tameiameiae) attracts and may provide an important carbohydrate source for yellowjackets, which primarily feed on insects including native pollinators (Gambino et al. 1987).

Large numbers of alien invertebrates such as earthworms, ants, slugs, isopods, millipedes, and snails can cause "significant changes in the nutrient cycling process even if their direct impacts are obscure" (Howarth 1985). Changes in composition and structure of plant communities can result. Hybridization of aliens with close native relatives, can result in eventual extinction of species endemic to Hawai‘i (Wells et al. 1983).

**Alien Plants**

Of all the disruptive forces present today in the Hawaiian Islands, feral ungulates and invasive alien plants are the two most serious threats to natural areas that have so far escaped destruction by development, agriculture, or grazing. The native flora of the Hawaiian Islands is comprised of nearly 1,000 species, 89% of them endemic (Wagner et al., in press). By contrast, nonnative plants, introduced into the Hawaiian Islands either accidentally or intentionally for human use, number approximately 4,600 (St. John 1973). More than 800 alien plant species have become naturalized (i.e., reproducing and establishing themselves without human assistance), amounting to 47% of Hawai‘i's flowering plant flora (Wagner et al., in press). Of these naturalized aliens, Smith (1985) recognized 86 as serious problems in native ecosystems. He suggested that 28 of these plant pests were capable of invading undisturbed native systems.

Alien plant introductions to Hawai‘i began with the arrival of the Polynesians, who brought with them about 32 species (Nagata 1985). Most of these were cultivars, and less than 25 types escaped cultivation and became naturalized (Smith 1985). Some have had such a long history in the Hawaiian Islands and have become so much a part of lowland forests that they were considered native by many earlier writers. These include *kukui* (*Aleurites moluccana*), mountain apple or ‘ōhi‘a ‘ai (*Syzygium malaccense*), and wild yam (*Dioscorea pentaphylla*) (Handy and Handy 1972). A few additional species previously thought to be either indigenous or later European introductions may actually be accidental Polynesian introductions (St. John 1978; Wester, in press).

With the arrival of Capt. Cook in 1778 and the subsequent influx of ships from Europe and North America, plant introductions accelerated. The first recorded western introductions into Hawai‘i were pumpkins, melons, and onions (*Cucurbita pepo, Cucumis melo, Allium cepa*), planted during Cook’s visit. In the 60 years following Cook’s voyage, at least 111 nonnative plant species were introduced (Nagata 1985). These were primarily fruit trees, vegetables, and ornamentals, many of which remained in cultivation and did not become naturalized. Notable exceptions are guava (*Psidium guajava*), strawberry guava (*P. cattleianum*), and koa haole (*Leucaena leucocephala*), which became major invaders of lowland ecosystems. Most of the alien plant species threatening the integrity of native systems today are introductions of the 20th century. Wester (in press) noted that the rate of plant introduction increased sharply at the end of the 19th century, and about five species per year have naturalized during the 20th century.
The effects of alien species on native species and ecosystems are not well understood or documented (Loope, in press), although it is clear that their impact has been major in Hawai‘i. Many of the component species of island biota are thought to exhibit "reduced aggressiveness or increased vulnerability to extinction," and the characteristic low number of species per unit area in island ecosystems, as well as "certain genetic properties" of island organisms, may contribute to this vulnerability (Loope and Mueller-Dombois 1989). Because of limited ranges, consequent small population sizes, and habitat specialization, endemic Hawaiian species are vulnerable to disturbance (Simon 1987). Alien plants (and animals) have contributed to the extinction of native species in the lowlands of Hawai‘i and have been a "primary cause" of such extinction in upland habitats (Vitousek et al. 1987).

The most-often cited effects of alien plants on native plant species are competition and displacement; competition may be for water or nutrients, or it may involve allelopathy (chemical inhibition of other plants) (Smith 1985). Alien plants may displace natives by preventing their reproduction, usually by shading and taking up available sites for seeding establishment (Vitousek et al. 1987). Alien plant invasions may also alter entire ecosystems by forming monotypic stands, changing fire characteristics of native communities, altering soil-water regimes, changing nutrient cycling, or encouraging other nonnative organisms (Smith 1985; Vitousek et al. 1987). Hawai‘i is now recognized as an important "natural laboratory" to study interactions among native and alien species and the invasions of aliens in natural ecosystems (Simon 1987; Vitousek et al. 1987).

The following discussion will cover only a few of the most widespread and disruptive of alien plant invaders: those primarily found in wet zones, then those characteristic of dry and mesic habitats. For a more inclusive discussion of disruptive alien plants in Hawai‘i, see Smith (1985).

Invaders of Wet Habitats

**Banana Poka and Other Passiflora.** A vine or liana in the passionflower family, banana poka (*Passiflora mollissima*) was introduced to the island of Hawai‘i before 1921, when it was observed growing wild at Pu‘uwa‘awa‘a Ranch in the North Kona District (LaRosa 1984). Native to the Andes (St. John 1973), the vine was probably brought to Hawai‘i as an ornamental because of its showy, pink flowers. Banana poka was apparently planted at most of the sites that currently have heavy infestations: Hualalai; Keanakolu/Laupahoehoe on Mauna Kea; Wright Road/Ola’a on Mauna Loa; Kula on Maui; and Kōke‘e, Kaua‘i (Warshauer et al. 1983; LaRosa, in press). As of 1981, banana poka was distributed over 520 km² (200 mi²) in wet and mesic forests on Hawai‘i and Kaua‘i, of which 190 km² (73 mi²) represented continuous distribution. On Maui, banana poka is found on 200 ha (500 a) at upper Waiakoa, near Kula, where it was previously thought to have been eradicated by State workers (Smith 1985). The relatively small Maui population has undergone dramatic expansion since 1971 (LaRosa, in press) but has recently received attention from volunteer groups, and there is hope that complete eradication is still possible with the aid of special funding to the Department of Land and Natural Resources from 1989 to 1991 (B. Gagné and L. Loope, pers. comm. 1989).

Most of the poka-infested acreage occurs on the island of Hawai‘i, with three main areas of concentration and several isolated populations. The oldest infestation is in the North Kona District in mesic forests of the slopes of Hualalai and Mauna Loa, where
banana poka vines are scattered rather than continuously distributed over most of its range. Warshauer et al. (1983) interpreted the distribution pattern to mean that banana poka is still spreading into adjacent suitable habitat.

The second-oldest infestation is in koa/ʻōhi'a (Acacia koa/Metrosideros polymorpha) forest of the windward slopes of Mauna Kea and is called the "most advanced or severe" infestation surveyed. When banana poka distribution in 1983 was compared with that found in a survey ten years earlier, the size of this population seemed relatively stable.

The third major infestation occurs in montane wet ʻōhi'a/hāpu'u (Metrosideros polymorpha/Cibotium spp.) forest in 'Ola'a Tract of Hawaii Volcanoes National Park, where the vine increased in cover 18-fold between 1971 and 1981 (Warshauer et al. 1983). Warshauer and others blamed the banana poka extension and intensification on a major wind storm in 1980, which opened up the tree fern (Cibotium spp.) canopy in 'Ola'a. LaRosa (in press) considered the 'Ola'a population to be the youngest and most rapidly expanding infestation on the Island. In a recent (1988) ground survey of 'Ola'a in Hawaii Volcanoes National Park, banana poka was found to be widely distributed, with dense cover in the western third of the Tract (S.J. Anderson et al., unpubl. data).

Banana poka is extremely detrimental to native forests and can change their structure and species composition (Warshauer et al. 1983). Heavy loads of vines may cause damage or death to native trees, and when branches break or trees fall, the tree canopy is opened, changing understory conditions. Dense cover of banana poka in forest canopies reduces the sunlight reaching trees and may decrease photosynthesis (LaRosa, in press). Much of the Hawai'i Island infestation is in koa forests; here koa reproduction is affected, as banana poka outgrows koa seedlings and saplings and competes with them (Scowcroft and Nelson 1976). Bird populations of infested forests may be adversely affected by the disturbance brought about by banana poka, particularly those endangered bird species whose primary habitat is the koa/ʻōhi'a forest (Warshauer et al. 1983; Scott et al. 1986). One native bird, the 'i'iwi (Vestiaria coccinea), has become "associated with Passiflora" and has been observed feeding on banana poka nectar (Scott et al. 1986).

Banana poka is spread by alien fruit-eating birds such as the kalij pheasant (Lophura leucomelana) (Lewin and Lewin 1984), as well as by the feral pig (Sus scrofa) (Warshauer et al. 1983). Feral pigs open up forest understory, create bare disturbed areas in which weeds can establish, and topple tree ferns, encouraging the sun-loving banana poka. LaRosa (1984) reported that banana poka invaded and increased in density in 'Ola'a forest plots after the tree fern cover was artificially decreased or removed. Dispersal of this plant by alien animals, as well as natural perturbations such as windstorms and treefalls, make banana poka difficult to control. Previous attempts to reduce banana poka with herbicides on heavily infested State lands were largely unsuccessful (LaRosa 1984). Efforts are currently under way to control the plant in certain relatively intact portions of Hawaii Volcanoes National Park (Cuddihy et al. 1988), and preliminary research into mechanical and herbicidal control methods has been completed (Santos et al. 1989a). Biological control research is ongoing, and one potential control agent, an iridescent blue moth (Cyanotricha necyria), whose larvae defoliate poka leaves, has already been released on Hawai'i Island (Markin et al., in press).
If left unchecked, the future distribution of banana poka could be far greater than it is today. Jacobi and Warshauer (in press) modeled potential banana poka range on the island of Hawai'i based on rainfall and elevation and concluded that the species could become established over a vast region between 500 and 2,500 m (1,640-8,200 ft) elevation on the slopes of all five of the Island’s volcanoes.

Other species of Passiflora are also pests in natural areas of the Hawaiian Islands, although they currently have a more limited distribution than banana poka. Sweet granadilla (P. ligularis) has habits similar to banana poka and is a problem on Hualalai, Hawai'i Island; significant populations also exist in the South Kona and Puna Districts. Huehue-haole (P. suberosa), a vine capable of smothering small trees and understory plants, has infested drier areas, such as the Wai'anae Mountains of O'ahu and leeward East Maui (Smith 1985).

Guavas: Strawberry Guava, Common Guava. An early European introduction native to Brazil (St. John 1973), strawberry guava or waiawi (Psidium cattleianum) has been in the Hawaiian Islands since 1825, when several species of guava and other cultivated fruit plants arrived on the British ship Blonde (Nagata 1985). Today strawberry guava occurs on all the main Hawaiian Islands except Ni'ihau and Kaho'olawe, generally in wet and mesic lowland regions between 150 and 1,300 m (495-4,260 ft) elevation (Smith 1985; Wagner et al., in press). During the U.S. Fish and Wildlife Service Forest Bird Survey, strawberry guava was found at 8% of the stations sampled above 500 m (1,650 ft) elevation on the island of Hawai'i, suggesting that it is a very widespread alien species (Jacobi and Warshauer, in press).

In Hawaii Volcanoes National Park, strawberry guava is found from near sea level to above 1,220 m (4,000 ft) elevation and is acknowledged as a disruptive invader of native systems (Tunison et al., in press). In a recent (1988-89) survey, strawberry guava was found to be distributed at low densities over nearly half of 'Ola'a Tract and was even more widespread and abundant in 'ōhi'a (Metrosideros polymorpha) forests on the Park's East Rift (S.J. Anderson et al., unpubl. data). Strawberry guava is recognized as the most serious plant pest of the Kipahulu District of Haleakala National Park, Maui (National Park Service 1986b), where it has invaded koa (Acacia koa) and 'ōhi'a rain forests up to an elevation of 1,140 m (3,740 ft) (Yoshinaga 1980) and forms very dense stands in lower-elevation forests. Strawberry guava potentially threatens numerous low- to middle-elevation rain forest plant species in Haleakala National Park with extirpation through displacement. Species particularly threatened in this way include hame (Antidesma platyphyllum), po'olā (Claoxylon sandwicense), kuenui (Cyanea grimesiana), 'ohe (Joinvillea ascendens), 'aiea (Nothocestrum longifolium), and nuku'i'iwi (Strongylodon ruber) (Loope et al., in press b).

Smith (1985) considered strawberry guava to be the "worst pest in Hawai'i's rain forests." In favorable habitats, the tree is capable of forming nearly monotypic stands, excluding almost all native ground cover and other understory species. In forests with serious strawberry guava infestations, most native plants do not regenerate for several reasons: very dense shade, digging by feral pigs (Sus scrofa) that are attracted by strawberry guava fruit, and allelopathic effects of strawberry guava leaf litter (Smith 1985; Wagner et al., in press). Even where strawberry guava does not form dense stands, its presence encourages higher pig populations and may adversely affect rare plant species with limited distributions. The seeds of strawberry guava are known to be spread by both feral pigs and frugivorous birds. L.F. Huenneke and P.M. Vitousek (unpubl. data) found that strawberry guava has very high rates of fruit production and
seed germination; they also attributed some of the plant's success in native forests to its ability to readily produce sturdy, fast-growing suckers, which are less injured by falling branches than are more delicate native tree seedlings.

Strawberry guava is considered to be a pest by cattle ranchers because of its ability to form dense stands and crowd out forage plants (Hosaka and Thistle 1954). The species is currently recognized as a forest pest by State foresters (Hostetler 1970), but in the past (1928-52) more than 26,000 strawberry guava trees were planted in O'ahu and Kaua'i forest reserves (Skolmen 1979).

Control of strawberry guava may be possible with herbicides applied to cut stumps or frilled trunks (Gardner 1980; Santos et al., 1989b), at least in limited areas with important natural resources (Tunison, in press a). However, such control is extremely labor intensive and cannot be applied to dense and extensive infestations. The economic value of common guava (Psidium guajava) was previously thought to preclude biological control research (Gardner and Davis 1982); however, several potential insect control agents were recently located on native stands of strawberry guava during exploration in Brazil, suggesting the possibility of future successful biocontrol (Hodges 1988). Control of this pest is critical; Jacobi and Warshauer (in press) predicted that the plant may be able to invade nearly half the area surveyed by the U.S. Fish and Wildlife Service Forest Bird Survey on the island of Hawai'i, by "filling in" areas among current infestations below 1,300 m (4,260 ft) elevation.

Common guava, another early introduction from tropical America, is also considered a pest by many. Like strawberry guava, it can form dense thickets in wet and mesic habitats of the lowlands. However, unlike its congener, common guava is usually found in previously disturbed areas and is abundant primarily below 500 m (1,650 ft) elevation (Smith 1985; Wagner et al., in press). This is the plant that Isabella Bird (1966) found so common on the sides of ridges and in the understory of forests she traversed on Hawai'i Island in 1873. Other trees of the myrtle family have also become pests and lowland forest invaders in the Hawaiian Islands, most notably two species in the genus Syzygium (formerly Eugenia): Java plum (S. cumini) and roseapple (S. jambos).

**Melastomes: Clidemia, Tibouchina, Others.** The melastome family (Melastomataceae) is not represented in the native flora of Hawai'i, but there are 14 naturalized species in 11 genera, and at least five have become disruptive invaders of native ecosystems (Wagner et al., in press). One of the worst is Clidemia hirta or Koster's curse, which has caused great concern because of its rapid spread in the Hawaiian Islands. Native to tropical Central and South America, Clidemia has become widely naturalized in other tropical regions. In Fiji, Clidemia was introduced in the late 19th century and became a serious pest in pasturelands and rubber and coconut plantations (Wester and Wood 1977).

Clidemia was first reported in the Hawaiian Islands in 1941 on O'ahu near Poamoho; it may have been originally planted in the nearby Wahiawa Botanic Gardens. Within the Ko'olau Mountains, Clidemia spread in the 1940s (Smith, in press), and by the early 1960s the pest had expanded from Tantalus north to Kawaiola Ridge (Wester and Wood 1977) but was not known to be on the other Hawaiian Islands (Plucknett and Stone 1961). By 1970, Clidemia had spread to the Wai'anae Mountains, where it is now widespread in the Honouliuli Forest Reserve (Smith, in press). The rapidity of the spread and intensification of Clidemia on O'ahu is remarkable: between 1977 and

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1988, the species tripled its range from 31,350 ha (77,430 a) (Wester and Wood 1977) to more than 100,000 ha (247,000 a) (Smith, in press).

*Clidemia* spread to five other Hawaiian Islands in the 1970s and 1980s. On Kaua‘i, there are now at least five small populations, totalling about 40 ha (100 a). Moloka‘i currently has an infestation totalling more than 6,000 ha (14,820 a), primarily in Wailau and Pelekunu Valleys of the North Shore; much more available habitat on Moloka‘i may be invaded in the next few years. East Maui was invaded in the mid 1970s, and by the early 1980s *Clidemia* was found on West Maui (Smith, in press). At present, there are two established populations of *Clidemia* on East Maui: at Makapipi (30 ha or 75 a), and at ‘O’opuola and adjacent stream drainages (1,200 ha or 3,000 a) (Medeiros et al. 1989). Recently (1988), two *Clidemia* plants were discovered on the island of Lāna‘i (Smith 1988). The island of Hawai‘i has at least four populations on windward slopes, all reported between 1972 and 1983 (Smith, in press).

The effects of *Clidemia* on native vegetation are devastating. The shrub replaces native plants of the forest understory, and almost nothing except a few hardy mosses can grow beneath its shade. As *Clidemia* replaces the multilayered understory of native forests, their value as watersheds may be greatly reduced (Smith, in press). Even the hardy and aggressive native matted fern uluhe (*Dicranopteris linearis*), a dominant cover on many slopes and in lowland forests, can be replaced by *Clidemia* in shaded areas (Wester and Wood 1977). Native insects and birds are probably lost along with the native vegetation in invaded areas. While not resistant to fires (Smith 1985), *Clidemia* can take advantage of such disturbances. Two years after a fire in Kawaiola Forest Reserve on O‘ahu, *Clidemia* had become the dominant cover and achieved heights greater than 2 m (7 ft) (Wester and Wood 1977). *Clidemia* is considered a pest by cattle ranchers, as it has no forage value (Hosaka and Thistle 1954).

Clearly, the plant is easily dispersed. Birds are known to eat the berries (Hosaka and Thistle 1954), and the ubiquitous Japanese white-eye (*Zosterops japonica*) may be a prime vector (Wester and Wood 1977). Hikers, hunters, and marijuana (*Cannabis sativa*) growers are responsible for accidentally dispersing *Clidemia*; feral pigs (*Sus scrofa*) both spread the weed and encourage its intensification through digging and soil disturbance (Smith, in press).

There is hope that biological control may help reduce the impacts of *Clidemia*. Nakahara et al. (in press) reported that of 14 insects from Trinidad recently evaluated as *Clidemia* control agents, three to five may be released in Hawai‘i in the near future. A fungus from Panama (*Colletotrichum gloeosporoides*), which attacks the leaves of *Clidemia*, was introduced into Hawai‘i in 1986 (Anon. 1986; TenBruggencate 1986b). A thrips (*Liothrips urichi*), introduced earlier (1953) to combat *Clidemia*, has been effective on ranchlands but not in forests (Reimer 1985). Other insects released to control the weed have been even less successful (Nakahara et al., in press). If biological control is not effective on this species, the future of remaining low-elevation wet and mesic forests is grim. Seeds in the soil may remain viable for more than four years. Mechanical control is very time consuming, and in rain forests, even the leaves of pulled plants may form new roots and reestablish (Smith, in press).

Three species of *Tibouchina* have become naturalized in the Hawaiian Islands, but only two currently represent a threat to native vegetation. Glorybush or princess
flower (*T. urvilleana*), native to Brazil, was introduced to Hawai‘i in 1910 as an ornamental. A shrub with showy, purple flowers, it is still cultivated around residences and has become naturalized in wet areas on Kaua‘i, O‘ahu, Maui, and Hawai‘i (Wagner et al., in press). In 1936-38, more than 1,000 glorybush were actually planted in forest reserves on O‘ahu and Kaua‘i (Skolmen 1979). Like other members of the melastome family, glorybush can form dense thickets and crowd or shade out native plants; the shrubs may achieve heights greater than 4 m (13 ft) (Smith 1985). In some areas, glorybush may penetrate less readily into forests than other melastomes (Plucknett and Stone 1961).

Glorybush spread rapidly from plantings in the Kōke‘e area of Kaua‘i and is considered a threat to native plants there (Hadley 1966). In Hawaii Volcanoes National Park, the species is localized in wet ‘ohi‘a (*Metrosideros polymorpha*) forests around the rim of Kilauea Crater, where it has proven difficult to eradicate (J.T. Tunison, pers. comm. 1988). Although it seems not to produce viable seeds (Wagner et al., in press), cut, broken, or decumbent branches of glorybush readily root in humid montane climates. A relatively effective cut-stump herbicide treatment for glorybush is known, but the shrub’s tangled growth form makes such control extremely laborious (Santos et al. 1986). A moth introduced as a biological control agent (*Seleca brunella*) has become established on glorybush (Gardner and Davis 1982) but is apparently not seriously affecting the species (Smith 1985).

*Tibouchina herbacea*, native to several South American countries, became established in the late 1970s on Hawai‘i Island and was noted on East Maui in 1982 and West Maui in 1986 (Wagner et al., in press). First collected on Hawai‘i Island near the Saddle Road, this shrub has been rapidly spreading in wet and mesic forests and has recently been reported from Pu‘u o ‘Umi Natural Area Reserve (Smith 1988), Kahauale‘a Natural Area Reserve (S. Perlman, pers. comm. 1989), and remote forests of the East Rift of Kilauea in Hawaii Volcanoes National Park (S.J. Anderson et al., unpubl. data). The disruptive capabilities of this species are not fully known, but *T. herbacea* density seems to be intensifying in forests with ‘ohi‘a dieback.

Other members of the melastome family are also potential threats to native forests. One recently naturalized species is *Miconia calvescens* or velvet tree, native to tropical America (Wagner et al., in press). *Miconia* has been planted as an ornamental in yards and botanic gardens on Hawai‘i Island (Davis 1987), Maui (Smith 1988), and O‘ahu (B. Gagné, pers. comm. 1989). On the island of Hawai‘i at least, *Miconia* has escaped cultivation and become established near Onomea, Hilo, and Pāhoa (Davis 1987). While the current infestations in Hawai‘i are small, *Miconia* has the potential to be seriously disruptive; after introduction to Tahiti and Moorea, *Miconia* rapidly invaded native forests up to 1,220 m (4,000 ft) elevation and became an understory dominant (B. Gagné, pers. comm. 1989).

Another member of the melastome family, *Oxyspora paniculata*, is a pink-flowered shrub that escaped from cultivation on Tantalus and became established on O‘ahu before 1954 (Wagner et al., in press). Although it was considered non-invasive for more than 20 years, recently *Oxyspora* has been spreading eastward along the summits of the Ko‘olau Mountains and has become the dominant plant along some mountain stream banks (Obata 1985a). Because of this "population explosion," Obata raised the specter of *Oxyspora* as another *Clidemia*. That *Oxyspora* is continuing its spread on O‘ahu is indicated by recent sightings on Wa‘ahila Ridge (Smith 1988) and Mānoa Valley (C.W. Smith, pers. comm. 1989).
Rubus: Yellow Himalayan Raspberry, Blackberry, Others. At least six species of *Rubus* have become naturalized in the Hawaiian Islands. Three of these species now infest wet and mesic ecosystems on several islands, and one is also a problem in subalpine grasslands. The three remaining species occur over smaller and more limited areas, but they have potential to invade natural areas (Wagner et al., in press).

*Rubus ellipticus*, or yellow Himalayan raspberry, is a recent introduction into the State (c. 1961) and is currently restricted to the island of Hawai‘i. Perhaps introduced as an ornamental or for its insipid, yellow fruit, the raspberry is native to India (Wagner et al., in press). There is a major infestation in the Volcano area and Hawaii Volcanoes National Park, but the plant may be seen between 700 and 1,700 m (2,300-5,580 ft) in wet habitats elsewhere on the Island (Smith 1985). Like other *Rubus* species, this raspberry may form large, tangled thickets and will grow in wet forests, pastures, open areas, and roadsides. Unlike Florida blackberry (*R. argutus*), yellow Himalayan raspberry is capable of growing to enormous size even in the dense shade of closed-canopy rain forests. In a recent (1988) survey of ‘Ola‘a Tract in Hawaii Volcanoes National Park, yellow Himalayan raspberry was found to be distributed over more than half of the 3,770-ha (9,300-a) Tract (S.J. Anderson et al., unpubl. data). This indicates a dramatic expansion since an earlier survey of ‘Ola‘a Tract by Jacobi and Warshauer (1975), when the species was recorded primarily in a 100-ha (250-a) kipuka (island of older vegetation) of *koa* (*Acacia koa*) and along nearby roadsides.

During the U.S. Fish and Wildlife Service Forest Bird Survey of 1976-81, yellow Himalayan raspberry was found on only 0.1% of the stations sampled (Jacobi and Warshauer, in press), but the authors considered this *Rubus* to be in "an incipient stage of invasion" and predicted that it had the potential to invade vast regions on both the windward and leeward slopes of Hawai‘i Island. *Rubus ellipticus* has been reported to be capable of nitrogen fixation on the island of Java (Becking 1979), although this has not been demonstrated in Hawai‘i. Biological control work has not been initiated on this species (Smith 1985), but research has been conducted on potential herbicidal control methods (Santos et al. 1986) with some success.

Another serious invader is prickly Florida blackberry (*R. argutus*, formerly known in Hawai‘i as *R. penetrans*), introduced from the eastern or central United States perhaps as early as 1894 (Haselwood and Motter 1976). Recognized as a pest in the 1930s (Doerr 1931), by 1962 blackberry was distributed over nearly 18,000 ha (44,460 a) on four Islands (Santos et al. 1986). It now occurs on five of the main Islands between 1,000 and 2,300 m (3,280-7,540 ft) elevation (Smith 1985) and is considered a noxious weed by the Hawaii State Department of Agriculture (1978).

Easily spread by fruit-eating birds and capable of vigorous vegetative reproduction, this plant can form dense, impenetrable thickets in open vegetation (Smith 1985). Montane mesic forests of Hawaii Volcanoes National Park have been heavily invaded, and Florida blackberry has also been found in nearly intact rain forest and subalpine grasslands of Haleakala National Park on Maui (Loope et al., in press b). On Kaua‘i, blackberry has invaded the rain forests of Kōkē’e, where it appears to be competing with the native plant understory and "smothering native ferns" (Hadley 1966).

*Rubus rosifolius*, locally known as thimbleberry, is perhaps the most widespread of all the introduced species of *Rubus*. This Asian plant was introduced to the island of Hawai‘i from Jamaica in the early 1880s, and despite regulations to prohibit
interisland transport (Degener 1936), thimbleberry became established in wet forests on all the main Hawaiian Islands (except Ni'ihau) from near sea level to above 1,730 m (5,670 ft) elevation (Wagner et al., in press). A relatively small shrub, thimbleberry seems to be much less aggressive than other members of the genus, although it may be very abundant locally, particularly in areas disturbed by feral pigs (Sus scrofa).

Other naturalized species of Rubus currently have more limited distributions in Hawai‘i. A relatively recent infestation of Rubus glaucus in ‘Ola‘a Tract of Hawaii Volcanoes National Park is growing in dense ‘ōhi‘a/tree fern (Metrosideros/Cibotium) rain forest. This species was apparently planted at a nearby agricultural experiment station and then abandoned as a potential crop, without destruction of the plants (Smith 1985). Two other cultivated raspberries, Rubus niveus and R. sieboldii, native to Asia, were introduced to one or more of the Islands for their fruit; these potential pests have escaped cultivation and become naturalized (Smith 1985; Wagner et al., in press). Bryan (1954) reported that seeds of R. niveus (then called R. albescens) were imported from Florida in 1952 and were germinated and distributed to the public by the territorial Division of Forestry; this Rubus was considered to be safe to distribute because plants did not "send out adventurous suckers from the root." Despite the history of Rubus infestations in the Hawaiian Islands, nearly 100 Rubus plants, unidentified to species, were planted in four forest reserves between 1938 and 1954 (Skolmen 1979).

**Gingers: White, Yellow, and Kahili Ginger.** Two members of the ginger family were brought to Hawai‘i by the Polynesians: ‘ōlena (Curcuma longa) and shampoo ginger or ‘awapuhi (Zingiber zerumbet). Both have become naturalized, usually in previously disturbed habitats. Among the many ornamental gingers that have been introduced to Hawai‘i in the last 200 years, six have become naturalized (Wagner et al., in press), and three are problems in native ecosystems.

White ginger (Hedychium coronarium) and the very similar yellow ginger (H. flavescens) were both introduced in the late 1800s, probably by Chinese immigrants (Degener 1934). White ginger is from China and the Himalayas, and its yellow-flowered relative is native to northern India. Both species escaped cultivation and are now established in lowland wet and mesic forests on most of the main Hawaiian Islands (Wagner et al., in press). In suitable habitats, these large, vigorous herbs are capable of forming a very dense ground cover that excludes all other growth. Although most common along roadsides, trails, and other disturbed areas, they have also invaded streambeds in many natural areas. A nearly pure stand of ginger was recently noted in streambeds of The Nature Conservancy’s Kamakou Preserve on Moloka‘i (Cuddihy et al. 1982b). Fortunately, neither white nor yellow ginger is a heavy producer of conspicuous fruit or seed, so vegetative reproduction predominates (Smith 1985).

Kahili ginger (H. gardnerianum) is a more serious threat to native forests because of its ability to produce abundant fruit, with orange seeds attractive to many alien birds. Native to the Himalayas, this ginger was introduced to Hawai‘i sometime before 1940 (Wagner et al., in press) and was first collected in Hawaii Volcanoes National Park in 1943 (Fagerlund 1943). At first, kahili ginger was not perceived as a threat to native forests of the Park because it remained restricted to gardens in the Park housing area, at least until 1947 (Fagerlund 1947). However, by the mid-1980s, kahili ginger had become established at high densities in the Kilauea area more than a mile from original plantings, and it was found at low densities in Kipuka Puaulu and ‘Ola‘a Tract, more than 5 km (3 mi) from the housing area. Additional isolated
populations occur on Kilauea's East Rift, more than 10 km (6 mi) from Park housing (S.J. Anderson et al., unpubl. data); these populations probably represent dispersal from other plantings.

Major infestations also occur at Kōke'e, Kaua'i, and windward East Maui (Smith 1985). In Kipahulu Valley on Maui, kahili ginger has been recently (1988) sighted as high as 1,220 m (4,000 ft) elevation, in nearly intact rain forest (Loope et al., in press b). This ginger was not found in previous surveys for weeds of the middle and upper Valley (Yoshinaga 1980), and it appears to be rapidly invading from more disturbed habitats at lower elevations. Kahili ginger is particularly insidious in native rain forest because, unlike most alien plants, it can form very dense stands even under the heavy shade of intact canopies of native trees and tree ferns. Kahili ginger typically achieves heights greater than 1 m (3 ft) and can displace all other understory plants in its vicinity (Smith 1985).

Although a number of diseases and insects are known to attack gingers, biological control is thought to be unlikely because of opposition from horticulturists and commercial ginger producers (Gardner and Davis 1982; Smith 1985). Herbicidal control methods have been investigated but are not completely effective (Santos et al. 1986). Mechanical removal of ginger by digging up rhizome masses has been accomplished in small portions of Hawaii Volcanoes National Park, but this method of control is extremely time consuming and labor intensive (Cuddihy et al. 1988; Tunison, in press a).

**Australian Tree Fern.** Ferns are not generally considered to be disruptive to native systems in Hawai‘i; only two species, *Athryopsis japonica* and *Blechnum occidentale*, are listed by Smith (1985) as potential pests. Australian tree fern (*Cyathea australis*) is a recent introduction into Hawai‘i that has been planted in private and hotel gardens at low elevations on O‘ahu, Maui, and Hawai‘i. It is naturalized on O‘ahu (C.W. Smith, pers. comm. 1988), and in 1988 naturalized populations of this fern were found in Kipahulu Valley, Haleakala National Park, at elevations as high as 1,040 m (3,400 ft) (Anderson et al., in press). Some of the individuals seen were taller than 4 m (13 ft), which undoubtedly represents the growth of many years, even though this alien plant was not listed on previous plant surveys of the Valley (Lamoureux 1967; Yoshinaga 1980). The future of this infestation and possible competition between this fern and native tree ferns (*Cibotium* spp.) in the Valley are unknown, but monitoring and control have been recommended (Loope et al., in press b). Ironically, another recently introduced *Cyathea* (*C. cooperi*), is being promoted for commercial propagation in West Hawai‘i with the reasoning that production of the faster-growing *Cyathea* will decrease exploitation of native tree ferns and reduce destruction of native forests (Anon. 1989 b).

**Grasses:** *Hilo Grass, Palmgrass, Meadow Ricegrass.* Alien grasses have long been perceived as threats to Hawaiian forests. Three species, Hilo grass, palmgrass, and meadow ricegrass (*Paspalum conjugatum, Setaria palrnifolia, and Ehrharta stipoides*, syn. *Microlaena*), are especially important in wet habitats in Hawai‘i. Two additional species, dallis grass and Vasey grass (*Paspalum dilataturn and P. urvillei*) are widespread in disturbed areas on the islands of Maui and Hawai‘i and have invaded some rain forests and montane mesic communities.

One of the earliest recognized invaders of wet forests was *Hilo grass* (*Paspalum conjugatum*). Native to tropical America, this grass was first noted near Hilo in 1840, hence the common name in Hawai‘i. Hilo grass spread rapidly and is now
distributed in wet habitats from near sea level to 2,000 m (6,560 ft) elevation (Smith 1985) on all the main Islands except Ni'ihau and Kaho'olawe (Wagner et al., in press). In the early 1900s, Hilo grass was recognized as a serious threat to watersheds and forest reserves, and its presence was considered an impediment to planting and reforestation (Giffard 1918). Able to form a dense cover even on poor soils (Smith 1985), Hilo grass was blamed for preventing the reproduction and spread of native ferns and other plants and for converting forest to open grassland (Lyon 1921; Merrill 1941). Lyon (1922) described how Hilo grass could overtop and eliminate herbaceous undergrowth and smother native tree seedlings. He observed that ‘ōhi'a (Metrosideros polymorpha) seeds scattered over Hilo grass never became established as seedlings.

Hilo grass remains a problem in forest reserves and national parks in Hawai‘i, particularly at lower elevations. In Kipahulu Valley of Haleakala National Park, Hilo grass is abundant below 1,000 m (3,280 ft) elevation and has apparently been increasing in cover over the last 20 years, perhaps because of disturbance from feral pigs (Sus scrofa) (Yoshinaga 1980). Between 1980 and 1986, Hilo grass expanded its range upslope in Kipahulu Valley by more than 180 m (600 ft) elevation and by 1986 was found to dominate ground cover below 1,140 m (3,740 ft) (Anderson et al., in press). Re-examination of an exclosure at 975 m (3,200 ft) elevation in Kipahulu Valley (after 10 years of protection and drastic reduction of the feral pig population) suggests a small decline in Hilo grass, recovery of fragile native ground ferns, and reproduction of native species (A.C. Medeiros and L.L. Loope, pers. comm.). The future recovery of forests heavily invaded by Hilo grass appears jeopardized, because it is not certain that infested areas will ever completely recover, even in the absence of disturbance from feral animals. Biological control research is not planned for this species (Smith 1985).

Palmgrass (Setaria palmifolia), introduced from tropical Asia before 1903, has become naturalized on O'ahu, Lāna‘i, Maui, and Hawai‘i (Wagner et al., in press). Currently, this tall (to 2 m or 7 ft) perennial grass may be found in wet forests to an elevation of 2,000 m (6,560 ft), with major infestations in the Ko‘olau Forest Reserve on Maui, in four reserves on windward Hawai‘i, and in the ‘Ola‘a Tract of Hawaii Volcanoes National Park (Smith 1985). In a recent survey of ‘Ola‘a (1988-89), approximately one quarter of the Tract was found to have a dense ground cover (>50%) of palmgrass (S.J. Anderson et al., unpubl. data). By contrast, when Jacobi and Warshauer (1975) surveyed the ‘Ola‘a Tract 14 years ago, they found the species only along trails and in very few disturbed localities. When this large grass invades forests, it may form very dense stands and shade out all other herbaceous vegetation (Smith 1985). Fosberg (1948b) noted that palmgrass, in less than 20 years, spread from a single colony to become the dominant herbaceous plant along a trail in Mānoa Valley on O‘ahu. He also observed that once established, palmgrass eradication is very difficult.

A third grass with disturbing potential in both wet and mesic habitats is meadow ricegrass (Ehrharta stipoides, syn. Microlaena), a native of Australia, New Zealand, and the Philippines; it arrived in Hawai‘i before 1916 and is now found on O‘ahu, Maui, and Hawai‘i (Wagner et al., in press). Meadow ricegrass is capable of rapidly invading disturbed sites and forming dense stands. It has become established over large areas in the Puna, Ka‘ū, and South Kona Districts of the island of Hawai‘i (Smith 1985) and has also invaded important endangered bird habitat at Hakalau Forest National Wildlife Refuge (Stone et al., in press). While the effects of meadow ricegrass have not been fully assessed, it is capable of growing in the moderate shade of many mesic forests, and its dense ground cover may be capable of inhibiting the reproduction of
native forest plants. The disappearance of colonies of the legume *Vicia menziesii*, Hawai‘i's first listed endangered plant species, has been associated with meadow rice-grass invasions subsequent to disturbance from clearing and logging activities (Clarke et al. 1983).

**Invaders of Dry and Mesic Habitats.** Most of the dry lowlands of the Hawaiian Islands are completely dominated by alien plants, particularly shrubs and grasses (Smith 1985). A few of the worst invaders of remaining native plant habitats in dry and mesic regions are discussed below.

**Trees: Firetree, Silk Oak.** Firetree, or faya tree (*Myrica faya*), has attracted a great deal of attention and concern for its recent explosive increase on several Islands and its ability to invade nearly intact native ecosystems. Native to the Azores, Madeira, and the Canary Islands, firetree occurs there in laurel (*Laurus azorica*) forests from sea level to 610 m (2,000 ft) elevation and is also found in pastures within its native range (Hodges and Gardner 1985).

Firetree was introduced to Hawai‘i before 1900, probably by Portuguese immigrants as an ornamental, a source of fruit for wine-making (Wagner et al., in press), or for firewood (Hodges and Gardner 1985). Subsequently, the Hawaii Sugar Planters' Association obtained seeds from a Portuguese farmer on Hawai‘i (Whiteaker and Gardner 1985), and the tree was planted in ten forest reserves on Kaua‘i, O‘ahu, and Hawai‘i, primarily in 1926-27 (Skolmen 1979). L.W. Bryan recorded collecting *Myrica* seed from Hawai‘i and sending it to a forest supervisor on Kaua‘i in 1926 (Bryan 1926a). By the mid-1980s, firetree had infested more than 34,000 ha (83,980 a) Statewide, with most of the infestation on the island of Hawai‘i and smaller acreages on Maui, Lāna‘i, O‘ahu, and Kaua‘i (Whiteaker and Gardner 1985). The Maui infestations occur primarily on ranchlands, and the O‘ahu firetree distribution includes remnant ʻōhia (Metrosideros polymorpha) cloud forest and mesic shrublands of the southern Wai‘anae Mountains. On Kaua‘i, firetree has invaded wet and mesic montane rain forests of Kōke‘e State Park. On Hawai‘i Island, firetree occurs in pasturelands of the Hāmākua and Ka‘ū Districts and on the slopes of Hualālai, as well as in native rain forests and open ʻōhiʻa forest and shrublands in and near Hawaii Volcanoes National Park (Whiteaker and Gardner 1985).

The history of firetree in Hawaii Volcanoes National Park illustrates the rapidity of its invasion and intensification. In 1961, only one individual tree was known to occur in the Park, at Kilauea Military Camp (Fosberg 1966), although others probably existed away from trails and roads. By 1966, approximately 90 ha (225 a) in the Park were infested with firetree, and in just 11 years (1977) this infestation increased explosively to 3,640 ha (9,000 a) (Smathers and Gardner 1979). By 1985, firetree covered more than 12,200 ha (30,150 a) of the Park (Whiteaker and Gardner 1985).

The impacts of firetree on native ecosystems are serious. Given enough time in a suitable site, firetree can form a dense, closed-canopy, nearly monospecific stand of trees 15 m (50 ft) tall, with virtually no plant cover on the ground (Smathers and Gardner 1979; Smith 1985). Lack of ground cover may be due to dense shade produced by firetree canopies, or to some allelopathic effect of firetree leaves (Smith 1985); a relative of firetree native to Florida (*Myrica cerifera*) is known to produce allelopathic substances (Ewel et al. 1982). More insidious is the ability of firetree to fix nitrogen and invade nutrient-poor volcanic substrates of recent origin. On recent cinder substrate, firetree litter adds about four times more nitrogen to the soil than is derived from all other natural nitrogen sources combined (Vitousek et al. 1987).
The addition of large quantities of nitrogen alters the nutrient balance of entire ecosystems and thus may eventually encourage the invasion of nonnative plants that would not otherwise be able to grow as well as native plants adapted to naturally low nitrogen levels in developing Hawaiian soils (Vitousek, in press). Nitrogen-fixing firetree grows far more quickly than does the native ʻōhiʻa, and firetree probably competes with ʻōhiʻa for both water and sunlight (Smathers and Gardner 1979).

Firetree was formerly considered to be a noxious weed by the Hawaii State Department of Agriculture (1962) before populations became "uncontrollable" and is of little value as forage on rangelands (Hosaka and Thistle 1954). The fruit of firetree is dispersed by alien birds, particularly Japanese white-eyes (Zosterops japonica) (Vitousek et al. 1987) and house finches (Carpodacus mexicanus); some native birds also take the fruit (LaRosa et al. 1985). Seeds are also dispersed by the feral pig (Sus scrofa), which may consume large quantities of fruit during periods of heavy production (C.P. Stone, unpubl. data).

Firetree can be killed by several treatments with herbicides (Santos et al. 1989c). The State has for more than 25 years been attempting to control firetree at a major infestation in the Hamakua District of Hawaiʻi Island, with some success (Whiteaker and Gardner 1985). Hawaii Volcanoes National Park managers have had success in controlling firetree in relatively intact systems, called Special Ecological Areas, over approximately 3,850 ha (9,500 a) (Cuddihy et al. 1988). However, the species is so widespread in the Park and elsewhere that biological control is considered the only hope for large-scale, long-term control. Previous attempts at biological control of firetree with several introduced insects were largely ineffectual (Clausen 1978). However, recent trips to the native habitat of firetree have resulted in the identification of potential insect and fungal control agents (Hodges and Gardner 1985; Markin et al., in press), and biological control research is ongoing. It is probable that the present range of firetree in Hawaiʻi, large as it is, is not the full extent of its potential range in the State (Whiteaker and Gardner 1985).

Approximately 15 additional alien tree species are listed by Smith (1985) as important invaders of native dry and mesic ecosystems. Some of these, such as New Zealand laurel (Corynocarpus laevigatus) on Kauaʻi and Leptospermum ericoides on Lānaʻi, are currently restricted to only one island. Other tree species are more widespread, such as silk oak (Grevillea robusta), introduced from Australia about 1880 and planted in more than 40 forest reserves on five Islands (Skolmen 1979). Silk oak has become naturalized at least on Kauaʻi, Oʻahu, and Hawaiʻi (Wagner et al., in press), in dry regions between 350 and 1,600 m (1,150-5,250 ft) elevation (Smith 1985). Although silk oak is a potentially valuable timber species, it is particularly threatening to native plant communities because of its ability to form dense stands and to produce allelopathic substances, which prevent the establishment of other species (Smith 1985). The tree grows well on shallow, rocky soil in low rainfall areas (Nelson 1960); thus, it has been able to invade relatively recent substrates (e.g., in and near Hawaii Volcanoes National Park ʻōhiʻa forests and shrublands).

**Shrubs:** Koa-haole, Lantana, Christmasberry, Gorse, Sourbush. One of the most widespread alien shrubs or small trees of the arid lowlands is koa haole (Leucaena leucocephala), introduced from the Neotropics before 1837 and formerly cultivated for cattle feed and firewood (Wagner et al., in press). Seeds were broadcast over the lowlands, and now this species is found in dry habitats to above 700 m (2,300 ft) elevation on all the main Islands (Smith 1985). Koa haole was apparently kept somewhat in...
check by feral animals in the late 1800s, but with their reduction in many areas, the shrub came to dominate dry lowlands, forming dense stands as tall as 9 m (30 ft) (Egler 1942). In such monotypic stands all other plants are excluded (Smith 1985). Like several other successful invaders of dry habitats in Hawai‘i, koa haole is a nitrogen fixer (Brewbaker and Styles 1984). Although often seen in very disturbed areas, the species also occurs in remnant dry and mesic forests that are important habitats for endangered and rare endemic plant species (e.g., on the upper slopes and ridges of the Wai‘anae Mountains on O‘ahu) (Wagner et al. 1985). Egler (1942) theorized that koa haole stands might be succeeded eventually by native dryland shrub and tree species, as had been observed to occur on Martinique, an island in the West Indies. Forty years later, Smith (1985) found no evidence for this in Hawai‘i and considered such recolonization unlikely, because of the exhaustion of native plant seed banks. Recently, an introduced insect, the leucaena psyllid *Heteropsylla cubana*, has greatly reduced the vigor of koa haole stands (Smith 1985), but the economic value of the shrub precludes a biological control research effort. Koa haole has proven intractable to control efforts using herbicides, at least in Hawaii Volcanoes National Park (J.T. Tuni-son, pers. comm.).

Lantana (*Lantana camara*), native to the West Indies and naturalized in many tropical countries, was introduced to Hawai‘i in the mid-19th century and became naturalized before 1871 (Wagner et al., in press). Apparently, lantana spread rapidly after the introduction of alien fruit-eating birds such as the spotted dove (*Streptopelia chinensis*) and the common myna (*Acridotheres tristis*) (Clausen 1978). Today, lantana is distributed from sea level to above 1,070 m (3,510 ft) elevation, primarily in dry and mesic forests and shrublands (Wagner et al., in press) but also in wet habitats (Smith 1985). The shrub can form very heavy cover and produces allelopathic substances (Smith 1985), so it may be capable of displacing native shrub and herb species. Since lantana is toxic to cattle and has long been viewed as a pest of pastures and dry range-lands (Hosaka and Thistle 1954), it was an early target of biological control efforts. Just after the turn of the century, 23 insect species were introduced from Central America to combat its spread, and eight of these became established. Another round of biological control introductions took place in the 1950s and 1960s, with the result that the spread of lantana was halted in the drier regions of the Islands (Clausen 1978). Even though lantana cover was greatly reduced during the first half of the 20th century, it was apparently replaced, at least in arid areas, by other alien shrubs and grasses (Egler 1942).

Christmasberry (*Schinus terebinthifolius*), native to Brazil, was introduced as an ornamental to Hawai‘i before 1911 and now occupies dry and mesic habitats in the Islands from near sea level to approximately 920 m (3,020 ft) elevation (Wagner et al., in press). By 1962, Christmasberry had invaded 42,000 ha (103,740 a) in the State (Clausen 1978), and its lack of forage value and aggressive nature caused it to be considered a noxious pest in pastures (Hosaka and Thistle 1954). Christmasberry is capable of invading native dry and mesic forests and shrublands, and by forming dense stands it can shade out other plants; it also produces allelopathic substances (Smith 1985). As early as the 1940s, Christmasberry was recognized as an important invader of dry slopes on O‘ahu (Egler 1942). Today, the shrub has replaced native forests and shrublands in much of the southern half of the Wai‘anae Mountains, which have suffered disturbance from fire and grazing animals (Frierson 1973). In the Wai‘anae Mountains, Christmasberry and other alien shrubs and vines threaten the last known population of the rare ‘ōpuhe (*Urera kaalae*) (Obata 1986). Christmasberry is also a threat to lowland plant communities in Hawaii Volcanoes and Haleakala National Parks and Kaloko-
Honokohau and Kalaupapa National Historic Parks. Herbicidal control methods have been developed for Christmasberry, also a pest in U.S. mainland parks (Ewel et al. 1982). In Hawai‘i, biological control insects have been introduced, but they have not been very successful against dense infestations (Clausen 1978).

Gorse (Ulex europaeus), introduced early this century (before 1910) from western Europe, is now found primarily at high elevations on Maui and Hawai‘i (Wagner et al., in press). In a recent survey of gorse distribution, the shrub was found over 8,260 ha (20,415 a) on the southeastern slopes of Mauna Kea, Hawai‘i Island, to about 2,400 m (7,870 ft) elevation, with isolated pockets down to 450 m (1,480 ft). Gorse was also distributed over nearly 5,985 ha (14,789 a) on the northwestern slopes of Haleakalā, East Maui, between 630 and 2,220 m (2,070-7,280 ft) elevation. In portions of its range with heavy infestations, gorse forms dense, tall stands with as many as 60,000 stems/ha (24,290/a) (Markin et al. 1988). Although currently a problem on grazing lands, it is possible that the species may be able to invade open upland forests and subalpine shrublands. Of particular concern are the native high-elevation communities of Haleakalā National Park, which are adjacent to gorse-infested rangelands (Loope et al., in press b). Gorse is a pasture pest in New Zealand, but there, when protected from fire and sheep browsing, the shrub can be succeeded by native forest trees (Allen 1936). Biological control research on gorse is under way, and two insects have been introduced to attack it: the gorse seed weevil (Apion ulicis) (Clausen 1978) and, more recently, a gorse moth (Agonopterix ulicitella) (Critchlow 1988).

Sourbush (Pluchea symphytifolia, formerly called P. odorata), a tropical American species, was introduced to Hawai‘i around 1931, probably accidentally (Hosaka and Thistle 1954). Within 20 years, it had become very common in the dry leeward lowlands of O‘ahu (Fosberg 1948b), and by 1962, 20,000 ha (49,400 a) throughout the Islands were infested with sourbush (Clausen 1978). Currently, this fast-growing shrub occurs on the main Islands from sea level to above 1,000 m (3,280 ft) elevation (Smith 1985). While not generally considered an invader of intact ecosystems on a par with such disruptive species as Christmasberry or koa haole, sourbush can be locally devastating to coastal plant communities and has been blamed for hastening the aging process of anchialine pools (brackish pools with no surface connections to the ocean) through input of massive amounts of litter (Chai et al., in press).

Vines and Herbs: German Ivy, Nasturtium, Coccinea, Mullein. While vines and herbs are typically less important than other life forms as invaders of dry systems, one vine in particular has great disruptive potential. German ivy (Senecio mikanioides), an ornamental from South Africa, was apparently naturalized in Hawai‘i by 1910. Since then, it has become common in North and South Kona Districts and on leeward slopes of Mauna Kea, Hawai‘i Island; the vine also occurs on Maui (Wagner et al., in press). In a survey of the island of Hawai‘i 8 to 10 years ago, German ivy was found between 500 and 2,500 m (1,640-8,200 ft) elevation, primarily on the leeward slopes of Mauna Loa and Hualalai. Although sighted on less than 2% of the area sampled, Jacobi and Warshauer (in press) predicted that German ivy may be capable of establishing on more than 63% of the lands covered by the U.S. Fish and Wildlife Service Forest Bird Survey, which would encompass vast acreages including most of Hualalai, both leeward and windward slopes of Mauna Loa, the summit area of the Kohala Mountains, and upper elevations of Mauna Kea. The impacts of this vine on native plants have not been well studied, but its habit of growing densely into the canopy of native trees (Smith 1985), such as māmane (Sophora chrysophylla) in upper-elevation forests, probably results in structural damage and reduction of available light. German ivy can form
significant ground cover in native koa/ōhi'a (Acacia koa/Metrosideros polymorpha) forests of South Kona on Hawai'i (Clarke et al. 1980), where it may interfere with native tree and shrub reproduction. Although mechanical control may be possible in localized areas (Smith 1985), if the predictions of Jacobi and Warshauer about Senecio distribution are realized, this vine will be a tremendous problem in native dry and mesic systems in Hawai'i in the future.

Another vine with disruptive potential in mesic forests is nasturtium (Tropaeolum majus). An herbaceous ornamental native to South America, nasturtium has been naturalized in Hawai'i since the 1870s and now occurs at middle elevations on the islands of Kaua'i, Moloka'i, Maui, and Hawai'i (Wagner et al., in press). At present, this vine is a problem only in very localized areas, such as the montane mesic forest of Kipuka Puauulu, Hawaii Volcanoes National Park. Prior to control efforts, nasturtium formed a monospecific cover in forest openings, grew into native tree canopies, and smothered small trees and shrubs (Tunison et al., in press).

A much more recent invader is the vine Coccinea grandis or scarlet-fruited gourd, a native of Africa, Asia, and Australia (Wagner et al., in press). Apparently introduced to the State in 1969 and collected first on O'ahu in 1985, Coccinea has become established at several O'ahu localities below 150 m (490 ft) elevation as well as near Kailua-Kona, Hawai'i Island, where it grows quickly, "smothering ground, shrubs, and trees in a solid blanket" (Linney 1986). While early naturalized populations were in disturbed lowland scrub, the vine has recently invaded native dryland vegetation containing endangered plant species (Linney 1989).

The high-elevation mountain slopes of Maui and Hawai'i have suffered less from alien plant invasions of the last 200 years than have the lowlands. However, during this century, common mullein (Verbascum thapsus) has become well established and abundant in Hawai'i’s native-dominated subalpine ecosystems. Mullein is a native of Europe, widely naturalized in North America, and was first reported in Hawai'i in 1932, near the summit of Hualalai (Lyon 1932; Wagner et al., in press). During the next few decades, mullein became established on Mauna Kea and Mauna Loa and now occurs over an area of 2,000 km² (770 mi²) (Juvik and Juvik, in press). Mullein has also recently (1986) invaded the slopes of Haleakala National Park, where two plants were found and removed from roadsides above 2,070 m (6,800 ft) elevation (Anon. 1989c; B. Gagné, pers. comm.). In a recent survey of roadsides on Mauna Kea, mullein was found between 1,625 and 3,300 m (5,330-10,820 ft) elevation, with densities of 160-190 plants per 100 m² near its lower limit (Juvik and Juvik, in press). These high densities may be related to the disturbed nature of the area sampled, but as mullein is reputedly distasteful to animals (Juvik and Juvik, in press), it may have had an unnatural advantage over native plants during past years when large numbers of feral sheep and goats roamed Mauna Kea. Juvik and Juvik (in press) speculated that mullein has usurped the niche of the endangered Mauna Kea silversword (Argyroxiphium sandwicense subsp. sandwicense) subsequent to that rare plant’s reduction by feral animals. Mullein is likely to be a permanent component of the vegetation in heavily invaded areas; the seeds of mullein and other species of Verbascum can germinate after 100 years of burial (Kivilaan and Bandurski 1981).

Grasses: Broomsedge, Molasses Grass, Fountain Grass, Kikuyu Grass. Hundreds of grass species have been introduced into Hawai'i over the last 200 years (St. John 1973), and many have become established in dry rangelands and undeveloped leeward
areas. Several grasses are notable for the amount of area they now cover and the great changes they have wrought in the plant communities they have invaded.

One of the most abundant alien grasses, at least on O'ahu and Hawai'i, is broomsedge (*Andropogon virginicus*). This tall bunchgrass, native to eastern North America, was first collected on Hawai'i in 1924 (Wagner et al., in press). By 1932, the grass was well established in the Kohala District of Hawai'i Island, where it spread from the Kohala Ditch Trail, the presumed site of introduction (Bryan 1977b). Broomsedge has become an important component of many lowland grasslands, ridge tops, and dry and mesic forests and shrublands. The grass ranges to above 1,600 m (5,250 ft) elevation, and at lower and middle elevations it is often the dominant ground cover (Smith 1985). The rapidity of broomsedge invasion may be illustrated by its increase over time in Hawaii Volcanoes National Park. In a 1947 checklist of exotic plants in the Park and a 1959 survey of the Kalapana extension, the grass was not even listed as present (Fagerlund 1947; Stone 1959); but a few years later, broomsedge was abundant from near sea level to 1,370 m (4,500 ft) elevation (Fosberg 1966). Currently, the grass is a dominant or co-dominant ground cover over thousands of hectares in the Park's coastal lowland and middle-elevation dry zones (Parman and Wampler 1977; J.T. Tunison, pers. comm. 1989). At some lowland sites, broomsedge apparently displaced the formerly dominant, native pili grass (*Heteropogon contortus*) (Stone 1959).

Perhaps the most important effects of this grass and the very similar bush beard-grass (*Schizachyrium condensatum*, syn. *A. glomeratus*) are the ability to carry fire, and adaptations to increase cover and range following fires (Sorenson 1977; Smith et al. 1980). In wet areas on O'ahu, broomsedge has been found to increase erosion because it generally is dry and does not transpire during the heavy rainfall periods of winter (Mueller-Dombois 1973). Broomsedge is allelopathic to other plants (Rice in Smith 1985), is an efficient $C_4$ photosynthesizer (Rundel 1980), and is capable of growing well in nutrient-poor soils and under drought conditions (Sorenson 1977). Herbicidal control is not deemed practical for large stands of broomsedge, and biological control research is considered unlikely in the near future (Gardner and Davis 1982).

Molasses grass (*Melinis minutiflora*), native to Africa, was intentionally introduced into Hawai'i in the early 1900s as cattle fodder (Wagner et al., in press). Subsequently, the grass spread into dry and mesic systems at low and middle elevations (Whitney et al. 1939). Besides being used as forage, molasses grass was planted out in dry areas of Moloka'i and Lana'i for erosion control (Munro 1930). This grass produces dense, perennial mats that are capable of smothering other plants (Smith 1985) and preventing seedling growth and native tree reproduction (Scowcroft and Hobdy 1986). Perhaps the most serious impact of fire-adapted molasses grass is its ability to carry fire into areas with native woody plants. While much of the area invaded by molasses grass is in the disturbed lowlands, it also grows on the sides of gulches containing remnant stands of dry and mesic forest with candidate endangered plant species, at least on Moloka'i (Cuddihy et al. 1982b) and O'ahu (The Nature Conservancy of Hawaii, unpubl. data). Molasses grass has also invaded Haleakala National Park on Maui, where in the aftermath of goat control, it has "spread explosively" and developed heavy cover locally. Because of its tendency to spread and fuel intense fires, molasses grass presents a serious threat to rare dry forest plants and all other native vegetation in Kaupō Gap of Haleakalā (Loope et al., in press b).

Fountain grass (*Pennisetum setaceum*), also native to Africa, is another fire-adapted bunchgrass that has dramatically spread since its introduction as an ornamental
into Hawai‘i in the early part of the 20th century. Fountain grass has invaded bare lava flows and open areas on Kaua‘i, O‘ahu, Lāna‘i, and Hawai‘i from sea level to an elevation above 2,000 m (6,560 ft) (Wagner et al., in press) and has been present on Maui for at least 25 years, the legacy of an ornamental planting in Wailuku (Loope et al., in press b). Fountain grass is already a dominant ground cover in dry ranchlands and open vegetation of North Kona and South Kohala on the island of Hawai‘i. Based on environmental conditions in its present range, Jacobi and Warshauer (in press) predicted that fountain grass will be able to invade much more area on the upper-elevation slopes of Mauna Loa and Mauna Kea.

Fountain grass is particularly insidious because it is able to invade lava flows previously dominated by native plants, where it interferes with native plant regeneration, upsets natural succession, and allows damaging fires to occur where native vegetation alone would not have supported fires (Tunison et al. 1989). Fountain grass, by increasing the likelihood of fire, is recognized as a serious threat to endangered plant populations in dry forests on Hawai‘i Island, for example *Kokia drynarioides*, *Caesalpinia kavaiense*, *Stenogyne angustifolia*, *Haplostachys haplostachya*, and *Lipochaeta venosa* (Wagner et al. 1985). Fears of fire in the habitat of endangered trees and other rare species (Powell and Warshauer 1985a) proved well founded when much of the proposed Pu‘uwa‘awa‘a Natural Area Reserve, infested with fountain grass, burned in 1986.

Fountain grass has proven very difficult to manually control in Hawaii Volcanoes National Park, where it infests more than 8,000 ha (19,760 a) of the coastal lowlands. After more than 10 years of concentrated effort at the center of the Park’s infestation, the species had not been eradicated, and Park managers developed a new strategy of confining the grass to the largely disturbed lowlands rather than attempting to remove it throughout its range in the Park (Tunison et al. 1989). If left unmanaged, it is feared that fountain grass could become distributed over all Park lands not covered by rain forest (Tunison, in press b).

On Maui, the State Department of Agriculture has successfully confined fountain grass to two small populations through persistent efforts in manual control, and the number of young plants recruited from the seed bank appears to be declining (Loope et al., in press b). If left untreated, fountain grass would be a serious threat to Haleakala Crater and other upslope natural areas.

Kikuyu grass (*Pennisetum clandestinum*), another African species, has been widely planted in upland pastures in Hawai‘i (Hosaka 1958), where it may grow up to 3,050 m (10,000 ft) elevation (Loope et al., in press b). This aggressive grass has also invaded dry and mesic habitats as well as disturbed wet forests on all the Islands, where it forms thick mats, spreads rapidly by stolons, and produces allelopathic substances (Smith 1985). Of particular concern are rich assemblages of native species, such as the dry forest at Auwahi, East Maui, where kikuyu grass has formed a dense mat that prevents the reproduction of native tree species, some of them extremely rare (Medeiros et al. 1986). Kikuyu grass, along with other alien grasses, is also a problem in Kaupo Gap of Haleakala National Park, where it has been expanding upslope from ranches for more than 40 years (Loope et al., in press).

Even in the largely disturbed leeward lowlands, alien grasses may threaten remnants of native vegetation. In a recent study of one of only two known populations of the rare Hawaiian water fern *Marsilea villosa*, several grasses were found to share the
fern's restricted habitat in a small crater on Koko Head, O'ahu. Of these alien species, ricegrass (*Echinochloa colona*) was recognized as the most serious competitor of *Marsilea* (Wester and Ikagawa 1988). At present, this site is managed by The Nature Conservancy of Hawaii, and volunteer groups are monitoring plant cover and removing encroaching alien grasses.

**Fire**

**Natural Fire Regime and Fire History.** Vogl (1969) proposed that naturally occurring fires, primarily from lightning strikes, have been important in the development of the original Hawaiian flora, and that many Hawaiian plants might be fire adapted. While accepting lightning as a potential ignition source, Mueller-Dombois (1981a) pointed out that most natural vegetation types of Hawai'i would not carry fire before the introduction of alien grasses. Native plant fuels typically have low flammability (Smith and Tunison, in press). However, fire probably influenced evolution of the montane ecosystems of Maui and Hawai'i, which contain grasslands of the native *Deschampsia nubigena* and stands of native shrub species and *koa* (*Acacia koa*). As for rain forest vegetation, Mueller-Dombois (1981a) cited evidence from charcoal layers in soil pits that indicates an extremely low frequency of fire once every 700-1,000 years prior to human occupation of the Islands. Since carbon in soil layers may also represent lava flows or phreatic eruptions rather than free-burning fires, the true incidence of natural fire may be even rarer (Smith and Tunison, in press). Smith and Tunison concluded that although natural fire regimes in Hawai'i are "difficult to reconstruct," they are for most areas "best characterized as fire-independent."

In historic times, a few large rain forest fires have been reported, although it is usually unclear whether burned forests were intact or contained aliens that might have encouraged fires. During a drought in 1901, a fire burned an area 24 km (15 mi) long and 3-6 km (2-4 mi) wide in the southern part of the Hāmākua District of Hawai'i Island (Hall 1904). The composition of the forest was described as ‘ōhi'a (*Metrosideros polymorpha*) and koa with an understory of ferns, which after the fire was apparently followed by a "growth of weeds." This fire is said to have burned for "months" and destroyed forests over an estimated 12,150 ha (30,000 a) (Bryan 1961a). Other large fires apparently occurred in Hilo and Hāmākua forests in the middle of the 19th century, and the native forest trees were able to regenerate (Hall 1904, Horner 1908). Another long-burning fire occurred near Kula, Maui, in the 1880s and reportedly burned for weeks (Nelson 1967).

In the early decades of the 20th century, forest fires in Hawai'i were usually blamed on the flammability of the indigenous matted fern *uluhe* (*Dicranopteris linearis*), which was considered a pest (Judd 1931) and thought to be a nonnative introduction (Bryan 1926b; Hostetler 1970). In 1926, an uluhe-fueled fire burned for a week, destroying much of the Pana'ewa Forest Reserve near Hilo and causing businessmen and homeowners of Hilo to fear for their property (Chamber of Commerce of Hilo 1926). This forest had somehow escaped development during the Hawaiian period despite its proximity to agricultural lands of Hilo (McEldowney 1979) and had been dominated, at least in its interior, by native trees when Isabella Bird passed through in 1873 (Bird 1966). After the fire, the burned area was planted with alien trees (Bryan 1926b), which today dominate most of the remaining forests of the Pana'ewa area.

Although large fires have certainly occurred in predominantly native vegetation (see Smith and Tunison, in press, for evidence of a large fire on Mauna Loa slopes 400 years