

Chapter 5. Wildland Fires and Greenhouse Gas Emissions in Hawai'i

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5.1. Highlights

- Across the islands of Hawai'i, Maui, Moloka'i, O'ahu, and Kaua'i the annual area burned by large wildland fires (≥ 0.4 km² burned) between 2002 and 2011 averaged 48 km²/yr and ranged between 5 and 119 km²/yr.
- For the same period, greenhouse gas emissions from wildland fires averaged 0.0271 TgC/yr (0.0942 TgCO₂-eq/yr) and ranged from 0.0016 to 0.0637 TgC/yr (0.0055 to 0.2202 TgCO₂-eq/yr).
- Average annual wildland fire emissions from 2002 to 2011 were equivalent to 0.5 percent of 2011 fossil-fuel emissions in Hawai'i (5.8 TgC/yr or 21.2 TgCO₂-eq/yr).
- Most of the area burned was in grassland (51 percent) and dry shrubland (22 percent) ecosystems.
- The potential for future increases in carbon storage in Hawaiian ecosystems is limited by fire and its controls on the recovery of forest vegetation in fire-prone nonnative shrublands and grasslands and the loss of forest vegetation when fire converts these systems to nonnative shrublands and grasslands after fires.

5.2. Introduction

Restoring, protecting, and maintaining functional ecosystems that can continue to sequester CO₂ from the atmosphere offers a powerful set of tools for mitigating climate change, which, paired with policies and regulations, can serve to directly offset anthropogenic greenhouse gas emissions. The energy budgets of ecosystems are controlled by albedo, evapotranspiration, and resulting changes to solar heating and water balance. These drivers, in turn, regulate ecosystem capacity to fix and retain carbon, and so modify

atmospheric CO₂ concentrations (Bonan, 2008). The total amount of carbon entering ecosystems through the process of photosynthesis is gross primary production (GPP). The difference between GPP and carbon losses from autotrophic and heterotrophic respiration is net ecosystem production (NEP) or net ecosystem carbon balance (NECB) when additional carbon losses from disturbances and lateral fluxes are also considered (Chapin and others, 2006). When NECB is positive, carbon is sequestered from the atmosphere and stored in ecosystem carbon stocks, potentially offsetting anthropogenic greenhouse gas emissions (for example, fossil fuel combustion and other emissions).

Global estimates of greenhouse gas emissions in 2011 from fossil fuel combustion, cement production, and gas flares were 9.5±0.8 PgC/yr; forestry, wildland fires, and land-use change accounted for additional emissions of 0.9±0.8 PgC/yr from 2002 to 2011 (Intergovernmental Panel on Climate Change [IPCC], 2014). Approximately 40 percent of anthropogenic emissions from 1750 to 2011 have remained in the atmosphere where they increase atmospheric radiative forcing. The remaining 60 percent of anthropogenic emissions during this same period are thought to have been absorbed by ocean and vegetation carbon sinks (IPCC, 2014).

The amount of carbon stored and sequestered by specific ecosystem types varies widely (Whittaker and Likens, 1973; Bonan, 2008). Globally, forest ecosystems contain nearly 45 percent of all carbon stored in terrestrial ecosystems and account for nearly 50 percent of all carbon sequestered by ecosystems (Bonan, 2008). However, savannas, shrublands, and grasslands can also be highly productive (Lieth, 1973; Whittaker and Likens, 1973), and because of their large areal extent, non-forest ecosystems hold a large proportion of global carbon stocks (Bonan, 2008). For instance, temperate grasslands store 6 PgC and sequester 5.6 PgC/yr globally (Saugier and others, 2001). Among forests, tropical forests tend to store and sequester more carbon than other ecosystems (471±93 PgC and 1.02±0.47 PgC/yr, respectively [Pan and others, 2011]), even though they cover only 12 percent of the ice-free terrestrial surface area (Bonan, 2008). Boreal forests and temperate forests also have large carbon stocks (282±22.5 PgC and 118.6±6.3 PgC, respectively) and contribute substantially to global net primary production (NPP) (0.5±0.08 PgC/yr and 0.72±0.08 PgC/yr, respectively) (Pan and others, 2011). In certain cases, carbon stored in and sequestered

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by temperate forests can be as great or greater than boreal and tropical forests (Keith and others, 2009).

Hawai'i is climatically, and as a result biogeographically, very diverse because of the remarkably steep elevation and precipitation gradients created by fixed zones of uplift on windward slopes found across the main islands of the Hawaiian archipelago (Giambelluca and others, 2013). As a result of these gradients, Hawai'i provides a unique opportunity to understand the effects of climate, land-use, and land-cover change on carbon storage and flux. Land cover in Hawai'i is approximately 12 percent developed and agriculture, 19 percent barren or sparse vegetation on lava flows and alpine areas, and 69 percent natural and managed ecosystems (forests, shrublands, and grasslands). Approximately 36 percent (6,000 km²) of the land area in the main Hawaiian Islands is dominated by forest, of which 15 percent is classified as alien and (or) mixed forest, 1 percent is classified as alien tree plantation, and 20 percent is classified as native forest (Jacobi and others, this volume, chap. 2). Because these ecosystems cross wide topographic and climatic gradients, Hawai'i is home to a tremendous diversity of biome types with some ecosystems supporting high carbon densities and others capable of sequestering large quantities of carbon from the atmosphere (Aplet and Vitousek, 1994; Raich and others, 1997; Schuur and others, 2001; Ziegler, 2002; Litton and Kauffman, 2008; Asner and others, 2009; Vitousek and others, 2009).

The largest changes in aboveground carbon stocks in Hawai'i likely occurred within the past century when large areas of dry, mesic, and even wet forest were converted to ranchlands and plantation agriculture. A portion of these converted lands have since been abandoned, and the resulting nonnative shrublands and grasslands currently represent 33 percent of terrestrial ecosystem land cover. In the past three decades (1980–2010), decreases in agricultural production across Hawai'i have further increased the flammability of Hawai'i's landscapes (Trauernicht and others, 2015). These wildland fire-prone novel ecosystems greatly increase the potential for wildland fire impacts in adjacent forests, threatening the stability of carbon stocks.

For the purposes of this assessment, we define wildland fires as fires that burned wildland vegetation and were ignited by natural causes (for example, lightning and volcanoes) or by humans (arson or accidental ignitions). Globally, wildland fires are a critical driver of the carbon cycle because they consume biomass through combustion resulting in an immediate release of greenhouse gases including CO₂, CO, and CH₄ (Seiler and Crutzen, 1980). Wildland fires can also impact carbon cycling after combustion, by transferring biomass from live to dead pools, through decomposition of dead vegetation, but more so by influencing the establishment of new vegetation that can result in years to decades before carbon stocks return to pre-wildland fire conditions (Turner and others, 1998; Cleary and others, 2010; Hurteau and Brooks, 2011; Kashian and others, 2013). However, if wildland fires completely change the vegetation community that regrows in the post-wildland fire environment (for example, converting forest to grassland), then the impact on

the carbon cycle will be large and long lasting (Bachelet and others, 2001). This type of potential impact is particularly important in Hawai'i, where novel wildland fire cycles have become established with invasion by nonnative grasses that create and allow grasslands to persist, ensuring the continuation of the wildland–nonnative-grass fire cycle.

Wildland fires are thought to have been historically infrequent and to have had a small impact on most native Hawaiian ecosystems (and carbon cycling) relative to present day conditions. Before European contact, natural ignition sources were relatively rare, but Hawaiians likely used fire to clear native vegetation for agriculture or to promote growth of pili grass (*Heteropogon contortus*) which was used to thatch houses (Cuddihy and Stone, 1990). Hawai'i's native wildland fire regime was radically transformed with the introduction of wildland fire-prone and wildland fire-adapted nonnative grasses and shrubs (D'Antonio and Vitousek, 1992) and an increase in ignition sources from human activities (Trauernicht and others, 2015). Today, these nonnative shrubs and grasses and continued anthropogenic ignition sources drive a novel wildland fire disturbance regime, with a particularly large influence on carbon storage because invasion ultimately carries wildland fire that kills woody plants and consumes carbon stored in woody biomass (Ellsworth and others, 2014). The resulting nonnative grasslands have a much lower capacity to store carbon, and this limited capacity is further reduced by frequently recurring wildland fires. Nonnative grasses including *Cenchrus clandestinus* (kikuyu grass), *Cenchrus setaceus* (fountain grass), *Melinis minutiflora* (molasses grass), and *Megathyrsus maximus* (guinea grass) were originally introduced in the late 1800s and early 1990s for animal forage or as ornamentals and have since spread widely throughout dry-to-mesic and even wet areas of Hawai'i, particularly those areas that have been disturbed by human activities such as ranching and agriculture (Cuddihy and Stone, 1990). Wildland fire-adapted and -promoting nonnative shrubs include *Leucaena leucocephala* (haole koa), *Schinus terebinthifolia* (Christmas-berry), and *Ulex europaeus* (European gorse). In areas where the nonnative grasses are established, regeneration of native trees is commonly limited because of competition for limiting resources (Hughes and others, 1991; Litton and others, 2006), but also because cattle and other nonnative ungulates consume native seedlings and saplings which evolved without large mammalian herbivores present (Cole and Litton, 2014). Further, rodents like black rats reduce available seed for regeneration by consuming fruits (Shiels and Drake, 2011). Cattle grazing in Hawai'i has decreased by more than 60 percent within the past several decades (from 850,000 ha to 324,000 ha between 1960 and 2012 [Trauernicht and others, 2015]) and in the absence of grazing, grass fuel loads accumulate and the risk of severe wildland fires increases (Elmore and others, 2005; Evans and others, 2015), which, in turn, results in an additional loss of native forest and further promotes nonnative grasses (Hughes and others, 1991; Blackmore and Vitousek, 2000; Litton and others, 2006). This nonnative grass wildland fire cycle perpetuates the conditions necessary for future potential

wildland fires to spread into adjacent shrubland and forests where they can drive additional vegetation-type conversions and carbon losses to the atmosphere.

The drivers of wildland fire occurrence in Hawai‘i are different than those in other parts of the United States. Like many parts of the world, wildland fire occurrence in Hawai‘i has been linked to drought, which can intensify during El Niño events (Dolling and others, 2005; Chu and others, 2002). However, long-term records indicate that the four-fold increase in annual area burned in Hawai‘i within the past century is primarily related to human activity (Trauernicht and others, 2015). Given the scarcity of dry lightning strikes in Hawai‘i, nearly all wildland fires in Hawai‘i outside of active volcanic areas are human caused, and the introduction and expansion of wildland fire-prone plants with changes in land use has greatly increased the potential for wildland fire occurrence and spread. Based on spatial analyses of point-based wildland fire records, wildland fire ignitions are most prevalent in developed areas, indicating humans are the prime cause of wildland fire occurrence; most large wildland fires occur in nonnative shrublands and grasslands in the drier, leeward parts of the state (Trauernicht and others, 2015), indicating nonnative grasses are the prime cause of wildland fire spread.

Wildland fires are important to consider in assessments of carbon cycling in Hawai‘i because they produce greenhouse gas emissions as well as changing vegetation types, carbon-flux rates, and carbon-storage potential. The importance of wildland fire in driving land-cover conversions in Hawaiian ecosystems and the potential drivers influencing when and where wildland fires occur in Hawai‘i are well understood and documented in existing research. What is less understood are the impacts of wildland fire on carbon emissions across space and time in the Hawaiian Islands. Thus, this chapter focuses on quantifying (1) temporal patterns of area burned and greenhouse gas emissions by wildland fires for the State of Hawai‘i for the baseline period of 2002 through 2011, and (2) the implications of wildland fire-induced shifts in vegetation cover on carbon storage. Long-term effects of fires on carbon stocks in vegetation are assessed in Sleeter and others (this volume, chap. 8).

5.3. Input Data and Methods

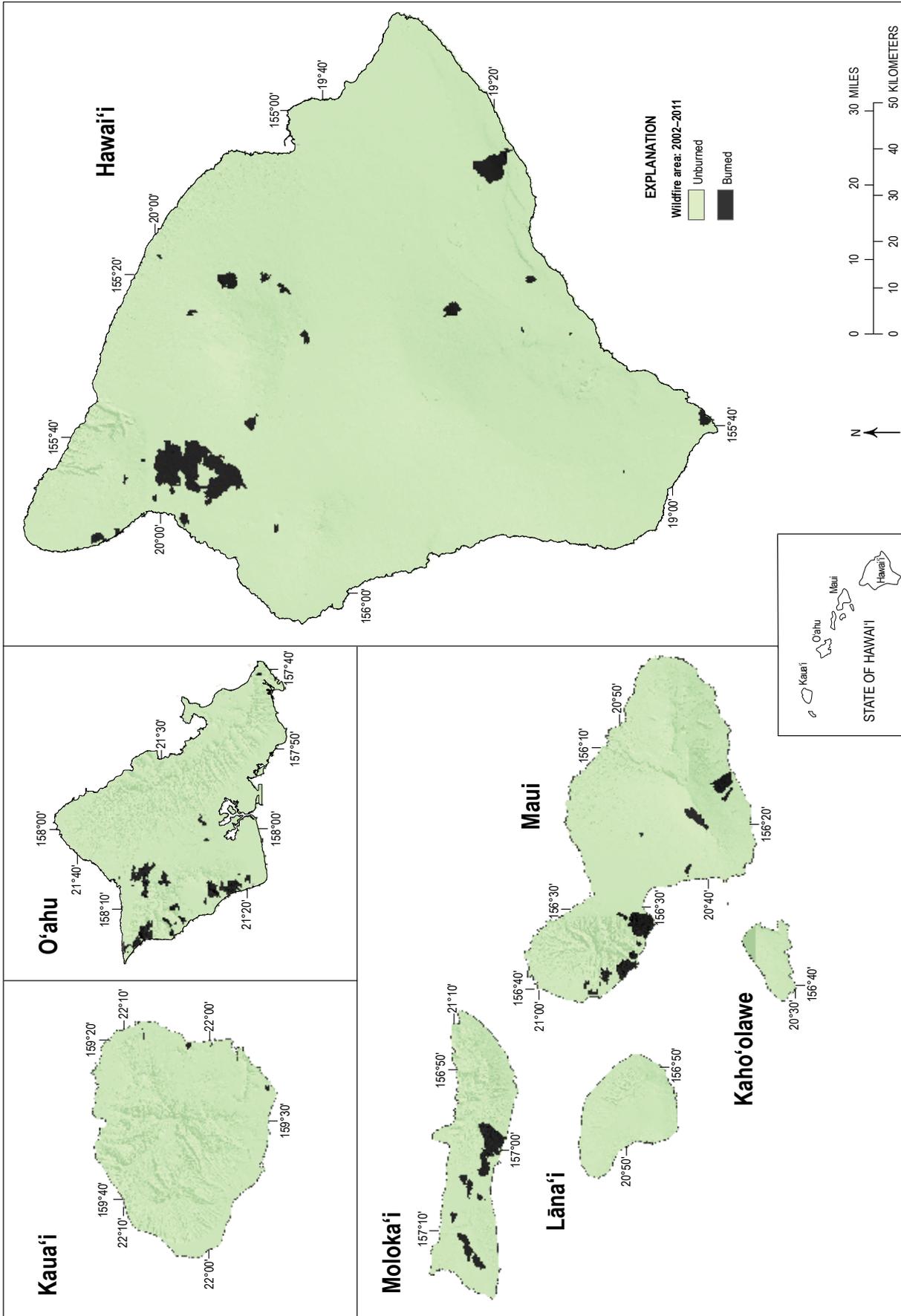
The approach for this analysis included several steps: (1) estimating annual area burned for Hawai‘i using satellite-derived data to determine the spatial extent of wildland fire incidents and to characterize patterns of burn severity, (2) developing a database of biomass loads for different vegetation types from existing studies, (3) linking values from the biomass data to a satellite-derived land-cover classification, (4) combining the raster burn-severity data with the land-cover data to estimate annual area burned by vegetation type, and (5) estimating biomass consumption and fire emissions using the combination of the burn severity with land-cover classes and their associated biomass loads. The individual steps in this analysis and the data they depended on are described in the following sections.

5.3.1. Wildland Fire Occurrence and Severity Data

Limited wildland fire perimeter data exist for the State of Hawai‘i to quantify the extent and frequency of wildland fires in different vegetation types. For example, the Monitoring Trends in Burn Severity (MTBS) database only contains 11 perimeters for Hawai‘i from the years 2002 to 2011 (Eidenshink and others, 2007), however, point locations of wildland fire ignitions are routinely recorded by State and Federal agencies (Pierce and Pickett, 2014). For the wildland fire component of this assessment, a new database was developed of burned areas with severity for wildland fires ≥ 100 acres (0.4 km²). Points of reported wildland fire locations from 2002 to 2011 were gathered from State and Federal agencies in Hawai‘i (Hawai‘i Wildfire Management Organization, 2013) and from the Short (2014) wildland fire occurrence database. These points were compared with points in the MTBS data and were removed if they had existing MTBS perimeters. This resulted in a total of 146 points, of which 135 lacked corresponding MTBS perimeters. At each point without an existing perimeter, Landsat 7 satellite imagery was visually examined for evidence of burned areas. If the burned area was visible in the Landsat imagery, burn severity was mapped following MTBS protocols (Eidenshink and others, 2007). After Landsat image interpretation, burned areas were found for 82 of the original 135 wildland fire points that did not have existing MTBS perimeters. An additional 11 wildland fires that were not reported were also found. Of the 53 wildland fire points not associated with burned areas visible in the Landsat images, 5 appeared to be duplicates. No visible burned areas were found for the other 48 points. No wildland-fire data existed for the islands of Kaho‘olawe, Lāna‘i, and Ni‘ihau, so they were excluded from this analysis. The final wildland-fire database contained perimeters and severity information for more than 104 wildland fires ≥ 100 acres (0.4 km²) between 2002 and 2011 (fig. 5.1), which formed the basis for analyses of wildland fires in this assessment.

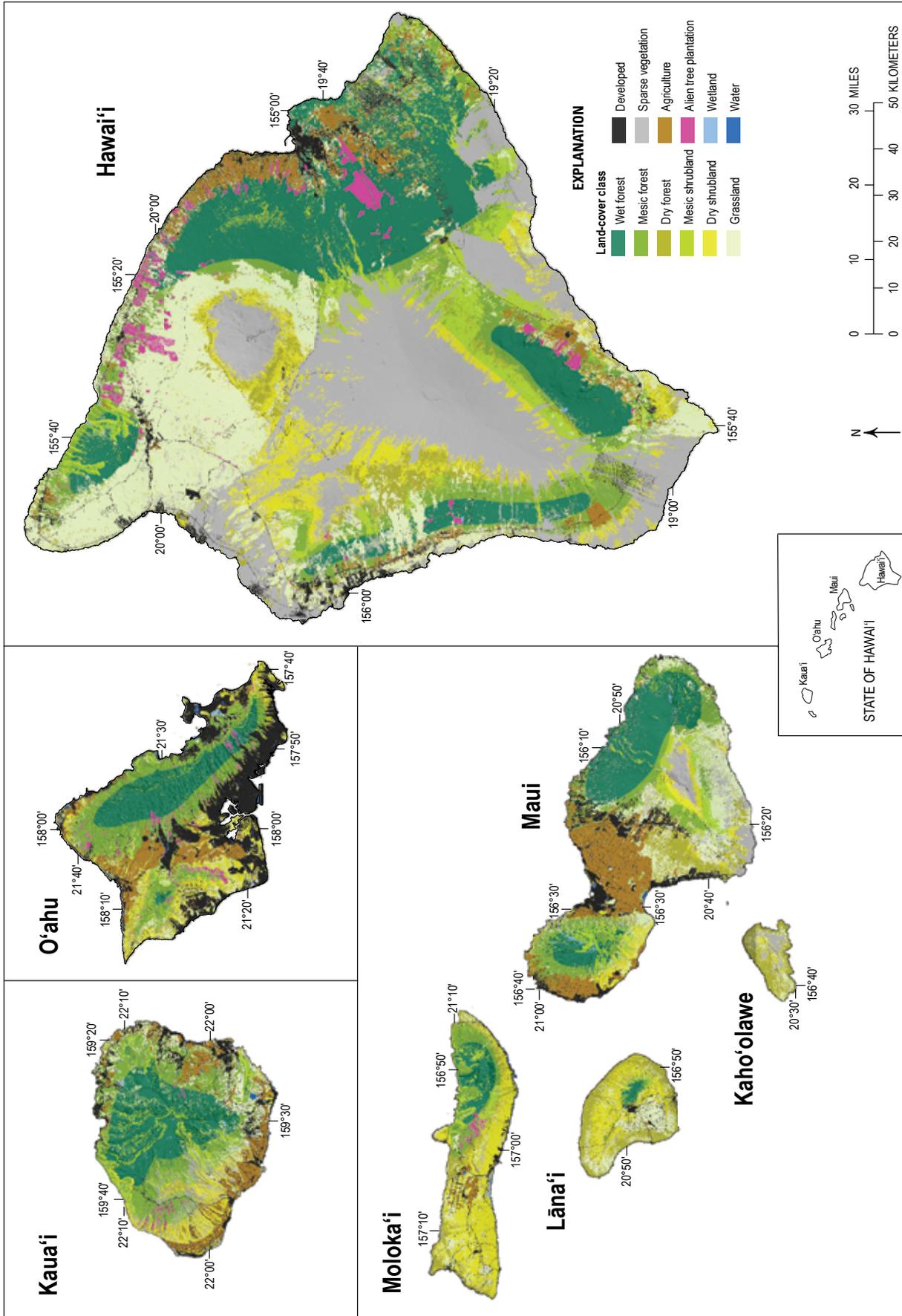
5.3.2. Land-Cover Data

Spatial data depicting vegetation types for Hawai‘i were required to estimate emissions. For this analysis and other components of the carbon assessment of Hawai‘i, a simplified version of the carbon assessment for Hawai‘i (CAH) land-cover map combined with moisture zones was used (fig. 5.2; Jacobi and others, this volume, chap. 2). These data were generated by reducing the CAH land-cover map to the following land-cover classes: forest, shrubland, grassland, wetland, developed, and agriculture. These classes were then subdivided using moisture-zone data from Jacobi and others (this volume, chap. 2). For instance, forest classes were split into dry, mesic, or wet. Classes were further refined to represent vegetation classes where enough field-based information from previous studies was available to assign biomass fuel loads (see section 5.3.3). Burn-severity information from the wildland-fire database was combined with spatial information delineating the different land-cover classes. A few burned pixels were located in anthropogenic (developed), wetland, and open-water land-cover



Shaded-relief base modified from U.S. Geological Survey National Elevation Dataset, 2015.

Figure 5.1. Maps showing burned areas of Hawai'i from 2002 to 2011. There is no wildland-fire observation data for the islands of Kaho'olawe and Lāna'i.



Shaded-relief base modified from U.S. Geological Survey National Elevation Dataset, 2015.

Figure 5.2. Maps showing land-cover classes, simplified from the carbon assessment of Hawai'i land-cover map (Jacobi and others, this volume, chap. 2) used in this assessment.

categories. This was most likely a result of errors in the land-cover classification and those burned pixels were removed prior to analysis.

5.3.3. Pre-Wildland Fire Biomass Data

Estimating biomass consumption and fire emissions required pre-wildland fire biomass levels specific to individual vegetation classes. To meet this need, a database of biomass

measurements from field data gathered across Hawai'i was compiled. Data were gathered from existing published and unpublished studies (table 5.1) with a focus on shrubland and grassland ecosystems because these systems experience most of the wildland fires, have highly variable biomass loads, and are poorly represented in existing biomass and fuels databases used for wildland fire behavior and emissions modeling (for example, LANDFIRE; Rollins, 2009). Biomass component names were used inconsistently across the different studies and required normalization to a common naming scheme. For

Table 5.1. Studies from which biomass data were compiled for this assessment.

Ainsworth, A., and Kauffman, J.B., 2010, Interactions of fire and nonnative species across an elevation/plant community gradient in Hawaii Volcanoes National Park: <i>Biotropica</i> , v. 42, no. 6, p 647–655, available at http://dx.doi.org/10.1111/j.1744-7429.2010.00636.x .
Ansari, S., Hirsh, H., and Thair, T., 2008, Removal of invasive fire-prone grasses to increase training lands in the Pacific: Department of Defense Legacy Project #07-362.
Beavers, A., Burgan, R., Fujioka, F., Laven, R., and Omi, P., 1999, Analysis of wildland fire management concerns at Makua Military Reservation: Fort Collins, Colo., Center for Environmental Management of Military Lands.
Beavers, A., Unpublished biomass data from Dole Plantation, Helemano, O'ahu.
Beavers, A., Unpublished biomass data from Keamuku, Hawai'i.
Beavers, A., Unpublished biomass data from Schofield Barracks.
Blackmore, M., and Vitousek, P.M., 2000, Cattle grazing, forest loss, and fuel loading in a dry forest ecosystem at Pu'u Wa'aWa'a Ranch, Hawai'i: <i>Biotropica</i> , v. 32, no. 4a, p. 625–632.
Castillo, J.M., Enriques, G., Nakahara, M., Weise, D., Ford, L., Moraga, R., and Vihnanek, R., 2007, Effects of cattle grazing, glyphosphate, and prescribed burning on fountain grass fuel loading in Hawai'i, in Masters, R.E., and Galley, K.E.M., eds., <i>Proceedings of the 23rd Tall Timbers Fire Ecology Conference—Fire in Grassland and Shrubland Ecosystems</i> : Tallahassee, Fla., Tall Timbers Research Station, p. 230–239. [Also available at http://talltimbers.org/tall-timbers-fire-ecology-conference-proceedings-volume-23/ .]
Ellsworth, L.M., Litton, C.M., Taylor, A.D., and Kauffman, J.B., 2013, Spatial and temporal variability of guinea grass (<i>Megathyrsus maximus</i>) fuel loads and moisture on Oahu, Hawaii: <i>International Journal of Wildland Fire</i> , v. 22, no. 8, p. 1083–1092, available at http://dx.doi.org/10.1071/WF12051 .
Freifelder, R.R., Vitousek, P.M., and D'Antonio, C.M., 1998, Microclimate change and effect on fire following forest-grass conversion in seasonally dry tropical woodland: <i>Biotropica</i> , v. 30, no. 2, p. 286–297.
Fujioka, F.M., and Fujii, D.M., 1980, Physical characteristics of selected fine fuels in Hawaii—some refinements on surface area-to-volume calculations: Berkeley, Calif., U.S. Department of Agriculture, Forest Service, Pacific Southwest Forest and Range Experiment Station, Research Note PSW-348, 7 p.
Iwashita, D.K., Litton, C.M., and Giardina, C.P., 2013, Coarse woody debris carbon storage across a mean annual temperature gradient in tropical montane wet forest: <i>Forest Ecology and Management</i> , v. 291, p. 336–343, available at http://dx.doi.org/10.1016/j.foreco.2012.11.043 .
Litton, C.M., Sandquist, D.R., and Cordell S., Unpublished biomass data for Kaupulehu Dry Forest. [See Litton and others (2006) for a description of the study site.]
Litton, C.M., Sandquist, D.R., and Cordell, S., 2006, Effects of non-native grass invasion on aboveground carbon pools and tree population structure in a tropical dry forest of Hawaii: <i>Forest Ecology and Management</i> , v. 231, no. 1–3, p. 105–113.
Litton, C.M., Ainsworth, A., and Kauffman, J.B., Unpublished biomass data for Hawaii Volcanoes National Park [See Ainsworth and Kauffman (2010) for a description of study site.]
McDaniel, S., and Ostertag, R., 2010, Strategic light manipulation as a restoration strategy to reduce alien grasses and encourage native regeneration in Hawaiian mesic forests: <i>Applied Vegetation Science</i> , v. 13, p. 280–290, available at http://dx.doi.org/10.1111/j.1654-109X.2009.01074.x .
Pierce, A., Giardina, C.P., and Litton, C.M., Unpublished biomass data for Pohakuloa Training Area, Hawaii Island.
Schuur, E.A.G., Chadwick, O.A., and Matson, P.A., 2001, Carbon cycling and soil carbon storage in mesic to wet Hawaiian montane forests: <i>Ecology</i> , v. 82, no. 11, p. 3182–3196.
Selmants, P.C., Litton, C.M., Giardina, C.P., and Asner, G.P., 2014, Ecosystem carbon storage does not vary with mean annual temperature in Hawaiian tropical montane wet forests: <i>Global Change Biology</i> , v. 20, no. 9, p. 2927–2937, available at http://dx.doi.org/10.1111/gcb.12636 .
Wright, C.S., Ottmar, R.D., Vihnanek, R.E., and Weise, D.R., 2002, Stereo Photo Series for Quantifying Natural Fuels: Grasslands, Shrubland, Woodland, and Forest Types in Hawaii: Portland, Oreg., U.S. Department of Agriculture, Forest Service Pacific Northwest Research Station General Technical Report PNW-GTR-545, 91 p.

reporting purposes in this study, biomass pools were grouped into the following categories: (1) Duff and Litter, (2) Small Downed Woody Debris (DWD—Small), (3) Large Downed Woody Debris (DWD—Large), (4) Grass and Herbaceous, and (5) Standing Live Trees. The two downed woody debris classes were defined based on the diameter of logs: large downed woody debris included any logs with 7.6-cm (3-inch) diameter and greater, everything else was considered small downed woody debris. All biomass values were converted from reported units to kilograms of carbon per square meter using a 0.47 biomass-to-carbon conversion factor (Litton and others, 2006). Ultimately, biomass data for 419 plots at 204 different sites were collected from the 19 different studies (table 5.1). Unless there was clear indication that measurements represented different vegetation types, all biomass measurements were averaged to the site level for each study.

Coordinates for individual plot locations were gathered when possible and combined with the land-cover raster data to crosswalk the plot data to different land-cover classes. Expert opinion was used to assign the plot to a land-cover class when plot locations were not available or when the vegetation composition and structure described by the study did not match the land-cover class in the raster data at the plot location. The expert-opinion vegetation-state-class assignment was based on published descriptions of the dominant vegetation types and the authors' familiarity with the vegetation at the study area where the data were collected.

Land-cover classes were further simplified because there were a limited number of plots representing certain vegetation classes in the biomass database. For a given vegetation type (for example, shrubland), plots in the biomass databases tended to fall within one invasion status or another (native versus invaded). Therefore, invasion status was removed from all land-cover classes. In general, the grassland class was primarily invaded and other vegetation classes were primarily native.

5.3.4. Biomass Consumption and Wildland Fire Emissions

Biomass consumption and emission rates were generated using the First-Order Wildland Fire Effects Model (FOFEM) (Reinhardt and others, 1997) for each vegetation class and

burn-severity category using the information on wildland fire perimeters by vegetation class and the pre-fire biomass loads described above. The FOFEM uses biomass loads along with fuel moistures to estimate the amount of litter and downed woody debris consumed during combustion (Albini and others, 1995; Albini and Reinhardt, 1995; Albini and Reinhardt, 1997). In the FOFEM, the consumption of duff (decaying litter), trees, plants, and shrubs is estimated as a function of the land cover class, season, fuel moistures, and biomass loads. Canopy fuel consumption is estimated as a function of the burn severity provided by the MTBS data: 50 percent for high severity, 30 percent for moderate severity, and 10 percent for low and very low severity. The emissions of CO₂, CO, and CH₄ were then calculated on the basis of the amount of fuel consumed, the organic-matter content of the fuel, and how efficiently it burned. The required input data for FOFEM included biomass loads, burn severity, and dead- and live-fuel moistures. To simplify the reporting of results, emission estimates were summarized for all carbon-containing constituents to units of carbon and units of CO₂-equivalents (CO₂-eq) using their 100-year global warming potentials (IPCC, 2007) and the following equation:

$$\text{CO}_2\text{-eq} = \text{CO}_2 + (2.33 \times \text{CO}) + (21.0 \times \text{CH}_4) \quad (1)$$

Three different fuel moisture levels (very low, low, and moderate) were used to generate biomass consumption and emission estimates with FOFEM, corresponding to different burn severities (high, moderate, and low; table 5.2). Variability in biomass consumption and emissions across fuel moisture levels were largely related to the proportion of canopy fuels, woody debris, and litter and duff consumed. The amount of shrub and grass fuels tended to vary little across the moisture level range because these are estimated using fixed proportions of biomass components in FOFEM. Final FOFEM-based estimates for biomass consumption and emission rates by vegetation class, burn-severity class, and biomass pools are shown in table 5.6. Values from this table were multiplied by the area burned in each vegetation class and year to produce estimates of the annual area burned and emissions. Reported results include annual area burned and emissions (total and by vegetation class).

Table 5.2. Fuel moisture levels and values used to estimate biomass consumption and emissions for different land-cover classes.

Parameter	Fuel moisture level (percent)		
	Very low	Low	Moderate
10-hour fuel moisture	4	7	10
1,000-hour fuel moisture	6	9	12
Duff fuel moisture	30	60	90
Canopy consumption	50	30	10

5.4. Results

Between 2002 and 2011, across five of the main islands of Hawai'i (excluding Kaho'olawe, Lāna'i, and Ni'ihau), the average annual area burned by large wildland fires (≥ 0.4 km² burned) was 48 km²/yr (fig. 5.3A). The amount of area that burned in any given year fluctuated widely, ranging from 5 km² in 2004 to 119 km² in 2007. The grassland vegetation class had the greatest percentage of its area burned annually (0.85 percent/yr), followed by dry shrublands (0.72 percent/yr), then dry forests (0.35 percent/yr), and mesic shrublands (0.24 percent/yr). Other vegetation classes tended to have a smaller percentage of their total area burned each year (table 5.3). The percentage of available area of a vegetation class that burned in a given year was also highly variable and that

variability tended to increase with the average annual percentage of a vegetation class that burned. For example, grasslands had the highest variability in the annual percentage of area burned and wet forests had the lowest (table 5.4). On a per-unit-area basis, forest vegetation classes contained the greatest amount of biomass (all pools combined; fig. 5.4). Wet forests had the most biomass with 11.7 kgC/m², followed by mesic forests with 10.8 kgC/m², and then dry forests with 5.2 kgC/m². Other, non-forest land-cover classes had less biomass, typically less than 1 kgC/m² (fig. 5.4). Similarly, total carbon stored in non-forest land-cover classes was low (7 percent of total carbon stocks) even though non-forest land-cover classes cover much more than 7 percent of land in Hawai'i. Forests cover 36 percent of Hawai'i land, but hold 93 percent of all carbon in terrestrial ecosystems (table 5.5).

Table 5.3. Annual percentage of area burned by vegetation class from 2002 to 2011 across the main Hawaiian Islands, excluding Kaho'olawe, Lāna'i, and Ni'ihau.

Vegetation class	Area burned (km ²)	Annual percent of area burned										
		2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	Mean
Dry forest	29.4	2.5	5.7	12.6	3.3	3.4	6.5	31.5	8.5	6.3	0.1	6.2
Mesic forest	15.5	9.4	6.0	0.0	0.1	0.1	5.3	2.3	0.5	0.4	14.3	3.2
Wet forest	16.6	9.6	14.7	0.0	0.0	0.0	0.0	0.1	0.0	0.0	23.6	3.5
Alien tree plantation	1.4	0.0	0.0	0.0	0.1	0.0	0.3	0.2	2.0	0.0	0.0	0.3
Dry shrubland	105.1	11.2	21.9	43.4	8.8	23.4	21.0	1.4	72.8	34.9	0.3	22.0
Mesic shrubland	31.0	45.5	17.9	0.0	0.2	0.0	1.9	8.6	3.7	0.9	20.1	6.5
Grassland	243.5	20.9	26.8	35.6	84.0	60.1	51.5	56.0	6.3	53.9	40.4	51.0
Sparse vegetation	35.2	0.9	6.9	8.4	3.4	13.1	13.4	0.0	6.2	3.6	1.2	7.4

Table 5.4. Annual percentage area of each vegetation class that burned from 2002 to 2011 across the main Hawaiian Islands, excluding Kaho'olawe, Lāna'i, and Ni'ihau.

Vegetation class	Total area (km ²)	Annual percent of vegetation class burned										
		2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	Mean
Dry forest	812	0.0563	0.5642	0.0769	0.4758	0.1771	0.9620	0.6969	0.4116	0.2017	0.0009	0.3624
Mesic forest	1,720	0.0985	0.2842	0.0000	0.0039	0.0026	0.3651	0.0242	0.0110	0.0061	0.1060	0.0902
Wet forest	3,221	0.0539	0.3694	0.0000	0.0001	0.0000	0.0000	0.0006	0.0000	0.0000	0.0930	0.0517
Alien tree plantation	258	0.0000	0.0066	0.0000	0.0620	0.0000	0.1421	0.0118	0.3052	0.0000	0.0000	0.0528
Dry shrubland	1,452	0.1383	1.2183	0.1475	0.7027	0.6865	1.7247	0.0171	1.9705	0.6275	0.0027	0.7236
Mesic shrubland	1,312	0.6250	1.1034	0.0000	0.0217	0.0006	0.1750	0.1179	0.1113	0.0176	0.1944	0.2367
Grassland	2,849	0.1319	0.7616	0.0618	3.4173	0.8984	2.1602	0.3536	0.0866	0.4937	0.1804	0.8545
Barren	3,280	0.0048	0.1711	0.0126	0.1195	0.1696	0.4892	0.0002	0.0740	0.0288	0.0046	0.1075

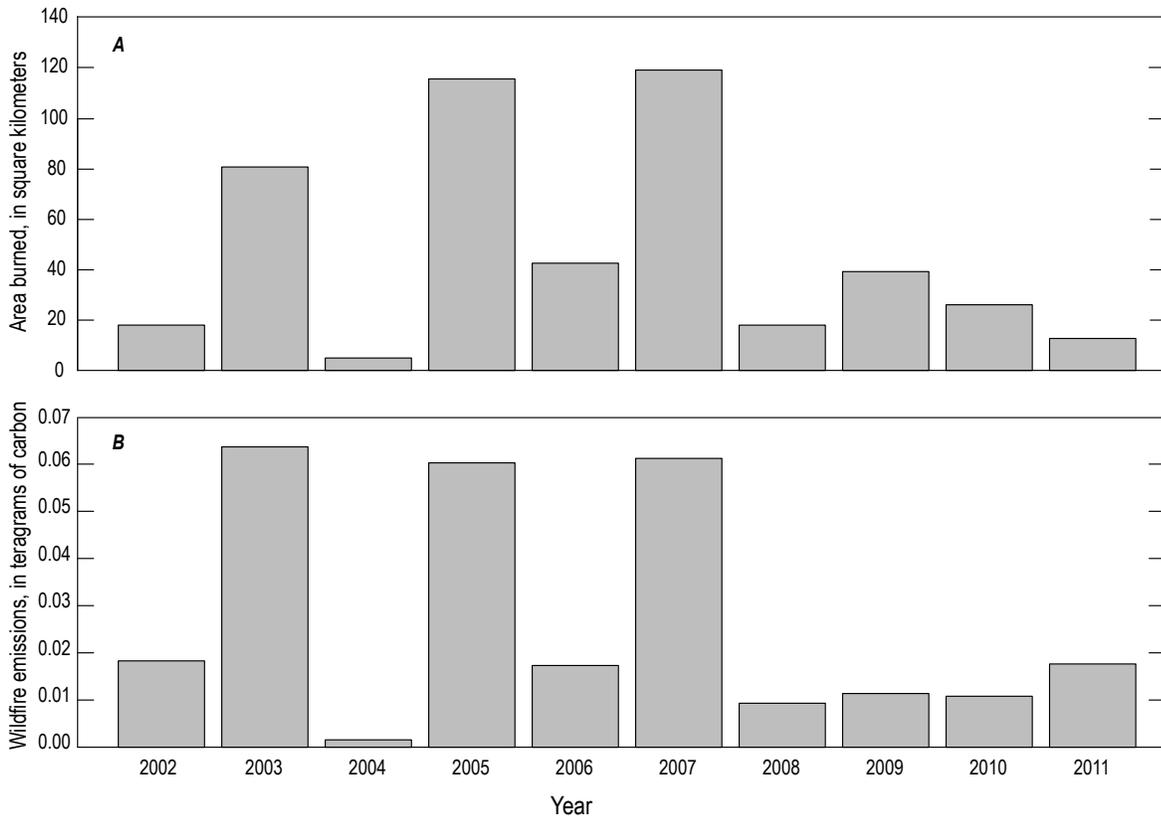


Figure 5.3. Plots of annual area burned (A) and wildland fire emissions (B) from 2002 to 2011 across all of the main Hawaiian Islands, excluding Kaho’olawe, Lāna’i, and Ni’ihau.

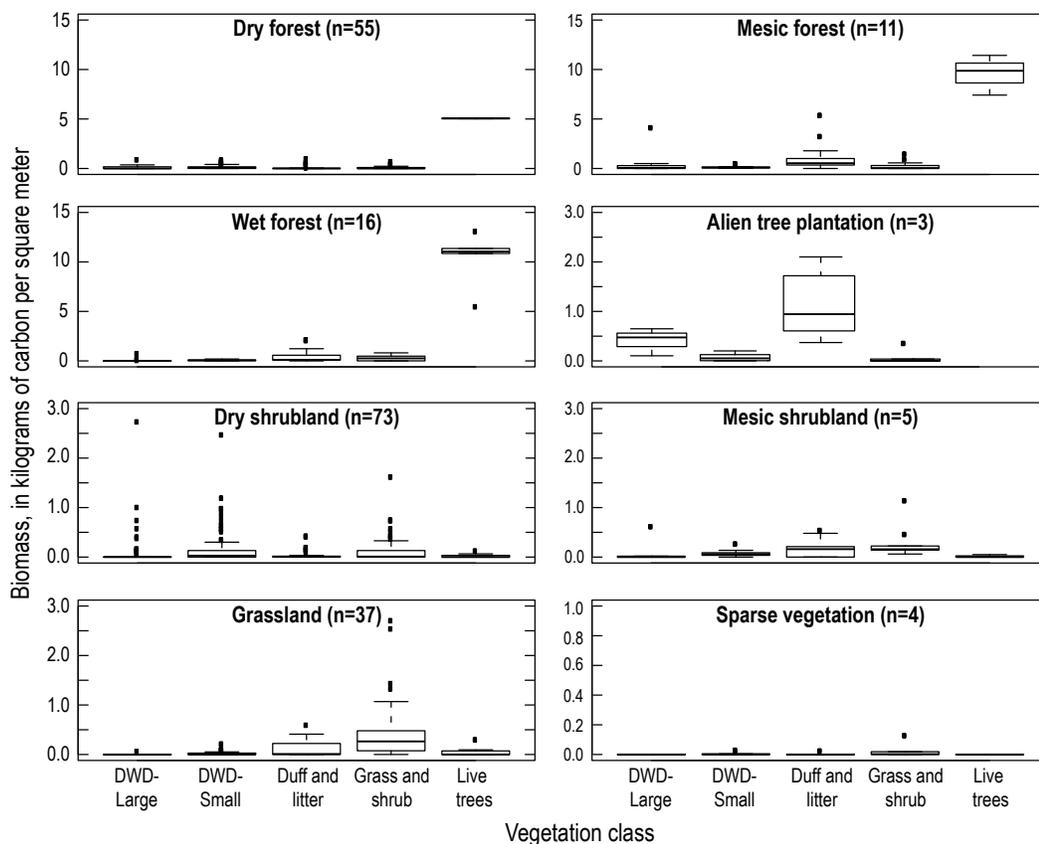


Figure 5.4. Boxplots showing the range of values for each biomass pool for the different vegetation classes of Hawai’i. The number of plots in the biomass database per vegetation class is shown in parentheses. DWD, downed woody debris.

Table 5.5. Median biomass, surface area, total biomass, and percentage of total biomass by vegetation class.[kgC/m², kilograms of carbon per square meter; TgC, teragrams of carbon]

Vegetation class	Biomass (kgC/m ²)	Area (km ²)	Total biomass (TgC)	Percent of total biomass
Dry forest	5.22	1,061	5.9	8
Mesic forest	10.83	1,640	17.3	25
Wet forest	11.71	3,039	42.0	59
Alien tree plantation	1.50	243	0.4	1
Dry shrubland	0.04	1,505	0.6	1
Mesic shrubland	0.39	1,243	1.2	2
Grassland	0.28	2,790	3.2	4
Sparse vegetation	0.00	3,155	0.1	0
Total		14,675	70.8	100

Table 5.6. Median proportion of biomass consumed by vegetation class, moisture scenario, and biomass pool.

[Data produced by the First-Order Wildland Fire Effects Model (FOFEM). AGB–LT, Aboveground biomass—live trees; AGB–GS, Aboveground biomass—shrub and grass; LDWD, Large downed woody debris; SDWD, Small downed woody debris]

Vegetation class	Moisture scenario	AGB–LT	AGB–GS	LDWD	SDWD	Duff and litter
Dry forest	Very low	0.35	1.00	0.19	0.70	0.71
	Low	0.21	1.00	0.18	0.67	0.60
	Moderate	0.07	1.00	0.16	0.65	0.45
Mesic forest	Very low	0.35	0.96	0.44	1.00	1.00
	Low	0.21	0.96	0.41	1.00	1.00
	Moderate	0.07	0.96	0.37	1.00	1.00
Wet forest	Very low	0.35	1.00	0.35	1.00	1.00
	Low	0.21	1.00	0.33	1.00	1.00
	Moderate	0.07	1.00	0.30	0.99	1.00
Dry shrubland	Very low	0.34	1.00	0.21	0.43	0.72
	Low	0.21	1.00	0.19	0.39	0.60
	Moderate	0.06	1.00	0.18	0.36	0.45
Alien tree plantation	Very low		0.80	0.60	1.00	0.81
	Low		0.80	0.53	1.00	0.72
	Moderate		0.80	0.43	1.00	0.64
Mesic shrubland	Very low	0.34	0.78	0.14	0.87	1.00
	Low	0.21	0.78	0.13	0.85	1.00
	Moderate	0.06	0.78	0.08	0.84	1.00
Grassland	Very low	0.35	1.00	0.05	0.45	1.00
	Low	0.21	1.00	0.04	0.42	1.00
	Moderate	0.07	1.00	0.04	0.39	1.00
Sparse vegetation	Very low		0.92		0.30	0.99
	Low		0.92		0.26	0.99
	Moderate		0.92		0.26	0.99

Across vegetation classes the median percentage of biomass consumed in wildland fires was 21 percent of live tree, 93 percent of shrub and grass, 25 percent of large downed woody debris, 70 percent of small downed woody debris, and 86 percent of litter and duff (table 5.6). Simulated biomass consumption was greater for the very low moisture scenario (corresponding to high burn severity) and low for the moderate moisture scenario (corresponding to moderate burn severity). Wildland fire emissions from different vegetation types were largely dependent on the amount of pre-wildland fire biomass available for consumption (fig. 5.4; table 5.7). Wildland fire emissions, averaged across moisture scenarios, were greatest for mesic forests (3.5 kgC/m² burned), followed by alien tree plantations (2.3 kgC/m² burned), wet forests (1.5 kgC/m² burned), and mesic shrublands (1.0 kgC/m² burned) (table 5.7).

Wildland fire emissions averaged 0.0271 TgC (0.0942 TgCO₂-eq/yr) across Hawai'i (excluding Kaho'olawe, Lāna'i, and Ni'ihau) between 2002 and 2011 (fig. 5.3B). The minimum and maximum annual values of carbon loss in wildland fire emissions were 2004 and 2003, respectively, and ranged from 0.0016 to 0.0637 TgC (0.0055 to 0.2202 TgCO₂-eq). The greatest amount of total emissions resulted from wildland fires in the grassland vegetation class (0.0142 TgC/yr or 0.0479 TgCO₂-eq/yr on average; table 5.8), largely because of the extensive grassland area burned each year. However, wildland fires in other land-cover classes also resulted in substantial emissions because of their high biomass loads. For instance, average annual emissions were 0.0045 TgC (0.0164 TgCO₂-eq/yr) for mesic forests, 0.0032 TgC (0.0109 TgCO₂-eq/yr) for mesic shrublands, and 0.0041 TgC (0.0079 TgCO₂-eq/yr) for wet forests.

Table 5.7. Median carbon and CO₂-equivalent emissions by vegetation class, moisture scenario, and biomass pool produced by the First-Order Wildland Fire Effects Model (FOFEM).

[C, carbon; CO₂-eq, CO₂-equivalent; gC/m², grams of carbon per square meter]

Vegetation class	Moisture scenario	C emissions (gC/m ²)	CO ₂ -eq emissions (gC/m ²)
Dry forest	Very low	149	586
	Low	144	566
	Moderate	139	545
Mesic forest	Very low	4,912	17,032
	Low	3,349	11,690
	Moderate	2,231	8,427
Wet forest	Very low	1,616	5,456
	Low	1,615	5,453
	Moderate	1,285	4,333
Alien tree plantation	Very low	2,554	10,337
	Low	2,280	9,141
	Moderate	1,993	7,887
Dry shrubland	Very low	215	919
	Low	212	856
	Moderate	208	794
Mesic shrubland	Very low	1,031	3,526
	Low	1,029	3,518
	Moderate	1,026	3,514
Grassland	Very low	584	1,969
	Low	584	1,969
	Moderate	584	1,969
Sparse vegetation	Very low	10	39
	Low	9	37
	Moderate	9	36

Table 5.8. Wildland fire emissions by land-cover class from 2002 to 2011 across all islands, except Kaho'olawe, Lāna'i, and Ni'ihau.

Vegetation class	Annual wildfire emissions (TgC)									
	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011
Dry forest	0.0001	0.0006	0.0001	0.0005	0.0002	0.0011	0.0008	0.0005	0.0002	0.0000
Mesic forest	0.0046	0.0151	0.0000	0.0002	0.0001	0.0159	0.0009	0.0004	0.0002	0.0079
Wet forest	0.0026	0.0166	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0041
Alien tree plantation	0.0000	0.0000	0.0000	0.0003	0.0000	0.0007	0.0001	0.0016	0.0000	0.0000
Dry shrubland	0.0004	0.0037	0.0004	0.0021	0.0021	0.0052	0.0001	0.0060	0.0019	0.0000
Mesic shrubland	0.0084	0.0149	0.0000	0.0003	0.0000	0.0024	0.0016	0.0015	0.0002	0.0026
Grassland	0.0022	0.0127	0.0010	0.0568	0.0149	0.0359	0.0059	0.0014	0.0082	0.0030
Barren	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
Total	0.0183	0.0637	0.0016	0.0603	0.0173	0.0612	0.0093	0.0114	0.0108	0.0176

5.5. Discussion

Based on the estimates generated for this assessment, 0.29 percent of the total land area in Hawai'i burned annually between 2002 and 2011. This percentage is slightly less than in the conterminous United States, where 0.39 percent of the total land area burned based on MTBS data for the same period (Eidenshink and others, 2007). This is lower than estimates using point-based agency records (Pierce and Pickett, 2014; Trauernicht and others, 2015) and likely reflects the fact that not all reported wildland fires were detected using remote sensing.

Average annual fossil-fuel emissions for Hawai'i (2002–2011) totaled 5.8 TgC (21.2 TgCO₂-eq) (U.S. Energy Information Administration, 2014). Most fossil-fuel emissions were from the transportation sector (51.3 percent) and electrical power generation sector (38.5 percent). Industrial, commercial, and residential sectors contributed less to the fossil-fuel emissions (8.4 percent, 1.5 percent, and 0.3 percent, respectively). In turn, the annual estimate of wildland fire emissions generated for this assessment was 0.0271 TgC (0.0942 TgCO₂-eq/yr), which amounts to only 0.5 percent of fossil-fuel emissions.

GPP of ecosystems in Hawai'i averaged 2.23 TgC/yr from 2002 to 2010 (Selmants and others, this volume, chap. 6), and wildland fire emissions in Hawai'i were equivalent to only 1.2 percent of GPP. Because this proportion is small, wildland fires did not substantially reduce the overall ability of Hawaiian ecosystems to sequester carbon for the baseline period of this assessment (2002–2011).

5.5.1. Limitations and Uncertainties

The analysis of burned area and wildland fire emissions for this chapter relied on newly developed data where burned area and severity were mapped from Landsat 7 imagery, guided by point locations for 135 known wildland fires. However, no visible burned area was observed in the Landsat 7 imagery for 48 of these

wildland fires. The reported area burned for the missing wildland fires totaled 152 km². When compared to the 480-km² total area burned of the mapped wildland fires, this suggests that the results presented here may only represent 76 percent of the total area affected by wildland fires. Other uncertainties were introduced in this analysis because small wildland fires (<0.4 km² burned) were not included, although they typically contribute little to the total area burned (Strauss and others, 1989; Stocks and others, 2002) and, consequently, emissions.

Another source of uncertainty in this analysis of wildland fire emissions for Hawai'i is the pre-fire biomass data collected for this assessment. Prior to the assessment, the only spatially complete data source representing biomass in all Hawaiian ecosystems was the LANDFIRE data layers (Rollins, 2009), and the general feeling among the scientific and management community in Hawai'i was that these data did not adequately represent fine-scale biomass values or spatial variability in biomass across ecosystem types in the state. Therefore, a database was compiled of biomass measurements from a variety of published and unpublished studies for this assessment. This effort highlighted a general lack of field-based measurements available to fully represent biomass values across the diverse climate and topographic gradients and land-use histories in Hawai'i. Light detection and ranging (lidar) data were used to generate aboveground biomass estimates for this assessment, but lack detailed information about other biomass components such as downed woody debris and litter (Selmants and others, this volume, chap. 6). However, estimates of the amount of biomass for non-forest ecosystems (shrublands and grasslands) remain low. Future efforts to understand carbon and wildland fire dynamics in Hawai'i would benefit from a large-scale systematic sampling effort to quantify biomass in shrublands and grasslands across moisture and productivity gradients and in relation to the presence of nonnative species.

Wildland fire is widely used for sugar cane agriculture in Hawai'i to remove biomass before harvest. Agricultural wildland fires were not included in this assessment because complete data characterizing their occurrence are lacking. However, based on previous studies in Brazil, sugar cane biomass can vary between 5 and 15 kg/m² and burning removes approximately 20 percent of

that biomass in the form of cane tops, leaves, and straw that are not used for sugar or energy production (Ripoli and others, 2000). Sugar cane is typically harvested on a 3-year rotation in Hawai‘i, and approximately 65.0 km²/yr of sugar cane was harvested in Hawai‘i on average from 2012 to 2014 (National Agricultural Statistics Service, 2015). Assuming that sugar cane biomass was between 5 and 15 kg/m², a 50 percent carbon content, that 20 percent of cane biomass is lost to burning, and that all the sugar cane area was burned, emissions from sugar cane burning would range between 0.03 and 0.10 TgC/yr. This is 20 percent less to 260 percent more than the average annual emissions for wildland fires and, as such, represents a potentially important component of wildland fire emissions that were not considered in this study.

Active volcanoes can also be a potential source of CO₂, SO₂, and other greenhouse gasses. Globally, CO₂ and SO₂ emissions from volcanoes are equivalent to about 0.22 percent of anthropogenic emissions (Williams and others, 1992), but volcanoes may be a substantial emissions source at local to regional scales. Volcanic emissions were also not included in this assessment because they do not represent a flux from ecosystems. However, active volcanoes in Hawai‘i are a source of wildland fire ignitions and they offer an interesting comparison with other types of fires. Gerlach and others (2002) estimated emissions from Kīlauea Volcano to be 8,500 Mg of CO₂/day from 1995 to 1998. On an annual timestep, this daily rate is equivalent to 3.1025 TgCO₂/yr, or 3,300 percent greater than the mean rate of wildland fire emissions (0.0942 TgCO₂-eq/yr) and 38 percent greater than the GPP (8.2 TgCO₂-eq/yr) estimated for this assessment.

5.5.2. Mitigating Negative Impacts of Wildland Fires on Carbon Cycling Through Aggressive Fire and Vegetation Management

Based on the summaries of the results generated for the wildland fire portion of this assessment, the immediate impacts of wildland fire on carbon cycling from wildland fire emissions is small in Hawai‘i, particularly compared to fossil-fuel emissions. However, the long-term effects of wildland fire in Hawai‘i should be considered when assessing the regional role of wildland fires in carbon storage, because the importance of fire in Hawai‘i is somewhat unique from its importance in the conterminous United States. In particular, vegetation type can convert from woody-dominated ecosystems to nonnative grassland after wildland fire in forest and shrubland ecosystems in Hawai‘i, resulting in a potentially large loss of carbon-storage and carbon-sequestration capacity. For example, Litton and others (2006) found that transformation of native Hawaiian dry forest to nonnative grasslands reduced aboveground carbon storage in live biomass by a factor of 14, with native dry forest ecosystems storing 5.12 kgC/m² and converted grasslands dominated by *C. setaceus* containing only 0.37 kgC/m².

On a landscape level, Ellsworth and others (2014) examined the rate of land-type conversion from both wildland fire occurrence and wildland fire suppression during the period from 1950 to 2010 for two military bases on the island of O‘ahu that vary in wildland fire prevention capacity:

Schofield Barracks (297 m above sea level [ASL]; mean annual precipitation [MAP], 1,000 mm; mean annual temperature [MAT], 22 °C) and Makua Military Reservation (108 m ASL; MAP, 864 mm; MAT, 23 °C). Schofield instituted improved, onsite wildland fire management in the 1990s, whereas at Makua the remoteness of the site coupled with common anthropogenic ignitions on and just outside of the base have complicated wildland fire suppression efforts. Both bases are divided into core active-training areas with abundant ignition sources and outside buffer areas with fewer ignitions. Over the 50–60-year period of the Ellsworth and others (2014) study, there were marked differences in land-cover change associated with wildland fire activity at the two bases, related to both ignition sources and wildland fire prevention activities. At Schofield, in the core area of active training (which has abundant ignition sources), recurring wildland fires expanded grassland cover from 48 to 63 percent over 61 years, with concurrent reductions in woody cover (fig. 5.5A). Outside of the wildland fire break, active wildland fire mitigation (in other words, having a wildland fire crew onsite) resulted in the opposite pattern where woody cover expanded from 79 to 98 percent during the same period (fig. 5.5B). In Makua, however, where wildland fire suppression is more challenging, there was

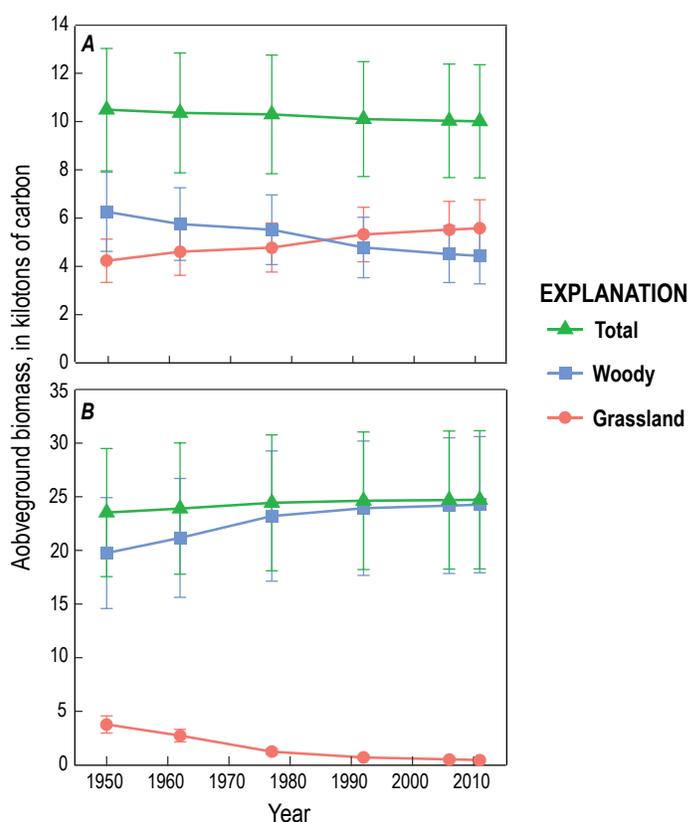


Figure 5.5. Plots of aboveground carbon partitioned into grassland and woody vegetation contained in the active wildland-fire training area at Schofield Barracks, O‘ahu, Hawai‘i (A), an area subject to frequent wildland fires, and in the watershed area outside of the training area (B), where wildland fire breaks and suppression have limited wildland fire incursion. Mean values are shown with ± 1 standard deviation error bars.

a reduction in woody cover and expansion of grassland cover over 49 years in both the area of active training (from 13 to 66 percent grassland expansion) and outside of the wildland fire break (from 30 to 41 percent grassland expansion). Conversion of forest to grassland alters fuel heights and increases modeled surface wildland fire spread and intensity, which represents not only forest loss but also a positive feedback to grassland dominance. Importantly, these conversions of woody-dominated vegetation to grasslands from wildland fire represent a loss in aboveground carbon storage of 26–41 percent (Ellsworth and others, 2014).

The results of the studies highlighted above demonstrate large-type conversions from forest to grassland where wildland fires have occurred during the past 50 or more years which represents a loss of carbon-storage and carbon-sequestration potential. On a smaller scale, they demonstrate a corresponding conversion from grassland back to forest in areas of active wildland fire suppression, which represents carbon sequestration and increased carbon storage. The results generated for this assessment suggest that an average of 62 km² of forest and 136 km² of shrubland burn each year in Hawai'i. Without active restoration efforts, based on the results highlighted above, much of that area will convert to and persist as nonnative grassland ecosystems. If wildland fires and vegetation-type conversions in Hawai'i continue in the future at current or increased rates, the potential for carbon sequestration in woody-dominated ecosystems in Hawai'i will be greatly reduced. However, these results also suggest that more active wildland fire management in the form of onsite wildland fire suppression can be an effective approach to controlling wildland fires, even allowing for type conversions back to woody ecosystems. Onsite wildland fire management will not always be feasible, however, and there is an urgent need to explore other options for limiting wildland fire spread, such as the use of greenbreaks and domestic livestock grazing in areas with little or no native species component (Evans and others, 2015).

The risks and impacts of wildland fire can be reduced through prevention, pre-wildland fire preparedness and vegetation management, increased wildland fire suppression capacity, and enhanced post-wildland fire response. Given that nearly all wildland fires in Hawai'i are human-caused, public outreach and education are critical. Local organizations, including the Hawai'i Division of Forestry and Wildlife and the Hawai'i Wildfire Management Organization are working to adapt national resources such as Ready, Set, Go! and Firewise Communities to local contexts (Hawai'i Wildfire Management Organization, 2015). Wildland fire managers in Hawai'i point to landowner preparedness and resource mapping (for example, access, water, and valued resources) as key components of pre-suppression planning (Clay Trauernicht, University of Hawai'i at Mānoa, unpublished data).

Pre-wildland fire management also includes reducing wildland fire risk directly by reducing the quantity and continuity of available fuels. Fuels reduction has been shown to reduce costs and increase effectiveness of wildland fire suppression elsewhere (Hurteau and North, 2009; Syphard and others, 2011), but “traditional” mechanical and chemical means of fuels reduction are costly to implement in Hawai'i, owing to tropical, year-long growing seasons and difficult terrain. Managed grazing has been shown to effectively reduce fine fuel loads through field assessments (Blackmore and Vitousek, 2000; Evans and others, 2015), experimental prescriptive trials (Castillo and others, 2003; Warren and others, 2007; Ansari and others, 2008), and remote sensing (Elmore and others, 2005). Land managers are also experimenting with vegetated fuel breaks, or ‘greenstrips,’ that integrate wildland fire-resistant shrubs and grasses and (or) use trees to reduce herbaceous fuels by shading (Trauernicht and others, 2012). However, plant-species suitability and the effectiveness of greenstrip implementation in Hawai'i is only beginning to be quantified (Ellsworth, 2012; Ellsworth and others, 2015).

Wildland fire suppression in Hawai'i is challenging owing to completely novel fuels, high spatial and temporal environmental variability, and complex, challenging terrain. For example, because of limited access and dangerous conditions, suppression efforts heavily rely on helicopter support, which substantially increases suppression costs. Hawai'i's geography also prevents the rapid mobilization of heavy equipment like water trucks, brush trucks, and dozers among islands, although personnel are typically mobilized in the event of large wildland fires. Under these constraints, County, State and Federal wildland fire response agencies in the state cooperate frequently on wildland fire incidents and local incident commanders stress the importance of both personal relationships and mutual-aid agreements. The top recommendations from Incident Commanders in Hawai'i for improving suppression efforts include increased training, personnel, and equipment as well as the extent of pre-wildland fire mitigation efforts (Clay Trauernicht, University of Hawai'i at Mānoa, unpublished data).

Wildland fire impacts, such as the conversion of forest to grassland, can also be mitigated by increasing resources for, and effectiveness of, post-wildland fire assessment and response in Hawai'i. Rapid response measures, such as soil stabilization and weed management, and longer term revegetation and restoration are currently limited by funding and the availability of materials, especially native seeds. Woody plant recovery in burned areas is typically by nonnative species and, therefore, establishing and maintaining native diversity will require more intensive management and landscape-scale approaches to reforestation and restoration (Loh and others, 2009; Friday and others, 2015).

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