Descriptions and characteristics of these models are listed in Table 4.3.1-1.

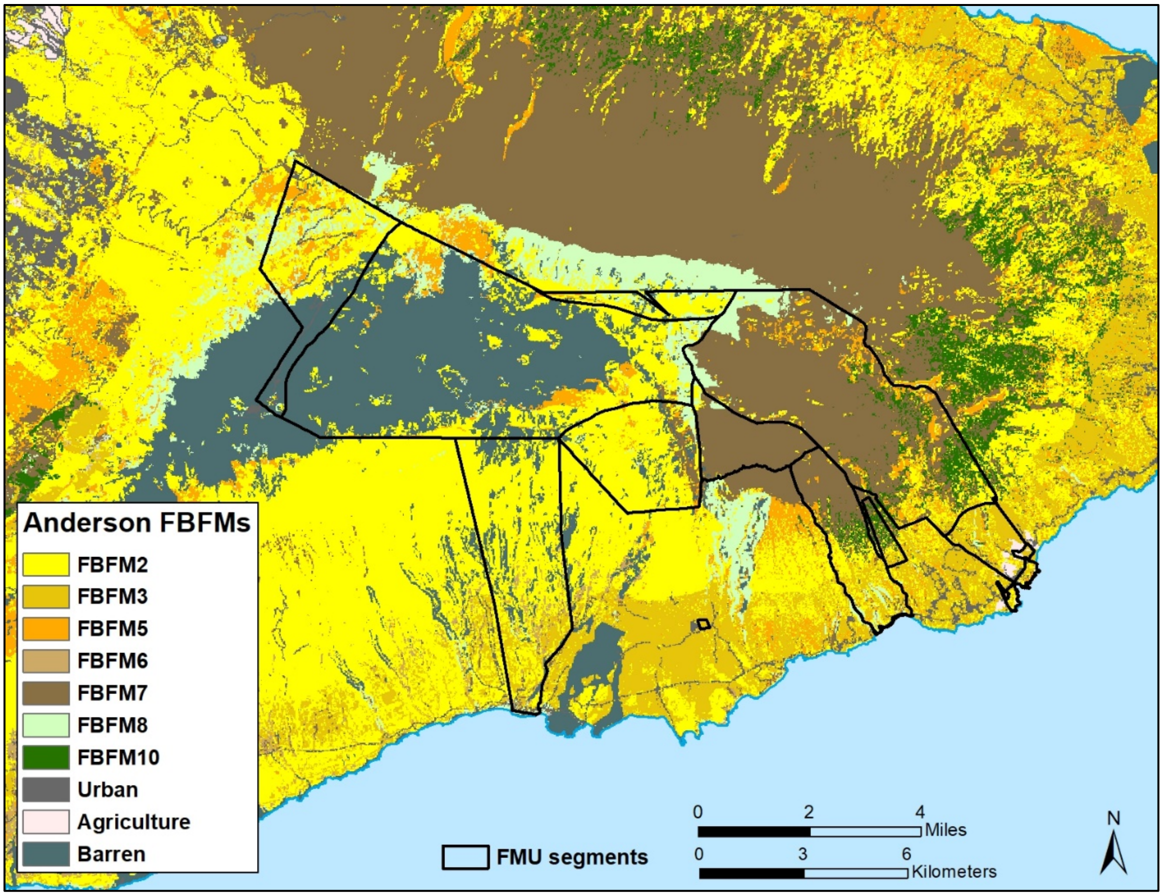

Figure 4.3.1-4. Distribution of Anderson fire behavior fuel model (FBFM) types that occur in and around HALE. Also shown are the nine designated FMU segments, plus the Nuʻu Parcel. Source: LANDFIRE (2017).

Table 4.3.1-1. Description and characteristics of the Anderson FBFM types occurring in and around HALE. Source: Anderson (1982), FBFM descriptions taken from LANDFIRE (2017) data dictionary.

FBFM	Description	Rate of Spread* (chains/hr)	Flame Length* (ft)
2	Burns fine, herbaceous fuels; stand is curing or dead; may produce fire brands on oak or pine stands	35	6
3	Most intense fire of grass group; spreads quickly with wind; one third of stand dead or cured; stands average 3 ft tall	104	12
5	Low intensity fires; young, green shrubs with little dead material; fuels consist of litter from understory	18	4
6	Broad range of shrubs; fire requires moderate winds to maintain flame at shrub height, or will drop to the ground with low winds	32	6
7	Foliage highly flammable, allowing fire to reach shrub strata levels; shrubs generally 2 to 6 ft high	20	5

* Rates of spread and flame length based on average expected conditions of 5 mph wind speeds, fuel moisture content of 8%, and live fuel moisture content of 100%

Table 4.3.1-1 (continued). Description and characteristics of the Anderson FBFM types occurring in and around HALE. Source: Anderson (1982), FBFM descriptions taken from LANDFIRE (2017) data dictionary.

FBFM	Description	Rate of Spread* (chains/hr)	Flame Length* (ft)
8	Slow, ground burning fires; closed canopy stands with short needle conifers or hardwoods; litter consist mainly of needles and leaves, with little undergrowth; occasional flares with concentrated fuels	1.6	1.0
10	Surface and ground fire more intense; dead-down fuels more abundant; frequent crowning and spotting causing fire control to be more difficult	7.9	4.8

* Rates of spread and flame length based on average expected conditions of 5 mph wind speeds, fuel moisture content of 8%, and live fuel moisture content of 100%

Because of the uncertainty of the applicability of these models to Hawaiʻi, their prediction of fire behavior in HALE should be interpreted with caution. However, generally speaking, fuel models 2 and 3, which characterize relatively high fire risk due to high loads of grasses and other fine fuels, may be relevant to relatively large areas of the park, especially lowland areas, most of Nuʻu, Kaupō Gap, and portions of the Crater and West Slope (Figure 4.3.1-4). Fuel model 2 areas may support open shrub canopy, while model 3 areas are dominated by grasses, often with high proportions of dead standing biomass (Anderson 1982, NPS 2005). In HALE, the latter are mapped primarily in lowland areas of Nuʻu, Kaʻāpahu, and Kīpahulu. Upland forested areas in HALE are classified mainly under fuel model 7, while much of the Crater is classified as barren and unlikely to represent high risk of wildfire. Fuel models 5, 6, 8 and 10 cover relatively small regions of the park (Figure 4.3.1-4).

Owing to the lack of more definitive fuel models for Hawaiʻi, fuel characteristics in HALE have typically been described more qualitatively. The most recent FMP (NPS 2005) discusses four main fuel types in the park, and excerpts of this summary are reproduced below:

- Grasses - The most significant fuel type at HALE is alien grasses, which have adapted to the regular fire regimes in their native habitat. Of particular importance are fire-tolerant/ stimulated grasses which maintain high dead-to-live biomass ratios throughout the year and burn at high relative humidities and high fuel moistures. The most combustible grasses at HALE are molasses grass (*Melinis minutiflora*) and broomsedge (*Andropogon virginicus*). Other alien grasses are not particularly combustible but will carry fire in dry conditions. Such grasses at HALE include: elephant grass (*Cenchrus purpureus*) which can reach heights of 12-14 ft, California grass (*Urochloa mutica*), natal grass *(Melinis repens*), velvetgrass (*Holcus lanatus*), sweet vernalgrass (*Anthoxanthum odoratum*), and Kikuyu grass (*Cenchrus clandestinus*). *Cenchrus setaceus* (syn. *Pennisetum setaceum*), a highly invasive flammable grass, has been detected and controlled on the island of Maui, but has not been recorded within the park (Adkins et al. 2011, MISC 2017). Native bunchgrasses such as *Deschampsia nubigena* are also not particularly combustible, but will carry fire in dry conditions.

- Brush – This fuel type is comprised of native shrubs such as pukiawe (*Leptecophyla tameiameiae*), mamane (*Sophora chrysophylla*), ohelo (*Vaccinium reticulatum*), and ‘a‘ali‘i (*Dodonaea viscosa*) and alien shrubs such as Christmasberry (*Schinus terebinthifolius*), gorse (*Ulex europaeus*) and guava (*Psidium guava*). This fuel type forms closed or nearly closed stands with grasses forming an understory in all except the densest stands. This fuel type may produce intense fires with high rates of spread.
- Timber Litter – The fuel type occurs as leaf litter ground cover in closed forest areas of the park which are dominated by the native ‘ohi‘a lehua (*Metrosideros polymorpha*) and koa (*Acacia koa*). Areas of eucalyptus (*Eucalyptus* spp.) and pine (*Pinus* spp.) forests also occur within the park, as well as adjacent to the park. Fire intensity and rates of spread are expected to be low in this fuel type.
- Ferns – This fuel type occurs as an understory in upper elevation forested areas and consists of uluhe (*Dicranopteris linearis*) and a variety of other ferns and tree ferns. This fuel type is expected to burn only in extended drought conditions and spread in very windy conditions.

The main fuel types characterizing each of the FMU Segments are summarized in Table 4.3.1-2. These correspond reasonably well with fire risk inferred from the Anderson fire behavior fuel model map (Figure 4.3.1-4). In particular, areas of HALE most invaded by high-risk alien grasses, especially the highly combustible molasses grass (*M. minutiflora*), are lowland portions of Ka‘āpahu and Kīpahulu Valley, and Kaupō Gap. Elevated fire risk in Kaupō due to alien grasses has been recognized for some time (Loope et al. 1992). Shrubland fuels in the Crater and West Slope also pose a fire risk under dry and windy conditions, as do the native grasses on Kalapawili Ridge.

Table 4.3.1-2. Main fuel types in each FMU Segment in HALE, as summarized in the 2005 Fire Management Plan. Source: NPS (2005).

FMU Segment	Fuel Types
Coastal Kīpahulu	Predominately alien grasses and brush. Elephant grass (*Cenchrus purpureus*) comprises about 40% of the vegetation cover. Bermuda grass (*Cynodon dactylon*), California grass (*Urchola mutica*) and pangola grass (*Digitaria eriantha*) covers about 25% of the vegetative cover.
Lower Kīpahulu Valley	Predominately alien grasses and brush. In particular Elephant grass (*Cenchrus purpureus*), pangola grass (*Digitaria eriantha*), and West Indian dropseed (*Sporobolus indicus*).
Upper Kīpahulu Valley	Timber litter and ferns.
Ka‘āpahu	Timber litter, ferns such as the native uluhe (*Dicranopteris linearis*), brush and alien grasses. In particular Molasses grass (*Melinis minutiflora*), natal grass (*Melinis repens*) and broomsedge (*Andropogon virginicus*).
Manawainui	Timber litter and ferns.
Kalapawili Ridge	Native bunchgrass, *Deschampsia nubigena* and the alien velvetgrass (*Holcus lanatus*) and sweet vernalgrass (*Anthoxanthum odoratum*) predominate.
Crater West Slope	Native brush and alien grasses. In particular velvetgrass (*Holcus lanatus*) and sweet vernalgrass (*Anthoxanthum odoratum*). Kikuyu grass (*Cenchrus clandestinus*) growing around park developed areas.

Table 4.3.1-2 (continued). Main fuel types in each FMU Segment in HALE, as summarized in the 2005 Fire Management Plan. Source: NPS (2005).

FMU Segment	Fuel Types
Crater Interior Floor	Native brush and alien grasses. In particular velvetgrass (*Holcus lanatus*) and sweet vernalgrass (*Anthoxanthum odoratum*). Kikuyu grass (*Cenchrus clandestinus*) growing around park developed areas.
Kaupō Gap	Timber litter, native brush and alien grasses. In particular Molasses grass (*Melinis minutiflora*) Kikuyu grass (*Cenchrus clandestinus*), velvetgrass (*Holcus lanatus*), natal grass (*Melinis repens*) and broomsedge (*Andropogon virginicus*).

The highest risk of wildfire stemming from unnatural fuel loads is likely most strongly associated with vegetation communities dominated by introduced grasses and herbs, which provide fine fuels that ignite relatively easily and carry rapid fire. This is especially true for communities dominated by the highly combustible molasses grass, *M. minutiflora*. These communities are shown in Figure 4.3.1-5. It should be noted, however, that *M. minutiflora* distributions in Kaupō are substantially more extensive than indicated in the vegetation community polygons shown in Figure 4.3.1-5 (P. Welton pers. comm. 2018). Some areas, such as lower Kīpahulu Valley and the Palikū pasture, would likely catch fire only under very dry conditions, while others, such as those in Kaupō and Nuʻu, represent fairly persistent fire risks. One area of divergence from the general assessments inferred from the Anderson FBFM map and FMU Segment fuel descriptions concerns lowland portions of Kaʻāpahu. The vegetation communities mapped by Green et al. (2015) classify these as dominated by alien trees and other woody vegetation. While these are certainly capable of burning under dry conditions, they were not found to be dominated by alien fine fuels like grasses and herbs, which represent a higher fire risk. Introduced grasses and herbs occur to at least some degree in most vegetation communities in the park, however those identified in Figure 4.3.1-5 likely represent the highest loads of invasive fine fuels in HALE.

Fire risk in the newer Nuʻu Parcel may be the highest of any region of the park. Much of the region is comprised mainly of alien vegetation, including communities dominated by fire-prone non-native grasses and herbs (Figure 4.3.1-5). Fine fuel loads are likely to increase in the short-term after the completion of the fence encircling the upper portion of Nuʻu (Figure 4.2.9-3) and subsequent removal of browsing ungulates (NPS 2016). This risk is acknowledged in the Environmental Assessment for the Nuʻu fencing project, and specific mitigation efforts are being undertaken (NPS 2016). Over the longer-term, it is anticipated that the removal of feral ungulates, combined with vegetation restoration efforts, will allow the recovery of native woody plant communities and will eventually reduce the risk of fire in this region.

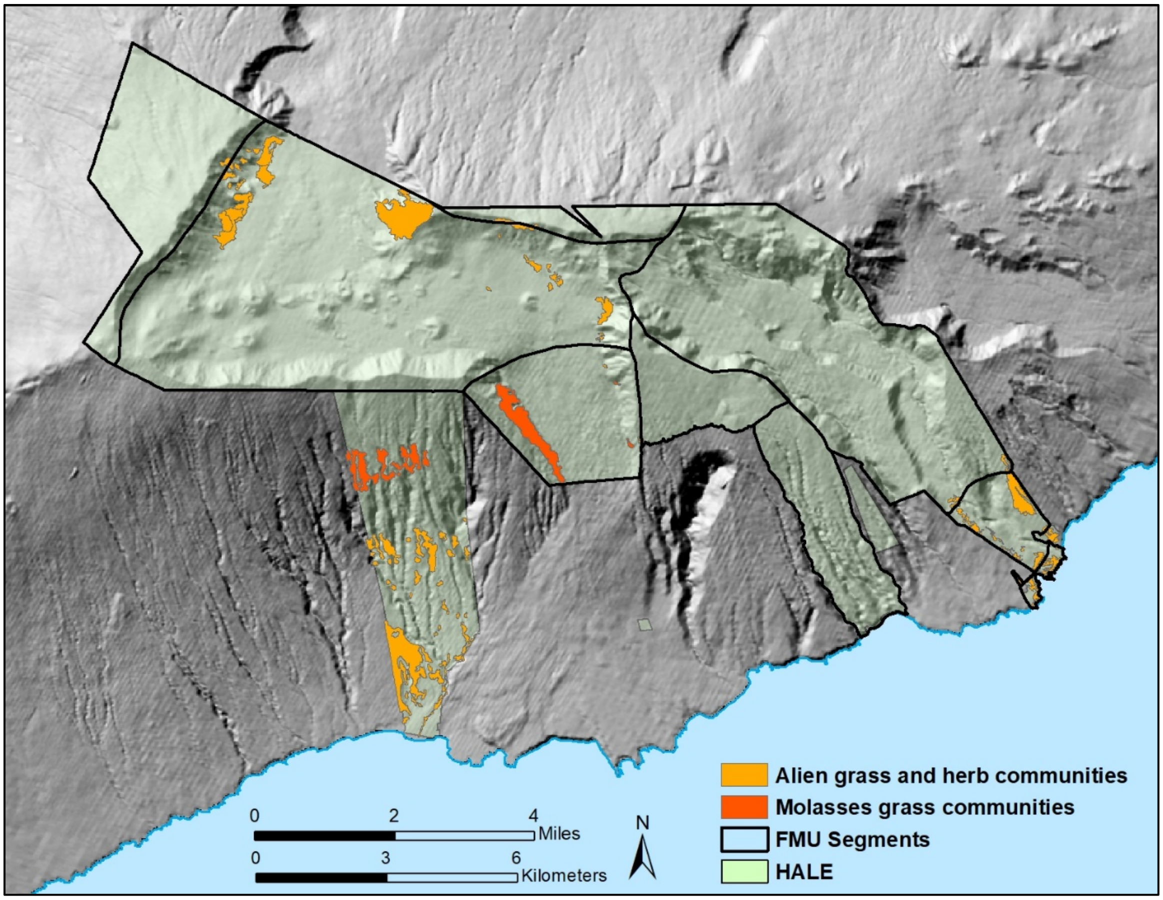

Figure 4.3.1-5. Vegetation communities in HALE dominated by alien grasses and herbs (light orange) and communities dominated by the highly combustible molasses grass (dark orange). These communities represent the highest loads of unnatural fine fuels in the park. Source: Green et al. (2015).

The condition of fuel characteristics in HALE is judged to warrant moderate concern. Although the 2005 HALE FMP judged that fire potential in the park is "generally low", some regions support relatively extensive areas of unnatural fine fuels comprised of non-native grasses and herbs, including species that are highly combustible and capable of regenerating quickly after fire. The Nuʻu and Kaupō regions, in particular, represent the highest fire risk areas of the park. Confidence in this assessment is medium, owing to an absence of fuel models specific to vegetation communities occurring in HALE. The trend of this condition is unknown. While past management efforts have targeted alien grass species, such as molasses grass, the degree to which the distribution of this and other non-native grasses has increased or decreased is unclear. Furthermore, fuel loads in the Nuʻu region may increase over the near-term future. Finally, if changing climate leads to drier future conditions, this may increase dead standing biomass and wildfire fuel loads.

Fire frequency and size

The most recent HALE FMP listed a total of 33 known fires occurring in and around the park between 1935 and 2008 (NPS 2005). The Hawaiʻi State Wildfire History Data Set listed an additional four fires in or around HALE through the year 2010 (HWMO 2013). Of these 37 fires, only 17 appear to have occurred within park boundaries, which together burned a total of 10.45 ac (4.23 ha) (Table 4.3.1-3). However, the size of two fires in the 1930s are unknown, and the description of the 1938 fire in Kaupō Gap suggests it was quite large. Aside from the latter fire, all other known fires were smaller than 1 ac (0.4 ha) in size, with the exception of a pair of fires in the eastern Crater in 1970, which together burned 7 ac (2.8 ha). Most of the fires occurred either in Kaupō Gap and the eastern Crater, or near park facilities (near visitor centers and campgrounds in both the Summit and Kīpahulu Districts) (Table 4.3.1-3). Nearly all of the fires are known to be started by human-caused ignitions.

Table 4.3.1-3. Known fires recorded in HALE between 1935 and 2010. Source: NPS (2005) and HWMO (2013).

Year	Location	Area burned (acres)	Cause/remarks
1935	Kaupō Gap	?	713 Tradition Fort Collins, CO 80526
1938	Kaupō Gap	?	Fire started below the park boundary at 3,800 foot elevation level and swept up the flat areas almost to the base of ʻOʻili puʻu (6,500 foot level).
1941	Halemauʻu	0.7	Class B fire caused by escaped campfire at Halemau'u trailhead.
1957	Kapalaoa	?	Class A grass fire near Kapalaoa Cabin.
1970	Waikane	4.0	South of Waikane Spring and east of Kaupō Gap. Fire burned at slow rate. Problem with fire recurrence. Separated 1 to 1.5 miles from ʻOʻili puʻu fire.
1970	ʻOʻili Puʻu	3.0	Apparently human caused. Fire died out naturally. Separated 1 to 1.5 miles from Waikane fire.
1986	Kaupō Gap	0.1	Escaped campfire in West Kaupō Gap at 4,700 foot level.
1992	Park HQ	0.6	Fire probably started from HQ-12 stove flue sparks.
1993	Hosmer Grove	0.1	Fire ignited by windblown sparks from Hosmer Campground grill.
1993	Summit	0.1	Escaped campfire near Haleakalā Visitor Center.
1997	Park HQ	0.1	Fire ignited by burning vehicle in parking lot.
1997	Kīpahulu	0.1	Slow burning fire in pile of moist logs and branches near trail leading to Makahiku Falls.
2000	Kīpahulu	0.3	Escaped campfire at Kīpahulu campground.
2002	Kīpahulu	0.1	Fire ignited by discarded cigarette behind the restrooms.
2007	Kīpahulu	<0.1	Appears to be roadside at ʻOheʻo.
2009	Kīpahulu	0.25	Appears to be roadside at ʻOheʻo.
2010	Kaʻāpahu	1.0	Appears to be roadside.

When grouped by decade, the number of fires appears to have increased in the last two decades on record, the 1990s and 2000s (Figure 4.3.1-6). However, it is unknown to what extent this increase may be attributed to more diligent record keeping, rather than increases in the frequency of ignitions.

In spite of the higher number of known recent fires, the total area burned has not increased in the past two decades (Figure 4.3.1-7).

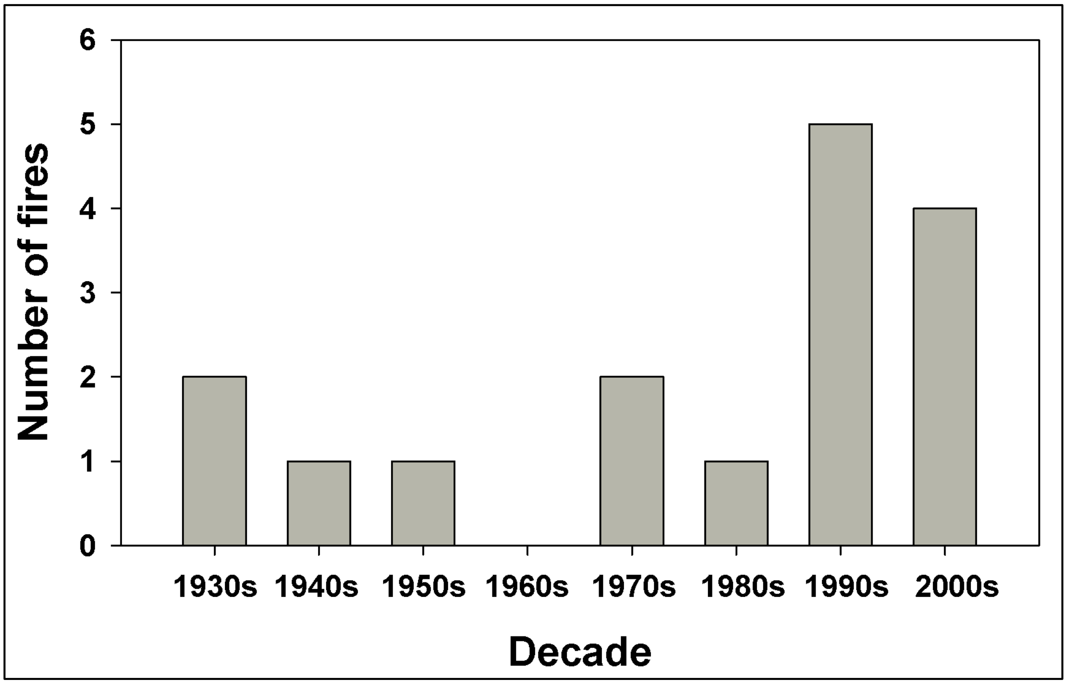

Figure 4.3.1-6. Number of fires per decade in HALE, from 1930 to 2009. One fire recorded in 2010 was excluded from the figure. Source: NPS (2005) and HMWO (2013).

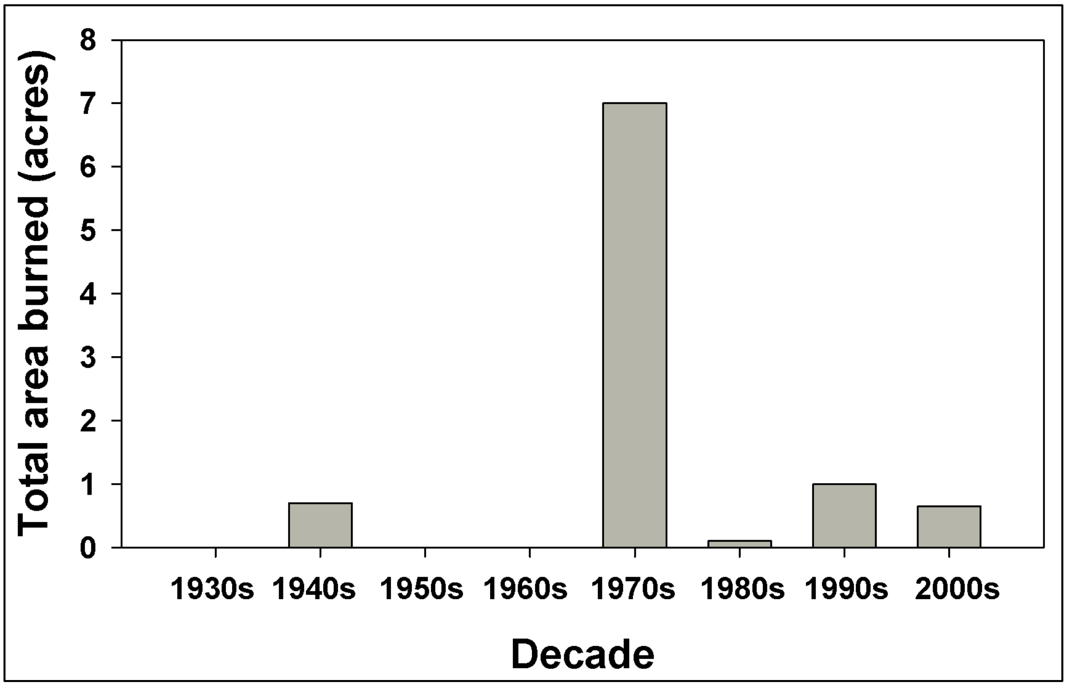

Figure 4.3.1-7. Total area burned per decade in HALE, from 1930 to 2009. One fire recorded in 2010 (1 acre in size) was excluded from the figure, and several fires of unknown size occurred in the 1930s. Source: NPS (2005) and HMWO (2013).

The 2005 HALE FMP assessed the degree to which fire regimes in each FMU Segment appear to have diverged from the natural state, according to the qualitative classification system laid out in Hardy et al. (2002). Based on the Hardy et al. "Current Condition Classes", it was judged that four segments (Coastal Kīpahulu, Lower Kīpahulu Valley, Ka'āpahu, and Kaupō Gap) fall within Class 3, meaning that "fire regimes have been significantly altered from its historical range; the risk of losing key ecosystem components is high; vegetation attributes have been significantly altered from their historical range; and fire frequencies have departed from historic frequencies by multiple return intervals" (NPS 2005). Three segments (Crater Interior Floor, Crater West Slope, and Kalapaili Ridge) were judged to fall within Class 2, meaning that "fire regimes have been moderately altered from their historical range; the risk of losing key ecosystem components is moderate; vegetation attributes have been moderately altered from their historical range; and fire frequencies have departed from historical frequencies by one or more return intervals." Only two segments (Upper Kīpahulu Valley and Manawainui Planeze) were judged to fall within Class 1, indicating that modern fire regimes "are within their historical regimes; the risk of losing key ecosystem components is low; and vegetation attributes are intact and functioning within their historical range" (NPS 2005).

On the whole, evidence suggests that fire frequency and size have increased substantially from conditions under natural fire regimes in most parts of HALE, with departures most severe in lowland and drier regions of the park. However, in recent decades, fires have generally been quickly extinguished, so that area burned has been very small after the 1970s despite a regular occurrence of ignition events (averaging approximately one every two years over the past two decades). The condition of fire frequency and size is therefore judged to warrant moderate concern. The goal of no wildfires occurring in the park has not been attained, however such a goal is not likely realistic, and the small size and short duration of recent fires suggests that their impacts have been minimal. An improving trend to the condition is assigned, because fire management strategies in recent decades appear to have effectively decreased the size of fires relative to earlier time periods. However, future drier climate conditions, if they occur, could reverse these gains. For example, a large wildfire burned several thousand acres to the southwest of the park as recently as 2007 (HWMO 2013), and was driven by exceptionally dry climatic conditions. Confidence in this assessment is high, because it is unlikely that any large and important fires in HALE have gone unrecorded in recent decades.

Overall assessment

Integration of the two indicators of fuel and fire dynamics yields an overall condition warranting moderate concern. The overall trend is designated as unknown, because of the unknown trend in fuel characteristics in the park. Although the size of area burned has declined in recent decades due to effective fire management, the contemporary departure from natural fire regimes in most regions of the park remains a cause for concern. Furthermore, as long as significant areas of unnatural fuel loads exist in HALE, comprised mainly of fire-promoting non-native grasses and herbs, the risk of ecosystem-damaging wildfire will persist. In the near future, this risk is likely to be highest in Nu'u, as well as in portions of Kaupō Gap where vegetation communities dominated by the highly combustible molasses grass occur. Ongoing vegetation management will be required in other park regions as well, to avoid conversion of native plant communities to non-native fire-adapted vegetation. This includes limiting the incursion of pines into West Slope and Crater areas, which are largely invading from outside the park and adding heavy fuels to native shrubland ecosystems. (return to Condition Summary).

Level of confidence

Overall level of confidence in this assessment is medium. The history of fires over the course of the park's history is known with fairly high confidence, especially over recent decades. However, certain aspects of fuel characteristics are more uncertain. In particular, changes in alien grass distributions are less well quantified, and fire behavior in vegetation types occurring in HALE has not been characterized.

Information gaps and research recommendations

Fuel models specific to Hawaiian vegetation types are lacking, with a few localized exceptions. Better quantification of the distribution and fuel loads of non-native fire-prone vegetation would better characterize fire risk in different regions of the park.

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4.3.2. Soundscape

Condition Summary

Overall resource condition warrants moderate concern with an unknown trend; confidence in condition determination is medium. Rationale is discussed in the "*Overall assessment*" section and following.

●

Relevance and Context

Sounds of nature are an integral part of the experience of visiting a national park, especially within designated wilderness areas. Unwanted human-generated sounds, or noise, can significantly detract from this visitor experience, and may impact wildlife as well (Lawson et al. 2007, Prasad and Tomonari-Tuggle 2008, Bell et al. 2010). NPS therefore recognizes the soundscape as a vital natural resource, and is mandated under a variety of authorities to maintain the integrity of natural soundscapes within National Park units (Lynch 2012, NPS NSNSD 2017). For example, section 4.9 of the NPS Management Policy guidelines is devoted to soundscape management (NPS 2006), and states that "The Service will restore to the natural condition wherever possible those park soundscapes that have become degraded by unnatural sounds (noise), and will protect natural soundscapes from unacceptable impacts."

Haleakalā Crater is frequently recognized as one of the quietest natural areas within the National Park system (Bell et al. 2010, McCusker and Cahill 2010, Lynch 2012). Ambient sound levels within portions of the Crater have been reported to be as low as 10 dBA, which is similar to the sound of a person breathing at a distance of 3 m (9.8 ft), and is exceptionally rare in nature (Bell et al. 2010, Lynch 2012). Preserving the natural soundscape is thus a core priority for HALE, as stated in the park's Resource Management Plan (NPS 1999) as well as in the park's Foundation Document (NPS 2015).

The principal source of noise at HALE is aircraft overflights, mainly in the form of commercial helicopter tours during the day and high altitude jets at night (Lynch 2012). Prior to an altitude restriction imposed over the park as part of the National Parks Overflights Act of 1987, helicopter tours regularly flew within the Crater and as low as 91 m (300 ft) above the Crater floor (Lynch 2012). This Act, and a 1998 Letter of Agreement between HALE and the Hawai‘i Air Tour Association on Maui, restricted commercial helicopter flights over the park to a path crossing over Waimoku Falls in lower Kīpahulu Valley (Lee et al. 2016). However, tours still flew along the park boundaries above the Crater rim, and continue to do so today, and these are audible within the Crater. The National Parks Air Tour Management Act (NPATMA) was subsequently passed in 2000, under which HALE is required to develop an air tour management plan (ATMP). The ATMP, which is not yet completed, will specify which operators will have authority to conduct tours within one-half mile (0.8 km) of the park boundary. In the meantime, existing operators have been granted interim operating authority to continue tours within this zone (Lynch 2012).

As part of the environmental assessment to produce the HALE ATMP, baseline ambient sound level data were collected at 10 locations in the park in 2003 (Lee et al. 2016). Supplemental sound level data were also collected at three locations in 2008 to estimate aircraft overflight frequencies and provide other related information (Lynch 2012). Although additional sound data have been collected in the park in the 1990s and up to 2013 (Wood 2015), the 2003 and 2008 efforts provide the most consistent and comprehensive sound level data sets for the park. These efforts characterized existing ambient sound levels and estimated sound levels under natural conditions (in the absence of anthropogenic noise), and further calculated the percentage of time that noise was audible at different locations. Focus is therefore placed on these data sources to assess the degree to which natural quiet conditions exist at HALE. This indicator, natural quiet, is the most relevant metric of the condition of the soundscape in the park.

Indicators

- Natural quiet

Data and Methods

A review of available data and information was used to assess conditions, and if possible, trends, in the indicator. No additional data analysis was performed.

Sources of information

Information on ambient sound levels in HALE was obtained mainly from Lynch (2012) and Lee et al. (2016), and secondarily from Wood (2015).

Reference condition

The reference condition for natural quiet sound levels are those absent anthropogenic noise. This noise is primarily generated by helicopter activity in or near the park. Estimated natural ambient sound levels approximate the noise-free reference condition.

Condition and Trend

Natural quiet

Lee et al. (2016) measured existing ambient sound levels at 10 locations in HALE between February 27 and May 27, 2003, for a total of 165 days of acoustic recording. However, recordings only covered one or two days at two of the locations owing to technical problems, and data from these locations were excluded from most of the resultant analyses. Lynch (2012) measured ambient sound levels at 3 locations in April, 2008, for 30 days at each location. The 11 locations that recorded usable data in these two studies are shown in Figure 4.3.2-1.

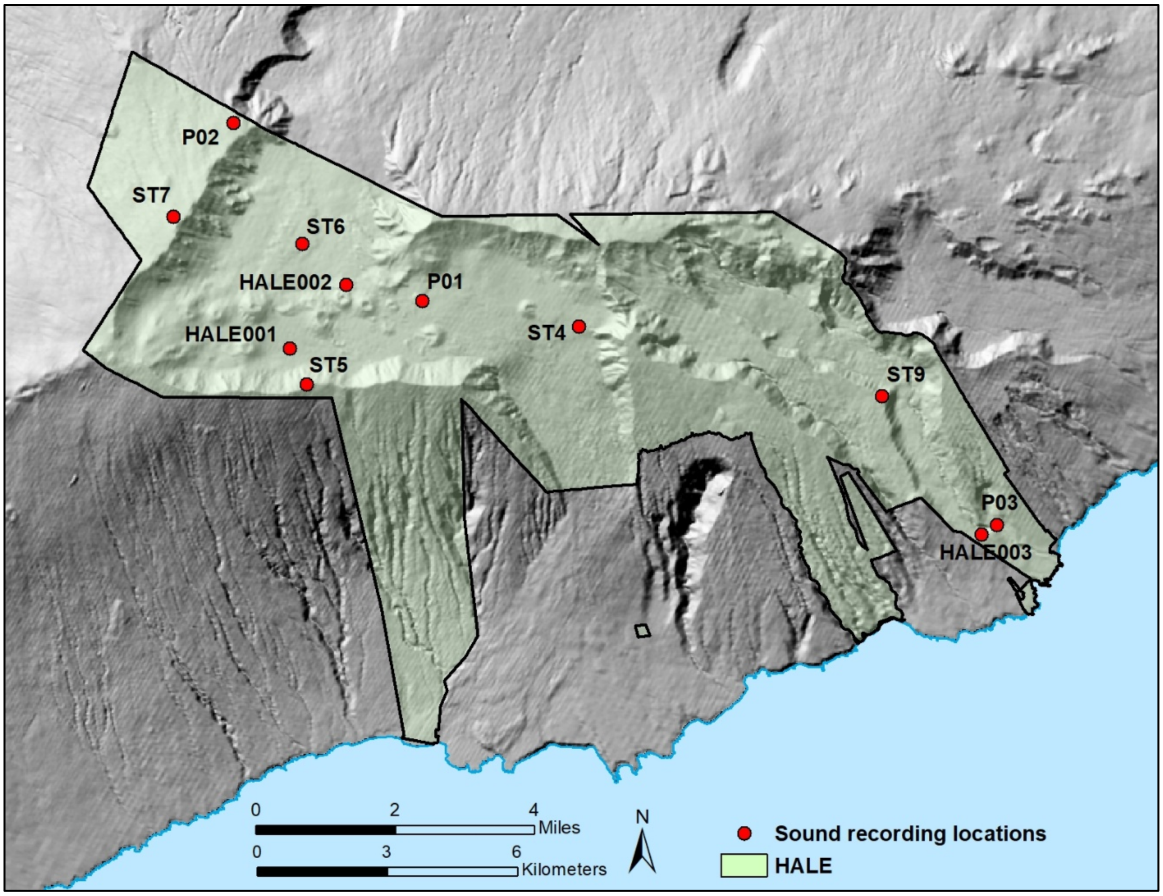

Figure 4.3.2-1. Locations where ambient sound levels were recorded in 2003 and 2008. Source: Lynch (2012), Lee et al. (2016).

Both efforts calculated a number of sound metrics, typically expressed as sound levels exceeded a given percent of the time. For example, the existing ambient sound level that is exceeded 50% of the time (or the median sound level) is indicated as L_{50}, while L_{10} is the existing sound level exceeded 10% of the time, and L_{90} is the existing sound level exceeded 90% of the time. "Existing" sound levels include all recorded sounds, both natural and anthropogenic. The median sound level (L_{50}) is a reasonable metric of typical conditions, while L_{90} indicates sound levels under very quiet conditions at a particular location. The latter is sometimes used as a proxy for "natural" ambient sound conditions (i.e., excluding anthropogenic noise) at a location (Wood 2015). However, both Lynch (2012) and Lee et al. (2016) provide alternative estimates of natural ambient sound levels under typical conditions, using different methods. Both estimates of natural ambient sound levels are expressed here as L_{nat} (L_{50} values for natural sound levels are used to represent L_{nat} for the 2003 data (Lee et al. 2016), while L_{nat} is used directly as reported for the 2008 data (Lynch 2012)).

Ambient sound levels during daytime hours for each of the 11 locations are shown in Table 4.3.2-1. All sound levels are expressed in dBA, or A-weighted decibels. A-weighting favors sound frequencies most easily detected by humans (Lynch 2012). The decibel scale is logarithmic, so increasing values indicate exponential increases in sound levels. For example, an increase of 3 dBA corresponds to an approximate doubling of sound energy (Lynch 2012). For reference, sound levels for a range of common sound sources are provided in Table 4.3.2-2.

Table 4.3.2-1. Ambient daytime sound levels recorded at 11 locations in HALE in 2003 and 2008. Source: Lynch (2012), Lee et al. (2016).

Station	Location	Year	L_{50} (dBA)[1]	L_{90} (dBA)[2]	L_{nat} (dBA)[3]
P02	West Slope, supply trail	2003	27.2	21.5	27.7
ST7	West Slope, Kalahaku Overlook	2003	28.4	21.9	23.6
P01	Crater, Namana o ke Akua	2003	24.5	18.6	22.5
ST5	Crater, notch of south rim	2003	24.6	19.3	21.8
ST6	Crater, Silversword Loop	2003	23.5	19.2	21.4
ST4	Crater, top of Kaupō Trail	2003	22.5	18.8	22.6
P03	Kīpahulu, lower valley	2003	43.5	38.2	45.3
ST9	Kīpahulu, upper valley	2003	34.9	30.0	30.7
HALE001	Crater, Sliding Sands Trail near hitching post	2008	18.2	15.9	17.3
HALE002	Crater, rain gauge near Pu'u Halali'i	2008	21.9	15.8	19.5
HALE003	Kīpahulu, lower valley	2008	38.9	35.1	38.0

[1] L_{50} of existing ambient sound levels (i.e. median existing sound levels)

[2] L_{90} of existing ambient sound levels (i.e. existing sound levels under quiet conditions)

[3] Estimated natural ambient sound levels (i.e. excluding anthropogenic noise)

As can be seen in Table 4.3.2-1, median existing ambient sound levels are quite low in within the Crater, typically 20-25 dBA. During quiet conditions (L_{90}), existing sound levels within the Crater can be very low, typically between 15-20 dBA. Measurements within the Crater tended to be lower in 2008 than in 2003; this may have been caused by windier conditions or other weather differences between the two years (Lynch 2012). Existing sound levels on the West Slope are somewhat higher, as they are affected by noise from traffic on the park road. Periodic construction at the Science City observatories is an additional source of noise at the summit, but data characterizing the levels of noise generated by these activities do not exist. The upper Kīpahulu Valley location had higher sound levels, owing to sounds from birds and insects. The two lower Kīpahulu Valley locations had the highest median existing sound levels, and this was attributed to sounds emanating from the nearby Waimoku Falls (Lynch 2012, Lee et al. 2016).

Comparison of L_{50} and L_{nat} values in Table 4.3.2-1 provides an estimate of the degree to which daytime anthropogenic noise elevates typical sound conditions at HALE. The 2008 L_{nat} values were 0.9 to 2.4 dBA lower than existing L_{50} values. For the 2003 measurements, L_{nat} values were in some cases up to 4.8 dBA lower than existing L_{50} values, but were actually higher than existing L_{50} values for a few locations. The latter makes little physical sense, and may be an artifact of the method used to estimate natural ambient sound levels for the 2003 data.

Table 4.3.2-2. Common sound sources and their approximate sound levels. Source: reproduced from Lynch (2012).

Source	Sound level (dBA)
Human breathing at 3 m	10
Whispering	20
Residential area at night	40
Busy restaurant	60
Curbside of busy street	80
Jackhammer at 2 m	100
Train horn at 1 m	120

Most of the daytime anthropogenic noise was caused by helicopter air tours (Lynch 2012, Lee et al. 2006). Both the 2003 and 2008 studies estimated the percent of time such tours were audible at each location, although they again used different methods to accomplish this. For the 2003 study, percent of time audible (PA) statistics were calculated from on-site observers visiting each station for durations of approximately 40 to 120 minutes (Lee et al. 2016). For the 2008 study, PA statistics were calculated by visually detecting signatures of helicopter overflights from the full audio digital recordings at each location (Lynch 2012), and therefore utilized a much longer time period for analysis and may be more accurate. The PA calculations for helicopters at each location are shown in Table 4.3.2-3. These data indicate that helicopter air tours are audible during significant portions of daytime hours in many regions of the park.

Table 4.3.2-3. Mean percent of time audible (PA) statistics for helicopter overflights at the 11 recording locations in HALE in 2003 and 2008. Source: Lynch (2012), Lee et al. (2016).

Station	Location	Year	PA for helicopters (%)
P02	West Slope, supply trail	2003	10.2
ST7	West Slope, Kalahaku Overlook	2003	2.5
P01	Crater, Namana o ke Akua	2003	30.1
ST5	Crater, notch of south rim	2003	31.8
ST6	Crater, Silversword Loop	2003	65.5
ST4	Crater, top of Kaupō Trail	2003	4.5
P03	Kīpahulu, lower valley	2003	16.0
ST9	Kīpahulu, upper valley	2003	27.8
HALE001	Crater, Sliding Sands Trail near hitching post	2008	14.6
HALE002	Crater, rain gauge near Puʻu Halaliʻi	2008	19.7
HALE003	Kīpahulu, lower valley	2008	16.6

Lynch (2012) further calculated PA statistics for three different types of aircraft for both daytime and nighttime periods at the three listening locations used in 2008. These are shown in Table 4.3.2-4, and indicate that helicopters were the most common source of noise during the day, while high altitude jets were the most common source of noise at night.

Based on the recordings during the 2008 sampling period, Lynch (2012) calculated that an average of 17.1 helicopter tours could be heard per day at the HALE001 location, 20.8 tours per day were audible at the HALE002 location, and 25.6 tours per day were audible at the HALE003 location. These were most frequent in mid-morning hours, as can be seen in Figures 4.3.2-2 to 4.3.2-4. The percent of time that helicopters were audible often approached or exceeded 40% during mid- to late-morning periods.

Table 4.3.2-4. Mean percent of time audible (PA) statistics for helicopters, fixed-wing propeller planes, and jets during daytime and nighttime hours at the three recording locations in HALE in 2008. Source: Lynch (2012).

		Day			**Night**		
Station	**Location**	**Heli.**	**Prop.**	**Jet**	**Heli.**	**Prop.**	**Jet**
HALE001	Crater, Sliding Sands Trail near hitching post	14.6	0.7	6.9	0.0	0.6	7.8
HALE002	Crater, rain gauge near Puʻu Halaliʻi	19.7	1.1	8.1	0.2	1.4	8.2
HALE003	Kīpahulu, lower valley	16.6	1.2	1.7	0.0	0.0	3.9

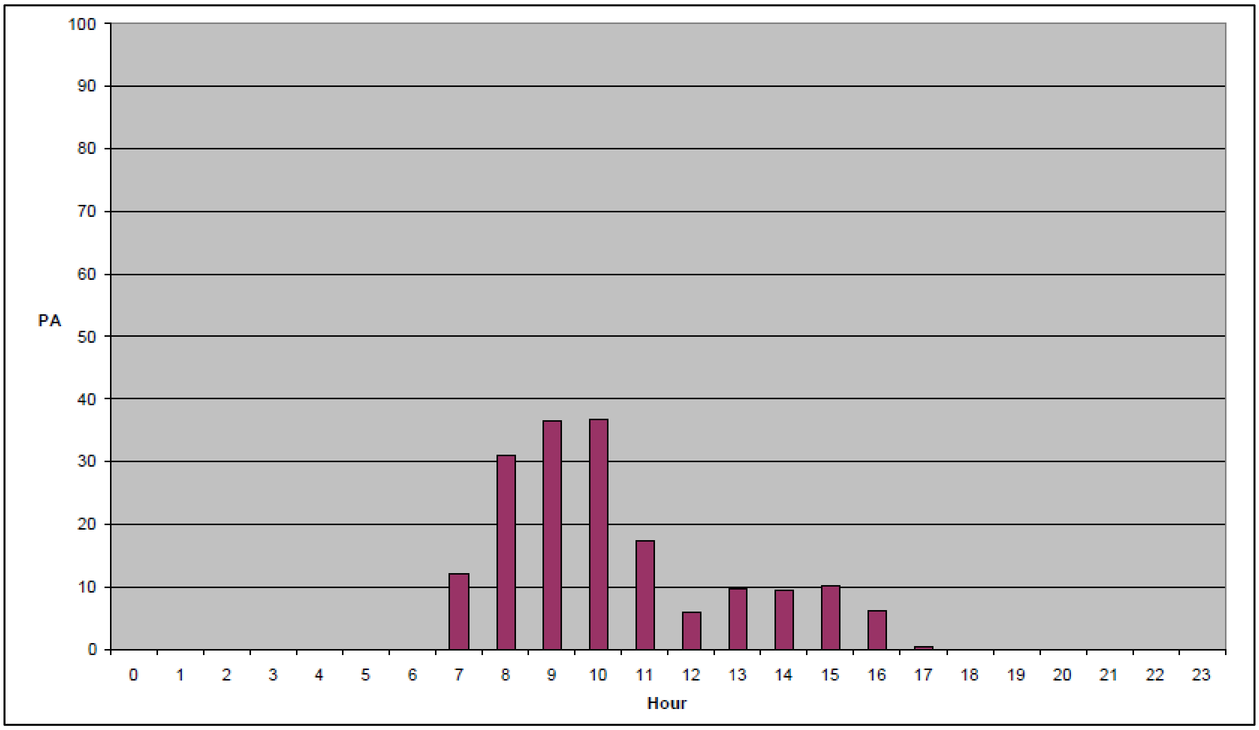

Figure 4.3.2-2. Mean percent of time that helicopters were audible (PA) per hour at HALE001 station (Sliding Sands trail, near hitching post) in 2008. Source: reproduced from Lynch (2012).

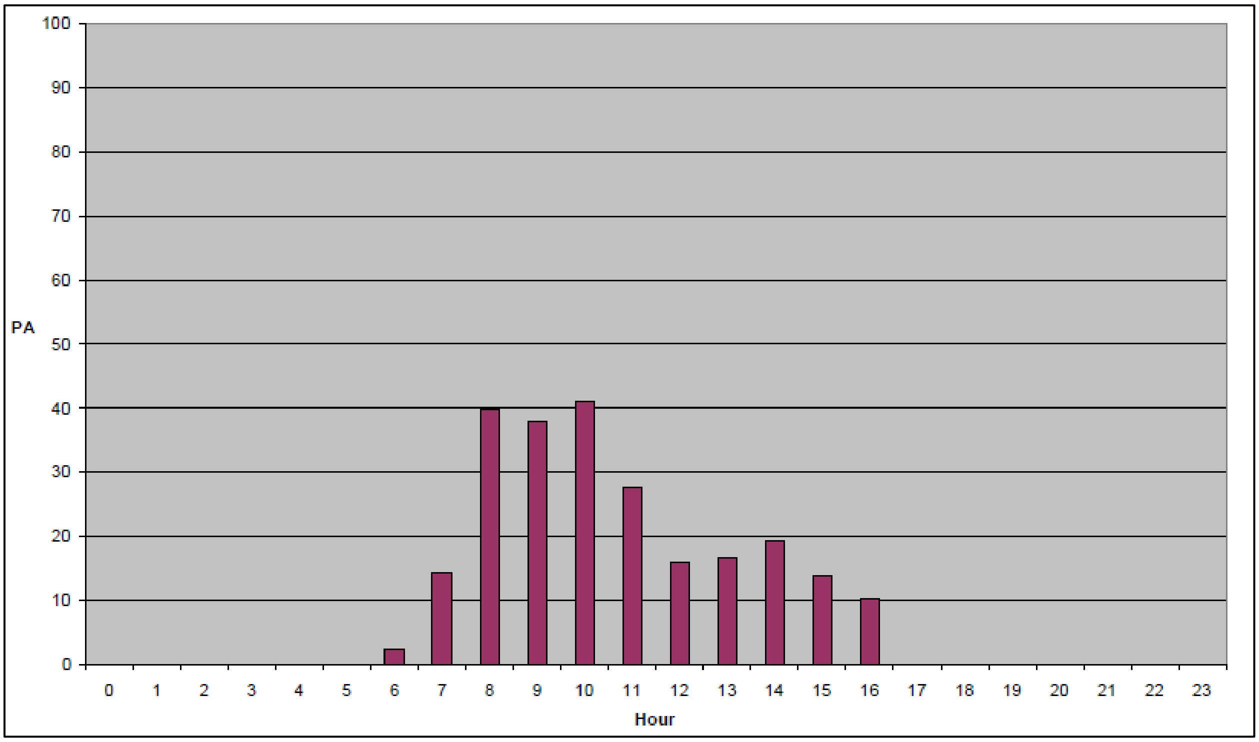

Figure 4.3.2-3. Mean percent of time that helicopters were audible (PA) per hour at HALE002 station (Crater, rain gauge near Puʻu Halaliʻi) in 2008. Source: reproduced from Lynch (2012).

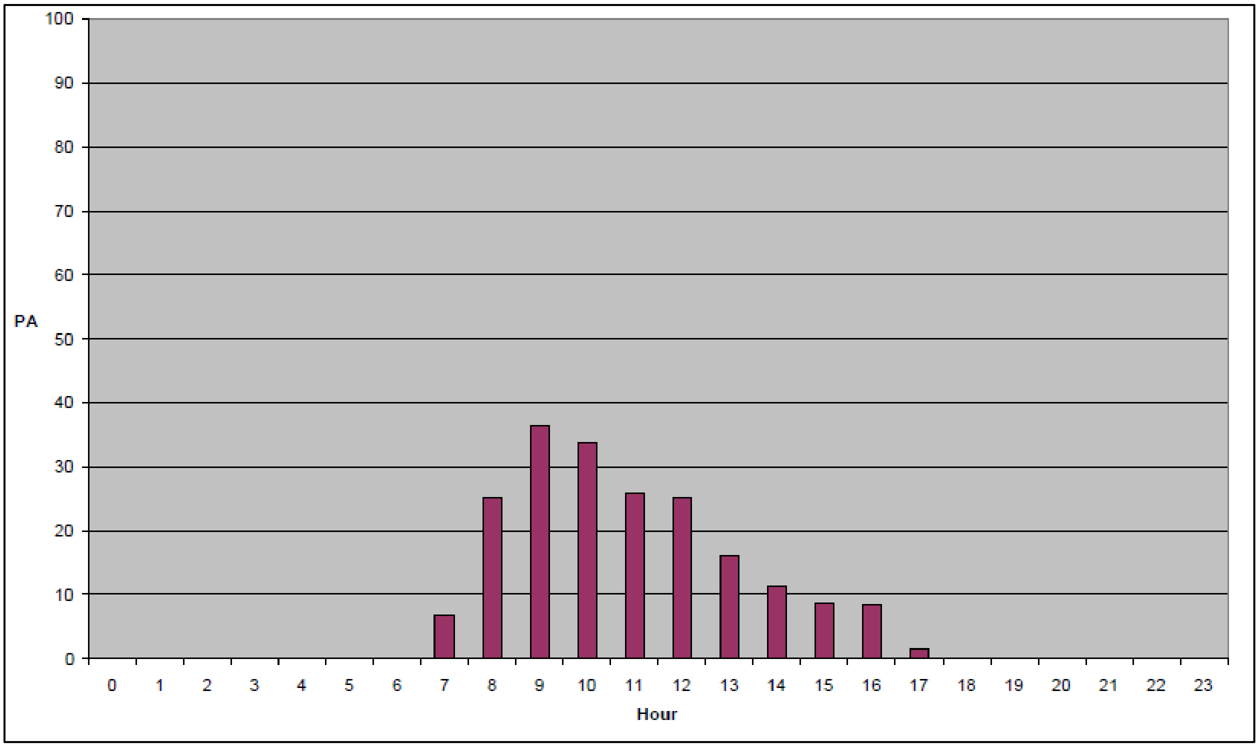

Figure 4.3.2-4. Mean percent of time that helicopters were audible (PA) per hour at HALE003 station (lower Kīpahulu Valley) in 2008. Source: reproduced from Lynch (2012).

Overall assessment

The overall condition of the soundscape at HALE is judged to warrant moderate concern. Sound recordings across HALE indicate that this wilderness area is unusually quiet in its natural state, and is quiet even under typical existing ambient conditions with the incursion of anthropogenic noise. However, daytime helicopter overflights remain a frequent source of noise that is audible in most regions of the park. The vast majority of these flights are commercial air tours, but NPS operations using helicopters also contribute to this noise. Backcountry hikers and native Hawaiian cultural practitioners hear this air traffic frequently, and perceive it negatively (Lawson et al. 2007, Prasad and Tomonari-Tuggle 2008). Evidence suggests that this negative perception persists regardless of whether helicopter flights are commercial in nature or conducted by park staff in the course of resource management or other efforts (Bell et al. 2010). Anthropogenic noise is much lower and less frequent during nighttime hours, and is dominated by jet overflights. The trend in this condition is unknown. While noise from helicopter tours is likely to have decreased substantially after the passage of the National Parks Overflights Act of 1987 and subsequent agreements between HALE and Maui tour operators, commercial operators are not required to report their flights to NPS, and most do not (Lynch 2012). Data from air tour fees paid to HALE suggest a declining number of tours from 2002 to 2008; however, fees are paid inconsistently, making it difficult to assess any longer-term trends in helicopter noise frequency (Lynch 2012). (return to Condition Summary).

Level of confidence

Overall level of confidence in this assessment is medium. Sound level data recorded at HALE are quite extensive, relative to other parks. However, the frequency of helicopter tours near the park, the principal source of noise, has not been estimated since 2008. This reduces confidence in the current level of noise impacts on the soundscape.

Information gaps and research recommendations

Data on the number and frequency of helicopter air tours near the park would allow a much better assessment of the magnitude and trend of this source of noise, which is the main factor negatively affecting the HALE soundscape. In addition, very little to nothing is known about the impacts of noise on park wildlife.

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4.3.3. Viewscape

Condition Summary

Overall resource condition is good with an unknown trend; confidence in condition determination is medium. Rationale is discussed in the "*Overall assessment*" section and following.

🟢

Relevance and Context

Natural landscapes and viewsheds represent core values to visitors of National Parks. These values motivate the restriction of development that might degrade the "untrammeled" nature of wilderness areas (Public Law 88-577, 16 U.S.C. section 1131-1136). Dark night skies are an integral part of viewscapes in National Parks, allowing for exceptional stargazing and other night activities (Smith and Hallo 2013), and are ecologically important for many nocturnal species (Rich and Longcore 2006). Preserving dark night skies, however, is in some ways more difficult than preserving landscapes, because light pollution is increasingly impairing dark night skies worldwide (Falchi et al. 2016). While national parks may successfully remove or minimize light sources within their boundaries, they may have less capacity to mitigate light pollution originating from urban areas outside. Such light pollution can affect night skies many miles away (Falchi et al. 2016).

According to the NPS Natural Sounds and Night Skies Division (NSNSD), light pollution, or the introduction of artificial light into the natural environment, can take two forms: Glare is the direct shining of light, while Sky Glow (or "light domes" or "fugitive light"), is the brightening of the night sky from human-caused light scattered in the atmosphere (NPS NSNSD 2017a). Both forms of light pollution are undesirable at HALE. Sky Glow from urban areas of Maui may degrade the quality of astrological viewing, and can otherwise diminish the sense of solitude and connection with nature in wilderness areas of the park. Glare from point sources of light may impact wildlife, even when such sources are distant from the park. For example, seabirds including ʻuaʻu (*Pterodroma sandwichensis*) and ʻaʻo (*Puffinus auricularis newelli*) are known to be attracted to lowland urban light sources in Hawaiʻi, and this may result in the downing and death of many fledgling birds as they depart their nests on their first flight out to sea (Reed et al. 1985, Gassmann-Duvall et al. 1988). At HALE, attraction to coastal lights was estimated to account for 8% of ʻuaʻu mortalities between 1991 and 2011 (NPS 2012).

The HALE Foundation Document (NPS 2015) makes specific reference to the importance of preserving viewscapes, including dark night skies: "At Haleakalā, the volcano's height, landscape, air quality, and location on Earth provide for excellent, clear night skies. From ancient Polynesian navigators to current day astronomers, people have and continue to use the summit of Haleakalā to study and view the night sky. Numerous light-sensitive species, whose lives are negatively impacted by artificial light, depend on Haleakalā's natural lightscapes for survival." Natural Sounds, Viewsheds, and Dark Night Skies are also identified as among the park's Fundamental Resources and Values (NPS 2015). With the exception of the astronomical observatories at Science City, there are no major human alterations of daytime viewsheds of the landscape, especially within designated wilderness areas. Dark night skies, and the degree of their impairment from light pollution, is therefore used as the most relevant indicator of the condition of the viewscape at HALE.

Indicator

- Dark night skies

Data and Methods

A review of available data and information was used to assess conditions, and if possible, trends, in the indicator listed above. No additional data analysis was performed.

Sources of information

Information on night sky quality and light pollution at HALE was obtained from the Night Sky Monitoring Database (NPS NSNSD 2017b) and from the HALE night skies and photic environment resource summary (Wood 2015).

Reference condition

The reference condition for dark night skies is the absence of anthropogenic light pollution, both in the form of sky glow from urban lights and point sources of light glare within the park.

Condition and Trend

Dark night skies

Measurements of the photic environment at HALE are limited to three nights of data collection by the NPS NSNSD in June of 2012. Night skies were assessed on two consecutive nights from Puʻu ʻUlaʻula at the summit, and one night at Kalahaku Overlook (NPS NSNSD 2017b). Each measurement night yielded a Night Sky Monitoring Quality Report, which included both panoramic images of the night sky and artificial sky glow, and summary statistics on visual and photometric indicators.

The three sets of night sky images are reproduced in Figures 4.3.3-1 to 4.3.3-3. In the top half of each figure is the mosaic image of the night sky rendered in false color, indicating calibrated sky brightness. The Milky Way is clearly visible on all three nights, appearing as the lighter blue, green and yellow regions arcing across the sky. Sky glow is visible along the horizon in the direction of Kihei and central Maui. This artificial glow is depicted in the lower half of each figure. While clearly visible, the light pollution from sky glow is for the most part restricted to this one region of the sky, and bright areas do not penetrate very far above the horizon.

Narrative accounts by the observers stated: “The city lights of Kahului down to the SW coast (Kihei and Wailea) produced a substantial glow below the horizon. The Milky Way was rich in detail and the Southern Cross (Crux) was clearly visible…The zodiacal light was prominent coming out of the glow from Kihei and Honolulu.” and “The Milky Way was again striking in its detail, though it was obscured at about 15° in the NNE by clouds and moisture/humidity.” (NPS NSNSD 2017b).

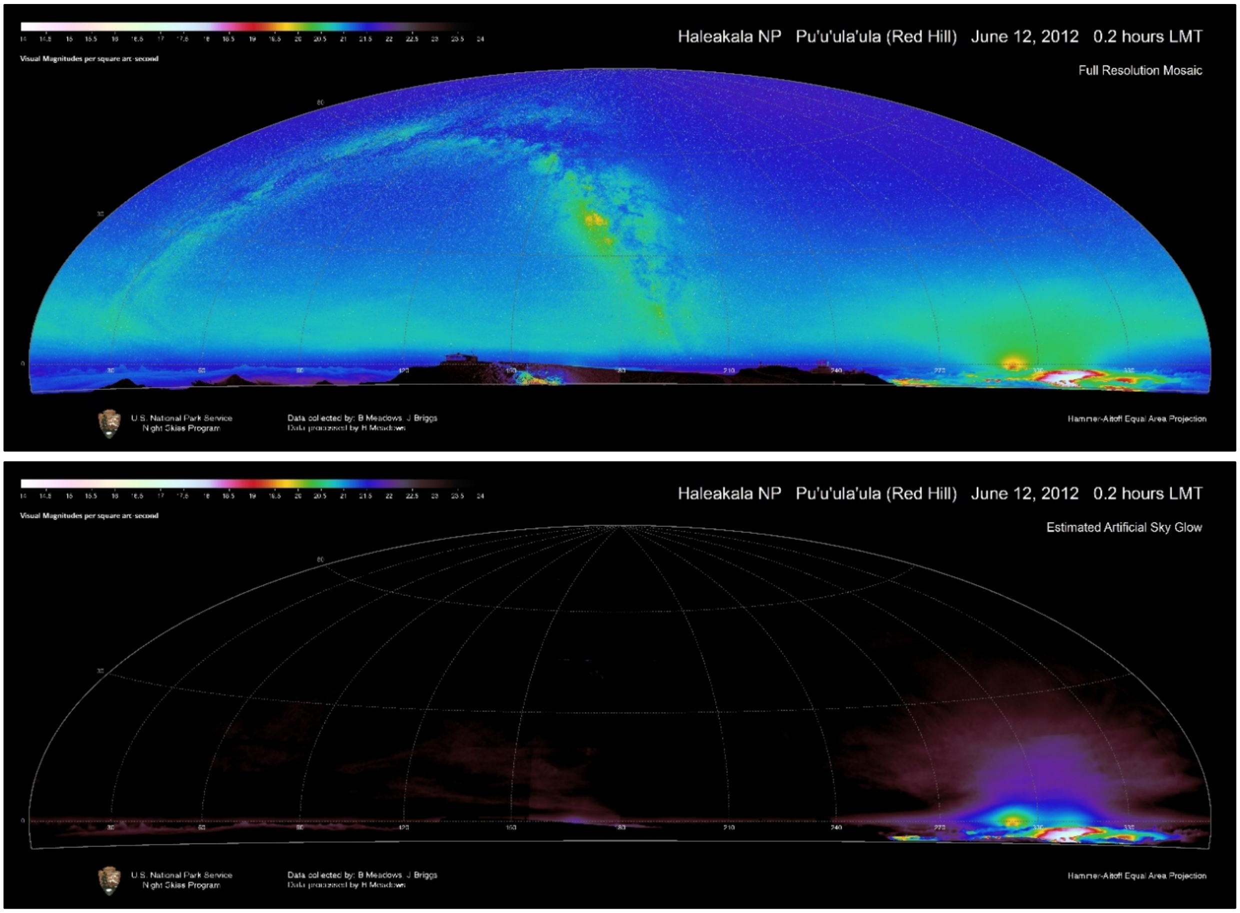

Figure 4.3.3-1. Night sky at summit, Puʻu ʻUlaʻula, on June 12, 2012, at 12:14 am. Top half shows full resolution mosaic image of night sky in false color depiction to indicate calibrated sky brightness. Bottom half shows image of estimated artificial sky glow from anthropogenic lights. Source: NPS NSNSD (2017b).

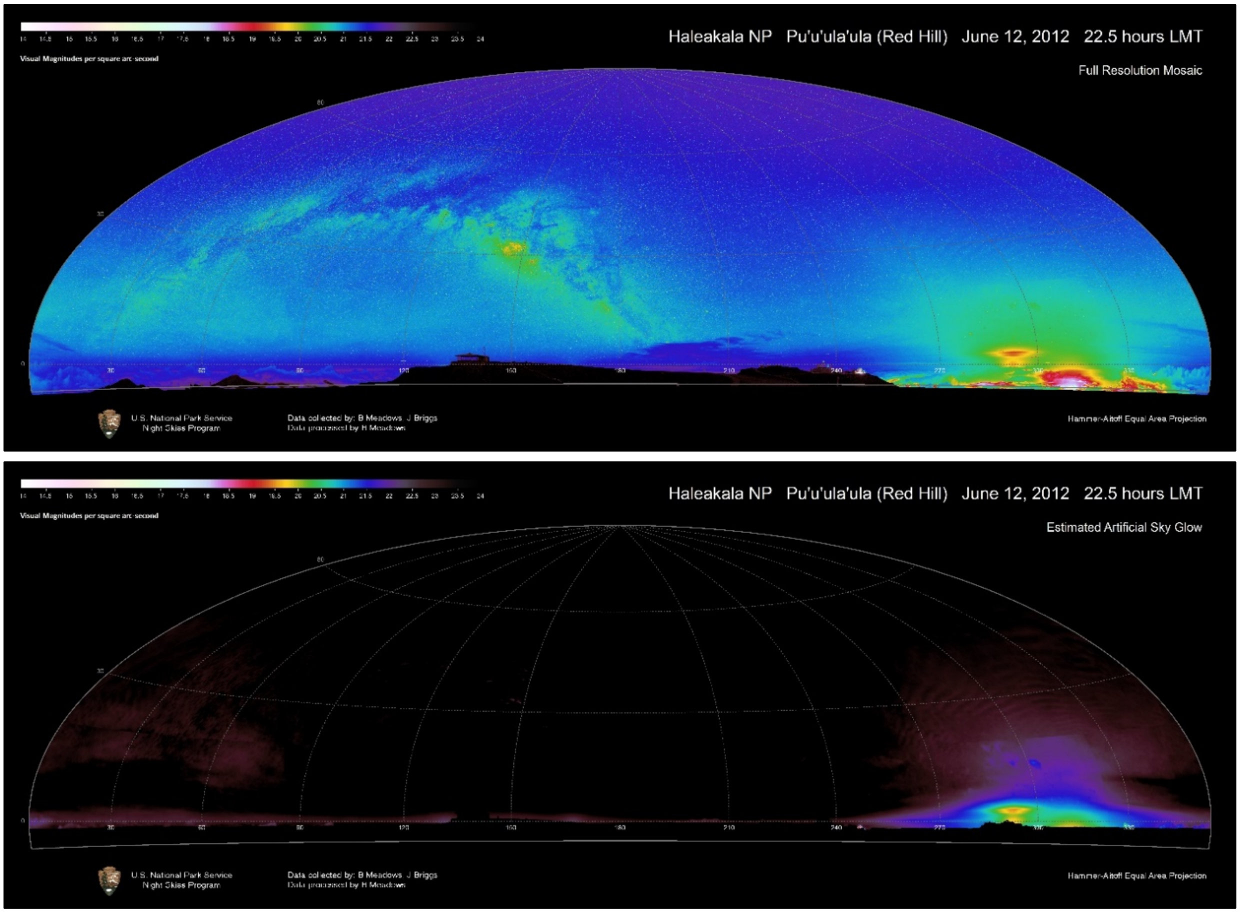

Figure 4.3.3-2. Night sky at summit, Pu‘u ‘Ula‘ula, on June 12, 2012, at 10:30 pm. Top half shows full resolution mosaic image of night sky in false color depiction to indicate calibrated sky brightness. Bottom half shows image of estimated artificial sky glow from anthropogenic lights. Source: NPS NSNSD (2017b).

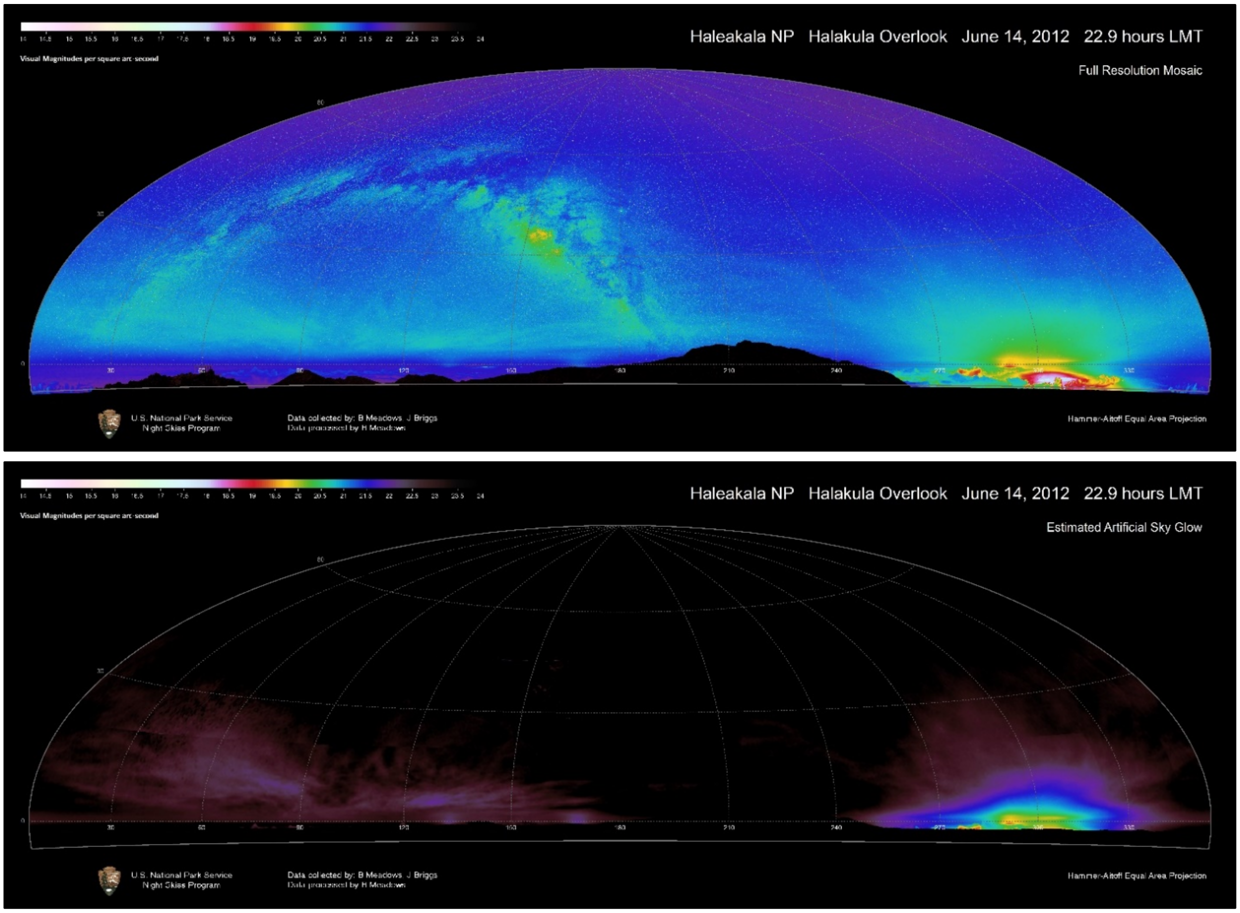

Figure 4.3.3-3. Night sky at Kalahaku Overlook, on June 14, 2012, at 10:56 pm. Top half shows full resolution mosaic image of night sky in false color depiction to indicate calibrated sky brightness. Bottom half shows image of estimated artificial sky glow from anthropogenic lights. Source: NPS NSNSD (2017b).

Some of the more relevant visual and photometric indicators measured on the three nights are summarized in Table 4.3.3-1. These data, and the following descriptions of the indicators, are reproduced from NPS NSNSD (2017b).

NELM is the naked eye limiting magnitude, or the faintest stars that can be observed with the naked human eye. This metric is somewhat subjective as it varies from observer to observer; however, 6.6 is considered near pristine under average conditions, 7.0 is achievable under good seeing conditions, and 7.4 is excellent. Values below 6.3 usually indicate significantly degraded sky quality.

Bortle Class is a semi-quantitative measure of the sky quality observed visually, as developed by astronomer John Bortle. Classes are whole numbers 1-9, with 1 the very best and 9 the poorest. Classes 2 and 3, scored at HALE, correspond to a “Typical truly dark site” and “Rural sky”, respectively. In the latter, “some indication of light pollution is evident along the horizon” (Bortle 2001).

Table 4.3.3-1. Summary of important visual and photometric indicators measured at HALE on three nights in 2012. Descriptions of indicators provided in text. Source: NPS NSNSD (2017b).

Visual or Photometric Indicator	Puʻu ʻUlaʻula June 12 12:14 am	Puʻu ʻUlaʻula June 12 10:30 pm	Kalahaku Overlook June 14 10:56 pm
NELM	6.8	6.5	6.8
Bortle Class	3	3	2
SQI All-sky	93.4	94.3	93.2
Zenith Sky Luminance LPR	<0.10	<0.10	<0.10
Mean All-sky Luminance LPR	0.18	0.11	0.14
Median Sky Luminance LPR	0.06	0.04	0.06
Horizontal Illuminance LPR	0.06	0.05	0.05

SQI All-sky is the sky quality index for the entire sky. It is a synthetic index derived from the distribution of sky luminance values in the artificial sky glow mosaic. Its range is 0-100, where 100 is a sky free of artificial sky glow. Values of 80-100 may be considered to represent skies that retain all of the natural characteristics throughout most of the sky, 60-80 retaining most of the natural sky features, but only in areas within 40 degrees of the zenith, 40-60 represents skies where the Milky Way is not visible or only the brightest parts are visible near the zenith, 20-40 represents skies only stars and planets remaining and the land is illuminated at a level of moonlight, and 0-20 indicates only the brightest stars remain, and the land is in perpetual twilight.

Zenith Sky Luminance LPR is the luminance of the sky in a one degree circle around the zenith, one of the most commonly reported sky quality indicators, and is expressed here in terms of the light pollution ratio (LPR). The LPR indicates the amount of artificial or anthropogenic light measured above a natural reference condition; for example, a value of 0.10 indicates the luminance value is 10% above natural, whereas a value of 0.50 indicates 50% above natural.

Mean All-sky Luminance LPR is the mean luminance measured over the entire sky. It is an unbiased measure of the amount of light reaching the observer from sky luminance, and is again reported here in terms of the LPR.

Median Sky Luminance LPR is the middle sky brightness value over the whole sky; a view of the whole sky will reveal most of the areas to be near this value. It is reported here in terms of the LPR.

Horizontal Illuminance LPR refers to the amount of light striking the ground from the sky. It is reported here in terms of the LPR.

All of the values summarized in Table 4.3.3-1 indicate good night sky conditions, with artificial luminance and illuminance levels less than 20% above natural conditions. Moreover, these measurements were made at the summit and along the west rim of the Crater, which have direct views of urban areas in central and southern Maui. Although no measurements have been made in the Crater or other wilderness areas, light pollution in the form of sky glow is presumably even lower in most if not all of these areas. Point sources of artificial light within wilderness areas are very few, and are essentially restricted to candle or lantern lights at the three visitor cabins, photovoltaic-powered electric lights at Palikū cabin, headlamps from campers and hikers, and occasional lantern lights at other backcountry shelters when they are in use. Although one dead ʻuaʻu found near Hōlua cabin may have been attracted to and grounded because of candle or lantern light within (NPS 2012), this type of event is likely to be exceedingly rare.

Overall assessment

The overall condition of the viewscape at HALE, based on the quality of dark night skies, is judged to be good. The data available on the nighttime photic environment suggest limited impairment from light pollution. The NSNSD resource summary for HALE (Wood 2015) stated that at the light levels experienced in HALE, most observers feel they are in a natural environment; the Milky Way is visible from horizon to horizon and fine details (such as the Prancing Horse) may be observed; that zodiacal light (or "false dawn," the faint glow at the horizon just before dawn or just after dusk) can be seen under favorable conditions; and there is negligible impact to dark adaptation in any direction. The summary further indicated that HALE's "unique landscape and location provide skywatching opportunities that are not found elsewhere" (Wood 2015). The trend in this condition is unknown. Although Maui continues to develop, there has been no repeated night sky monitoring that could measure the degree to which this development may be increasing light pollution at HALE, especially within designated wilderness areas. (return to Condition Summary).

Level of confidence

Overall level of confidence in this assessment is medium. Night sky photic measurements collected at HALE are robust and relatively recent, but they are limited in spatial and temporal extent.

Information gaps and research recommendations

Periodic monitoring of light pollution reaching HALE would allow for an assessment of a trend in the condition of dark night skies. Characterization of the nighttime photic environment within the Crater would also be useful information.

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Chapter 5. Discussion

Resource condition summaries for all of the focal resources, and their indicators, that are assessed in this report are presented in Tables 5-1 to 5-16, and are organized by resources pertaining to the life-supporting environment (Tables 5-1 to 5-3), biological integrity (Tables 5-4 to 5-13), and landscape condition context (Tables 5-14 to 5-16), respectively. An overall summary table, Table 5-17, distills the conditions and trends for the focal resources in each of the three categories. These tables may be viewed as a natural resource summary describing the current condition of the park's focal resources.

Table 5-1. Indicator summary for Air Quality focal resource.

Indicators of Condition	Measures or Criteria	Condition Status/Trend	Rationale
Ground-level ozone	NPS ARD criteria	◌	Ground-level ozone data were not available.
Wet deposition of N and S	NPS ARD criteria	◌	Although ecosystem sensitivity to acidification was estimated to be very high for HALE, no recent wet deposition data were available.
Visibility	NPS ARD criteria	🟢 ⇧	Visibility at HALE is generally very good, but can be affected by volcanic emissions from Hawai‘i Island. Resultant haze has been improving from the poorer conditions caused by invigorated activity in 2008-2010 at Kīlauea Volcano, but may now be affected by renewed eruptions in 2018.
Air Quality Overall		🟡 ⇧	–

Table 5-2. Indicator summary for Soil Quality focal resource.

Indicators of Condition	Measures or Criteria	Condition Status/Trend	Rationale
Erosion	Absence of apparent unnatural levels of soil loss	🟡 ⇔	Unnatural erosion appears to be an insignificant issue in most areas of the park, except for Nu‘u where erosion is currently severe. This is expected to improve following fencing and removal of ungulates. No data on rates of erosion at HALE are available, however.
Soil contamination	No anthropogenic pollutants or non-native animal inputs	◌	Although substantial or widespread soil contamination appears highly unlikely, there are no data to confirm this.

Table 5-2 (continued). Indicator summary for Soil Quality focal resource.

Indicators of Condition	**Measures or Criteria**	**Condition Status/Trend**	**Rationale**
Soil Quality Overall		🟡 ↔	–

Table 5-3. Indicator summary for Water Quality focal resource.

Indicators of Condition	**Measures or Criteria**	**Condition Status/Trend**	**Rationale**
Total Phosphorus	State of Hawai'i WQS	🟢	There have been no known violations of state water quality standards with respect to total phosphorus levels in HALE streams.
Total Nitrogen	State of Hawai'i WQS	🟢	There have been no known violations of state water quality standards with respect to total nitrogen levels in HALE streams.
Nitrate + Nitrite	State of Hawai'i WQS	🟡	Nitrate + Nitrite levels have exceeded state standards several times from 2007-2011, especially during the dry season in 'Alelele and Palikea Streams.
Turbidity	State of Hawai'i WQS	🟢	Turbidity levels exceeded state standards in only one year from 2007-2011. Turbidity can be expected to fluctuate strongly in Hawaiian streams, in response to natural high-flow events.
Water Quality Overall		🟢	–

Table 5-4. Indicator summary for Coastal Ecosystems focal resource.

Indicators of Condition	Measures or Criteria	Condition Status/Trend	Rationale
Coastal vegetation	Diversity and abundance of native species, degree of invasion by non-native species	🔴 ↔	Most coastal vegetation communities have been heavily modified, and are dominated by non-native plant species. Some nice examples of coastal strand vegetation exist in HALE, but most areas are low in native diversity.
Coastal vertebrates	Diveristy and abundance of marine animals, seabirds, shore birds and other birds	🔴 ↔	Shorebirds are present but not abundant. Almost no seabirds nest in coastal areas, likely owing to predation by non-native mammals and habitat degradation. Native passerines and other birds are completely absent from coastal ecosystems, due to disease, predation and habitat conversion.
Coastal invertebrates	Diversity and abundance of native species, degree of invasion by non-native species	🔴	Very few native invertebrate species appear to remain in coastal ecosystems. This loss results from competition and predation from invasive invertebrates, and habitat degradation.
Coastal Ecosystems Overall		🔴 ↔	–

Table 5-5. Indicator summary for Freshwater Ecosystems focal resource.

Indicators of Condition	Measures or Criteria	Condition Status/Trend	Rationale
Fish	Diversity and populations of native gobies	🟢 ↔	All five species of ‘o‘opu are present in the park, with known populations in several streams. Community composition and abundances appear stable over a several-decade period.
Aquatic invertebrates	Diveristy and abundance of freshwater molluscs, crustaceans and insects	🔴 ↔	Although aquatic insect communities appear to be in relatively good condition; native freshwater molluscs and shrimps both appear to have undergone dramatic population declines since measurements in the 1990s, with this situation unchanging over the most recent survey periods.
Freshwater Ecosystems Overall		🟡 ↔	–

Table 5-6. Indicator summary for Forest Ecosystems focal resource.

Indicators of Condition	Measures or Criteria	Condition Status/Trend	Rationale
High elevation vegetation (>4,000 ft)	Native plant diversity and cover, non-native plant diversity and cover	🟢⇔	High elevation wet forest richness and cover have increased slightly following control of feral animals, and native vegetation remains relatively intact and uninvaded by non-native weeds. In comparison, mesic forests of Kaupō are more highly invaded, but through management efforts have increased in extent and diversity over recent decades.
Low elevation vegetation (1,000-4,000 ft)	Native plant diversity and cover, non-native plant diversity and cover	🔴⇩	Low elevation forests have been heavily modified in areas, and over one third of original native cover has been replaced by non-native dominated vegetation. Native richness and cover increased following fencing and control of feral pigs, but subsequent invasion by habitat-modifying understory weeds threatens to reverse these gains. Cover of *Acacia koa* canopy has also declined, with the current trajectory of this measure uncertain.
High elevation bird communities (>4,200 ft)	Native forest bird diversity and abundance	🟢⇔	Recent data suggest the six extant forest bird species, including the two endangered species, have exhibited stable population trends. Current threats are generally lower than in low elevation forests. Assessment over a longer time period, however, is less favorable, with several known and putative species extinctions, and probable population declines.
Low elevation bird communities (1,000-4,200 ft)	Native forest bird diversity and abundance	🔴⇩	One third of the extant native forest bird species are absent, and half of the remaining species exist at lower densities compared to high elevation forests. Avian malaria and pox will continue to impact low elevation bird communities, and non-native rodents and possibly plants also contribute to their poor condition and declining trend.
Invertebrate communities	Native invertebrate diversity, degree of non-native invertebrate invasion	◌	High elevation wet forest invertebrate communities may be in relatively good condition, while mesic and low elevation wet forest communities are likely more degraded. However, most information on these communities is fragmentary and ~40-50 years old, making any definitive assessment of current condition or trend difficult.
Forest Ecosystems Overall		🟡⇔	–

Table 5-7. Indicator summary for Bog Ecosystems focal resource.

Indicators of Condition	**Measures or Criteria**	**Condition Status/Trend**	**Rationale**
Bog vegetation	Native plant diversity and cover	🟡 ↔	Some upper elevation bogs have increased in richness and cover following ungulate exclusion, but others remain moderately invaded. Lower elevation bogs are heavily modified and are unlikely to increase in native species richness or diversity.
Bog Ecosystems Overall		🟡 ↔	**–**

Table 5-8. Indicator summary for Cave Ecosystems focal resource.

Indicators of Condition	**Measures or Criteria**	**Condition Status/Trend**	**Rationale**
Cave-dwelling invertebrates and other fauna	Intact, uninvaded communities	🟢 ↔	A number of unique cave-adapted species occur in HALE, and communities appear relatively undisturbed. Invasive rats and arthropods are main current threats.
Vegetation	Undisturbed and uninvaded communities	🟢	No site-specific data on cave-associated vegetation communities exists, but overall patterns of plant invasion suggest that vegetation above caves at higher elevations should be relatively undisturbed and uninvaded, and should provide necessary root resources for cave communities.
Cave Ecosystems Overall		🟢 ↔	**–**

Table 5-9. Indicator summary for Shrubland Ecosystems focal resource.

Indicators of Condition	Measures or Criteria	Condition Status/Trend	Rationale
West Slope vegetation	Native plant diversity and cover, non-native plant diversity and cover	🟢 ⇧	Native plant diversity and cover is substantially higher than that of non-native plants, especially in the shrub canopy layer. This condition has improved since the fencing and removal of ungulates, although rigorous data to support this inference are lacking. Threats from habitat-modifying invasive plants persist, and changing climate represents a potential new threat.
Kaupō vegetation	Native plant diversity and cover, non-native plant diversity and cover	🟢 ⇧	Native plant diversity and cover is substantially higher than that of non-native plants, especially in the shrub canopy layer. This condition has improved since the fencing and removal of ungulates, although rigorous data to support this inference are lacking. Threats from habitat-modifying invasive plants persist, and changing climate may represent a new threat.
Invertebrates	Native invertebrate diversity, degree of non-native invertebrate invasion	🔴 ⇩	A large fraction of shrubland arthropod communities is composed of non-native species, including several highly damaging species of invasive ants and wasps. The number of established non-native arthropod species continues to grow, and the invasive Argentine ant continues to spread. Degradation of the native arthropod community may also impair important ecological interactions, such as pollination.
Shrubland Ecosystems Overall		🟡 ⇔	–

Table 5-10. Indicator summary for Sub-alpine Grassland Ecosystems focal resources.

Indicators of Condition	Measures or Criteria	Condition Status/Trend	Rationale
Grassland vegetation	Native plant diversity and cover, non-native plant diversity and cover	🟢 ⇔	Native plant diversity, and especially cover, is higher than that of non-native plants. This condition improved since the fencing and removal of ungulates several decades ago. Data on more recent trends are lacking, but vegetation condition appears to be fairly stable. Changing climate may represent a new influence on these ecosystems.
Sub-alpine Grassland Ecosystems Overall		🟢 ⇔	–

Table 5-11. Indicator summary for Craters focal resources.

Indicators of Condition	Measures or Criteria	Condition Status/Trend	Rationale
Crater vegetation	Native plant diversity and cover, non-native plant diversity and cover	🟢↔	Native plant diversity and cover is substantially higher than that of non-native plants. This condition has likely improved since the fencing and removal of ungulates, although data to support this inference are lacking. Conversely, recent evidence indicates that changing climate is strongly impacting ʻāhinahina plants, but it is unclear whether similar effects on other Crater plants are occurring.
Crater invertebrates	Native invertebrate diversity, degree of non-native invertebrate invasion	🟡↓	A moderate fraction of shrubland arthropod communities is composed of non-native species, including the highly damaging Argentine ant and western yellowjacket. The number of established non-native arthropod species has likely continued to grow, as in adjacent shrubland ecosystems, but comprehensive invertebrate surveys have not been conducted since the 1970s to quantify actual patterns. While the Argentine ant is currently limited in distribution to a small portion of the western Crater, it is spreading rapidly in this region.
Crater Overall		🟢↔	–

Table 5-12. Indicator summary for Nu'u Parcel focal resource.

Indicators of Condition	Measures or Criteria	Condition Status/Trend	Rationale
Vegetation	Native and invasive species richness and cover, soil retention	🔴	Most vegetation communities have been heavily modified by cattle grazing and feral ungulate browsing, and are now dominated by non-native plant species. Forests and shrublands are mostly converted to open or bare ground, resulting in severe soil erosion.
Birds	Native bird diversity and population size	🔴	Nine native bird species are thought to occur in Nuʻu, but abundance estimates exist only for ʻuaʻu, whose numbers of burrows are low. Most or all other native birds likely exist at low population densities. Various non-native predators occur in the parcel and are currently unmanaged.

Table 5-12 (continued). Indicator summary for Nu'u Parcel focal resource.

Indicators of Condition	**Measures or Criteria**	**Condition Status/Trend**	**Rationale**
Invertebrates	Native arthropod diversity, degree of non-native arthropod invasion	🔴	No direct information on arthropods or other invertebrates is known from Nuʻu. However, the highly degraded condition of the habitats they rely on makes it very likely that only small remnant populations and low diversities of native arthropods remain.
Nuʻu Parcel Overall		🔴	–

Table 5-13. Indicator summary for T&E Species focal resource.

Indicators of Condition	**Measures or Criteria**	**Condition Status/Trend**	**Rationale**
Plants	Population status and trends of T&E plants	🔴⇩	Estimation of population trends is not possible for nearly all plant taxa, but anecdotal evidence suggests that at least several are declining, while possibly only one may be increasing. Most taxa persist with very small and precarious populations, and while outplanting has augmented many of these, natural regeneration has been observed for less than half of the extant taxa. Prospects for delisting are few, if any, while the list of T&E plants in the park continues to grow.
Birds	Population status and trends of T&E birds	🟡	Out of seven resident extant bird species, three currently appear to have either stable or increasing population trends. At least two of the remaining four species exist at very low densities, and information on population trends is limited for all four species. Future warming may cause the conditions of listed forest bird species to decline strongly, due to elevational expansion of mosquitos and the diseases they vector.
Invertebrates	Population status and trend of listed damselfly	◌	Information on the sole listed invertebrate at HALE, the damselfly *Megalagrion pacificum*, is limited to two collecting events in 1980 and 1993. No estimates of population numbers or trends are possible, precluding an assessment of current condition.

Table 5-13 (continued). Indicator summary for T&E Species focal resource.

Indicators of Condition	Measures or Criteria	Condition Status/Trend	Rationale
Bat	Population status and trend of listed bat	◌	Bats are regularly detected traversing and foraging in HALE, and may roost in certain locations. However, no estimates of population size or trend are possible at this time, precluding an assessment of current condition.
T&E Species Overall		🔴	–

Table 5-14. Indicator summary for Fuel and Fire Dynamics focal resources.

Indicators of Condition	Measures or Criteria	Condition Status/Trend	Rationale
Fuel characteristics	Regional fuel types and unnatural fine fuel loads	🟡	General risk of wildfire at HALE is low, but regions with extensive areas of unnatural fine fuel loads, especially fire-prone non-native grasses, exist. Fire risks are likely to be highest in the Kaupō and Nuʻu regions.
Fire frequency and size	Departure from natural fire regime	🟡⇧	Fire frequency and size has likely increased substantially from the natural historic fire regime in most of HALE. However, area burned has been very small in recent decades, suggesting that this condition is improving through effective fire management.
Fuel and Fire Dynamics Overall		🟡	–

Table 5-15. Indicator summary for Soundscape focal resources.

Indicators of Condition	Measures or Criteria	Condition Status/Trend	Rationale
Natural quiet	Magnitude and frequency of anthropogenic noise	🟡	Existing ambient sound levels in wilderness areas of HALE are unusually low, especially during quiet conditions. However, noise from helicopters, mainly commercial air tours, are a frequent disruption to natural sound levels. It is unclear whether these tours are increasing or decreasing in frequency.
Soundscape Overall		🟡	–

Table 5-16. Indicator summary for Viewscape focal resources.

Indicators of Condition	**Measures or Criteria**	**Condition Status/Trend**	**Rationale**
Dark night skies	Degree of light pollution	🟢	Light pollution in the form of sky glow from Maui’s urban environment is low at the summit and west rim of the Crater, and is presumably even lower in backcountry wilderness areas. Point sources of light pollution are also minimal in wilderness areas. Dark night sky measurements have only been made once, precluding an assessment of trends in this condition.
Viewscape Overall		🟢	–

Table 5-17. Overall resource-level summary table.

Resource Category	**Focal Resource**	**Condition Status/Trend**	**Rationale**
Life-Supporting Environment	Air Quality	🟢⬆	Levels of ground-level ozone and wet deposition of N and S are unknown. Visibility at HALE is generally very good, but can be affected by volcanic emissions from Hawai‘i Island. Resultant haze has been improving from the poorer conditions caused by invigorated activity in 2008-2010 at Kīlauea Volcano, but may now be affected by renewed eruptions in 2018.
	Soil Quality	🟡↔	Unnatural erosion appears to be an insignificant issue in most areas of the park, except for Nu‘u where erosion is currently severe. This is expected to improve following fencing and removal of ungulates. Substantial or widespread soil contamination appears highly unlikely.
	Water Quality	🟢	Levels of total phosphorus, total nitrogen, and turbidity are generally good, with no or few known violations of state water quality standards. Nitrate + Nitrite levels have exceeded state standards several times from 2007-2011, but longer terms trends are unknown.

Table 5-17 (continued). Overall resource-level summary table.

Resource Category	Focal Resource	Condition Status/Trend	Rationale
Biological Integrity	Coastal Ecosystems	🔴 ↔	Most coastal vegetation communities have been heavily modified, and are dominated by non-native plant species. This habitat degradation, along with pressures from abundant invasive invertebrates, contributes to the poor condition of native coastal invertebrate communities. Shorebirds are present but not abundant, and almost no seabirds nest in coastal portions of the park, likely owing to predation by invasive mammals. Similarly, native passerines are absent from coastal ecosystems.
	Freshwater Ecosystems	🟡 ↔	All five species of ‘o‘opu are present and apparently stable in the park, with known populations in several streams. However, native freshwater molluscs and shrimps appear to have undergone dramatic population declines since measurements in the 1990s, with this situation apparently unchanging over the most recent survey periods.
	Forest Ecosystems	🟡 ↔	The condition of forest ecosystems differ sharply between low and high elevation zones, separated roughly by the 4,000 ft elevation level. Native plant diversity and cover has remained high, and non-native plant invasion has been comparatively low, in high-elevation forests. Similarly, the six extant forest bird species have exhibited stable population trends. Low elevation forests, in contrast, are much more heavily invaded by non-native plants, and support lower diversity and densities of forest birds. Similar patterns may apply to native invertebrate communities, but information is too limited to be sure.
	Bog Ecosystems	🟡 ↔	Some upper elevation bogs have increased in richness and cover following ungulate exclusion, but others remain moderately invaded. Lower elevation bogs are heavily modified and are unlikely to increase in native species richness or diversity.
	Cave Ecosystems	🟢 ↔	A number of unique cave-adapted invertebrates occur in HALE, and invertebrate communities appear relatively undisturbed. Plant communities above known caves should be relatively undisturbed and uninvaded, and thus provide necessary root resources for cave communities.

Table 5-17 (continued). Overall resource-level summary table.

Resource Category	Focal Resource	Condition Status/Trend	Rationale
Biological Integrity (continued)	Shrubland Ecosystems	🟡↔	In both West Slope and Kaupō shrublands, native plant diversity and cover is substantially higher than that of non-native plants, especially in the canopy layer. This good condition appears to have improved since the fencing and removal of ungulates. However, shrubland arthropod communities are heavily invaded, and are especially threatened by several highly damaging species of invasive ants and wasps. The poor condition of arthropod communities appears to be worsening, as the number of established non-native arthropod species continues to grow, and the invasive Argentine ant continues to spread.
	Sub-alpine Grassland Ecosystems	🟢↔	Native plant diversity, and especially cover, is higher than that of non-native plants. This condition improved since the fencing and removal of ungulates several decades ago. Data on more recent trends are lacking, but vegetation condition appears to be fairly stable. Changing climate may represent a new influence on these ecosystems.
	Crater	🟢↔	Native plant diversity and cover is substantially higher than that of non-native plants. This good condition has likely improved since the fencing and removal of ungulates. However, recent evidence indicates that changing climate is impacting at least some plants, like ʻāhinahina, and may be an important new influence in Crater ecosystems. A moderate fraction of shrubland arthropod communities is composed of non-native species, including the highly damaging Argentine ant and western yellowjacket. The distribution of the Argentine ant in the Crater is currently limited but expanding, and is contributing to a worsening condition of native invertebrate communities.
	Nuʻu Parcel	🔴	Most vegetation communities have been heavily modified by cattle grazing and feral ungulate browsing, and are now dominated by non-native plant species. Forests and shrublands are mostly converted to open or bare ground, resulting in severe soil erosion. Most or all native bird species likely exist at low population densities, and non-native predators are currently unmanaged. The highly degraded condition of habitats in Nuʻu makes it very likely that only small remnant populations and low diversities of native arthropods remain.

Table 5-17 (continued). Overall resource-level summary table.

Resource Category	Focal Resource	Condition Status/Trend	Rationale
Biological Integrity (continued)	T&E Species	🔴	Estimation of population trends is not possible for nearly all of the 42 listed plant taxa, but anecdotal evidence suggests that at least several are declining, while possibly only one may be increasing. Most taxa persist with very small and precarious populations, and natural regeneration has been observed for less than half of the extant taxa. Only three of seven extant listed bird species currently appear to have either stable or increasing population trends, with information on the remaining four being limited. Future climate warming may cause the conditions of listed forest bird species to decline strongly. Information on the single listed invertebrate and bat is too limited to assess current condition for either species.
Landscape Condition Context	Fuel and Fire Dynamics	🟡	General risk of wildfire is low, but regions with extensive areas of unnatural fine fuel loads, especially fire-prone non-native grasses, exist, especially in the Kaupō and Nuʻu regions. Fire frequency and size has likely increased substantially from the natural historic fire regime in most of HALE, but area burned has been very small in recent decades, suggesting that fire management strategies have been effective.
	Soundscape	🟡	Existing ambient sound levels in wilderness areas of HALE are unusually low, especially during quiet conditions. However, noise from helicopters, mainly commercial air tours, are a frequent disruption to natural sound levels.
	Viewscape	🟢	Light pollution in the form of sky glow from Maui's urban environment is low at the summit and west rim of the Crater, and is presumably even lower in backcountry wilderness areas. Point sources of light pollution are also minimal in wilderness areas.

Of the 16 focal resources assessed for HALE, only six (38%) were judged to be in good condition. These are Air Quality, Water Quality, Cave Ecosystems, Sub-alpine Grassland Ecosystems, Crater, and Viewscape. One of these six resources was assigned an improving trend, three were assigned a stable trend, while trends could not be assessed for the remaining two. Three of the 16 focal resources (19%) were judged to warrant significant concern. These are Coastal Ecosystems, the Nuʻu Parcel, and T&E Species. The poor condition of Coastal Ecosystems was judged to be stable, while no trend was assigned for the Nuʻu Parcel and T&E Species. The remaining seven focal resources at

HALE (44%) were judged to warrant moderate concern, the intermediate condition category. Five of these were judged to be stable, and trends were not assigned for the remaining two resources.

Collectively, this assessment suggests that natural resources at HALE are under considerable strain. The geographic resources that are most degraded, namely Coastal Ecosystems and the Nuʻu Parcel, experienced wholesale conversion many decades to centuries ago. Some areas within these have transitioned into communities dominated by non-native species, so-called "novel ecosystems" (Davis et al. 2011), while others are now recovering from former agricultural use. For substantial portions of both types of areas, it is unlikely that they will return to a condition in which native species dominate, at least given current management capabilities. Yet, certain native species can persist within them, giving them conservation value. Other portions of these ecosystems are somewhat less degraded, and native communities may passively recover after the removal of feral ungulates (Weller et al. 2018). Even within some highly degraded ecosystems, like former cattle pasture, restoration of native-dominated shrubland and forest is possible with intensive effort (Scowcroft and Jeffrey 1999, Medeiros et al. 2014). Notwithstanding, restoration of the most degraded regions of HALE would likely require resource management commitments substantially above present levels.

Currently, resource management efforts are targeted at the remaining focal resources related to biological integrity, to prevent those warranting moderate concern to degrade further, and to protect those still in good condition from incipient threats. A good example of the latter is the effort to halt the recent invasion of pine trees into otherwise relatively intact shrublands. Across HALE, in fact, the primary force impacting biological resources is the collective pressures exerted by non-native species. As a consequence, conserving these resources continues to be most effectively achieved by attacking new invasive species, such as incipient weeds, and maintaining adequate control of persistent invasive species. Examples of the latter include maintaining fences and hunting trespassing feral ungulates, utilizing traplines to suppress populations of predatory small mammals, and holding the line on invasive plants that may be fairly widespread but have not yet penetrated into more pristine regions of the park.

Addressing the impacts of non-native invertebrates remains one of the most difficult challenges, for HALE and for Hawaiʻi more broadly. This includes both reducing the influx of new pests and improving methods for controlling especially damaging established species, such as ants, yellowjackets and disease-vectoring mosquitos. Other major resource management challenges include the potential arrival of devastating forest pathogens like Rapid ʻŌhiʻa Death (*Ceratocystis* spp.), and understanding how to most effectively deal with climate change. Regarding the latter, minimizing the pressures from invasive species will only become more important as stresses from changing climatic conditions intensify (Staudt et al. 2013). Maintaining collaborative ties with researchers will help ensure that the most effective management tools for all of these challenges are developed and employed.

Confidence for most of the resource assessments was not high, largely as a result of insufficient information. A substantial amount of the available information is qualitative in nature, and many information gaps exist, which is not surprising for such a large and complex natural area as HALE. Important information gaps identified within each focal resource assessment are summarized in

Table 5-18, which may be useful for resource management planning. Aside from an absence of information in certain areas, the lack of repeated monitoring data for all but a few resources made it difficult to assess trends in many cases. The relatively new NPS I&M program is an important step in improving this situation, and will undoubtedly make future resource assessments more robust.

Table 5-18. Summary of important information gaps for each focal resource.

Resource Category	Focal Resource	Important Information Gaps
Life-Supporting Environment	Air Quality	Data on ozone concentrations at HALE are not collected, and would provide a clearer picture of threats to human health and plants. There is also little information on N and S deposition at HALE. Given the high estimated ecosystem sensitivity to acidification, collection of on-site deposition data is recommended.
	Soil Quality	There are no monitoring programs or data on rates of erosion, soil contamination, or other metrics of soil quality at HALE.
	Water Quality	Various water quality parameters have not been measured recently, or with sufficient frequency, to assess their impacts on water quality at HALE. Some of these, such as concentrations of toxic compounds, can be highly detrimental to aquatic life. Relatively little information exists regarding human health pathogens in HALE streams. Little is known about water quality of the three montane lakes in the park.
Biological Integrity	Coastal Ecosystems	Repeated quantitative vegetation data are generally lacking, and no NPS I&M FTPC or EIPS monitoring plots are located in coastal ecosystems. Little is known about the life history of *Ischaemum byrone*, currently the only listed Endangered plant known to occur within the coastal region of the park. No surveys of invertebrates have been conducted in coastal ecosystems at Nuʻu or Kaʻāpahu, and no bird surveys from Nuʻu have been reported. Information on invertebrates at ʻOheʻo is also very limited.
	Freshwater Ecosystems	Continuous data on trends in aquatic biota are lacking for long periods of time, but should be improved with continued monitoring by the I&M program. Information on aquatic insects, especially at higher elevations, including the three montane lakes in the park, is very limited. Information on status, trends, and ecological effects of invasive species in and around freshwater ecosystems is limited.
	Forest Ecosystems	Repeated quantitative vegetation information is fairly limited, but will be remedied via the FTPC and the EIPS monitoring protocols of the I&M program. However, mesic forests of Kaupō are not covered in these monitoring systems. Consistent monitoring is not conducted for most of the threatened, endangered, or otherwise rare plant species in forest ecosystems. Repeated standardized monitoring of forest birds by the I&M program will provide greater certainty to future assessments of population trends. Recent quantitative surveys of native forest invertebrate communities are lacking, leaving a large information gap for this highly diverse and ecologically important forest component.

Table 5-18 (continued). Summary of important information gaps for each focal resource.

Resource Category	Focal Resource	Important Information Gaps
Biological Integrity (continued)	Bog Ecosystems	Repeated quantitative vegetation data are limited spatially and temporally. No I&M FTPC or EIPS monitoring plots are located in bog ecosystems. Information on invertebrate communities in bog ecosystems is very limited.
	Cave Ecosystems	No vegetation monitoring plots are known to be located directly above identified cave ecosystems, which would provide site-specific information on the status and trends of native and non-native plants that provide energy inputs for the resident invertebrate community. More extensive and repeated surveys of cave faunas would provide more information on their status and trends, and identification of plant roots within caves would help inform their management. However, the sensitive nature of these ecosystems requires careful consideration of the potential impacts of such monitoring.
	Shrubland Ecosystems	Repeated quantitative vegetation information is lacking, but will be remedied via the I&M FTPC and the EIPS monitoring networks. Consistent monitoring is not conducted for most of the threatened, endangered, or otherwise rare plant species in shrubland ecosystems. While shrubland arthropod communities are fairly well characterized on the West Slope, less is known about communities in other shrubland areas, and other invertebrates like molluscs are even less poorly known. No monitoring protocol exists for shrubland invertebrate communities.
	Sub-alpine Grassland Ecosystems	Repeated quantitative vegetation data are limited spatially and temporally. No I&M FTPC monitoring plots or EIPC fixed transects are located in subalpine grassland ecosystems. Information on invertebrate communities or other biota in grassland ecosystems is limited or lacking.
	Crater	Repeated quantitative vegetation information is lacking, but will be remedied via the I&M FTPC and the EIPS monitoring networks. Consistent monitoring is not conducted for many of the threatened, endangered, or otherwise rare plant species in the Crater. While Crater arthropod communities were fairly well characterized in the 1970s, this information is now very dated, and other invertebrates like molluscs are even less poorly studied. No monitoring protocol exists for Crater invertebrate communities.
	Nuʻu Parcel	No NPS I&M vegetation plots or observation points were located above 4,000 ft elevation, and existing data under-represent the upper elevation remnant native communities. No I&M FTPC monitoring plots or EIPC fixed transects are located in Nuʻu. Information on vertebrate and invertebrate species is currently very limited or in most cases completely lacking.
	T&E Species	For a number of the T&E species, including the invertebrate, bat, and several birds and plants, current or past population sizes are unknown. For most of the T&E species, repeated monitoring data allowing estimation of population trends are not available.

Table 5-18 (continued). Summary of important information gaps for each focal resource.

Resource Category	Focal Resource	Important Information Gaps
Landscape Condition Context	Fuel and Fire Dynamics	Fuel models specific to Hawaiian vegetation types are lacking, with a few localized exceptions. Better quantification of the distribution and fuel loads of non-native fire-prone vegetation would better characterize fire risk in different regions of the park.
	Soundscape	Data on the number and frequency of helicopter air tours near the park would allow a much better assessment of the magnitude and trend of this source of noise, which is the main factor negatively affecting the HALE soundscape.
	Viewscape	Periodic monitoring of light pollution reaching HALE would allow for an assessment of a trend in the condition of dark night skies. Characterization of the nighttime photic environment within the Crater would also be useful information.

Despite its resource management challenges, HALE remains one of the most valuable natural areas in Hawai‘i, protecting a wide diversity of ecosystems and a large number of threatened and endangered species. This circumstance is in large part owed to its inclusion within the National Park system, and the concerted management efforts that were subsequently undertaken. Major past achievements, such as the fencing and exclusion of feral ungulates from the majority of the park, provide hope that current and future threats may yet be mitigated with sufficient determination, creativity, and collaborative effort.

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Appendix A. Study Scoping Meeting Notes

Meeting Summary

The project kick-off meeting took place on October 13, 2015 at Pukalani Community Center, Maui, from 9am to 2:30pm. In attendance were project team members Paul Krushelnycky, Charles Chimera, and Eric VanderWerf, and NPS RM staff members Patti Welton, Joy Tamayose, Bill Haus, Sean Birney, Woody Mallinson, Cathleen Bailey, Raina Kaholoaa, and Elizabeth Urbanski. Positions of NPS staff are as follows:

Patti Welton: Botanist, Resource Management Division, Vegetation Management Program

Woody Mallison: Botanist, Resource Management Division, Vegetation Management Program Lead

Bill Haus, Botanist: Resource Management Division, Vegetation Management Program

Cathleen Bailey: Wildlife Biologist, Resource Management Division, Endangered Wildlife Management Program Lead

Joy Tamayose: Wildlife Biologist, Resource Management Division, Endangered Wildlife Management Program

Raina Kaholoaʻa: Biologist (Entomologist), Resource Management Division, Endangered Wildlife Management Program

Sean Birney: Data Manager, Resource Management Division

Elizabeth Urbanski: PACN Inventory and Monitoring Program

The NPS assessment project committee consisted of Patti Welton (Project Manager), Cathleen Bailey, and Steve Robertson (RM Division Chief, not in attendance).

Patti Welton started the meeting with an overview of the general purpose and format of NRCAs, and the group then initiated discussion of topics aimed at advancing the HALE NRCA in particular. First and foremost, this included a review of the list of priority themes and resources to be covered by the HALE NRCA, as laid out in the project funding opportunity announcement (FOA), which also inherently suggested an assessment framework that could be used in the NRCA. It was agreed that the framework used to identify and organize the priority themes and resources, which is essentially a combination of the Heinze Center framework and the NPS Ecological Monitoring framework, would serve as the framework used for the NRCA, possibly subject to some modification if needed. Listed potential assessment resources had been assigned priority rankings of 1, 2 or 3; it was decided that the NRCA would likely focus mainly on priority 1 and 2 resources. The group also discussed which themes or resources may be addressed by the NPS Regional Office, such as air quality and possibly some of the themes under the Landscape Condition Context.

Next, the group began discussion of the major sources of data for the main priority resources to be addressed by the NRCA, including birds, plants, invertebrates and stream water quality. This included both completed summaries and reports, as well as unanalyzed data that might be useful and amenable to analysis with a reasonable amount of effort on the part of the assessment team. These notes are listed in bullets below. An overview of the information available on the IRMA site was also provided.

There was also some discussion regarding the format of the project study plan, which is meant to be completed before formal initiation of the NRCA production begins, by 2/1/2016. No specific guidance as to the format for this document appeared forthcoming, but Patti Welton initiated inquiry with the NPS regional office about information that may be available. The remainder of the project timeline, as described in the project FOA, was reviewed and agreed upon.

The meeting adjourned with a much clearer understanding of biological information that may be suitable for the NRCA, and specific contacts between the assessment team and HALE staff members best suited to provide the information. Although many details remained unresolved in this initial meeting, it was decided that follow-up questions would be addressed in personal communications between the assessment team (mainly the lead, Krushelnycky) and the NPS assessment committee (mainly the Project Manager, Welton), either via phone, email, or in periodic visits to HALE conducted by Krushelnycky in the course of ongoing research.

Potential Information Sources and Notes

Birds

- Shore bird inventory (ʻOheʻo and Kāʻapahu) – I&M website
- Sea bird inventory (all birds not ʻuaʻu) –I&M website
- T&E status report for birds
- Crater: nēnē and ʻuaʻu data, anecdotal sitings of forest birds; nēnē trend data, nesting data, mortality data, nest locations; pueo incidental sitings; ʻuaʻu – lots of burrow data
- Forest bird: data from USGS forest bird transects
- Small mammals: Josh Adams doing analysis, hopefully ready by Sept 2016
- Ungulates: data on number of animals removed would need to obtained from Timmy
- Bats: anecdotal, and Forest and Kim data; USGS people doing bat work at Kahikinui (outside park); maybe mostly distributional information
- Feral chickens: absence in crater as of now.
- RBI inventories in 70's by Sheila Conant: PCSU tech reports

Inverts

- ʻOheʻo inventory
- Greg Brenner reports posted on statpros.com
- David Foote *Megalagrion* work around bogs

- *Vespula* data?

Plants

- Rare plant locality information – some opportunistic observations, no regular monitoring
- Rare plant action plans – short docs for some rare plants, not all (about 35 spp)
- Rachel Brunner's redo of Alvin Yoshinaga's plots; could possibly be analyzed
- Invasive plant control data: effort, area covered, area treated, also broken down by management units (are 43 for the park; 15 in frontcountry).
- Kīpahulu plots along Charlie down were repeated a few times, could possibly be analyzed if data could be located.

Water Resources

- Mainly I&M and USGS data, available from IRMA

Appendix B. Initial Potential Resource List

The table below shows the initial list of potential resources to be considered for the HALE NRCA, as determined by the NPS assessment committee prior to project initiation. Resources were also ranked for priority, with ranks of 1 indicating highest priority.

Table B-1. Initial Potential Resource List I: Life-Supporting Environment.

Attribute	Resource	Indicator	NRCA Priority
A. Physical and Chemical Integrity of Air Quality, Weather and Climate	Air Quality	This resource will be assessed by the NPS and included in the NRCA.	1
	Weather and Climate	Address effects on species/systems under biological integrity	3
B. Physical and Chemical Integrity of Geology and Soils	Geomorphology (Surface Processes)	Hillslope Features and Processes	3
		Coastal/Oceanographic Features and Processes (Japan tsunami debris)	3
	Geomorphology (Surface Processes)	Stream Channel Characteristics	3
	Geomorphology (Surface Processes)	Lake/Bog Features and Processes (limnology)	3
	Subsurface Geologic Processes	Cave Features and Processes (lava tubes)	3
	Subsurface Geologic Processes	Volcanic Features and Processes	2
	Soil Quality	Soil Function and Dynamics	2
C. Physical and Chemical Integrity of Water Resources	Hydrology	Surface Water Dynamics, Lakes, and Perennial and intermittent streams	3
	Hydrology	Groundwater Dynamics	3
	Hydrology	Coastal Dynamics, Storm surges, and Sea level changes	3
	Water Quality	Surface Water (lakes and streams)	2
		Groundwater	3

Table B-2. Initial Potential Resource List II: Biological Integrity.

Concern	Resource	Indicator	NRCA Priority
A. Ecological Communities of Concern	Cave Ecosystems	• Cave-dwelling invertebrates • Other fauna • Water quality • Hydrology • Vegetation	2
	Coastal Communities	• Marine animals: sea turtles, monk seals, invertebrates, shore birds • Intertidal communities • Coastal vegetation	1
	Crater Ecosystem	• Native plants/animals • Invasive species • Weather and Climate	1
	• Shrubland Communities • Front Country • Kaupo	• Plant diversity • Community richness • Native and non-native animals • Weather and climate effects • diseases, pests	1
	• Forest Communities • Low elevation (< 4000’) • High elevation (> 4000’) • Bogs	• Plant diversity • Community richness • Invasive plants and animals	1
	Sub-alpine Grassland Community	• Plant diversity and community richness • Native and non-native animals • Weather and climate effects • diseases, pests	1
	Nuu Parcel	Ecological integrity	1

Table B-2 (continued). Initial Potential Resource List II: Biological Integrity.

Concern	Resource	Indicator	NRCA Priority
A. Ecological Communities of Concern (continued)	Freshwater Communities	• Lakes and Streams • Aquatic invertebrates • Amphibians and reptiles • Fish	1
	Non-vascular Plants	• Bryophytes • Mosses • Mycorrhizae	2
B. Animal Communities of Concern	Invertebrates	• Aquatic Invertebrates/Pollinators • Terrestrial Invertebrates/Pollinators	1
	Birds	• Native and non-native birds • Shore birds (kolea) • Forest birds • Avian diseases	1
	T&E Wildlife	• Bats • Endangered Birds	1
C. Stressors	Invasive Species	• Invasive/Exotic Plants • Invasive/Exotic Animals • Mammals • Birds • Amphibians • Reptiles • Invertebrates • Insects	2
	Disease and Infestations	• Plant and Animal Diseases (avian malaria, pox) • Insect Pests	2

Table B-3. Initial Potential Resource List III: Landscape Condition Context.

Resource	Indicator	NRCA Priority
Landscape Dynamics	Fuel and Fire Dynamics	2
	• Land Cover/Use and Habitat Connectivity • Housing density • Road density • Human footprint	3
	• Natural Disturbance Regimes • tsunamis • hurricanes • drought	3
Soundscape	Natural Quiet	2
Viewscape	Dark Night Sky	2

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National Park Service
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Natural Resource Stewardship and Science
1201 Oakridge Drive, Suite 150
Fort Collins, CO 80525

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