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Chapter 11:

Fire and Nonnative Invasive Plants in the Hawaiian Islands Bioregion

Introduction

The Hawaiian Islands are national and global treasures of biological diversity. As the most isolated archipelago on earth, 90 percent of Hawai`i's 10,000 native species are endemic (Gagne and Cuddihy 1999). The broad range of elevation and climate found in the Hawaiian Islands supports a range of ecosystems encompassing deserts, rain forests and alpine communities often within the span of less than 30 miles. Recent analyses suggest that species diversity may not differ between island and continental ecosystems once habitat area is taken into account (Lonsdale 1999); however there are a disproportionate number of threats to Hawai`i's ecosystems.

Invasion of nonnative species is a leading cause of loss of biodiversity and species extinctions in Hawai`i (Loope 1998, 2004). On average, islands have about twice as many nonnative plant species as comparable mainland habitats (Sax and others 2002) and the number of plant invasions in Hawai`i is great. Over 8,000 plant species and cultivars have been introduced to Hawai`i and over 1,000 of these are reproducing on their own. Many watersheds in Hawai`i are now dominated by nonnative invasive species (Gagne and Cuddihy 1999).

Available evidence suggests that fire was infrequent in Hawaiian lowlands before human settlement, increasing in frequency with Polynesian and European colonization, the introduction and spread of invasive species, and most recently, the cessation of grazing by feral and domestic ungulates. Currently, most fires in the Hawaiian Islands occur in lowlands, communities that have been modified the most by fire.

Our understanding of fire ecology in Hawaiian ecosystems is limited by the relative scarcity of published information regarding fire and its effects on the native biota (but see reviews by Mueller-Dombois 1981; Smith and Tunison 1992; Tunison and others 2001). While anecdotal information is available, detailed studies are few and very little is known about the long term consequences of fire for Hawaiian ecosystems. But available evidence indicates that the altered fuel characteristics of many communities resulting from invasion by introduced grasses in Hawai`i and elsewhere in the tropics, coupled with an increase in ignitions, often results in frequent fires with severe consequences to the native biota

(D'Antonio and Vitousek 1992; Mueller-Dombois 2001; Mueller-Dombois and Goldhammer 1990).

In the following sections we review the interactions of fire and invasive species within grasslands, shrublands, woodlands, and forests of lowland and upland environments in Hawai`i (fig. 11-1; table 11-1). Where possible, distinctions are made between dry and mesic communities. Wet forests are discussed separately. For each environment we describe the affected vegetative communities and discuss how native fire regimes and native ecosystems have been affected by nonnative species invasions. Where information is available, we also discuss potential opportunities for the use of prescribed fire to manage invasive plants and restore native plant communities. We conclude with a summary of the current state of fire and invasive species in Hawai`i, the outlook for the future, and needs for further investigation.

Description of Dry and Mesic Grassland, Shrubland, Woodland, and Forest

Lowlands

Low-elevation forests, woodlands, shrublands, and grasslands generally occur above the salt spray zone and below 4,000 feet (1,200 m) in elevation and on a variety of soils ranging from relatively undeveloped, shallow soils to deeply weathered lava flows. The environment is dry to mesic, depending on soil type and precipitation, with annual rainfall from 4 to 50 inches (100 to 1,270 mm). Summer months are typically drier, but rainfall is variable and episodic both within and among years. Rainfall is less variable for mesic communities (Gagne and Cuddihy 1999).

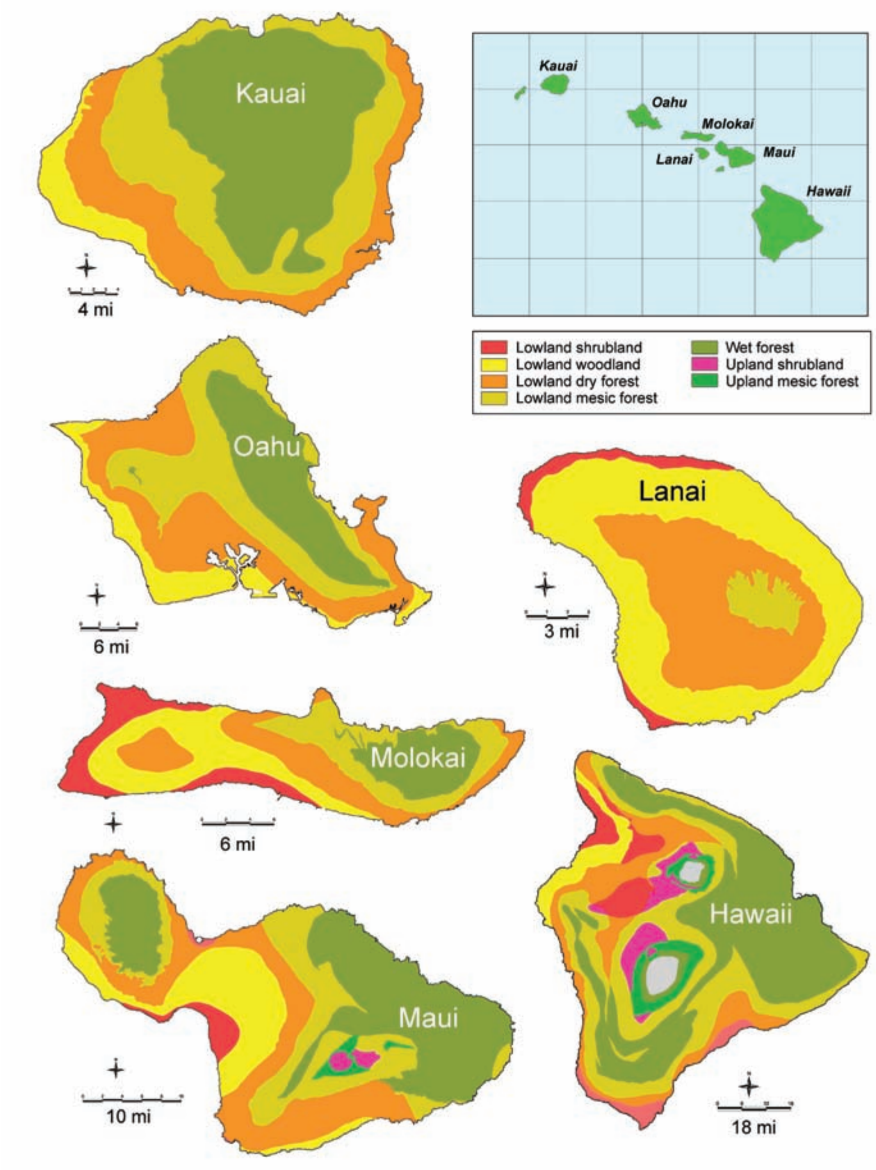

Figure 11-1—General distribution of shrublands, woodlands, and forests of lowland and upland environments in Hawai`i. No grasslands are shown on this map, although limited areas of native grasslands and extensive areas of nonnative grasslands are found in Hawai`i today. Submontane `ōhi`a woodlands are not differentiated and are lumped with mesic forest types. Map units shown here are based on the Bioclimatic Life Zone Maps (LZM) of Potential Natural Vegetation in Hawai`i (Tosi and others 2002), with modifications by Thomas Cole, Institute of Pacific Islands Forestry, USDA Forest Service, Hilo, Hawai`i.

Table 11-1—Principal nonnative species influenced by fire or influencing fire behavior, and their estimated threat potential in major plant communities in Hawai`i. L= low threat; H = high threat; PH = potentially high threat. Empty boxes reflect either a lack of information or no perceived threat.

		Community types								
		Lowland dry & mesic				Wet	Upland dry & mesic			
							Montane		Subalpine	
		Grassland	Shrubland	Forest	Woodland[a]	Forest	Shrubland	Forest	Shrubland	Forest
		Elevation range (feet)								
Scientific name[b]	**Common name**	- - - - - - - - up to 1,500 - - - - - - -			1,000-4,000	300-6,600	- - - 3,800-9,500 - - -		- - - 6,500-9,500 - - -	
Acacia farnesiana	Klu			PH						
Andropogon virginicus	Broomsedge	PH	H	H	H	PH	L	L		
Anthoxanthum odoratum	Sweet vernal grass							L	PH	PH
Ehrharta stipoides	Meadow ricegrass						PH	PH		
Festuca rubra	Red fescue								PH	
Holcus lanatus	Velvetgrass								H	H
Hyparrhenia rufa	Thatching grass	PH								
Leucaena leucocephala	Koa haole			PH	PH					
Melinis minutiflora	Molasses grass		PH		H					
Melinis repens	Natal redtop	H	H							
Morella faya	Firetree		PH	PH	PH	PH				
Nephrolepis multiflora	Asian sword fern				H	H				
Paspalum conjugatum	Hilo grass					H				
Pennisetum ciliare	Buffelgrass	PH	H	PH	PH					
Pennisetum clandestinum	Kikuyu grass				L			L		
Pennisetum setaceum	Fountain grass	H	H	H	H		PH	PH		PH
Prosopis pallida	Kiawe			PH	PH					
Schizachyrium condensatum	Bush beardgrass	PH	H	H	H		L	L		

[a] Includes submontane `ohi`a woodland and kiawe/koa haole woodlands.
[b] Scientific nomenclature follows Wagner and others (1999) and Palmer (2003).

Lowland dry and mesic forests consist of an array of communities with diverse canopy tree species, but many lowland forested areas are now dominated by introduced tree species, and remnants of native lowland forest communities remain primarily where human disturbances have been limited. Most native tree canopies are relatively open when compared to other tropical areas. Dominant native tree species across this broad spectrum of habitats include `ōhi`a (*Metrosideros polymorpha)*, wiliwili (*Erythrina sandwicensis*), lama (*Diospyros sandwicensis*), olopua (*Nestegis sandwicensis*), sandalwood (*Santalum freycinetianum*), koa (*Acacia koa*), and āulu (*Sapindus oahuensis*). Associated understory shrub species include many of the same species found in native shrublands described below (Gagne and Cuddihy 1999). `Ōhi`a forests are widespread and comprise 80 percent of all Hawai`i's native forests. `Ōhi`a occurs in nearly monotypic stands with closed to open canopies or as scattered trees (fig. 11-2), creating a gradient of communities that includes closed forests, woodlands, and savannas (D'Antonio and others 2000; Hughes and others 1991; Tunison and others 1995; Tunison and others 2001). Distinctions among types are often unclear, and most dry and mesic `ōhi`a forests have relatively open canopies; therefore, `ōhi`a community types will be referred to as "`ōhi`a woodlands." The composition of nonnative lowland forests is variable and dominated by such species as kiawe (*Prosopis pallida*), common guava (*Psidium guajava*), christmas berry (*Schinus terebinthifolius*) ironwood (*Casuarina equisetifolia*) and silk oak (*Grevillea robusta*) (Gagne and Cuddihy 1999). Kiawe and koa haole (*Leucaena leucocephala*) can form stands ranging from dense thickets to open woodlands, with an understory of introduced, drought resistant grasses.

Dry and mesic shrublands have been altered by grazing and fire and many have been replaced by nonnative communities. Today, native dry shrub communities in Hawai`i occur on a variety of soil types in leeward areas of most Hawaiian Islands. Shrub canopies are relatively open and consist of two or more dominant, widespread native species, including `a`ali`i (*Dodonaea viscosa*), `akia (*Wikstroemia* spp.), pūkiawe (*Styphelia tameiameiae*), and `ulei (*Osteomeles anthyllidifolia*). These shrub species often co-occur with an understory of nonnative grasses (Cuddihy and Stone 1990; Gagne and Cuddihy 1999) (fig. 11-3). The nonnative koa haole (*Leucaena leucocephala*) forms dense shrublands in coastal areas throughout the islands. Today native mesic shrublands occur in marginal habitats, such as on thin soils and exposed aspects, and consist of open- to closed-canopy communities with few understory herbs and shrubs up to 10 feet (3 m) tall. Dominants include: ohi'a pūkiawe, `a`ali`i, `ulei, and iliau (*Wilkesia* spp.) (Gagne and Cuddihy 1999).

Most dry and mesic lowland grasslands are floristically simple and likely anthropogenic. Gagne and Cuddihy (1999) recognize five dry, low-elevation grassland communities and three mesic lowland grasslands. Only two of these communities are dominated by native grasses: dry pili grasslands (*Heteropogon contortus*) and mesic kawelu grasslands (*Eragrostis variabilis*). Extensive pili grasslands are thought to have been maintained by burning in the lowlands prior to the 1700s (Daehler and Carino 1998; Kirch 1982) but they are now limited to small, scattered remnants (Gagne and Cuddihy 1999). The remaining grassland types are a result of past disturbances, including grazing, browsing, and fire, and are now dominated by one or two of

Figure 11-2—`Ōhi`a woodland with native shrub and uluhe (*Dicranopteris linearis*) understory invaded by nonnative grasses in Hawai`i Volcanoes National Park. (Photo by A. LaRosa.)

Figure 11-3—Low elevation dry shrublands dominated by `a`ali`i and nonnative grasses in Hawai`i Volcanoes National Park. (Photo by B. Kauffman.)

the many nonnative invasive grasses (fig. 11-4) that occur in these communities and fuel most of the fires occurring in Hawai`i today (table 11-2).

Montane and Subalpine

Montane and subalpine communities occur on the leeward slopes of Mauna Loa, Mauna Kea and Hualalai on Hawai`i Island and Haleakala, on Maui, between approximately 3,800 and 9,500 feet (1,500 to 3,000 m), as well as at higher elevations above the tradewind inversion layer (Gagne and Cuddihy 1999). Rainfall varies from 12 to 80 inches (300 to 1,970 mm) per year and is seasonal, coming largely during winter cyclonic storms. Summers are typically dry. Soils throughout the zone are either cinder, loams derived from volcanic ash, or thin soils over pahoehoe lava (Gagne and Cuddihy 1999).

Open- to closed-canopy shrubland communities of montane and subalpine areas are dominated by drought-tolerant native shrubs, including pūkiawe, `ōhelo (*Vaccinium calycinum*), and ‘a`ali`i. The `ama`u fern (*Sadleria cyatheoides*) is also abundant at higher elevations on Maui. Shrubs often occur in a matrix of native grasses such as alpine hairgrass (*Deschampsia nubigena*) or hardstem lovegrass (*Eragrostis atropioides*). Invasive nonnative grasses, such as velvetgrass (*Holcus lanatus*), sweet vernal grass (*Anthoxanthum odoratum*), and dallis grass (*Paspalum dilatatum*) are locally abundant. Montane and subalpine shrublands on Mauna Loa are dominated by the native shrubs `a`ali`i or naio (*Myoporum sandwicense*) with lesser amounts of ko`oko`olau (*Bidens menziesii*), `aheahea (*Chenopodium oahuense*) and `akoko (*Chamaesyce multiformis*). Grasses include hardstem lovegrass and nonnative fountain grass (*Pennisetum setaceum*) (Gagne and Cuddihy 1999).

Figure 11-4—Dense fountain grass (*Pennisetum setaceum*) on the leeward slopes of Hualalai, Hawai`i. (Photo by C. Litton.)

Table 11-2—Nonnative species that are primary carriers of fire in Hawaiian ecosystems. All are drought tolerant, perennial, C4 grasses[a].

Common name	Origin	Habit	Date[b] of introduction	Elevational range (feet)
Broomsedge	Eastern North America	Bunchgrass	1924(c)	150 to 4,000
Molasses grass	Africa	Mat forming	1914(c)	400 to 4,000
Buffelgrass	Africa & Tropical Asia	Mat forming	1932(c)	0 to 400
Fountain grass	North Africa	Bunchgrass	1914	0 to 7,000
Bush beardgrass	Tropical/subtropical America	Bunchgrass	1932	650 to 4,500

[a] As reported in the literature.
[b] (c) = Date first collected, otherwise date first reported.

Many areas of montane open and closed forests on Hawai`i contain nearly monotypic stands of koa with a continuous understory of invasive nonnative pasture grasses including meadow ricegrass (*Ehrharta stipoides*) (fig. 11-5), dallis grass, and Kikuyu grass (*Pennisetum clandestinum*). Native brackenfern (*Pteridium acquilium)* is found in the understory of some forests. Subalpine forests are open-canopied, with either `ōhi`a or mamane (*Sophora chrysophylla*) as the dominant species. A mamane-naio closed forest subtype is also present in some areas (Gagne and Cuddihy 1999). Many upper-elevation koa forests were logged beginning in the mid 19th century and then grazed by domestic and wild cattle. Other forests were subject to uncontrolled grazing and browsing by feral animals (cattle, goats, sheep, and pigs) during this time period (Cuddihy and Stone 1990).

Figure 11-5—Montane koa forest on Mauna Loa invaded by meadow ricegrass. (Photo by P. Scowcroft.)

Fire History and the Role of Nonnative Invasives on Fire Regimes in Dry and Mesic Ecosystems

Fires of volcanic origin occurred in Hawaiian forests prior to human habitation and continue today, although volcanism is intermittent and highly localized (fig. 11-6). While the lack of annual growth rings in tropical trees precludes use of dendrochronology for detailed analysis of early fire history, some data from sediment cores and evidence from charcoals in soils exist. Palynological studies from a high elevation bog on Maui indicates that fires occurred infrequently in forested areas in both dry and wet periods, during the last 9,000 years, and were probably volcanic in origin (Burney and others 1995). Soils from mesic and wet forested areas on Mauna Loa show charcoal from 2,080, 1,040 and 340 years ago, consistent with a volcanic origin of most fires and a long fire-return interval (Mueller-Dombois 1981). Charcoal has also been found, but not dated, from montane forest soils on Mauna Kea (Wakida 1997), indicating the occurrence of fire but not the nature of the fire regime in that area. Some prehistoric fires may also have started from occasional lightning strikes (Tunison and Leialoha 1988; Vogl 1969).

Several authors suggest that presettlement fires in Hawai`i were so infrequent (in other words, long fire-return intervals) that they had little effect on the evolution of the Hawaiian flora (Mueller-Dombois 1981, 2001; Smith and Tunison 1992) and most native plant species do not exhibit specific fire adaptations (for example, thick bark or serotiny). However, Mueller-Dombois (1981) also suggests that fire was an important ecological and evolutionary factor in upland communities because of prevailing dry conditions in leeward and high-elevation environments, the abundance and continuity of fine fuels (native grasses), and the ability of the common native species in these communities, such as koa, mamane, and bracken fern, to regenerate after fire.

Prehistoric Polynesians dramatically altered vegetation in Hawai`i's lowlands when they burned to clear land for agriculture. Reviewing archeological and palynological studies, Kirch (1982) and Cuddihy and Stone (1990) outline the general impacts of early Polynesians, including those from burning, on Hawaiian ecosystems. Slash and burn agriculture is used commonly in the

Figure 11-6—Lava ignited fire in Hawai`i Volcanoes National Park, 2002. (Photo courtesy of Hawai`i Volcanoes National Park.)

tropics (Bartlett 1956), and Polynesians burned to clear vegetation since they first occupied Hawai`i around 400 AD (Cuddihy and Stone 1990; Kirch 1982). During the early years of settlement, population levels were low and fire was probably used occasionally to clear shrublands and forests for cultivation. A population spike between 1100 and 1650 AD led to large-scale expansion of agriculture throughout much of Hawai`i. Kirch (1982) presents observational and physical evidence from several sources that suggests widespread burning of lowland areas by early Polynesians to clear land for shifting agriculture, maintain grasslands to provide pili grass thatch for shelter and, according to McEldowney (1979), promote the growth of ferns and other plants used for famine food and pig fodder. Montane and subalpine areas were not cultivated by early Polynesians and early anthropogenic fire was improbable there (Kirch 1982).

Fire size and frequency increased in the lowlands following European contact (beginning ~200 YBP). Sediment cores from a high-elevation bog on Maui suggest an appreciable increase in fire frequency during the last 200 years (Burney and others 1995). The introduction and spread of ungulates and nonnative pasture grasses accompanied European settlement and transformed some Hawaiian native forest and shrubland ecosystems into mixed communities with a significant herbaceous component (Cuddihy and Stone 1990; Gagne and Cuddihy 1999), which increased fuel loading and continuity (D'Antonio and Vitousek 1992; Hughes and Vitousek 1993). Pollen records from the last 200 years contained a high proportion of grasses and charcoal derived from grasses. However, the charcoal may have come from burning sugar cane (a common practice) rather than wildfires (Burney and others 1995).

Changes in fuels, in conjunction with increases in human-caused ignitions, contributed to the sizeable increase in fire observed in the 20th century in many ecosystems. Throughout the state, the average acreage burned increased five-fold and the average number of fires increased six-fold from the early (1904 to 1939) to the mid (1940 to 1976) part of the 20th century (table 11-3) (Cuddihy and Stone 1990). A similar, but more pronounced, trend occurred in the lowlands of Hawai`i Volcanoes National Park (HVNP); fires were three times more frequent and 60 times larger, on average, from the late 1960s to 1995 when compared to the period 1934 to the late 1960s (Tunison and others 2001). The increase in fire frequency and size in HVNP coincides with a period of increased volcanic activity from 1969 to the present from Mauna Ulu and Pu`u O`o flows, the establishment and spread of nonnative invasive grasses, and the removal of nonnative goats from HVNP in the late 1960s (Hughes and Vitousek 1993; Tunison and others 2001). Over 90 percent of all fires in the lowlands of HVNP occurred after taller, fire-adapted, nonnative perennial grasses replaced nonnative short-stature annual and perennial grasses following goat removal (Tunison and others 1994; Williams 1990).

The existence of a grass/fire cycle fueled by nonnative invasive species has been well established in some of Hawai`i's `ōhi`a woodlands (fig. 11-7). There was a dramatic increase in fire size and frequency in `ōhi`a woodlands within HVNP when compared to earlier in the century: annual fire frequency increased more than 3-fold and annual fire size more than 100-fold (table 11-3) (Tunison and others 1995). Prior to the invasion of nonnative grasses, `ōhi`a woodlands consisted of open stands of shrubs and `ōhi`a with few native grasses (Mueller-Dombois 1976) and the discontinuous surface fine fuels would rarely have carried fire. Now, nonnative broomsedge (*Andropogon virginicus*), bush beardgrass (*Schizachryium condensatum*) and molasses grass (*Melinis minutiflora*) constitute over 30

Table 11-3—Increases in the size and number of fires in Hawai`i from the early to latter part of the 20th century.

Location	Time period	Average area burned [a] (acres)	Average number of fires/year [a]
State of Hawai`i[b]	1904 to 1939	1,044	4
	1940 to 1976	5,740	24
Hawai`i Volcanoes National Park, `ōhi`a woodlands[c]	1920 to ~1970	< 2.5	11
	~1970 to 1995	430	39

[a] Data are from historic fire records available for the period 1904 to 1995.
[b] Data from Cuddihy and Stone (1990).
[c] Data from Tunison and others (1995).

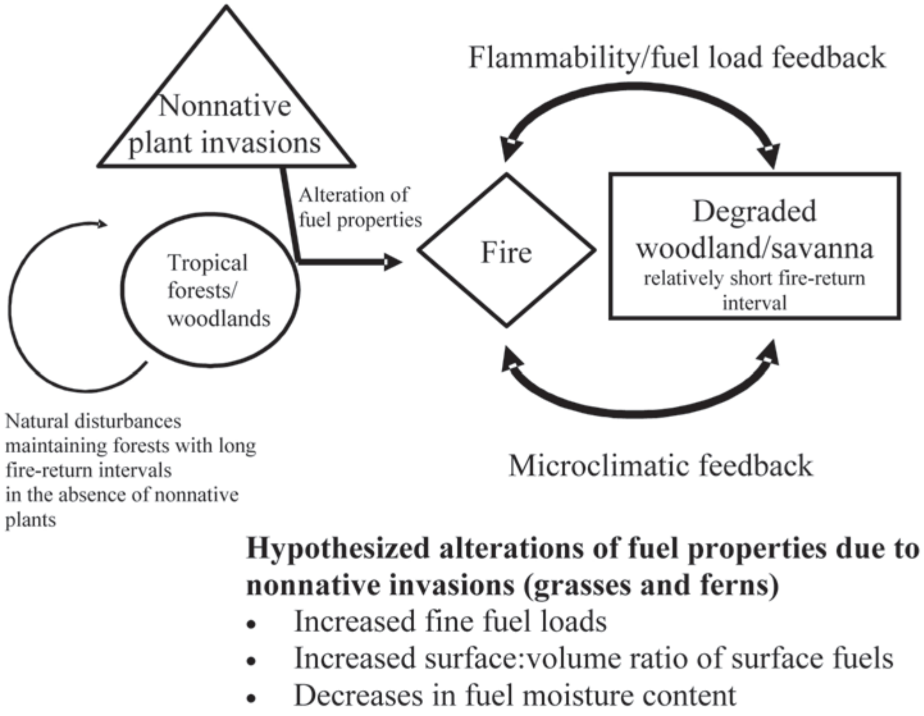

Figure 11-7—The potential effects of grass and fern invasions into Hawaiian forest and woodland ecosystems. (Modified from D'Antonio and Vitousek 1992.)

percent of the understory biomass and 60 to 80 percent of the understory cover in HVNP's `ōhi`a woodlands (D'Antonio and others 1998), forming a continuous matrix of fine fuel between native shrubs (fig. 11-8) (D`Antonio and others 2000; Hughes and others 1991; Tunison and others 1995, 2001). Similarly, fountain grass initially invaded lama forests without the aid of fire, and it now forms a nearly continuous layer of surface fuels in many areas of former lama forest (Cordell and others 2004). Introduced Asian sword fern (*Nephrolepis multiflora*) also invades mesic `ōhi`a woodlands and forms a continuous understory of fine fuels (fig. 11-9) that carry fire (Ainsworth and others 2005).

These nonnative species possess characteristics that facilitate fire spread, including a high standing biomass and a high dead-to-live biomass ratio throughout most of the year. All recover rapidly after fire with increased vigor, by resprouting or seedling recruitment (Ainsworth and others 2005; D'Antonio and others 2000). Burned sites are predisposed to more severe and repeated fires due to increased fuel loads and higher wind speeds in the more open postfire savannas compared to unburned areas (D'Antonio and Vitousek 1992; Freifelder and others 1998; Tunison and others 1995). This invasive plant-fire cycle can be persistent (Hughes and others 1991; Tunison and

Figure 11-8—Principal nonnative grass fuels in `ōhi`a woodland: broomsedge (left), bush beardgrass (middle), and molasses grass (right). (Photos by A. LaRosa.)

Figure 11-9—Submontane `ōhi`a woodland with understory of nonnative Asian sword fern. (Photo courtesy of Hawai`i Volcanoes National Park.)

others 1995) and may represent a long lasting shift in plant community composition (fig. 11-7).

We have no information on the impacts of lowland nonnative woody species such as kiawe, koa haole, and klu (*Acacia farnesiana*) on fuel characteristics and fire regimes of other invaded lowland forest communities in Hawai`i. These species are legumes capable of symbiotic nitrogen-fixation. Kiawe stands ranging in height from 6 to 30 feet (2 to 10 m) are now prevalent in many dry lowland areas, forming grass and tree mosaics ranging from dense forest to open woodlands and savannas. Koa haole and klu often occur within or adjacent to kiawe stands. Increases in fire frequency, size, and severity are possible where invasions of these species result in increased aboveground biomass.

Historic fire records are available but inadequate to characterize changes in the fire regimes of montane and subalpine habitats, where historic disturbances have been fewer and native grasses are significant components of some plant communities (Loope and others 1990; Wakida 1997). Most documented high-elevation fires were small and of probable or known human origin; lightning fires occurred rarely (Loope and others 1990; Wakida 1997). Over 40 percent of fires in and around the Mauna Kea Forest Reserve (MKFR) on Hawai`i, were less than 10 acres (25 ha) in size and over 80 percent were less than 200 acres (80 ha) (Wakida 1997). On Haleakala most fires were less than 1 acre (0.4 ha) in size but several large fires, estimated at greater than 1,000 acres (400 ha), started in "pasture lands" (probably nonnative grasses) and moved into subalpine shrublands (Loope and others 1990). Although fires are reported from nearly every month of the year, most fires in high-elevation habitats occured during the dry summer months between May and August (52 percent for MKFR) (Wakida 1997), or during periods of drought (Loope and others 1990). Fuels in montane forests of MKFR are now composed of light to heavy accumulations of dead and downed wood and nonnative grasses. Within Haleakala National Park, Maui, nonnative grasses comprise a significant part of the biomass and standing fuel accumulation (Loope and others 1990). Although fire frequency might have increased in some upland areas with the removal of feral ungulates and the accompanying increase in nonnative grass cover, insufficient data are available to support this assumption.

Fire and Its Effects on Nonnative Invasives in Dry and Mesic Ecosystems

Most fires in Hawai`i's grasslands, shrublands and forests are caused by one or more species of introduced grasses (table 11-2). The dominant nonnative grasses recover rapidly after fire, often increasing in abundance, although dominance may shift from one nonnative species to another (Daehler and Carino 1998; Daehler and Goergen 2005; Tunison and others 1994, 2001). Detailed fire effects studies are few and limited to the last 20 years. Most have been conducted in HVNP on the island of Hawai`i in areas where fire is now frequent, detailed fire records have been kept, and nonnative invasive grasses are present (D'Antonio and others 2000, 2001a; Freifelder and others 1998; Hughes and others 1991; Hughes and Vitousek 1993: Ley and D'Antonio 1998; Mack and others 2001; Tunison and others 2001). Recent studies in low-elevation grasslands represent a wider geographic area, including several islands (Daehler and Carino 1998; Daehler and Goergen 2005; Goergen and Daehler 2001, 2002; Nonner 2006).

Fire potential is highest in certain dry and mesic ecosystems where substrate age, precipitation and plant productivity do not limit fuel continuity and biomass, and where dry periods create conditions of low fuel moisture suitable for combustion and fire spread (Asner and others 2005). On the Island of Hawai`i, Asner and others (2005) found that these areas generally receive between 30 and 60 inches (750 to 1,500 mm) of rainfall annually and often are dominated by nonnative grasses.

A majority of Hawai`i's grasslands are nonnative and often maintained by fire. Tunison and others (1994) studied the impacts of fire in nonnative grasslands in the central coastal lowland portion of HVNP. The canopy cover of Natal redtop (*Melinus repens*) generally decreased and that of thatching grass (*Hyparrhenia rufa*) increased within 2 years following fire. Cover of other nonnative species, such as molasses grass, changed little after 2 years from prefire values (Tunison and others 1994). On Oahu, Kartawinata and Mueller-Dombois (1972) noted the postfire encroachment of molasses grass into nonnative broomsedge communities.

Fountain grass is present in grasslands and as an understory species in forests over a wide elevational range. It is well adapted to persist after fire. Live fountain grass culms can sprout rapidly following top-kill and set seed within a few weeks (Goergen and Daehler 2001). Seeds are also blown in from neighboring unburned populations and rapidly germinate under favorable conditions (Nonner 2006). Unusually wet years result in rapid growth and expansion of fountain grass, which has a high net photosynthetic rate (Goergen and Daehler 2001, 2002). When dry conditions return, fires of high severity often result from the high fuel loads and low standing fuel moistures associated with fountain grass (Blackmore and Vitousek 2000; Nonner 2006).

Nonnative grasses can contribute to the decline of native pili grasslands where fire is excluded. Daehler and Carino (1998) compared the current (1998) and historic (1965 to 1968) cover at 41 sites on Oahu that were dominated by pili grass in the late 1960s and where fires had generally been suppressed. At these sites, nonnative grass monotypes, particularly buffelgrass, guineagrass (*Panicum maximum*) and fountain grass replaced pili grass in 2/3 of sites sampled, and pili was absent in the remaining 1/3 of the sites. In burned areas, however, the fire-adapted native pili grass competes well with some nonnative fire adapted grasses. In low-elevation grasslands within HVNP, pili grass cover was the same or higher 2 years after fire in areas where it was dominant or co-dominant (30 percent to 50 percent of total cover) prior to the fires and nonnative grass cover did not increase from preburn levels (Tunison and others 1994, 2001).

Establishment of nonnative invasive grasses, such as broomsedge and beardgrass, initiates a grass/fire cycle (fig. 11-7) that can increase the cover of nonnative grasses, inhibit shrub and tree colonization and growth, and result in changes in the composition of native woody plant communities (D'Antonio and Vitousek 1992; Hughes and Vitousek 1993). Although this cycle is best documented for \`ōhi\`a woodlands in Hawai\`i (Tunison and others 1995) it is probable in other native plant communities here. Monocultures of molasses grass in \`ōhi\`a woodlands and fountain grass in montane shrublands have been documented following repeated fires (Hughes and others 1991; Shaw and others 1997).

Fire effects were monitored 2 to 5 years after fire on paired burned and unburned transects in open-canopy shrublands between 130 and 600 feet (40 to 190 m) in HVNP (D'Antonio and others 2000; Tunison and others 1994). Fires were fueled by nonnative broomsedge and bush beardgrass. Nonnative grasses maintained dominance of the sites following fire; composing a majority of the total cover on burned and unburned sites, although nonnative grass cover was slightly lower (5 percent to 15 percent) in four of the five burned sites. Total native plant cover was the same or slightly higher in burned areas (four out of five sites) but this was largely due to an increase in pili grass rather than woody species (D'Antonio and others 2001a; Tunison and others 2001).

The cover of fountain grass was reduced but recovered quickly following a low- to moderate-severity fire in montane shrublands in the saddle between Mauna Kea and Mauna Loa. The aerial cover of fountain grass was 50 percent of total herbaceous cover prior to the fire, but was less than half of preburn values 6 months after. Within 1 year, fountain grass had regained over half of its original aerial cover and one-third of its basal cover (Shaw and others 1997). At the same time, total shrub cover was much lower (4 percent total cover) than prefire shrub cover (20 percent) and stem densities were at half of preburn values. Shaw and others (1997) noted that fountain grass became a monoculture in nearby areas that had burned repeatedly.

Anecdotal accounts of historic fires in east Maui (Loope and others 1990) and observations from Haleakala and Hawai\`i Volcanoes National Parks (Smith and Tunison 1992) suggest that severe fires may convert native subalpine shrublands into nonnative grasslands (fig. 11-10). Anderson and Welton (unpublished data) observed a four-fold increase in cover of

Figure 11-10—The result of wildfire in the Haleakala subalpine shrubland is a conversion of fuels from closed canopy of endemic shrubs (background) to nonnative grassland with sparse shrub recovery (foreground) as shown on the flank of the 1992 fire scar. (Photo by S. Anderson.)

nonnative grasses including velvetgrass, sweet vernal grass, and red fescue (*Festuca rubra*) in a burned area when compared to an adjacent unburned area 6 years after a 0.6 acre (0.2 ha), high-severity fire in February of 1992 at 8,000 feet (2,630 m) on Haleakala (Anderson, personal communication, 2005). The dominant native shrubs in the area, pūkiawe and `ohelo, do not tolerate fire well and are slow to reestablish (Anderson and Welton, unpublished data; Smith and Tunison 1992).

A fire fueled by fountain grass converted a lama woodland on the west side of Hawai`i Island into a nonnative grass-dominated savanna with few postfire sprouts and seedlings of lama after 3 years (Takeuchi 1991). Fountain grass had invaded the lama woodlands in previous decades to become a nearly continuous understory dominant (fig. 11-11). Fountain grass recovered rapidly after fire, returning to dominance. Little postfire recruitment of common native woody plants was observed after fire, but three native species, ʻilima (*Sida fallax*), kulu`i (*Nototrichium sandwicense*), and mamane, regenerated from seed or vegetative sprouts in scattered, localized patches within the vigorously recovering fountain grass matrix.

Nonnative grasses and Asian sword fern can carry fire in the ungulate-disturbed understory of mesic `ōhi`a forests (Tunsion, personal observation, May 2002, Kupukupu Fire, Holei Pali, Hawai`i Volcanoes National Park.). Short-term fire effects in mesic `ōhi`a forest with an understory of nonnative Asian sword fern were documented after two fires in HVNP (Ainsworth and others 2005; Tunison and others 1995). The response of sword fern was comparable to many nonnative grasses; shortly after fire it recovered vegetatively and quickly dominated the understory. Survivorship of native woody plant species was generally high, with sprouting observed in most tree and shrub species, including `ōhi`a. However, a second fire occurring 1 year later dramatically increased `ōhi`a mortality (Ainsworth and others 2005).

Figure 11-11—The highly flammable fountain grass forms a nearly continuous fuel bed in dry submontane lama forests on the island of Hawai`i. (Photo by S. Cordell.)

Fire effects have been studied most frequently in `ōhi`a woodlands of HVNP. Wildland fire resulted in a rapid increase in cover and biomass of nonnative grasses following six wildfires on nine sites in submontane `ōhi`a woodlands of HVNP (D'Antonio and others 2000; Tunison and others 1995). All fires were fueled by the invasive grasses broomsedge, bush beardgrass and molasses grass. Total nonnative grass cover was about 30 percent higher and total native species cover lower for all burned transects when compared with unburned transects (nine out of nine sites). Molasses grass showed the greatest increase (D'Antonio and others 2000; Tunison and others 1995). Hughes and others (1991) also studied the effects of fire in `ōhi`a woodlands in HVNP, from 1 to 18 years later, with similar results. After 15 months, the total cover of nonnative grasses was the same or greater in burned areas than in unburned areas (table 11-4), indicating that nonnative grasses can quickly reclaim a site and increase in cover over time. The cover of individual grass species changed, however, as molasses grass replaced broomsedge and bush beardgrass as the dominant grass species (Hughes and others 1991). Cover of molasses grass was even higher in areas that had burned twice (table 11-4).

This cycle of repeated fires increases the potential that fires will burn into and through native woody plant communities further reducing cover of trees and shrubs, altering forest structure and composition, and depleting native seed banks (Blackmore and Vitousek 2000; D'Antonio and others 2000, 2001a; D'Antonio and Vitousek 1992; Hughes and Vitousek 1993; Tunison and others 2001). Changes in forest structure further alter the microclimate to warmer, drier, and windier conditions (Freifelder and others 1998), creating additional barriers to the recovery of native communities. Total cover of native woody species in `ōhi`a woodlands was lower by nearly two orders of magnitude in burned areas as fire top-killed three of the four common native shrubs (pūkiawe, `ulei, and `akia). An average of 55 percent of `ōhi`a was top-killed among all sites studied, although some surviving trees resprouted within 1 year after fire (Tunison and others 1995). More trees were killed with higher fire intensity (as estimated by char height) (D'Antonio and others 2000), and no `ōhi`a seedling recruitment was observed, resulting in conversion of open-canopied woodlands to savannas (Tunison and others 1995). Hughes and others (1991) noted that postfire sprouting of native shrubs occurred infrequently and the dominant shrub,

Table 11-4—Average cover of native and nonnative species and dominant nonnative grasses following fire in submontane `ōhi`a woodlands in Hawai`i Volcanoes National Park.*

	Fire history and sampling scheme			
Species	**Unburned**	**Burned once and sampled 15 months after fire**	**Burned once and sampled 18 years after fire**	**Burned twice and sampled 1 year after second fire**
Bush beardgrass	63.9^{a**}	38.5^{b}	39.1^{b}	21.4^{c}
Broomsedge	8.8^{a}	3.6^{b}	0.3^{c}	0.1^{c}
Molasses grass	7.2^{a}	49.7^{b}	62.1^{b}	79.3^{c}
Native species subtotal	117.0^{a}	5.8^{b}	31.6^{c}	0.7^{d}
Nonnative species subtotal	80.3^{a}	$92.4^{a,b}$	101.9^{b}	101.2^{b}

* Adapted from Hughes and others (1991).

** Numbers with different superscripts are significantly different (n = 5; $P > 0.05$).

pūkiawe, was absent from most burned areas 18 years after fire. The native shrub `ākia is highly intolerant of fire and was absent from burned sites of all ages in both shrublands and woodlands (D'Antonio and others 2000). A similar result was noted with a related species of `ākia (*W. oahuensis*) by Kartawinata and Mueller-Dombois (1972) on Oahu.

The vigorous postfire response of molasses grass, a mat-forming species, creates an environment that inhibits regeneration of many species, primarily due to low light levels under the thatch (fig. 11-12). In addition to reducing the cover of other nonnative grasses, total native understory shrub cover in `ōhi`a woodlands was lower by two orders of magnitude, from over 60 percent cover in unburned plots to 0.6 percent in twice-burned plots containing molasses grass (Hughes and others 1991). Some native shrubs, such as `a`ali`i, may persist because of rapid germination and growth, and tolerance of low light levels beneath grass canopies (Hughes and Vitousek 1993).

Molasses grass has also replaced other grasses, including nonnative broomsedge and bush beardgrass (Hughes and others 1991) and native pili grass. Pili grass was absent from the burned areas of native low-elevation shrublands where molasses grass occurred at very high postfire cover levels (96 percent) (D'Antonio and others 2000).

A long-term study of the effects of a single large wildfire in montane shrublands and forest on Mauna Loa (Haunss 2003) supports Mueller-Dombois' assertion that fire does not appreciably alter the composition of native communities that contain a high proportion of

Figure 11-12—Molasses grass (*Melinus minutiflora*) forms a dense mat preventing regeneration of other plants. (Photo by A. LaRosa.)

fire-tolerant species. There was no increase in cover of nonnative velvetgrass, dallis grass, and broomsedge in burned native shrublands 27 years after fire compared with unburned shrublands, although the native shrub, pukiawe, was replaced by the fire-tolerant native shrub `a`ali`i. Similarly, forest composition differed little in burned and unburned koa forests, and koa remained the dominant canopy tree species 27 years after fire (Haunss 2003). Koa recovered rapidly after the burn by sprouting from roots, and formed a canopy within a decade that was similar to that in unburned sites. The understories of both burned and unburned koa forests were dominated by nonnative meadow ricegrass, suggesting that disturbances other than fire may play a role in determining postfire community composition (Haunss 2003; Tunison and others 2001).

Banko and others (2004) studied the effects of fire on mortality and regeneration of mamane in MKFR and surrounding areas following a small (5.6 ha) fire in 1999. They reported higher fire severity (percent burned area) in open-canopy mamane forests with an understory of nonnative grasses (velvetgrass, sweet vernal grass) than in adjacent grazed pasture with scattered mamane. They attributed this difference largely to the lower fuel loading of grasses in heavily grazed pasture when compared to the forest. Relative grass cover and biomass were not reported, however, and the effects of fire on the nonnative grasses cannot be determined. Fire appeared to stimulate root suckering of mamane trees in both forest and pasture, but suckering was much higher in pasture areas where fire severity was reportedly lower. An increase in the number of mamane saplings in burned areas 10 months after fire may be attributed to the decrease in competition for light or moisture with the removal of the dense, nonnative grasses (velvetgrass, sweet vernal grass). This has been suggested for molasses grass in other systems and warrants closer study.

The potential for, and consequences of, the loss of threatened and endangered (T/E) species in Hawaiian forests is particularly serious. Hawai`i has nearly 280 listed T/E plant taxa (Bishop Museum 2004) and many of these occur in fire prone environments. Studies of the effects of fire on rare plants are few, and the documented effects are variable, depending upon species and, to a certain extent, life form. There was virtually no regeneration of rare small trees, such as koki`o (*Kokia drynarioides*), kauila (*Alphitonia ponderosa*), or uhiuhi (*Caesalpinia kavaiensis*) following a fire in lama forest with fountain grass (Takeuchi 1991). In high-elevation shrublands on Hawai`i, several endangered tree and shrub species showed no signs of regeneration 1 year after fire, while certain subshrubs and prostrate species with rhizomes or tuberous root systems (for example, the native mint *Stenogyne angustifolia*) survived and sprouted soon after fire (Shaw and others 1997).

Lowland and Montane Wet Forests

Wet forests are found on all the main Hawaiian Islands except Niihau and Kahoolawe, in lowland habitats ranging from 300 to 3,600 feet (100 to 1,200 m), and montane habitats ranging from 3,600 to 6,600 feet (1,200 to 2,200 m). Lowland wet forests occur across a wide range of substrate and soil conditions with annual rainfall from 60 to 200 inches (500 to 5,000 mm). Montane wet forests occur on windward aspects with more than 100 inches (2,500 mm) of rainfall per year. Fog and fog drip are common in montane wet forests (Gagne and Cuddihy 1999).

Wet forests have been altered by anthropogenic activities including logging, introductions of feral ungulates, and invasions by nonnative plant species. Despite widespread human-caused alterations, many wet forest habitats remain dominated by native species. Gagne and Cuddihy (1999) recognize seven lowland and four montane wet forest communities. `Ōhi`a is the dominant canopy species in all but two communities: the lowland nonnative wet forest community types, composed of java plum (*Syzygium cumuni*), rose apple (*S. jambos*), mountain apple (*S. malaccense*), common guava, strawberry guava (*Psidium cattleianum*), and others; and the montane nonnative firetree (*Morella faya*) forest community (Gagne and Cuddihy 1999).

Fire has been documented in `ōhi`a lowland wet forests and `ōhi`a / hapu`u (*Cibotium* spp.) tree/fern montane forests in and near HVNP (Tunison, personal observation, 30 September 1995, Volcano Dump Fire, Volcano, HI.; Tunison and others 2001). `Ōhi`a lowland wet forests vary widely in composition and structure, ranging from nearly monotypic `ōhi`a stands to older `ōhi`a forests with a diverse assemblage of trees, shrubs, ferns (including native uluhe, *Dicranopteris linearis*) and other herbs in the understory. Introduced shrubs, grasses and fern species often dominate disturbed areas. `Ōhi`a lowland wet forest grades into `ōhi`a/hapu`u tree/fern montane forest as elevation increases. `ōhi`a trees are present as scattered emergents above a lower closed canopy of tree ferns and other native trees (Gagne and Cuddihy 1999).

Fire History and the Role of Nonnative Invasives in Wet Forest Fire Regimes

The historic role of fire in the Hawaiian wet forests is not well understood. Some scientists have speculated that disturbances such as fire may set wet forest communities back to an earlier stage of succession dominated by `ōhi`a and uluhe ferns and that the presence of these communities on the landscape is evidence of past fire disturbance (Gagne and Cuddihy 1999). Vogl (1969) suggested that the capacity of tree fern (hapu‘u, *Cibotium glaucum*) to withstand disturbance may be an evolutionary adaptation to fire. Conversely, some scientists suggest that wildfires have had little influence on the development of tropical wet forests (Mueller-Dombois 1981). Studies have shown that fuel moisture content stays above the moisture of extinction in intact evergreen tropical forests with frequent, high precipitation coupled with constantly high relative humidity (Uhl and Kauffman 1990).

Few wildfires were recorded in Hawaiian wet forests prior to the 1990s (Tunison and others 2001). However, the record of fires in Hawai`i is relatively short (less than 100 years) compared to probable long fire-return intervals for forests with these fuel and weather patterns. Naturally ignited lava and lightning fires in wet forests are possible during extended droughts. Fuels are abundant and continuous in these wet forests. The pantropical climbing fern, uluhe, for example, can attain heights greater than 30 feet (9 m) (Kepler 1984), creating a laddered fuel bed (Holttum 1957) (fig. 11-2).

Since the early 1990s, at least 13 wildfires have occurred in relatively young (400- to 750-year-old) wet forests on the Island of Hawai`i. Following prolonged drought, five small fires (less than 30 acres (12 ha) each) occurred in wet montane forests near the Volcano community on the Island of Hawai`i. In HVNP, eight small to medium (less than 5,000 acres (2020 ha) each) wildfires were documented in lowland and montane wet forests from 1995 to 2004 (Loh, unpublished data, 2004). At least three of these wildfires occurred following severe droughts associated with El Niño events (Ainsworth and others 2005). Native and nonnative plant species fueled these wildfires (fig. 11-13). Disturbance of tropical wet forests can create in-stand weather and fuel conditions (altered loading and arrangement) that increase fire spread (D’Antonio and Vitousek 1992; Uhl and Kauffman 1990).

Fire and Its Effects on Nonnative Invasives in Wet Forests

Many nonnative species in wet forests of Hawai`i appear able to survive fire and/or recolonize after fire. Our information comes from a few studies conducted 1 year following the 2003 Luhi Fire in the southeastern region of the Island of Hawai`i on 400- to 750-year-old lava flows (Ainsworth, unpublished data, 2005). The nonnative trees, firetree and strawberry guava, sprouted after fire. On average, burned sites had four times as many nonnative species as neighboring unburned reference sites, and cover of many nonnative herbaceous species was significantly greater in burned sites than unburned reference sites. For example, cover of nonnative Hilo grass (*Paspalum conjugatum*), was 2 percent in unburned plots and 30 percent in burned plots (Ainsworth, unpublished data, 2005). Due to their capacity for invasion and rapid postfire site occupation, nonnative ferns and grasses may reduce the regeneration of native species following fire.

Many native species in wet forests possess traits that enable them to survive fire, but these may be adaptations to other disturbance factors such as volcanism and hurricanes. For example, following the recent lava-ignited fires in 2002 and 2003 at Hawai`i Volcanoes National Park, many native species including `ōhi`a

Figure 11-13—Lowland wet forest with native and nonnative dominated understory fuels. Photo on left shows native tree fern (*Cibotium glaucum*) in unburned area; on right is nonnative Hilo grass (Paspalum conjugatum) 2 years after fire. (Photo courtesy of Hawai`i Volcanoes National Park.)

(fig. 11-14), `ilima (*Sida fallax*), `iliahi (*Santalum paniculatum var. paniculatum*), and `a`ali`i (*Dodonaea viscosa*), resprouted and/or established from seed during the first 2 years after fire. Over 90 percent of the tree ferns, hapu`u, also survived wildfire by sprouting (Ainsworth and others 2005).

Nonnative species invasions may pose increasing threats to native wet forest ecosystems in Hawai`i because of their ability to dominate postfire environments. However, fire effects are site-specific and depend on microclimate, fire severity, and prefire species composition. Long-term research is needed to elucidate the interactions of climate, nonnative species, and fire on composition of Hawaiian wet forests.

Use of Fire to Control Invasive Species and Restore Native Ecosystems

While little is known about the long-term effects of fire on native species, many lowland native woody species appear to regenerate poorly after fire. There are two major areas of research on restoration of biologically and culturally important ecosystems degaded by fire: (1) researchers are experimenting with fire to enhance pili grass in formerly Polynesian-maintained grasslands, and (2) in native forest and woodlands, fire and other methods are being tested in an attempt to reduce nonnative grass dominance and maintain native woody plant communities.

Current research indicates that prescribed fire, in combination with other treatments, may be used to maintain or restore pili grasslands invaded by nonnative grasses. Managers at HVNP have used prescribed fire successfully to enhance pili grass in areas dominated by Natal redtop but have been less successful where thatching grass is present. Park managers continue to refine burn prescriptions and identify fire-adapted native shrubs useful for rehabilitation of shrublands with pili grass in the understory (Tunison and others 2001). Daehler and Goergen (2005) successfully reestablished pili grass in areas dominated by buffelgrass using biennial, low-intensity backing fires in winter followed by herbicides or mechanical removal of buffelgrass, or fire without other treatments. In areas of combined treatments, pili grass became dominant. The absolute cover of pili grass in burned areas averaged 35 percent in plots where buffelgrass had been removed and 10 percent in plots where buffelgrass remained (Goergen and Daehler 2002). These results indicate that fire alone may be insufficient to restore pili grasslands where competition from nonnative grasses is substantial. In areas where pili grass has been absent for some time, the addition of seed becomes necessary for restoration due to the absence of a soil seed bank (Daehler and Goergen 2005). Burning pili grasslands also increases pili grass seed production and could provide seed for large scale restoration (Daehler and Goergen 2005; Nonner 2006).

Figure 11-14—`Ōhi`a trees resprouting from base following fire in 2003 in Hawai`i Volcanoes National Park. (Photo courtesy of Hawai`i Volcanoes National Park.)

One of the most widespread and problematic invasive species in dry and mesic communities in Hawai`i is fountain grass (Cordell and others 2004; Smith, C. 1985; Smith and Tunison 1992). Scientists are working to restore highly altered native dry forests invaded by fountain grass using ungulate exclusion and grass removal followed by direct seeding of fast-growing, weedy native species. Some fast growing species are able to establish and persist on their own. These early successional species may then create suitable microsites for the establishment of other native species (Cabin and others 2002; Cordell and others 2002). Attempts at controlling fountain grass with prescribed fire, herbicides, and grazing have met with varying degrees of success. To date, herbicide use (glyphosate) appears to be the most successful technique (Cordell and others 2002).

Managers at Hawai`i Volcanoes National Park are attempting to reduce impacts of fires fueled by nonnative grasses by establishing self-sustaining, fire-tolerant "near native" communities in place of communities dominated by formerly common fire-intolerant native species found on those sites. Between 1993 and 2000 they tested the capacity of native species to survive and recolonize after fire with laboratory heat trials and seven research burns. Fourteen native plant species, among them mamane, `a`ali`i, and `iliahi, were identified as fire-tolerant (Tunison and others

2001). Intensive restoration efforts were initiated in 2001 on 2,000 acres (815 ha) of `ōhi`a woodlands burned by wildfires. Outplantings and direct seeding of shrubs and trees were completed in the 1,000-acre (405-ha) Broomsedge Fire area. Survival of outplantings after 1 to 3 years was greater than 80 percent, and direct seeding has resulted in establishment of some species, including koa, mamane, naio, `a`ali`i, naupaka (*Scaevola kilauea),* and ko`oko`olau (*Bidens hawaiensis*) (Loh and others 2004). Park managers intend to keep fire out of these rehabilitation sites for 15 to 20 years to allow maturation of fire-tolerant plantings and development of a soil seed bank. In other sites, small prescribed fires have been used to reduce biomass of nonnative grasses and promote the successful establishment of individuals via outplantings and direct seedings. Other methods to temporarily control grasses, including the use of herbicides, are under investigation (Loh and others 2007).

We could find no published descriptions of attempts to use prescribed fire for control of invasive species and restoration of upland communities. Results of the effects of wildfires on montane and subalpine shrublands with nonnative grasses suggest that fire can increase the cover of introduced grasses and decrease that of many native shrub species. The limited data also suggest that some shrubs may recover in the long term. Some species, notably koa and mamane, tolerate fire and resprout readily after fire in these upland ecosystems but the abundance of invasive grasses after wildfire may hamper recovery of native understory species. Experimental restoration of the understory in unburned montane seasonal koa forest at HVNP uses herbicidal control of grass with planting and direct seeding of understory species to develop a soil seed bank (McDaniel, unpublished, 2003).

Given the probable long fire-return intervals in wet forests and the facilitation of nonnative invasive species spread, few ecological benefits are expected to accrue from prescribed burning of these forests.

Conclusions

The historic fire regimes of many Hawaiian ecosystems have been altered from typically rare events to more frequent, and in some instances, more severe fires today. This is largely a result of the increase in continuous, fine fuels associated with the spread of nonnative fire-tolerant grasses into dry and mesic native plant communities that were once dominated by relatively open stands of shrub and trees (Gagne and Cuddihy 1999; Smith and Tunison 1992). Invasive grasses are altering fuel loading and continuity, microsite conditions, and the postfire recovery rate of native species.

Fire is now an important disturbance factor in Hawai`i (Smith and Tunison 1992). It contributes to the maintenance and spread of many nonnative grasses, even though some of these grasses invaded ecosystems following other human disturbances and can persist without fire (Tunison and others 2001; Williams 1990). A few species of nonnative grasses are the primary carriers of most fires in Hawai`i today (table 11-2). Many of these nonnative pyrophytic species alter community structure and ecosystem properties including resource availability, primary productivity, decomposition, and nutrient cycling (D'Antonio and others 2001a; Hughes and Vitousek 1993; Ley and D'Antonio 1998; Vitousek and others 1996).

Much of the available information on the effects of fire on invasive species in Hawai`i is anecdotal. Fire effects studies have been conducted primarily on the Island of Hawai`i and are few, narrowly focused, and generally of short duration (1 to 5 years after fire). In reviewing the available literature, it is clear that plant responses are complex, not well understood, and often site specific. For example, molasses grass cover increased very little in dry, low-elevation grasslands in HVNP while in higher and wetter grasslands and woodlands, molasses grass cover increased appreciably (Hughes and others 1991; Tunison and others 1994, 2001). Even within HVNP, where most fire research has occurred, many questions remain.

As nonnative grasses continue to spread, damage to and potential loss of native shrublands and forests in Hawai`i will increase. The best documented examples in Hawai`i come from lama forests and `ōhi`a woodlands, where several species of nonnative grasses readily carry fire and where relatively few native species are fire-tolerant. In the submontane `ōhi`a woodlands, nonnative grasses have set in motion a grass/fire cycle resulting in changes in plant community structure and composition, fuel properties, and microclimate. Many of these areas have been converted to savannas dominated by invasive grasses (D'Antonio and Vitousek 1992; D'Antonio and others 2001a; Hughes and others 1991; Hughes and Vitousek 1993). This invasive grass/fire cycle can be long lasting (Hughes and others 1991) and may represent a permanent shift in plant community composition (D'Antonio and others 2001a). A similar cycle is occurring with the nonnative Asian sword fern and may be found in other plant communities upon further study.

In ecosystems where the dominant native species exhibit some fire tolerance, for example koa and a`ali`i, the detrimental effects of fire at the community level may be limited as these species often recover from fire by sprouting or establishing from seed. In montane shrublands and koa forests on Mauna Loa, where fire-promoting grasses are fewer, native communities have changed little, even three decades after a fire (Haunss 2003; Tunison and others 2001).

One of the most problematic nonnative species in Hawai`i today is fountain grass, which is spreading in dry and mesic shrublands and forests, particularly in western Hawai`i (Island). Fountain grass has a wide ecological range in Hawai`i. It has invaded many site types, from bare lava flows to late successional forests, from near sea level to over 9,000 feet (2,800 m), in areas with rainfall ranging from 10 to 50 inches (25 to 125 cm) (Jacobi and Warshauer 1992). Fountain grass forms a nearly continuous bed of highly flammable fuel in many areas and has the potential to dramatically alter entire landscapes and contribute to the loss of remnant native dry forests (Cordell and others 2004; Shaw and others 1997). There is a critical need for more research on fire and fountain grass interactions.

The cycle of invasive grasses and fire must be broken to protect native plant communities and ecosystems. The potential for loss of threatened and endangered species in Hawaiian forests is of particular concern. With little history of fire in many ecosystems and limited knowledge of fire and its effects, managers in Hawai`i have been cautious with their use of fire to manage disturbed environments. However research into the use of fire to manage some lowland ecosystems, for example native grasslands, is increasing. Prescribed fire is not likely to be a useful tool in most Hawaiian environments where fire-return intervals were historically long and most native species are intolerant of fire. Attempts to reduce fuel loads and restore native ecosystems using a variety of methods other than fire are underway. More research is needed on the effects of fire in native and altered ecosystems to understand the mechanisms of invasion, the nature of competitive interactions among native and nonnative species after fire, and the use of fire and other techniques in restoration of degraded ecosystems. Longer term studies are needed to determine if the short-term competitive advantage observed for many fire-promoting nonnative grasses over native woody species persists over time.

Notes