# The invasion of Jamaican montane rainforests by the Australian tree *Pittosporum undulatum*

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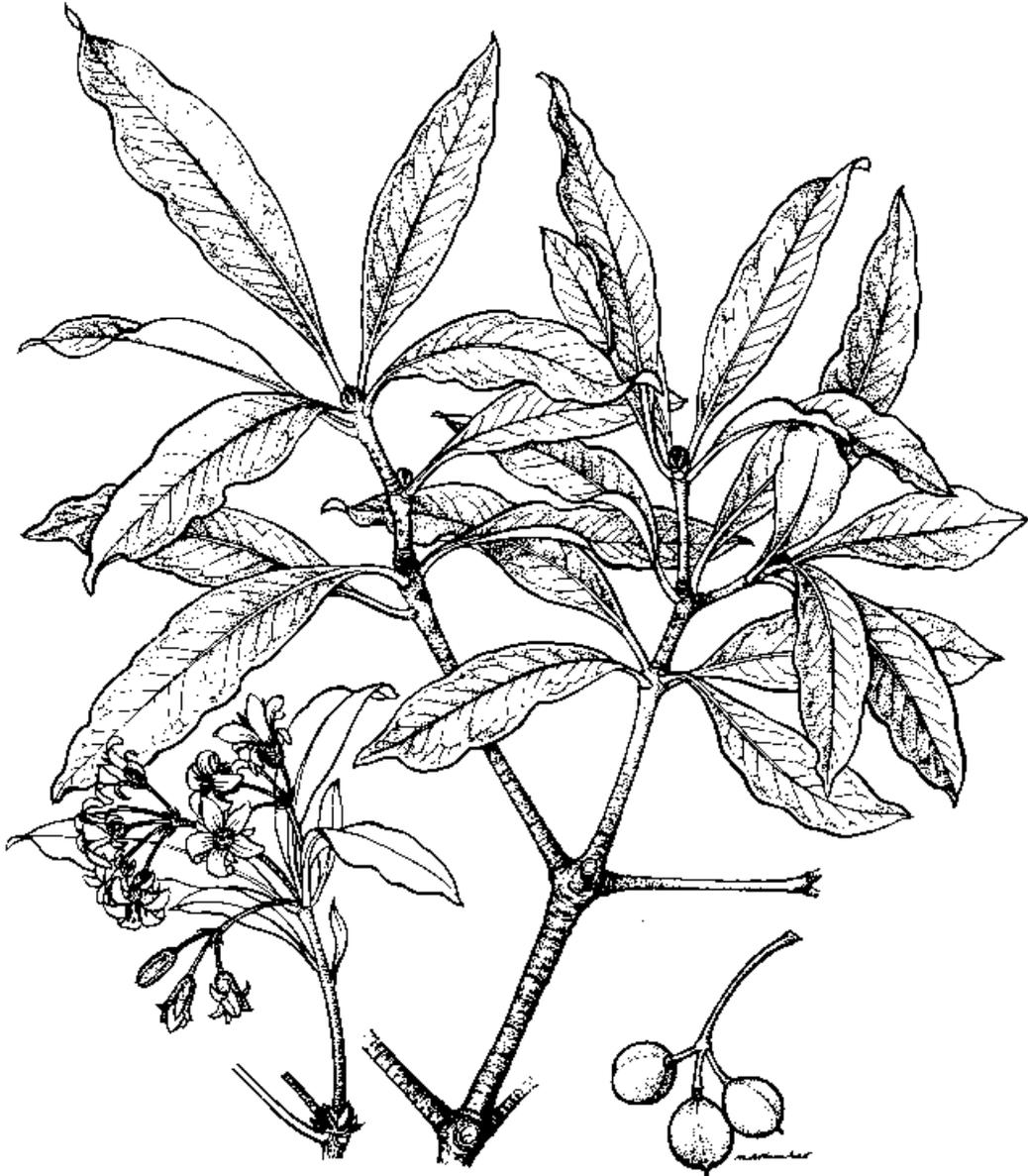
May 1996

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1.

Introduction

This report deals with the problem of the invasive Australian tree *Pittosporum undulatum* in the Blue Mountains of Jamaica. *P. undulatum*, which has the local names wild coffee and mock orange, was introduced to the Blue Mountains in 1883 and has now spread through about 1300 hectares of forest, most of it primary montane tropical rain forest. The report has five chapters. In the first we give necessary background information on the Blue Mountains and on *P. undulatum* itself. In the second we examine the effect of *P. undulatum* on the Blue Mountains - its native biodiversity and ecosystem functioning. In the third we explore the role of the species in the local economy. Chapters 2 and 3 roughly correspond to the second half of a cost:benefit analysis, that is to say the benefits of not having *P. undulatum* in the area, with the complication that *P. undulatum* may have significant positive benefits, especially to local people. The fourth chapter considers the control and management of *P. undulatum*, corresponding to the cost half of the equation. In contrast to the benefits of removing *P. undulatum*, the cost of control is fairly accurately quantifiable, and will be much more so after actual control operations have started. In the final chapter, as a conclusion, we attempt to weigh the costs and benefits, and reach policy recommendations.

There is a considerable body of scientific knowledge on the control of weeds in productive systems and of biological invasions of natural environments. During the development of this subject it has become clear that cost-effective control of weeds and other biological invasions is dependent on the establishment of a scientific understanding of key aspects of the invasive species, the invaded system and the ecology of the invasion process (e.g. Drake *et al.* 1989). Standard procedures now exist for utilising this scientific information in the development of an effective control programme. The need for careful scientific study before drastic control measures are implemented is especially important for areas of high conservation importance containing populations of endangered species (Drake *et al.* 1989). Therefore, it is recognised by both scientists and managers that it is essential for this approach be adopted for the *P. undulatum* invasion of the Blue Mountains.

1.1   The Blue Mountains

1.1.1 Topography, geology and soils

The Blue Mountains are geologically young, steep and highly dissected. The Grand Ridge forms the backbone of the range and extends for approximately 16 km east-west. Whilst Blue Mountain Peak reaches 2256 m, (the highest point in Jamaica), the mean height of the Grand Ridge is about 1800 m. Slopes are sometimes steeper than 70^0 and frequently steeper than 50^0 (Muchoney *et al.* 1994). The Blue Mountain Inlier is dominated by volcanic rocks (basalt) of the early Cretaceous period into which granodiorites were intruded in the late Cretaceous and Paleocene, with minor sedimentary (limestone and shales) and metamorphic units (blue and green schists and amphibolites). The soils are generally thin, porous, subject to heavy leaching, rather deficient in P, but not in N and K (except perhaps for Mor Ridge forest sites) (Tanner 1985). Decomposition of organic matter is slow, especially at higher altitudes (Nisbit 1994). Landslides are common and widely distributed (Dalling 1992). Earthquakes may have triggered massive landslides in the past, as a small one (5.5 on the Richter scale) in January 1993 was sufficient to cause small landslides and branch fall from trees.

1.1.2 Climate

The mean annual rainfall at the Cinchona Botanic Gardens, the place of introudction of *P. undulatum*, is between 2500-3000 mm (Tanner 1980). It rises to about 5000 mm on the Grand Ridge of the Blue Mountains to the north-east, and falls to about 2000 mm in the Yallahs Valley below Cinchona. There are two wet seasons, the main one in October and November and a minor one in May, but the seasonality of rainfall is rather unpredictable. The higher levels of the range are usually cloud-covered during the afternoon.

Table 1.1 Temparature data (^{o}C) for Cinchona (altitude 1500 m) and forest between 1555 - 1615 m (modified from Tanner 1980**).**

	Cinchona[1]	Forest[2]
Mean of monthly maxima	21.1	19.6
Mean of monthly minima	13.6	11.5
Extreme maximum	27.5	22.6
Extreme minimum	8.0	9.2

1. The mean of readings from the years 1891-1905, 1944-55 and 1 March 1974 - 28 February 1975.
2. The mean of four forest types in Tanner (1977), between 19 February 1974 - 7 March 1975.

Hurricanes occur irregularly, the last major one, Hurricane Gilbert, struck the area in 1988. It caused severe disturbance to the natural forest, including defoliation, snapping and uprooting of trees, and landslides (Bellingham 1991).

gure 1.1. Jamaica, showing the Blue and John Crow Mountains, Kingston and parish boundaries

1.1.3 Vegetation of the western Blue Mountains

There is no generally agreed classification of the forests of the Blue Mountains. Grubb & Tanner (1976) recognised seven types for the western end. Bellingham (pers. comm., 1994) carried out a classification, Twinspan, using data from 88 plots, and also found seven forest types, though not exactly the same types. A recent and comprehensive broad scale classification, based on forest structure and species composition, is given in Muchoney *et al.* (1994). Forest below about 1300 m on the southern slopes has been largely removed, and now the land is a complex mixture of remnants of natural forest, pine and coffee plantations, shifting agriculture, and land in all degrees of abandonment. Much of this deforested land within a few kilometres of Cinchona is either covered by the introduced grass *Melinis minutiflora* or by relict pockets of disturbed native vegetation much invaded with alien weeds.

There have been no published analyses of the species richness (flowering plant or otherwise) of the Blue Mountains, though some people have made more or less accurate calculations. The latest estimate is that there are 600 species of flowering plants in the Blue and John Crow Mountains National Park, 275 species and 14 varieties of which are endemic to Jamaica and, of these, 87 are endemic to the four easternmost counties of Jamaica (which contain the park) (Iremonger in press). Another study (Wood 1992) estimated that there are 110 tree species (including 11 tree fern species) in the Blue Mountains over 4500 feet (1370 m, the approximate lower altitudinal limit of Upper montane rain forest in the Blue Mountains). According to Adams (1972) and Proctor (1985) fifty-two of these species are endemic to Jamaica and ten of these are also endemic to the Blue Mountains. According to Lack (1976), the islands of the West Indies, including Jamaica, are unusual in having oceanic elements (i.e. high endemicity) despite being relatively close to mainland - North, Central and South America. Jamaica has 822 (27%) endemic flowering plant species, compared with 51% endemicity in Cuba and 30-33% in Hispaniola (Kelly 1988), though future monographic work may well alter (probably reduce) the Jamaican figure (C.D. Adams, pers. comm., 1992). Jamaica has only 4-7 endemic plant genera, low compared with truly oceanic islands.

1.1.4 Current use of natural resources in the area

The lower slopes of the Blue Mountains are quite densely populated (with an average population density of about 100 people km^{-2}), with most people heavily dependent on farming. Farmers clear areas, of up to a couple of hectares, of mostly secondary forest and scrub all around the intact core of more primary forest, usually for the cultivation of vegetables for the Kingston markets. They then abandon the land after a few years. Larger areas have been cleared by larger landowners for coffee, especially within the last ten or so years on the northern slopes, near the villages of Shirley Castle and Claverty Cottage, and in the Buff Bay Valley. The effect of these coffee planting projects on the spread of invasive weeds is not known. Fuelwood is gathered from accessible land, though most of it is dead or live branchwood, so the activity seems to have little impact on forest cover. A more serious threat is the cutting of large *Juniperus lucayana* trees for timber, though this illicit practise has been stopped in the Cinchona area since 1994 by park authorities. Little local use is made nowadays of the more inaccessible areas of forest, apart from occasional pig-trapping. Pigs, introduced by the Spanish, have been hunted for hundreds of years, with a population of perhaps only a few hundred. Setting traps for pigs seems to be getting less popular than even five years ago, so that perhaps as few as five men do so regularly in the forest of the western end of the range (R. Bryan, pers. comm., 1993).

1.1.5 Blue and John Crow Mountains National Park

The Blue and John Crow Mountains National Park was officially opened in February 1993, and is Jamaica's only terrestial national park. It covers a planimetric area of 48,284 hectares, an actual area (i.e. taking topography into account and based on a Digital Elevation Model developed by The Nature Conservancy) of 79,666 hectares, and a perimeter of 289 km (Muchoney *et al.* 1994). The park's management plan proposes the setting up of Special Conservation Sites (areas of less than ten hectares) and Special Conservation Zones (larger areas) to safeguard the habitats of particularly threatened biota, and/or areas of particular research importance. In more accessible areas within the park, the establishment of Sustained Yield Zones is proposed where the use of native species for productive use and reforestation is specifically recommended (Kerr *et al.* 1993). A buffer zone around the park is also planned, (provisionally about one kilometre wide, though this may vary in width depending on need and land ownership).
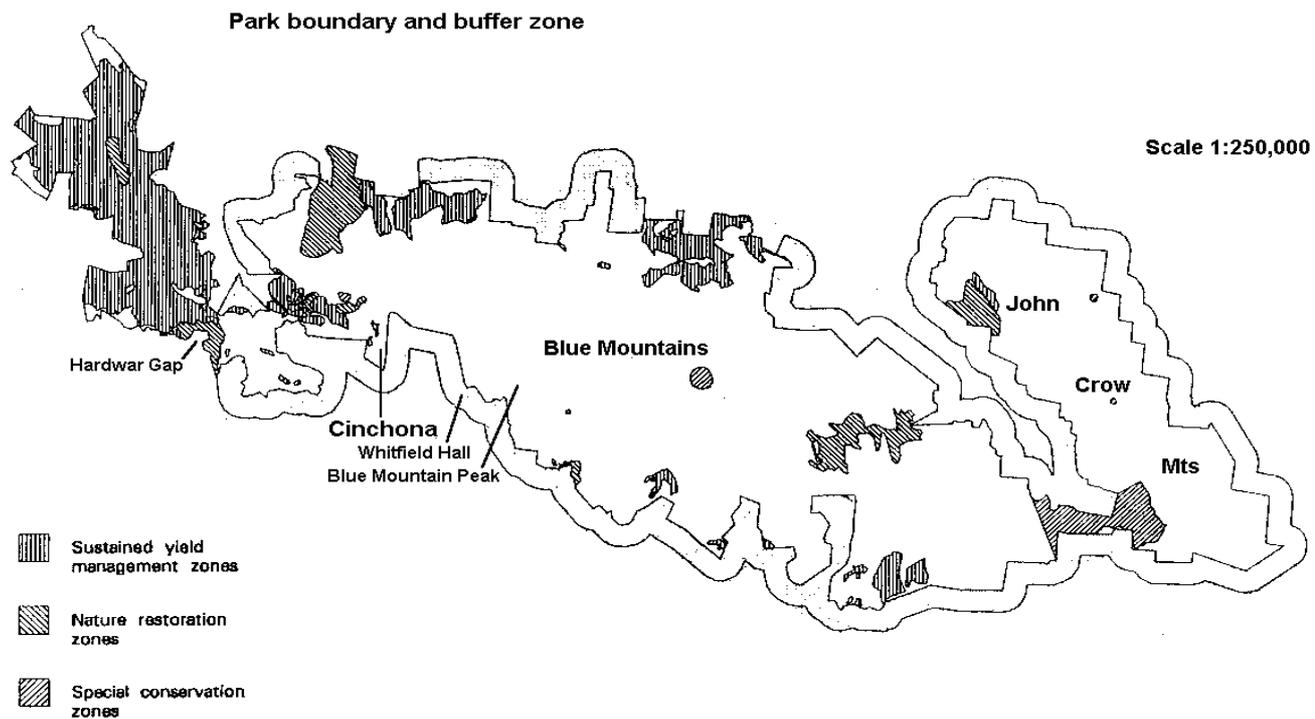
Figure 1.2 Blue and John Crow Mountains National Park. The buffer zone, three management zones and the location of the Cinchona Botanic Gardens are shown.

1.1.6 Threats

There are many threats to the forests of the Blue Mountains and these are described to give context to the threat from *Pittosporum undulatum*.

Deforestation. A priority of park managers has been to control the cutting of trees within the park. This has been partly a result of the recognition that deforested areas are frequently colonised by invasive alien plants.

Fire. Regular fires are a major problem where tree cover is absent or sparse, particularly on the drier southern slopes and especially where the alien grass *Melinis minutiflora* has established.

Exotic animals. Rats and pigs were introduced during the early years of European settlement, and the former are now common in the forest. Mongooses were introduced to control the rats in sugar plantations and are now also common in the park. These species have decimated bird and reptile populations on some oceanic islands (OTA 1993) and the damage they may have caused in the Blue Mountains is not known. They may have been a factor in the probable extinction of the Black-capped petrel (*Pterodroma hasitata* or "Blue Mountain duck") last seen in the 19th century. Introduced pigs have caused significant destruction to native vegetation in many islands and have been cited as one of the main factors leading to the invasion of alien plants in Hawaii (Stone *et al.* 1992), though the pig population is too low in the Blue Mountains to cause much damage. Goats are commonly kept, and some have escaped to become feral, though they do not tend to go far into forest.

Hunting and collecting. People shoot birds, often out of the legal hunting season and sometimes of rare and protected species such as the Mountain witch (*Geotrygon versicolor*), an endemic ground-living pigeon. Although

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there are about 60 orchid species in the montane forests of the western Blue Mountains (Adams 1972) few of them are spectacular and there is not thought to be much collecting of them or any other plant species, (although orchid collecting is a serious problem in other parts of Jamaica) (J. Littau, pers. comm., 1994). Tree fern (*Cyathea* species) caudices ("trunks") are collected in Jamaica for horticulture, and this happened a few years ago in the Cinchona area. Also, a burnt patch of forest of about a hectare that is clearly visible on the southern slopes of the Blue Mountains was apparently caused by pig hunters who started a fire to drive their quarry out of thickets (L. Stamp, pers. comm., 1992).

Tourism. Visitor pressure is high along the trail leading to Blue Mountain Peak but is otherwise low. There are a few old trails in the mountains, sometimes well constructed, but almost all these are now overgrown. The park authorities are trying to promote hiking and ecotourism and this may lead to the opening up some of these trails, but this now seems less likely because of the risk of letting illegal loggers and alien plants deeper into the park (D. Lee, pers. comm., 1994).

Invasive plants. We think that three alien weeds seriously threaten the forests of Blue Mountains, *P. undulatum*, *Hedychium gardneranum* (a shade tolerant rhizomatous herb) and *Polygonum chinense* (a scrambling gap coloniser). Further information on *H. gardneranum* and *P. chinense* is given in the appendix, together with less detailed information on other weeds that may be potentially threatening.

1.2   Pittosporum undulatum

1.2.1 Taxonomy and reproductive biology

Pittosporum undulatum Vent. is in the family Pittosporaceae; sub-family Pittosporeae, series Bivalvae. It is a fairly well-defined species, though it hybridises with *P. bicolor* in Victoria, Australia to a limited degree, and may also do so with *P. revolutum* further north in Australia (Cooper 1956). Vigorous growth of the *P. undulatum* × *P. bicolor* hybrid in Victoria is resulting in "serious swamping" of *P. bicolor* (Carr *et al.* 1992). Weedy species usually have a depauperate genetic structure (Burdon & Marshall 1981) and this can be particularly pronounced when introduced to a new location in small numbers because of a "bottleneck effect" (Harper 1977). We think that *P. undulatum* was introduced to Jamaica in very small numbers, so presumably the population is likely to have a narrow genetic range. We do not know from which part of its native range the *P. undulatum* in the Blue Mountains originated. A formal taxonomic description is given below, from Cooper (1956).

> **Leaves** alternate, elliptic-oblong to oblanceolate, acuminate at apex, attenuate at base, entire, 6 - 16 cm long, 1.5 - 5.0 cm broad, green above, paler beneath, tomentulose when young, membranous, margins undulate or flat, sometimes recurved; **main vein** sunken or raised adaxially, raised abaxially, secondary nerves about 12 per side, anastomising, distinct; **petioles** 7 - 26 mm long, 1 - 2 mm wide, tomentulose when young, glabrate. In the Blue Mountains the leaves of *P. undulatum* are usually between 8-15 (sometimes 20) cm long and 3-4 (up to 7) cm broad; the flowers are about 12 mm long and the capsule also 10-14 mm in diameter (from Adams (1972) and additional observations). The population in the Blue Mountains appears to rather uniform, with any variation in such parts of the plants as leaves or capsule size seemingly related to the local environment of the plant.
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> **Inflorescences** terminal, 4 - 15 flowered, subumbelliform; **peduncles** and **pedicels** 1.9 - 3.1 cm long, accrescent in fruit, tomentose; peduncles subtended by a whorl of leaves and numerous caducous, brown-tomentose, ciliolate bud scales 1 - 2 mm long; **sepals** frequently connate in a tube which splits into 2 parts, one 1 - 2-lobed, the other 3- to 4- lobed, lanceolate, acuminate 6.5 - 10.5 mm long 1.2 - 2.5 mm broad, tomentoluse, falling before the petals; **petals**, linear-lanceolate, to linear-oblong, obtuse, 11 - 17 mm long, 3 - 4 mm broad, coherent at the base, spreading to recurved above, creamy-white; **stamens** 5 - 11 mm long, sometimes reduced to sterile rudiments 0.5 - 1.0 mm long; **anthers** lanceolate-oblong to sagittiform, 2.5 - 5.0 mm long, 1.0 - 1.5 mm broad. Pistils at anthesis slightly shorter or longer than the stamens; **ovary** 4 - 6 mm long, 1.8 - 4.0 mm broad, tomentulose; **style** 2.0 - 4.5 mm long; stigma capitate and obscurely 2- to 4-lobed or almost truncate. **Capsules** sub-globose, 2-valved, 10 - 14 mm in diameter yellow to brown, smooth to slightly rugose, glabrous; **valves** convex in transverse section, less than 1 mm thick, coriaceous, with placenta thickened at the base and bearing 2 - 3 rows of short stout peg-like funicles from the base to just above the middle; **seeds** 12 - 22 (-35), orangy red to dark red to black, irregular.

Flowers which may be female have capitate obscurely lobed stigmas, styles 2 - 3 mm long, and ovaries 4.0 - 4.5 mm long and 2 - 3 mm broad. The stamens are rudimentary, gland-like structures 0.6 - 1.0 mm long. Flowers which may be male or bisexual have weakly capitate to truncate stigmas, styles 3.0 - 4.5 mm long, ovaries 3.5 - 5.0 mm long, 1.5 - 2.0 mm broad, stamens 8.5 - 11.0 mm long, and functional anthers 3.5 - 5.0 mm long and 1.0 - 1.5 mm broad.

The flowers are insect pollinated, bees being important pollinators in the Blue Mountains (Tanner 1982). The capsules dehisce to reveal about 30 seeds which birds eat and may disperse. During May 1995 we carried out a survey of 60 *P. undulatum* trees, mostly in secondary forest on Bellevue Peak, to try to determine their sexuality. Thirteen (21.6%) had only what seemed to be female flowers, with stamens less than one third the length of the style. The remaining 78.4% of the *P. undulatum* trees had only flowers that appeared to be hermaphroditic, stamens longer than the style. The reproductive characteristics of a single *P. undulatum* plant was monitored in Auckland, New

Zealand by Cooper (1956). The flowers had plump ovaries and apparently aborted anthers; the fruits were fully developed and the seeds were viable; there was no male plant in the neighbourhood and the source of the pollen was not known, but presumably come from the same plant.

1.2.2 Growth habit and longevity

Pittosporum undulatum has a very variable growth habit. In Australia, when growing in dry exposed locations, it is a shrub with a maximum height of 2-3 m, with branches from the base as well as the main trunk. In favourable situations it can reach 10-30 m tall, often with a straight trunk and regular branch arrangement. It apparently reaches 30 m height only in the north of its range in Queensland (Cooper 1956). In sheltered sites in the Blue Mountains we have measured trees up to 23 m high. We have little information on the longevity of *P. undulatum* trees. We have found a few (less than 10) dead *P. undulatum* trees in the lower Clydesdale valley (within a kilometre of Cinchona), which have "died back" either as a result of senescence, disease or adverse environmental conditions. The species has a striking ability to extend branches towards higher light levels, particularly noticeable when growing beside a longstanding open area, such as a road or landslide. Based on the account given in Cooper (1956), and our own observations in Jamaica, the growth pattern of the shoot system is as follows. In December to February inflorescences develop in terminal positions on leading shoots. A few weeks later, buds appear in the axils of leaves beneath the inflorescences, and some of these develop rapidly and assume a leading position, while others develop more slowly and form whorls of lateral branches. If flowers abort they are forced to one side, and shed, by an axillary shoot gaining a leading position. If the flowers are fertilised, the vegetative buds remain dormant in the axils of the subtending leaves and the cluster of capsules persists (for as long as 1-3 years) as a terminal structure.

1.2.3 *P. undulatum* in its native habitat

There are between 140 and 200 species of *Pittosporum* in the world, ranging from the Canary Islands, through West and East Africa, eastern Asia to Hawaii and Polynesia, but with the main concentration in Australasia (Heywood 1985), where there are 49 species (Beadle 1981). *P. undulatum* occurs naturally in the sub-humid forests seawards of the Great Dividing Range, from southernmost Victoria (39^0 south) to the Macpherson Range in southern Queensland (24^0 south), extending inland up to 280 km (Cooper 1956). Rainfall regimes vary considerably throughout its native range (with annual means between 800 and 1500 mm), but in all cases annual rainfall is well distributed throughout the year. It is found in Thornthwaites climatic types B B'r, C B'd and C B'r, meaning temperate rainforest with rainfall well-distributed throughout the year, temperate grasslands with rainfall well-distributed throughout the year and temperate grasslands dry at all seasons (Cooper 1956). It is found on a range of soil types, including alluvial soils, krasnozems, red or yellow podzolic soils, skeletal soils (Gleadow and Ashton 1981) and also old sand dunes, sometimes within range of severe salt spray (Cooper 1956). It is able to withstand drought in south-east Australia, so is often dominant in edaphically drier communities (Narayan 1993). At the other extreme it is able to withstand seasonal waterlogging (Bedggood *et al.* 1989).

P. undulatum occurs sporadically in notophyll vine forests in New South Wales, but is more common on their margins in tall open (wet sclerophyll) forests. In eastern Victoria it occurs in microphyll vine forests (especially those in drier gullies), as well as in wet and dry sclerophyll forests (Gleadow and Ashton 1981). A more detailed description is given by Beadle (1981):

Eucalyptus saligna Alliance. *P. undulatum* is the second most abundant understorey species in *E. pilularis* forest south of the Hunter River, NSW, and is a common species in *E. botryoides* hind-dune forest less than 15 m tall.

E. pilularis-E. saligna-E. paniculata Suballiance. *P. undulatum* is the most abundant understorey species on the wetter portion of the Wianamatta Shale area near Sydney.

E. pilularis-E. piperita Suballiance. *P. undulatum* occurs on valley sides chiefly on Sydney sandstones. It is usually stunted because of poor soils.

E. maculata Alliance. *P. undulatum* is an occasional understorey plant in this widely distributed alliance on less fertile and drier soils. In stands south of Sydney, mesomorphic species are eliminated by fire, but *P. undulatum* is not one of these.

P. undulatum also occurs in *Ceratophyllum-Diploglottis* forest; *Ceratophyllum-Schizomeria* forest; *Argyrodendron* forest; and *Drypetes* forest.

In the Sydney area *P. undulatum* grows in deep sandstone gullies on the fringes of depauperate temperate rainforest with a canopy height of 10 m. Higher up the slopes it is present in moist locations adjacent to or on sandstone outcrops (where it is smaller) and on shale-capped ridgetops. It is only able to persist on these slopes if there is a low fire frequency or if it is in a low fuel zone. *P. undulatum* on the floor of the gully may live for decades as even severe fires usually jump these areas. In the past it was only a minor component of open, grassy, tall forests but is now common and shades out almost everything underneath. It is the most abundant of the several rainforest species invading bushland on shales and sandstones. This shift from a dry sclerophyll to a mesic system seems to be caused by a reduced fire frequency in many reserves, increased water and nutrients carried by urban storm water

runoff and increased dispersers (especially the Pied Currawong). *P. undulatum* is now considered to be seriously out of balance in its natural habitat, and is being controlled (R. Buchanan, pers. comm, 1994).

In East Gippsland in eastern Victoria:*P. undulatum* was absent from those sites that had burned in extensive forest fires in 1965. Elsewhere, the species seemed to be largely confined to drier, higher light environments, being the commonest tree seedling in the centre of gaps and one of the two principal colonising tree species of open talus slopes and landslide-disturbed areas. Where landslide debris contains an appreciable amount of soil or fine material, pioneer species establish and this vegetation in turn is then invaded by *P. undulatum* and *Acmena smithii*. The species was dominant in much of the forest below the talus slopes, but was not a conspicuous component of the taller valley bottom forest. The largest *P. undulatum* trees in the area reached about 40 cm DBH. *P. undulatum* is virtually absent from the mature forest communities on the Bemm River. It is possible that the severe grazing of the species by insects in the microclimate of the mature *A. smithii*-forest at this site may drastically reduce its competitive ability under shade (Melick and Ashton 1991).

In summary, in its native range, *P. undulatum* is characteristically a species of forest margins, open forest and scrub and is less common in forest understorey; is moderately shade tolerant; casts a dense shade, dense enough to shade out most understorey species; is moderately susceptible to fires; and has recently increased in abundance and range where the soil fertility and moisture is high enough and the fire frequency low enough.

1.2.4 Invasions outside Jamaica

We have found reports of *Pittosporum undulatum* spreading through forest and scrub in fourteen countries, in montane areas of the tropics, sub-tropics and warm temperate areas. Additionally, it is or has been cultivated in Chile, Columbia, Bolivia, France, Israel, China, India (Cooper 1956), California (P.J. Bellingham, pers. comm., 1994) as well as milder parts of the British Isles (Bean 1976). *P. undulatum* is probably the most widely cultivated *Pittosporum* species (Cooper 1956). There has been quite recent interest in the use of *P. undulatum* as a plantation tree in Columbia (Montero & Estevez 1983). It has been reported as being "invasive" in the following countries.

Australia. *P. undulatum* is spreading outside its "natural range" at several locations in western Victoria, on King Island in Bass Strait (Gleadow and Ashton 1981), at Albany in Western Australia (Kloot 1985) and in eastern and western Tasmania (Narayan 1993). It is expanding its range in Victoria, where it is widely planted in gardens (Gleadow and Ashton 1981) and in the Dandedong Ranges National Park where it has been the subject of a control programme (Gillespie 1991). *P. undulatum* invades relatively open canopied forest and forms a dense understorey. In a large plot in a wet sclerophyll forest near Melbourne undergoing the early stages of invasion, Gleadow and Ashton (1981) found only one native plant species beneath dense clumps of *P. undulatum*. It has invaded the Mornington Peninsular where it was described as very dominant, occurring in about 90% of quadrats in one study, and was considered the most serious threat to the forest, being the most invasive alien weed out of 42 weed species (Bedggood *et al.* 1989). The depressive effect on native vegetation, and the considerable changes to forest structure, has lead to *P. undulatum* being controlled in many nature reserves and parks (Narayan 1993). The success of the species in Victoria seems to be in part due to the fire-suppression policies in the state, widespread planting in gardens and the introduction of the European blackbird (*Turdus merula*), which is a more effective seed disperser of *P. undulatum* than any of the native bird species (Gleadow and Ashton 1981).

Atlantic islands. *P. undulatum* is known in the Canary Islands, and Bermuda (Cooper 1956). *P. undulatum* and *P. crassifolium* were planted on Tristan da Cunha (Anon 1960). *P. undulatum* has a wide distribution on St Helena where it has had considerable value as a source of fodder. Although it has a deleterious effect on the island's flora, it is too common for control to be contemplated, nor would control be the highest priority, as there are other serious alien weeds on the island with no economically redeeming features (T. Kendle, pers. comm., 1994). *P. undulatum* dominates once extensive *Myrica faya* woodlands on the northern slopes of Pico in the Azores (Marler and Boatman 1952; Haggar 1988; J. Haggar, pers. comm., 1988). It is a significant problem and local people feel that it should be controlled. *P. undulatum* plantations were established on some of the other islands in the Azores, together with *Cryptomeria japonica*, *Eucalyptus* and *Acacia* species. Plantations of the latter two have a subcanopy of native trees, whilst no native seedlings survive under *P. undulatum* or *C. japonica*. The only other area where native trees survive in the lowlands is in coastal scrub, where *P. undulatum* seems to be relatively intolerant of exposure. Also in the Azores *P. undulatum* is used as a coppice below a tree native to the Azores, *Picconia azorica* (Haggar *et al.* 1989). The following description of vegetation along an altitudinal transect on Pico in the Azores is taken from Marler and Boatman (1952).

"Below 100 m there was little soil left and *P. undulatum* becomes increasingly dominant up to 1200 m. There is a gradual transition through a rather open, cleared woodland characterised by *Pinus pinaster*, to heavily *P. undulatum*-invaded forest. At 200 and 300 m altitude *P. undulatum* trees comprised 56 and 57% respectively of all trees. The *P. undulatum* dominated woodland was 5-7 m tall with a very closed canopy and sparse ground flora, lots of leaves and twigs, and soil only about 75 mm deep. It was evident from the frequency of *P. undulatum* seedlings that it could maintain its populations. Further up, an open *Myrica faya*-*Persea azorica* woodland with a well developed ground flora became dominant. This type of orchard-like woodland owed its perpetuation to sheep grazing, and without this grazing pressure *P. undulatum* would extend its range faster than it was doing. It was occasional at 1050 m where frost occurs".

New Zealand. *P. undulatum* has naturalised in New Zealand (Gleadow and Ashton 1981) though apparently is not a serious problem there (P.J. Bellingham, pers. comm., 1992). The species has for many years been wild along about 1 km of roadside scrub in Quarry Rd, Kaitaia, N. Auckland. The colony probably originated from long-distance dispersal by birds, either from Norfolk Island (where it is an introduced weed of some importance) or from Australia; therefore it is not clear whether it should be regarded as indigenous or naturalised (Webb et al. 1988).

Pacific islands. The species was introduced to Lord Howe Island in 1898 with the intention that it be used for windbreak purposes, (this was never carried out), but it has spread slowly into undisturbed forest (Pickard 1984). It has been introduced to the Norfolk Islands where it has naturalised (Turner *et al.* 1968). Known as 'Snowdrop-tree' in the island [Norfolk]. It occurs on the site of the old Government garden at the Cascades and in other places where it has been doubtless planted." p 783. one tree thirty inches in diameter (Maiden 1903). *P. undulatum* was first recorded in Hawaii in 1875 and is now present in the wild on the islands of Maui and Kaho'olawe (Wester 1992). It was one of 41 alien plant species targeted for control in the Hawaii Volcanoes National Park on Hawaii, species often very localised but thought to be potential threats. Twenty-five *P. undulatum* plants were found in December 1986 and treated by uprooting seedlings and cutting trees and applying the herbicide Tordon RTU to stumps. All these plants had died by November 1986, but it was felt that it was too early to say the species had been entirely eliminated from the park (Tunison 1992; T. Tunison, pers. comm., 1994).

South Africa. *P. undulatum* is spreading into tall scrub and riparian forest in the Cape Province of South Africa (Richardson & Brink 1985). *P. undulatum* was planted in the south-western Cape at least as early as 1901 when a seedling was planted in the Tokai Arboretum (Conservator of Forests, Cape of Good Hope, 1902). The species has been widely planted in gardens around Cape Town and has spread in the south-western Cape particularly. In Southern Africa it occurs in areas with winter rainfall (temperate), all-year rainfall (temperate) and summer rainfall (sub-tropical) (Wells et al. 1986). Two main vegetation types are being invaded in the Jonkershoek State Forest, a mountainous area of sandstone and granite - tall scrub dominated by *Olea europaea*, *Kiggelaria africana*, *Rhus angustifolia* and *Maytenus oleoides*; and indigenous riparian forest patches in mountain ravines along the Eerste River and its tributaries. *P. undulatum* plants are typically clumped around the butts of native trees that bear fruit attractive to birds. The species is considered a minor weed in South Africa because its distribution is severely limited by the lack of suitable microclimatic conditions required for establishment, principally moisture away from the shade of established trees (Kruger *et al.* 1986). It has not yet invaded indigenous forests in the southern Cape, although it appears to have the ability to become a major weed in that biome (Richardson & Brink 1985).

Mauritius. *P. undulatum* is invading montane evergreen forest in Mauritius and forms a forest shorter than the native one (Q. Cronk, pers. comm., 1986).

Portugal. *P. undulatum* is abundant in several hill forest areas in mainland Portugal (H. Scrase, pers. comm., 1992; J.R. Healey, pers. obs., 1993). It is the most frequent species in wet hill forest near Lisbon (J. Haggar, pers. comm., 1988).

1.2.5 Introduction of *Pittosporum* species to the Blue Mountains

Pittosporum undulatum was introduced to Jamaica by Daniel Morris in 1883 (Anon 1883). It was transported as seeds from the Nilgiri Hills in India and these were presumably taken to the Cinchona Botanic Gardens shortly after arrival in Jamaica. *P. viridiflorum* or Green-flowered Pittosporum was introduced at the same time. Three *Pittosporum* species from the Nilgiri Hills, St. Helena, and Australia (so presumably *P. undulatum*, *P. viridiflorum* and an unidentified *Pittosporum* species) were said to be growing "well, and in quantity" at Cinchona in 1884 (Morris 1884). Altogether we have found records of 16 *Pittosporum* species being introduced to Jamaica. It can probably be assumed that most or all of these were taken to the Cinchona Botanic Gardens, rather than any other, lower altitude, botanic gardens in Jamaica, as the genus is predominantly a temperate or sub-tropical one. We have found plants of only two of these species, *P. undulatum* and *P. viridiflorum*, both now common in forest near Cinchona.

2.

The effect of *Pittosporum undulatum*

This chapter examines the effects that *Pittosporum undulatum* is likely to have on all aspects of the natural environment of the Blue Mountains. The first part describes and assesses the current and potential range of *P. undulatum*, the rate at which it is spreading, the degree of dominance it is likely to achieve in different environments and the threat it poses to different vegetation types. This part employs data from 144 sample plots in forest and deals with the invasion and the threatened communities on a rather broad scale. These 144 plots belong to eight "series" of plots (series meaning a set of monitoring or experimental plots involving the enumeration of all trees) established by six people in the western end of the Blue Mountains (see Table 2.1).

Table 2.1 Information on the eight plot series used in this report

Series	Initiator	Date started	Number of plots Undist-urbed	Exper-imental	Plot area[1]	Number of enumerations[2] Trees	Saplings	Seedlings	Perma-nent?
Permanent forest plots	E.V.J. Tanner	1974[3]	46	0	100	5/(4)	None	None	Yes
Experimental gap study	J.R. Healey	1985	3	3	240	5?	5?	5	Yes
Undergraduate plots	T. Goodland	1989	20	0	121	1	1	1	No
Stratified plots	P.J. Bellingham	1991	16	0	200	2	None	None	Yes
HIFE	T. Goodland	1991	10	15	144	2	2	3	Yes
Deforestation experiment	M.A. McDonald	1992	4	12	120	1	1	1	Maybe
$1^0/2^0$ forests comp.	S. Morin	1992	6	0	225	1	1	None	Maybe
SIFE	T. Goodland	1994	6	6	630	1	1	1	Yes
Total			**111**	**36**					

1. The plot areas given are those in which all trees have been recorded, and exclude treatment areas around experimental plots.
2. The size definitions were: seedlings - plants <3 m tall; saplings 3 m tall to 3 cm DBH; trees >3 cm DBH.
3. Ten plots were set up in 1984.

The second part investigates the comparative performance of *P. undulatum* and native species, crucial to an understanding of the effect the alien has on native vegetation. The third part assesses the effect *P. undulatum* has on native vegetation more directly, using a correlative approach and by examining various mechanisms by which *P. undulatum* may have the effect it does. The fourth part investigates the effect of *P. undulatum* on animals and the physical environment of the Blue Mountains. We recognise that it can be very misleading to simply extrapolate the past increase in the extent and density of *P. undulatum* far into the future, so we have made an assessment of what factors may slow and, conversely, accelerate the invasion. We have tried to use our data and other sources as much as possible, but for several aspects of the invasion we have no data, because of lack of time or expertise, or because the subject is intrinsically unquantifiable or poorly defined. Therefore we have had to be somewhat speculative about issues that may be important, but for which we have no data.

2.1   Magnitude of the invasion

2.1.1 Present extent and density

We have been able to determine the present distribution and density of *Pittosporum undulatum* within about four kilometres of the Cinchona Botanic Gardens (the place of introduction) fairy accurately and give this information below. Our information on the extent of two separate *P. undulatum* invasions in the Blue Mountains, at Hardwar Gap and Whitfield Hall, is less complete.

Methods. We have assessed the distribution and density of *P. undulatum* by observation from vantage points (often with binoculars or telescope), extensive exploration of the area, and by calculation of the *P. undulatum* density in sample plots. The distributional information has been recorded as follows.

The two 1:12 500 scale maps (sheets 105B and 115A) that cover most of the western end of the Blue Mountains were joined together. A 16×16 mm grid on clear acetate (giving "cells" of 4 hectare (planimetrically)) was superimposed over all the land which we either have evidence for or strongly suspect that *P. undulatum* occurs as a tree, the southern boundary coinciding with Cinchona; (boundaries: western 76^0 40' 30'; eastern 76^0 37' 40''', southern 18^0 03' 58''', northern 18^0 06' 14'''). Four hectares was used as we have sufficient information at present to make good estimates of density for areas this size, though managers in future may need information at a finer scale, so the cells can of course readily be a divided into four 1 ha cells. The number of *P. undulatum* trees (i.e. stems ≥ 3 cm DBH) was estimated within each of these cells and put into four classes: Class 1 - 1-9; Class 2 - 10-99; Class 3 - 100-999; Class 4 - 1000-9999 individuals per cell. The accessibility of each cell from Cinchona was estimated by calculating the straight line distance between the two. The mean slope angle of this end of the Blue Mountains was estimated, using the Digital Elevation Model of the Blue Mountains and the data on the slope angle in

the 144 plots, to be approximately 35^0. This was used to increase the *P. undulatum* density from the planimetric value, by multiplying it by 1.26.

The frequency of *P. undulatum* was also compared with that of native species by calculating the number of plots that each of the 120 woody species, that occurred in at least one plot, was present in as a tree.

Results. Our estimate is that *P. undulatum* occurs in 330 four hectare cells, i.e. in a total of 1320 hectares, (see Figure 2.1.) Note that the class of a given cell does not necessarily equate with density in that cell, as some cells on the forest boundary are only part forest so may be in a low class even if very heavily invaded. Of the 330, we estimate that 108 are predominantly in secondary forest, some of it old and not necessarily heavily invaded. The positive correlation between past forest disturbance, (mostly around Cinchona and the *Cinchona* plantations in the Sir Johns Peak-Bellevue Peak area), and the density of *P. undulatum* is very striking. Concentrations also occur far down the north slopes, in steep landslide-prone areas.
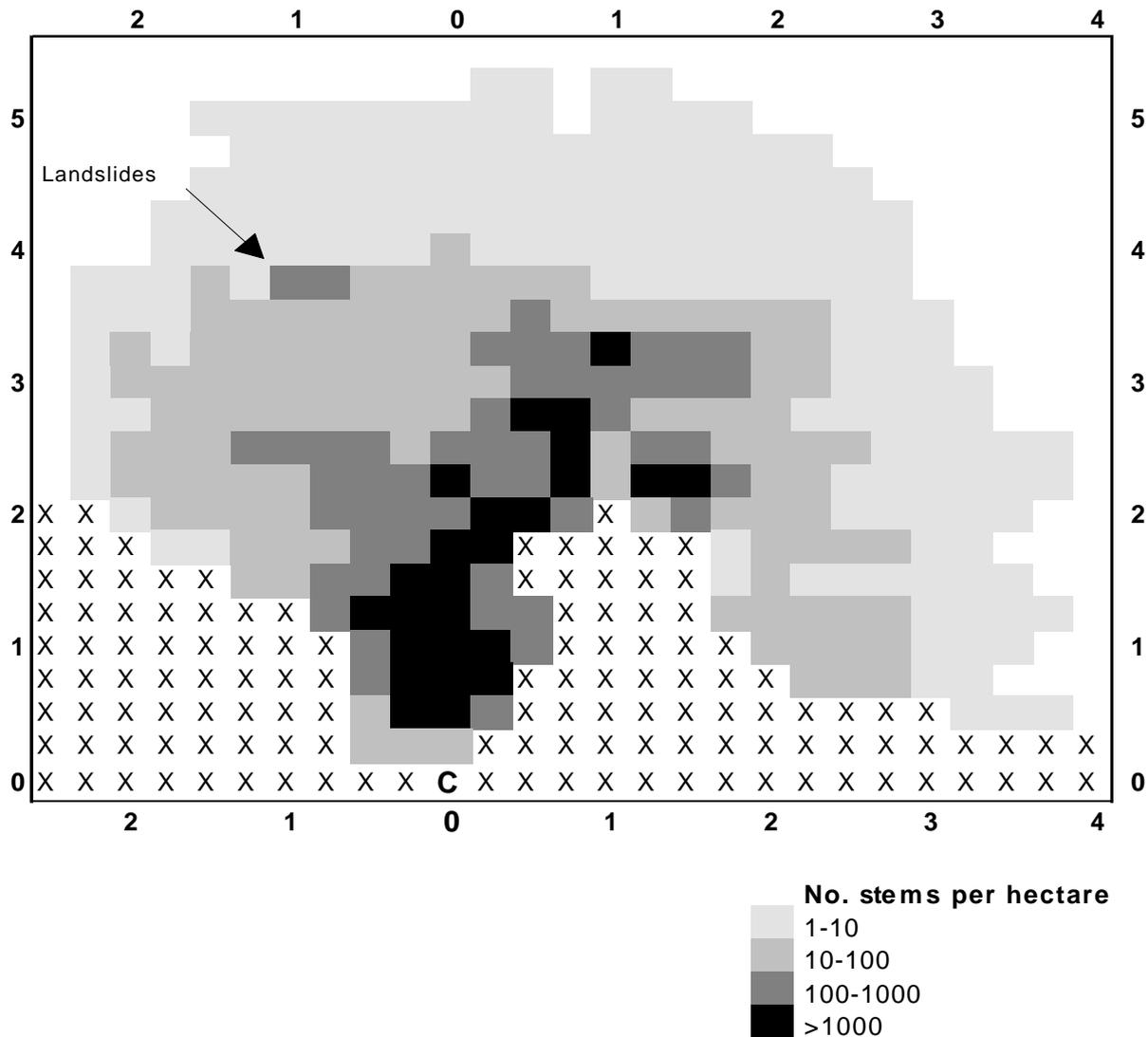
Figure 2.1. The assessed density of *P. undulatum* trees in four hectare cells north of Cinchona (C). "X" indicates that the land is partially or wholly deforested. The distance from Cinchona along an east-west and a north-south axis (in km) is shown around the perimeter of the map.

To put this area in perspective, we have estimated the total area of forest in the Blue Mountains that we think is invadable by *P. undulatum*. We assume that all non-limestone primary and modified montane and sub-montane forest is invadable. The area of these forest types is 44,026 ha (Muchoney *et al.* 1994), which would mean that about 3.0% of *P. undulatum's* potential range has already been "invaded". However, these calculations should be taken as an approximation only, bearing in mind that isolated *P. undulatum* seedlings may occur several hundreds of metres beyond the boundary. And whilst *P. undulatum* seedlings are generally common within it, some areas of up to several hectares will be free of *P. undulatum* seedlings. It is quite possible that there are isolated trees or clumps of trees well outside the area, on the north slopes, in addition to *P. undulatum* spreading from the Whitfield Hall and the Hardwar Gap invasions. *P. undulatum* occured in 83 out of the 144 plots (57.6%), the third most frequent species, remarkable bearing in mind the fact that it has only been in the country 110 years. Of the plots where it was not

present as a tree, it was already present as a seedling in 15 of 16 of the plots for which we have seedling data and altogether we estimate that it occurs as a tree or seedling in 123 (85%) of the plots.

2.1.2 Rate of invasion

Initial spread

Pittosporum undulatum first flowered at the Cinchona Botanic Gardens in 1887, approximately four years after its introduction (Anon 1887). There is no mention of the species in Shreve (1914), a comprehensive account of the plant ecology of the forests of the western end of the Blue Mountains near Cinchona. The first mention that we have found that *P. undulatum* had spread from the gardens was in January 1941, when it was described as regenerating profusely and dominating the woodland understorey near Cinchona (Swabey 1941a). *P. undulatum* "had spread immensely" by 1947 (Porten 1947). In South Africa *P. undulatum* did not start to spread until several decades after its introduction (Knight 1986). In Jamaica the species did not need a similar period of adaptation, as within 62 years from first flowering it had become "perhaps the commonest tree in the (Cinchona) area" (Bengry & Serrant 1949).

Dispersal agents

Planting (presumably as an ornamental in gardens) was probably the source of *P. undulatum* now naturalised near Hardwar Gap (about 6 km WNW of Cinchona) and near Whitfield Hall (about 5 km east of Cinchona). It was being offered for sale at Cinchona (together with *P. tenuifolium* and another member of the Pittosporaceae, *Sollya heterophylla)* for 1 pence per plant between 1887-1892 (Jamaica Bulletin 1892). There is no clear evidence that *P. undulatum* was ever planted nearer to the Grand Ridge of the Blue Mountains than Cinchona. However, other exotic tree species were, most noticeably *Eucalyptus* species that still dominate (by their height) some of the secondary forests north of Cinchona. As *P. undulatum* was planted as a hedge at Cinchona (Public Gardens and Parks 1886) it is quite possible that it was used as such in or near plantations.

P. undulatum seeds are dispersed by birds and the readiness with which birds eat its seeds is common to detailed reports of its invasion elsewhere in the world (Gleadow & Ashton 1981; Richardson & Brink 1985). The distribution of most of the bird species seen or suspected of dispersing *P. undulatum* seeds is through most of the altitudinal range of the Blue Mountains, so we think little can be inferred about its final distribution from knowledge of which species do disperse its seeds. Little is known about changes in populations of bird species in the area, though some species suffer from overhunting (Rhema Kerr, pers. comm., 1992). There are no alien bird species in the montane rainforests of the Blue Mountains, often a factor in the invasion of alien plants. The following birds, all native, were seen by Lack (1976) to take *P. undulatum* seeds: Jamaican oriole (*Icterus leucopteryx*); Greater Antillean elaenia (*Elaenia fallax*); Rufous-throated solitaire (*Myadestes genibarbis*); White-eyed thrush (*Turdus jamaicensis*); Greater Antillean bullfinch (*Loxigilla violacea*). Flocks of Cedar waxwings (migrants) also feed on *P. undulatum* seeds (J.W. Dalling, pers. comm., 1991). If the Ring-tailed pigeon (*Columba caribaea*) is a dispersal agent it may be very important, as pigeons often fly long distances (N. Varty, pers. comm., 1992). The germinability of two hundred *P. undulatum* seeds collected from bird droppings and two hundred collected from trees in the Cinchona area did not differ significantly (T. Goodland, unpublished data). Three separate tests in Australia found no effect of passage through European blackbirds, *Turdus merula*, a close relative of the two *Turdus* species thought to be major dispersers in the Blue Mountains (Cooper 1959; Gleadow & Ashton 1981).

The role of pigs in dispersing *P. undulatum* capsules or seeds is probably not significant as they are only infrequent visitors to the more accessible forest that tends to be more heavily invaded by *P. undulatum*. Piles of up to several thousand *P. undulatum* seeds and gnawed capsules occur near parent trees during the time of seed fall (T. Goodland, pers. obs., 1993). These hoards are probably amassed by rats, but they are unlikely to be major factors in dispersal, as rats are not likely to travel far.

Disturbance history of the Blue Mountains since the introduction of *P. undulatum*

Human disturbance. Much of the original forest in the Cinchona area was cleared, starting in 1868, for *Cinchona* plantations. At their maximum only 75 hectares were ever planted with *Cinchona* and these plantations were largely abandoned by 1883 (Anon 1884). (The same source mentions that the original *Cinchona* plantings had been "completely masked" by the abundant growth of self-sown *Cinchona offinalis* seedlings which had reached "several feet high". *C. offinalis* is now only occasional as a seedling.) Coffee, and to a lesser extent tea, then replaced *Cinchona* as the main plantation crop, but their cultivation was gradually abandoned from the late 19th century and had completely ceased by 1942. Most of the land between Cinchona and the natural forest was pine plantation in 1947 (Porten 1947). Since the 1940s coffee has again become dominant, high prices for Blue Mountain coffee leading to the clearance of most of the forest on Bellevue Peak (the "Sir Johns Peak plantation") during the 1970s. But this was ill advised as the land turned out to be too high for economic coffee production, and most of this area has now become overgrown with alien plants. Large *P. undulatum* trees are common next to trails in the area and occur about a kilometre further north along the Vinegar Hill Trail than in surrounding forest, and they may have played a large part in hastening the spread of *P. undulatum* and perhaps other weeds. The trails were once used

much more than they are at present and were kept open on a regular basis until the 1970s (R. Bryan, pers. comm., 1989). Frequent use led to canopy openness, soil and litter disturbance and perhaps enhanced dispersal by birds using the trails as flyways, and by sticky *P. undulatum* seeds adhering to people. Bird-deposited *P. undulatum* seeds are frequent on logs and stones along paths, and watercourses, during time of capsule dehiscence.

Natural disturbance. Even before the introduction of alien plants to Cinchona in the nineteenth century, disturbance often led to areas becoming dominated by weeds which slowed or prevented succession to forest cover (Harris 1909). For example, the native ferns *Pteridium aquilinum* and *Dicranopteris flexuata* dominated drier ridges following patchy deforestation (Bryant 1949) and a dense cover of ferns (mostly *Gleichenia* species) still dominate patches in the high mountains, said to have been created by forest clearance in "ages gone by" (Hooper 1886). Therefore, to a certain extent, the alien weeds have successfully competed against and competitively excluded native species in certain environments. However, they have probably expanded the range of environments colonised by weeds (especially in more shady and damper habitats) and have probably increased the persistence of weed thickets in these areas. *Chusquea abietifolia*, Jamaica's only native bamboo, can form extensive blankets 2-3 m thick, especially in exposed areas such as ridge tops and areas affected severely by hurricanes. At regular intervals of approximately 32 years the whole population flowers and dies, due to happen again in about 2014. It seems quite possible that the spread of *P. undulatum* may well receive a further boost once this happens, as *C. abietifolia* is very widespread and rampant in some places following Hurricane Gilbert, and there is a very sparse ground layer beneath it when dense, so its death could lead to these areas becoming highly vulnerable to invasion.

Hurricanes have probably played a large part in determining the present distribution of *P. undulatum*, especially in the forest. Twelve hurricanes followed a course that led their centres to pass within 50 kilometres of the study area between 1871 (when accurate records began) and 1994, a mean return period of 10.25 years (Neumann *et al.* 1978; Healey 1990). Between 1903-16 five hurricanes "affected" Cinchona but between 1958 and 1988 there was a prolonged period when hurricanes had little effect on Jamaica. Pigs can cause quite severe local disturbance to the ground flora and soil as they root around for food, triggering the recruitment of *P. undulatum*. In 1989, 11 months after Hurricane Gilbert had caused severe defoliation of the canopy trees, *P. undulatum* seedlings were very dense (about 500 m^{-2}) in an area of a few tens of square metres near Sir Johns Peak that had recently been disturbed by pigs, with only sparse seedlings where the soil was undisturbed. However by September 1991 most of these had died.

Disturbance history of the western Blue Mountains, 1884-1988. The year, nature of the disturbance and estimated distance of hurricane centres from the Morces Gap/John Crow Peak area are given.

1884 Mass flowering of *Chusquea abietifolia* starts, lasting rather more than a year.
1886 20 December. Hurricane; much damage to *Cinchona* plantations; 570 mm of rain (Fawcett 1887); 20 km.
1899 Floods in the Blue Mountains area.
1900 October-November. Many storms, uprooting trees (Fawcett 1900).
1903 Hurricane uprooted 500 plantation trees and damaged >1600 (inc. *Pittosporum* spp)(Fawcett 1904); 40 km.
1905 "Storm"; 50 km.
1909 Torrential rain, including 343 cm in 7 days at Silver Hill; devastating gravity landslide at Whitfield Hall.
1910 Hurricane; 40 km.
1915 Hurricane; 20 km.
1916 Hurricane; 30 km; mass flowering of *C. abietifolia* between 1916-1918 (Seifriz 1950).
1944 Hurricane; 10 km.
1948 Mass flowering of *C. abietifolia* 1948-50 (Proctor 1950).
1951 Hurricane Hazel; led the Yallahs valley to be declared a Disaster Area; 20 km.
1963 Hurricane Flora.
1980 Hurricane Allen. Minor disturbance to natural forest.
1982 Mass flowering of *C. abietifolia* (P.J. Bellingham, pers. comm., 1995).
1988 Hurricane Gilbert.

Estimation of rate of spread in forest

A very crude estimate of the rate at which *P. undulatum* has spread can be made from an assumption about when *P. undulatum* might first have escaped from the Cinchona Botanic Gardens (1887, the year it first flowered), and the maximum extent of its present distribution (excluding the two outlying invasions probably a result of planting). The maximum distances of known *P. undulatum* trees from Cinchona at three perpendicular directions (NW, NE and SE) are about 3.0, 4.5 and 3.0 km respectively, a mean of 3.5 km. Therefore, in 109 years it has spread a mean distance of 3.5 km, equivalent to 32 m yr^{-1}. At this rate the species would reach Cuna Cuna Pass (the eastern limit of the mountain range), 28.5 km from High Peak (at the present eastern limit of the Cinchona invasion) in about 890 years. There are several possible reasons why this simple calculation could be very inaccurate:

- It does not allow for a possible timelag before *P. undulatum* started to spread.

- Birds may have already dispersed seeds much longer distances without us having detected the resulting outlying trees.
- Changes in land use, possible planting of *P. undulatum* near the forest and the preferential spread of the species along trails within the forest (notably the Vinegar Hill Trail) were ignored.
- The effect of Hurricane Gilbert may not have been fully allowed for, so that newly recruited trees on the fringe of the current invasion could soon be producing seed that gets dispersed into extensive uninvaded areas.
- The disturbance regime, primarily from hurricanes, is more likely to increase than decrease from what it has been during the last century, because of climatic change.

The calculation of the rate of spread may be made more accurate with data on the ages of *P. undulatum* trees, especially large ones. This is presently difficult, because of the absence of distinct growth rings in *P. undulatum* tree trunks (Meir 1991); lack of data from large trees of the species (there is only one *P. undulatum* tree over 20 cm DBH in any of the permanent sample plots); the fact that we have only two enumerations of trees from HIFE (so cannot calculate the change in growth rate), and the fact that most of the HIFE plots are in secondary forest, so the growth rate may differ from that in primary forest. With more enumerations from our permanent sample plots we will be able to estimate the rate of spread with greater accuracy, so the next enumeration should provide extremely useful data.

Analysis of the factors affecting the present distribution and rate of spread

We have data on the number and diameter of all *P. undulatum* trees (≥3 cm DBH) in all 144 plots from the western end of the Blue Mountains. Seventy-five of these plots are in forest that has not been experimentally altered and for which we have data from more than one enumeration, enabling a calculation of *P. undulatum* recruitment and growth rates. For all these plots we have data or have made estimates of the following parameters: distance from Cinchona, altitude, aspect, slope angle, soil pH, forest type, degree of Hurricane Gilbert disturbance and degree of past human disturbance (both of these assessed as High, Intermediate or Low) and degree of *P. undulatum* dominance. Time since introduction is another factor, as the enumerations of the plots took place between 1989 and 1994, but this factor is probably quite minor. Although this data set is accurate and complete, the use of multivariate analysis to determine what factors have been most important in controlling the spread of *P. undulatum* has not been attempted. This is mainly because there is clear confounding between some parameters (for example distance from Cinchona and disturbance history), and the range of some parameters is very narrow (such as altitude or pH). Therefore we have explored the relationships between the distribution, recruitment and growth rate of *P. undulatum* and these parameters individually. In summary, we have failed to find many clear relationships. The only strong ones were a positive one between the growth rate and the degree of Hurricane Gilbert disturbance, and a negative one between the growth rate and the degree of *P. undulatum* dominance.

Figure 2.2 shows the relationship between, for each of the 144 plots, the relative basal area of *P. undulatum,* distance from Cinchona, and past human disturbance, classified into three classes: *Great* - definitely highly disturbed by man, i.e. secondary forest; *intermediate* - human disturbance either slight or long ago or disturbance could have been natural; *slight* - no sign of any significant human disturbance. There is a cluster of plots between 2.6-3.0 km distant, along the Grand Ridge of the Blue Mountains, which is roughly tangential to Cinchona. A relationship between increasing distance from Cinchona and decreasing degree of disturbance and *P. undulatum* invasion is obvious. Another cluster of plots at about 1.7 km from Cinchona are in secondary forest but are very little invaded, probably because the forest established soon after the abandonment of *Cinchona* plantations in about 1880, i.e. before *P. undulatum* had started to spread.
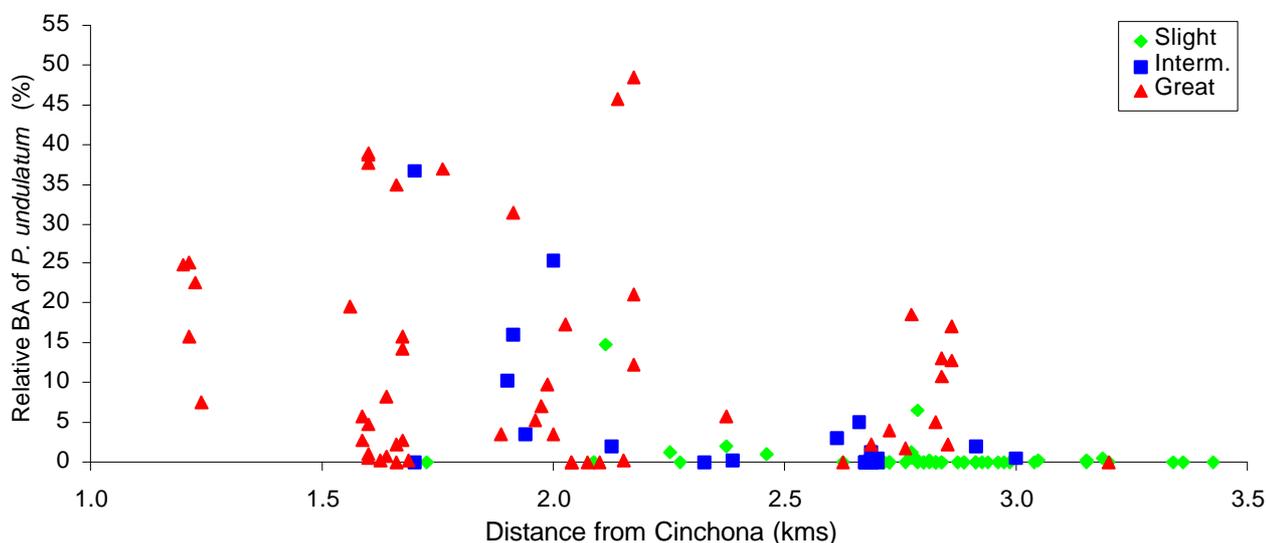
Figure 2.2. Relationship between the distance from Cinchona and the relative basal area of *P. undulatum* in 144 plots. See text for explanation.

2.1.3 Potential extent of the invasion

Here we examine the many factors that may limit the ultimate extent and/or degree of the *P. undulatum* invasion. Some of the factors may seem rather far-fetched, but if the invasion would take hundreds of years before it reaches the east of the range, there is time for the Blue Mountains and the biotic relations of *P. undulatum* to change.

Temperature. In its native range *P. undulatum* experiences a wide range of climates. At sea level in the northernmost part of its range, in southern Queensland (24^0 south), the mean temperature of the warmest month (January) is approximately 26^0C, slightly warmer than the mean temperature of the warmest month (August) on the coast to the north of the Blue Mountains in Jamaica (Clarke 1974). However, Gleadow (1982) found in a laboratory experiment that *P. undulatum* seed germination was at a maximum between 18 and 21^0C and dropped off steeply at higher temperatures, declining to zero at 30^0C. In Victoria, Australia, the species occurs up to 400 m (Cooper 1956) where the mean July temperature is approximately 8^0C, with occasional frosts. Therefore, it does not seem likely that low temperatures higher in the Blue Mountains will limit the species - we have found a healthy *P. undulatum* tree near the top of High Peak at about 2050 m.

Water availability. The rainfall regime varies considerably throughout *P. undulatum's* native range, but annual rainfall is usually well distributed throughout the year. In more inland areas of its native range, and in drier summers, droughts can greatly reduce the survival of seedlings (Gleadow & Ashton 1981). In the Blue Mountains it is relatively resistant to the effects of drought. During the dry summers of 1991 and 1994 most seedlings of many native species wilted noticeably beneath forest canopies. *P. undulatum* was one of the species whose seedlings did not, except when on logs or raised root plates. However, water shortage may be one of the most important factors limiting its distribution outside the forest on the southern slopes.

Climate change. The climate of the Blue Mountains might change so that *P. undulatum* (or any alien species) is unable to invade or survive. Rising temperatures could restrict *P. undulatum* to higher and higher altitudes in the mountains, though most models of climatic warming do not predict large temperature rises in tropical areas. A substantial reduction in the frequency of hurricanes would slow the invasion but probably not stop it, but models predict an increase in the frequency and strength of hurricanes (O'Brien 1992). If climatic change does lead to a much drier climate, or one with longer dry seasons, the forests could become combustible and this could lead to their replacement by alien herbs, such as *Melinis minutiflora*. *P. undulatum* regeneration is severely curtailed by fire.

Soils. *P. undulatum* seedlings grow on limestone outcrops and on limestone-derived soils on John Crow Peak, where the soil pH is in the range 6.4-6.8 (Grubb & Tanner 1976). As the species is only starting to invade this area, it is too early to say how serious a threat it is there. The species can also grow in the acidic soils of Mor Ridge forest, pH in the range 2.8-3.5 (Grubb & Tanner 1976), though apparently its establishment is difficult in the thick humus layer. There was only one *P. undulatum* tree in the 8 permanent sample plots established in Mor Ridge forest, recruited between 1991 and 1994 (E.V.J Tanner and P.J. Bellingham, unpublished data). It establishes, also with difficulty, on bare subsoil such as that exposed on landslides or road cuttings, the seedlings usually being chlorotic and slow growing. Lack of water may be a limiting factor, as the few *P. undulatum* seedlings that can establish on these substrates almost always do so on north-west to north-east facing slopes, though Dalling and Tanner (1995) found that water availability was high throughout the year on landslides in the area - the critical period could be the early establishment of seedlings.

Biotic factors. We have no data on the growth and health of *P. undulatum* lower down in the range, below 1300 m. Increasing competition from a wider range of faster growing native trees and perhaps more debilitating native pests or pathogens may limit the species lower down. Browsing goats and cattle, which are confined to areas outside the forest, do not readily eat *P. undulatum,* though goat browsing does restrict the distribution of the species in Pico in the Azores (Marler & Boatman 1952). Eighty-one years after its introduction to South Africa, a lethal (and unidentified) agent has largely eliminated *P. undulatum* from parts of the Cape Province (S. Neser, pers. comm., 1993). A similar event could, of course, happen in Jamaica. With time, and as *P. undulatum* gets more widespread and abundant, there will be an increasing chance that native predators will start to affect it more significantly, but it is more likely that *P. undulatum* plants will be damaged rather than killed.

2.1.4 Degree of invasion

The summed heights of all seedlings in each plot or sub-plot is a useful measure of dominance (Healey 1990). The maximum dominance of *P. undulatum* as a seedling can be very great, 52 metres per m^{-2}, 2.5 times the dominance of the most dominant native species, *Alchornea latifolia*. Dominance of trees and saplings as expressed by basal area is less illuminating, as *P. undulatum* is present in most plots only as a small tree, because of the early stage of the invasion. In one plot *P. undulatum* comprised 68% of the total plot basal area, though in most plots in heavily invaded forest, larger native trees, possibly left when the original forest was cleared, dominate in terms of basal area. *P. undulatum* trees had a maximum density of over 6000 stems per hectare, almost twice that of the most dense native species, *Eugenia virgultosa*.

2.2　The comparative performance of *Pittosporum undulatum* and native species

This is a very complex subject to research, analyse and present, and for which we have many data[1]. We have spent a large amount of time during this project on one study, the Heavily invaded forest experiment (HIFE), which we will describe fully elsewhere. It was a removal experiment whose main objective was to find out the best method of managing and ultimately eradicating *P. undulatum* in heavily invaded forest. Before treatment the density of native seedlings ranged from 0.55 to 24.2 m^{-2}, and that of *P. undulatum* from 0.4 to 101.0 m^{-2} - in 14 plots there were more *P. undulatum* seedlings than all native tree seedlings combined. In the first 28 months following the removal of all *P. undulatum P. undulatum* recruitment was very dense in some plots, up to 317.2 seedlings m^{-2}, whilst that of native species was generally very sparse.

2.2.1　Gaps created by the winching down of trees

In 1985-6 J.R. Healey set up an experiment in the forest west of Morces Gap in the Blue Mountains to investigate the differences in the regeneration ecology of the native tree species in an area of montane tropical rain forest (Healey 1990). Data from five enumerations provide a valuable insight into the success of *Pittosporum undulatum*.

Methods. Three pairs of plots, each 15.5×15.5 m, were established in an area of closed canopy forest. In one of each pair, all trees over five metres in height were winched over from the central 90 m^2. The other three plots were left as undisturbed controls. All seedlings in sixteen 1×1 metre sub-plots in the central area of each plot were identified, labelled and measured before the experimental treatment, and after 9, 18, 52 and 83 months. From these data, rates of recruitment, survivorship and growth were calculated for each species, in addition to the resulting changes in their density, frequency and dominance. *P. undulatum* trees and saplings (>3 m) were absent from the area encompassing the six experimental plots, but at least one isolated seed parent tree was observed within 200 m of the plots. At the initial enumeration the density of *P. undulatum* seedlings (all <1 m in height) in the plots was 0.42 m^{-2}, which is equivalent to only 0.8% of the dense seedling bank. The area was greatly affected by Hurricane Gilbert in 1988.

Results. In the undisturbed control plots the seedling dominance (defined here as the sum of the heights of seedlings present) of *P. undulatum* increased to 156% of its original value in 18 months before the strike of Hurricane Gilbert; this led to an increase in position of *P. undulatum* from 17th to 16th in the rank order of species. In the subsequent 32 month period, which included 23 months after the impact of H. Gilbert, the seedling dominance of *P. undulatum* increased to 1474% of its previous value, which led to an increase in the rank order of species to 7th position. *P. undulatum* represented 2.5% of the seedling dominance of all species combined. In the gap plots, *P. undulatum* seedling dominance increased to 871% of the pre-treatment value in the 18 months following experimental gap creation; this took *P. undulatum* from 17th to 5th in rank position. Then, in the subsequent 32 month period (including the impact of Hurricane Gilbert), *P. undulatum* seedling dominance increased to 422% of its previous value which led to an increase in rank order position to 2nd and to *P. undulatum* representing 10.6% of the seedling dominance of all species combined.

Conclusion. *P. undulatum* was initially a relatively unimportant species in the advance regeneration of this area of forest. During undisturbed conditions its seedling dominance increased slowly, but steadily. *P. undulatum* regeneration benefitted from the disturbance created by Hurricane Gilbert both in terms of absolute seedling dominance and relative to other species. *P. undulatum* regeneration benefitted to an even greater extent from the more severe experimental disturbance caused by winching down all trees taller than 5 m: after four years it had become the second most dominant species in these plots.

2.2.2　Trees

Data on the growth of every tree (stems ≥3 cm DBH) in 75 permanent forest sample plots (with an area of 2.04 hectares) were collected during the course of this project and were analysed to compare the basal area growth rate and change in populations of *P. undulatum* and native species. The results will be published elsewhere. In summary:

- There were no *P. undulatum* trees in any of the Tanner and Healey plots before the 1991 enumeration, six at the 1991 and 18 at the 1994 enumeration.
- The population of *P. undulatum* trees within the stratified plots of Bellingham increased from 198 to 297 (a 59.1% increase) between 1991-94, following Hurricane Gilbert in 1988, the second largest of any species with more than 10 individuals in 1991. The species with the largest increase in population (*Palicourea alpina*) is a short-lived native pioneer tree.

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[1] We have collected data on 20,055 trees and saplings, and 53,437 seedlings and herbs.

- The basal area increment of small trees (DBH <5.0 cm) between 1991 and 1994 in 74 permanent sample plots increased in the order shade-tolerant species<shade-intolerant species<*P. undulatum*. The increment of *P. undulatum* was higher at all degrees of *P. undulatum* density.

- No *P. undulatum* trees died in any of the enumeration periods in the Tanner, Healey, Bellingham or HIFE plots.

The impact of Hurricane Gilbert on the *P. undulatum* population

One of the reasons for the initiation of this project was the concern that *Pittosporum undulatum* would greatly benefit from the disturbance caused by Hurricane Gilbert in 1988. The performance of *P. undulatum* in the experimental gap plots before H. Gilbert and in the control plots since H. Gilbert (Healey 1990, Vernon 1991), as well as observations of natural gaps caused by the hurricane, confirm that *P. undulatum* is very competitively successful in gaps, becoming the dominant species in a majority of the recent gaps falling within its present range. As the invasion progresses, the reaction of *P. undulatum* trees to future hurricanes will become more important. The effect of H. Gilbert was assessed in 91 plots totalling 1.10 ha between February 1989 and August 1990, i.e. 5-23 months after the hurricane; namely the E.V.J. Tanner (Tanner 1977); J.R. Healey plots (Healey 1990); 26 non-bounded plots along a transect in the Mabess River valley and the 16 systematically placed plots of P.J. Bellingham (Bellingham 1993). A total of 5242 native and 53 *P. undulatum* trees were sampled. This work was conducted by P.J. Bellingham and was partly funded by another ODA project, R4611, *The protection role of Jamaican catchment forests and their resistance to and recovery from the impact of Hurricane Gilbert.*

Data on the 47 commonest species were analysed. *P. undulatum* was one of nine species that had no stems killed by the hurricane. *P. undulatum* was also one of only five species which had no stems that were completely defoliated and no stems broken. However, 11.4% of *P. undulatum* trees were uprooted (the ninth highest species percentage, the mean for all species was 0.49%). Bellingham (1993) classified the species into five categories of resistance to the hurricane according to levels of non-fatal damage and mortality. *P. undulatum* was placed into the most resistant category, though the relatively small number of stems (53) makes the classification tentative. In contrast to all other species in the resistant category, *P. undulatum* is readily recruited into hurricane caused gaps. Because of this, Bellingham considered the species to have no ecological analogue in the native tree flora.

In heavily invaded forest we have fewer data, though two small studies are illuminating. The ridge between Newhaven Gap and Sir Johns Peak is a likely illustration of what heavily invaded forest may look like after a major hurricane. The ridge, about 500 m long and 30 m wide, is very heavily invaded with *P. undulatum* (as a result of a road being bulldozed then abandoned in the early 1970s (R. Bryan, pers. comm., 1989)) and was severely affected by H. Gilbert. In the central 150 m section over 50% of all trees are of *P. undulatum* and about 75% of these were blown over, mostly leaning rather than horizontally, because of the high tree density. All *P. undulatum* trees blown down were living, and putting up vigorous epitrophic sprouts several metres high (T. Goodland, unpublished data, 1994). These findings are supported by data collected from a less heavily invaded forest with larger but more scattered *P. undulatum* trees. In an area of about 3 ha 11 *P. undulatum* trees over 22 cm DBH were found; of these 7 had been blown over, 2 were leaning and 2 were upright. In the same area there were estimated to be about 200 native trees >22 cm DBH but only 3 had been blown over (T. Goodland, unpublished data, 1992).

2.2.3 Effect on native plants

We used a variety of methods to try to find out what effect *P. undulatum* is having on native plants and our results will be published elsewhere. In summary there is a strong linear negative correlation between the dominance of *P. undulatum* and the density of native tree seedlings. The relationship is so clear and consistent that it strongly suggests that *P. undulatum* is causing, or at least contributing, to a major decline in the native seedling layer. The alien species reduced the recruitment and growth rate even of shade-tolerant native species. The main mechanisms by which *P. undulatum* may affect native plant species are discussed below.

Shading. *P. undulatum* trees have a substantially larger mean total leaf area per tree than the mean of native species of the same DBH (T. Goodland, unpublished data), thus shade probably accounts for a large part of the suppressive effect of the species. Light levels in heavily invaded forest are about 0.5% of full daylight (T. Goodland, unpublished data). We cannot determine to what extent the dense canopy has its effect because of a reduction in light or probable reduction in throughfall. *P. undulatum* could also affect native trees by changing their allometry through greater competition, thus changing their vulnerability to windthrow. *P. undulatum* trees do not appear to get covered with lianes as frequently as native species (though this could be because most climber species seem to be restricted to primary and therefore less invaded forest), so would not pull down other trees when blown down, (though we have no evidence as to the importance of this phenomenon in the Blue Mountains).

Below ground competition. We have some evidence that the below ground competitive ability of *P. undulatum* is high in comparison with the species native to the Blue Mountains, but further research would be needed to provide a clearer picture.

Allelopathy. Allelopathy has been suggested as a factor depressing the number of native seedlings beneath scattered *P. undulatum* trees in the Blue Mountains, where the light levels would have indicated a higher seedling density (J. Dalling, pers. comm., 1991). In Australia, Gleadow and Ashton (1981) found that leachates from *P.*

undulatum leaves appeared to inhibit the germination of several *Eucalyptus* species; for example, germination of *E. obliqua, E. melliodora* and *E. gonocalyx* was 47.1, 8.1 and 48.3% of untreated seeds. However, they stated that no inhibitory effects, other than that expected from deep shade, have been shown under canopies in the field. In South Africa, Richardson and Brink (1985) found no seedlings of *P. undulatum* or native species beneath established *P. undulatum* trees, and they thought that this was due to an allelopathic effect.

***P. undulatum* trees as a habitat.** The greatest effect that *P. undulatum* may have as a different habitat to native trees is on animals, but the structure of its crown or nature of its bark may have an effect on epiphytic plants independent from the density of its foliage. A large proportion of the non-woody plant species in the Blue Mountains are epiphytic (Adams 1972). Our observations and data collected by Mitchell (1989) suggest that epiphytic vegetation is much reduced both in the crowns of *P. undulatum* trees in comparison with native trees of similar size and on native species beneath dense *P. undulatum* canopies. This could be due to many factors such as reduced light levels and rain throughfall, the upright growth habit, different branch arrangement and bark characteristics of *P. undulatum* trees, allelopathic leachates from *P. undulatum* foliage and the faster growth rate of *P. undulatum* trees (hence less time for establishment).

Indirect effects on native plant species. We use the term indirect effect to mean an effect that *P. undulatum* has on a species by affecting the distribution or abundance of another species. These indirect effects could be very important, but they are hard to determine. The most obvious example is the effect *P. undulatum* may have on pollinators and seed dispersers of native plants. If *P. undulatum* is relatively successful in attracting pollinators and dispersers, those tree species neglected as a consequence could find their regeneration threatened.

2.2.4 Relative threat to different communities

There is a considerable amount of information on the vascular plant composition of most communities in the western end of the Blue Mountains, so the threat of *P. undulatum* towards different vegetation types can be assessed by examining the conservation importance of the more threatened types and the readiness with which the species invades. Those communities with particular conservation value are thought to be:

High altitude scrub forests. These are under threat because of their small size and the considerable visitor pressure on Blue Mountain Peak, the location of the largest patch. Of 48 species of flowering plants found in plots on the peak in this type of forest, 18 were endemic to Jamaica and some apparently endemic to this small area (Iremonger 1992). The top of Blue Mountain Peak is probably well within the climatic tolerance of *P. undulatum* and it occurs beside the trail to the peak above Portland Gap (M.A. McDonald, pers. comm., 1995).

Mor Ridge forest. This type of forest, which is restricted to certain ridge tops, is quite unique to the Blue Mountains. The thick humic soil supports many plant species which are only found as epiphytes elsewhere in the range. Areas of Mor Ridge forest within 3-4 km of Cinchona are being invaded by *P. undulatum*, slowly, so this type of forest is certainly threatened.

Montane rain forest over limestone. In 1992 there was estimated to be 10,106 ha of montane rainforest on limestone within the Blue and John Crow Mountains National Park and adjoining 1 km wide buffer zone (Muchoney *et al.* 1994). Over 99% of this is in the John Crow Mountains at the eastern end of the park and most of this is below 600 m, so it seems unlikely (though possible) that *P. undulatum* could ever be a serious problem there. There is also an important outcrop of limestone on John Crow Peak at the western end of the Blue Mountains. This area of only a few hectares supports several localised endemic plants and may be vulnerable to invasion by *P. undulatum*, as the species now occurs on the flanks of the mountain, on limestone-derived soil (T. Goodland, pers. obs., 1993).

As an initial attempt to quantify the threat that *P. undulatum* poses to biodiversity in the Blue Mountains we have selected three forest types (as described in Grubb & Tanner (1976)) and scored all three aginst five factors. They are the rate at which each forest type can be invaded, the ultimate degree of invasion that *P. undulatum* can attain (for example, maximum basal area), the extent to which the community is changed, the plant species richness and the degree of endemicity. For each of the five factors the three forest types have been ranked, then these ranks were converted to numbers (assigning 3 for High/Large, 2 for Intermediate and 1 for Low/Small) and summed. This assessment does not take into account the relative areas of different forest types. At present we have little detailed data on the cover of different forest types in the Blue Mountains, beyond such broad definitions as "Upper montane forest on shale". The arbitrary weighting given to the factors and imperfect information make this a preliminary assessment.

	Mor Ridge	Gully	Mull Ridge
Rate of invasion	Intermediate	Low	High
Ultimate "degree" of invasion	Low	High	Intermediate
Effect on community	Large	Small	Intermediate
Species richness	Low	Intermediate	High
Degree of endemicity	High	Low	Intermediate

The totals for the three forest types are therefore Mor Ridge 10, Gully 8 and Mull Ridge 12, suggesting that a strategy to preserve biodiversity should control *P. undulatum* in the order Mull, then Mor, then Gully (other things being equal).

2.3   Effect on other aspects of the Blue Mountains

2.3.1 Animals

The number of insects associated with a tree species gives a good indication of its conservation value. Insects are comparatively rare on *P. undulatum* foliage, so insectivorous birds may be badly affected and the dense canopy might make it harder for birds to find those present. *P. undulatum* trees produce a large quantity of nectar and seeds relative to that of most native tree species, so its spread may lead to decreasing populations of those animals unable to feed on these food sources. The streamertail hummingbird (*Trochilus polytmus*), endemic to Jamaica and the country's National Bird, is seen in great concentrations around flowering *P. undulatum* trees. The bananaquit (*Coereba flaveola*) is "particularly abundant where a favourite tree was in flower, ... the introduced *P. undulatum* on the trail from Cinchona through montane forest to Morces Gap" (Lack 1976, p. 355). But the flowers are only produced for about three months a year, and seed production is variable from year to year, so *P. undulatum* may actually benefit few native bird species. Most of the common tree species with bird dispersed seeds produce their seeds during July to September (Tanner 1982) and as *P. undulatum* exposes its seeds in October to December this might give it an advantage over competition for dispersers and increase the population level of some bird species. Seeds are present on the ground for a greater proportion of the year, and may be eaten by some species there, such as the endemic pigeon *Geotrygon versicolor*. Birds or other animals may be affected indirectly. For example, epiphytic bromeliads are important sources of food for the endemic and threatened Jamaican blackbird (*Nesopsar nigerrimus),* but are infrequent on *P. undulatum* trees. It is not known to what extent food is a limiting factor for any of the bird species in the Blue Mountains, so making it difficult to make predictions about the possible effect of *P. undulatum* on bird populations. Mitchell (1989) found that there was a slight inverse relationship between the dominance of *P. undulatum* and bird density in plots affected by a range of *P. undulatum* densities, but that bird diversity was not significantly affected. He suspected that his data on bird observations might have been too few to detect such a difference.

2.3.2 Water yield

Most of Kingston's water comes from the southern slopes of the Blue Mountains, and any reduction in this supply would have very serious economic costs, as there is already widespread water rationing in Kingston in most years. The northern slopes are wetter and do not supply any large population centres, so *P. undulatum*'s presence there is less likely to have any significant economic impact, unless in the long term water is piped from purpose-built reservoirs in northern catchments to the Greater Kingston area. The canopy of forest heavily invaded by *P. undulatum* is denser than that of uninvaded forest, potentially leading to increased evapotranspirational losses in the canopy. But the sparser understorey and reduced air turbulence beneath *P. undulatum* could counterbalance the effect, so the overall effect of *P. undulatum* could be small (R. Nik, pers. comm., 1989). Although the effect of *P. undulatum* on water yield could be a very important factor in the Cost-Benefit Analysis we have not had the resources to investigate the subject further. We collected data on the soil moisture content of soils in every HIFE plot twice before treatment and twice after treatment, and have analysed data from the experiment set up by M.A. McDonald, to try to find out what effect *P. undulatum* may be having on the hydrology, but neither sets of data have provided any solid indication. To obtain reliable information, a comparison between forest that has been heavily *P. undulatum*-invaded and forest that has not been invaded would be needed. Heavily invaded forest from which all *P. undulatum* has been removed, as in HIFE, has a much sparser understorey than forest that has never been invaded.

2.3.3 Sediment yield

As with water yield, the possible effect of *P. undulatum* on nutrient cycling and erosion was not directly addressed in this project, but some conclusions can be drawn from our observations, the work of M.A. McDonald, and other published data. There are three reasons for thinking that *P. undulatum* may influence rates of sediment loss.

- The regular branching habit of *P. undulatum* means that the stemflow of the species is high compared with native species (M.A. McDonald, pers. comm., 1993). This may in theory enhance erosion around the butts of *P. undulatum* trees, but little evidence of this has been found.
- Because of the typical poorly developed understorey beneath *P. undulatum* trees, it is likely that a large proportion of the rain intercepted by the tree canopy falls and impacts the ground at terminal velocity, without being intercepted by understorey vegetation.
- Data and observations after Hurricane Gilbert (Bellingham 1993; T. Goodland, pers. obs., 1989) indicate that a higher proportion of *P. undulatum* trees were blown over than native trees.

The seedlings of some common native species, such as *Eugenia virgultosa,* have strong tap roots, whereas those of *P. undulatum* are highly variable in direction and length, (suggesting an "exploratory" type of root growth), and are usually shallow or extending downwards only a few centimetres (T. Goodland & J.R. Healey, unpublished data). *P. undulatum* trees blown over by Hurricane Gilbert also had shallow root systems as displayed by root plates, though the trees blown down are a biased sample of the *P. undulatum* population. The root system of *P. undulatum* in its native range is very variable in form, depending on soil texture and compaction. In heavy compacted soils, such as silty loam podzols, large lateral roots radiate from the trunk in the uppermost layers, while in deep sandy soils the root system is extensive and the tap root is persistent to depths of one metre (Gleadow & Ashton 1981). There is a clear association between the distribution of landslides and *P. undulatum* in the Blue Mountains, the species forming dense fringes to old landslides and on the oldest landslides colonising the scar itself. Because of this association it might be thought that the relatively shallow root sytems of *P. undulatum* trees could be less effective at stabilising steep slopes than native trees. But it seems more likely that the disturbance around landslides promotes the establishment of *P. undulatum* and the species itself has no effect on slope instability.

2.3.4 Nutrient cycling

The nutrient dynamics of primary forest in the Blue Mountains has been well studied (Tanner 1977, 1980a, 1985), but information on the nutrient relations of *P. undulatum* is much less complete. Evidence suggests that *P. undulatum* leaves placed on the soil surface can decompose rather rapidly in the Blue Mountains (M.A. McDonald, pers. comm., 1993). In contrast, on Pico in the Azores, Marler & Boatman (1952) measured the organic matter content and Cation Exchange Capacity of soils along an altitudinal gradient in forest that was heavily invaded by *P. undulatum* in the middle elevations. There was an increase in base saturation from 1110 m to 600 m, then a decrease as the forest became dominated by *P. undulatum*. They ascribed these changes to a comparatively slow rate of decomposition of *P. undulatum* litter. In Victoria, Australia, Gleadow & Ashton (1981) found that fresh *P. undulatum* litter had a much higher nutrient content than *Eucalyptus* species as shown in the table.

Table 2.5. Nutrient analysis (% dry wt) of fresh *Pittosporum undulatum* and *Eucalyptus* leaf litter and fresh *P. undulatum* leaves (from Gleadow & Ashton 1981).

| | *P. undulatum* | | *Eucalyptus* |
	Fresh lvs	Litter	Litter
Mg	0.32	0.21	0.08
Na	0.22	0.28	0.04
K	2.47	0.21	0.27
Ca	1.23	1.17	0.34

Litterfall of *P. undulatum* was heavy in Victoria (2.6 t ha^{-1} yr^{-1}) in comparison with *Eucalyptus* litter (0.9 t ha^{-1} yr^{-1}) (Gleadow & Ashton 1981), suggesting, together with the higher nutrient content of *P. undulatum* litter, a much greater cycling of nutrients than with *Eucalyptus* species. Data from the experiment set up by M.A. McDonald show a slight negative relationship between the relative basal area of *P. undulatum* and litterfall, that is to say a result that seems to be contrary to that from Australia, but the relationship is weak. However, all the plots were in secondary forest so the results are difficult to interpret (M.A. McDonald, pers. comm., 1995).

The mycorrhizal relationships of *P. undulatum* in the Blue Mountains have not been properly investigated but we have found evidence of Vesicular Arbuscular mycorrhizae (J.R. Healey, pers. obs., 1986).

2.3.5 Forest combustibility

The undisturbed forests of the Blue Mountains are relatively resistant to burning, although forest that has been disturbed or is scrubby can burn (as for example in the summer of 1991). We do not know whether living *P. undulatum* trees are more flammable than native trees; the bark is resinous, the leaves seem quite volatile rich (in comparison with most Blue Mountain species) and the wood is dense and so would probably burn at a high temperature. There may be the possibility that areas with serious blowdown of *P. undulatum* could be vulnerable to fire. However, of the hundred or so *P. undulatum* trees seen after being blown down by H. Gilbert, only two had died, suggesting that the amount of dead and therefore flammable wood following a major blowdown of *P. undulatum*

would be small. A more serious threat perhaps is a pest or disease that kills *P. undulatum* plants of all sizes quickly. This would leave large amounts of dead wood in a few hundred hectares of forest which may leave this forest susceptible to fire for several years.

2.4   Conclusion

Pittosporum undulatum is now widespread in the western Blue Mountains but still only occupies about 3% of its potential range, and is present in most of this area at only a very low density. Its rate of invasion has been slow so far (about 30 m per year), but there are several reasons for thinking that the past rate is a poor guide to the potential future rate. As the species is so successful in canopy gaps, such as that caused by Hurricane Gilbert in 1988, it is likely to increase its rate of spread significantly over the next few years. It appears that *P. undulatum* trees are readily blown down by hurricanes but as their survivorship is very high the dominance of the alien species is not likely to be reduced. All the evidence indicates that the hurricanes will benefit the species.

P. undulatum is able to grow at higher densities than any native species and, combined with its dense crown, achieves a much greater dominance than any native species. It appears to reduce the survival and growth of even shade-tolerant native species. It has a serious impact on all native vegetation types, though invades some types (for example Mor Ridge forest) more slowly than others. Its effects on animals and ecosytem processes are likely to be detrimental, but are almost completely unresearched. Its greater productivity should have benefits though, particularly to local people, examined in the next chapter.

3.

Impact on people and use of *Pittosporum undulatum*

In this chapter we examine the direct impacts of *P. undulatum* on people and how the species can be managed to provide more benefits to the communities adjacent to the Blue and John Crow Mountains National Park. The recent establishment of the park, with its potential to bring significant tourist revenue to the area, provides one of the most promising long term solutions to the economic problems of local people, but this could be jeopardised by the spread of alien plants. Park managers have expressed their strong desire to discover ways of managing the *P. undulatum* invasion to protect the park's diversity whilst simultaneously benefiting the people who live in the surrounding area (D. Lee, pers. comm., 1991). The people of the Blue Mountains area are, in comparison with people in most other countries where alien trees are invading natural areas, heavily dependent on the natural resources of the mountains and farming land on its lower slopes. Poor people in Jamaica suffer from a serious lack of fuel and construction materials and, whilst the situation in the Blue Mountains is not at present chronic, it is likely to get worse. Therefore, the effect of management policy towards *P. undulatum* (whether to eradicate, manage, promote or leave it) will influence the future availability of work and wood, and needs to be considered carefully. In this chapter, firstly, we assess the present and future supply and demand of tree products in the Blue Mountains area; secondly, we examine *P. undulatum*'s qualities as a fuel and material; thirdly we estimate the quantity and accessibility of the existing and potential resource. Lastly we look at the potential for the use of the species outside the forest.

3.1   Use and availability of tree products in the Cinchona area

3.1.1 Fuel

A large proportion of the firewood used in the Cinchona area (approximately 10 km^2, roughly encompassing the villages of Westphalia, Halls Delight and Top Mountain) comes from about 150 ha of secondary forest and scrub near Cinchona. Almost all people in the area rely on firewood for their cooking needs, though many families have a backup cylinder gas stove for occasional use. There is no organised firewood market in the area, as most families have members who can collect it. Firewood usually has to be carried perhaps 3-4 km, though most of it is usually carried down at the end of the day's work on the coffee plantation or farm. Much of the wood collected is from dead native trees, rather than *P. undulatum* which is so common in the area. This may be because most local people are reluctant to cut green timber (though the branches are often lopped off living trees) and there are many dead native trees in the area because of forest clearance, fires and Hurricane. Additionally, most of the remaining native trees are of moderate size and with dense tough wood, making their cutting with machetes hard work. Little charcoal is produced in the Cinchona area, (though closer to Kingston, where the biggest market for it is, production is widespread). Remoteness is probably only part of the reason, the fact that the land is in or near the national park is probably as important.

There will probably be a greater fuelwood shortage in the area over the coming years, due to a number of reasons.

- Accessible supplies of dead wood are dwindling as the trees cleared or left isolated by forest clearance during the 1970s, or blown down by Hurricane Gilbert, are used up or rot *in situ*.
- The human population is likely to increase substantially through the next few years. The ratio of children to adults is high in the area (at it is in Jamaica generally), though it remains to be seen how great future migration to Kingston will be.
- The price of firewood and charcoal is influenced by the price of kerosene, which has gone up in recent years as a state supported subsidy has been reduced, with little prospect of it being restored to previous levels.
- Most local farmers have been reluctant to plant trees on their land because of a combination of factors, including very small land holdings (in 1982, 53% of Blue Mountains farmers farmed less than 0.4 ha (McGregor *et al.* 1985)) and tenurial insecurity (reviewed by Aldrich 1993).

3.1.2 Building materials

Local people cut pole-size trees from accessible forest for a variety of minor construction jobs, and preferred species include rodwood (*Eugenia* species), soapwood (*Clethra occidentalis*) and *P. undulatum*. This harvesting has started on a larger more organised scale within the last year or so. For example, one gang used chainsaws to cut about 200 small trees, mostly of *P. undulatum*, collected the following day by truck (T. Goodland. pers. obs., 1995). The only timber species that are extracted from the forests in the Cinchona area and transported to the lowlands for sawmilling are exotic pines and *Juniperus lucayana*. The pines were mostly those blown down by Hurricane Gilbert, and accessible pine is getting scarce. Poor roads and steep hillsides within the area have meant that all large trees have been sawn up into planks *in situ* and carried to the nearest road, and then transported to Kingston. The policy of

establishing tropical pine plantations for timber has been almost completely abandoned islandwide since H. Gilbert (R. Jones, pers. comm., 1991). The park authorities appear to have been successful in halting the illicit felling of native *J. lucayana* trees in the Cinchona area. However, as there is still the expertise and equipment in the area (and probably the desire too) to resume timber cutting in the forest, the selective felling of *P. undulatum* trees seems an attractive alternative. Jamaica imports almost all of its timber, the second largest single commodity in terms of foreign exchange expenditure, and there is a desire amongst many Jamaicans for the development of new sources, so that this loss of foreign exchange can be reduced. There is great demand for a species which produces good timber and is available in sufficient quantities for conversion and marketing in Kingston.

3.1.3 Use of *P. undulatum*

The present use of *P. undulatum* by local people has been quite hard to assess. A survey of 42 farmers to gather information on the use people make of the forest and forest fringe was carried out by J. Nenon in the Westphalia to Whitfield Hall area in 1991 (J. Nenon, pers. comm., 1991). At our request she included two questions about *P. undulatum* and the answers shed some light. In response to the question "are you familiar with wild coffee" 23 people said yes, (it would have been better to ask people if they could identify a sample of *P. undulatum*). Of these people, 19 said that they used it for firewood, and 11 said that they used poles of the species for small constructional jobs. The cutting of all trees within the national park is illegal without a licence, and the access to heavily invaded forest is poor, so little can be inferred about the potential of *P. undulatum* from a study of current use.

3.2   Qualities of *Pittosporum undulatum* as a source of goods

3.2.1 Firewood

The factors that determine suitability of a wood for firewood include density (closely related to calorific content), whether it needs drying before burning, ash content, degree to which the wood "spits" when burnt and "splittability" (M. Breese, pers. comm., 1993). Informal interviews with local people near Cinchona suggest that *P. undulatum,* one of the most popular firewoods in the Cinchona area, scores highly on all these counts, though no quantitative information is available apart from density. It burns well with a slow steady heat and little smoke, without the need for seasoning and is dense so gives out much heat. The data we have on the wood density of *P. undulatum* and native species is presented after a description of methods below.

Twenty-seven trees, up to 10 cm DBH, were cut in December 1991 as part of HIFE (Healey *et al.* 1992, p 19). Two samples of stems 30 cm long were selected from each tree. The samples were air dried at Cinchona for 25 months (at approximately 18°C in moderate humidity), taken to the University of the West Indies where they were weighed and their volume measured by the displacement method. Ideally the volumes should have been measured when fresh, however none of the samples showed any obvious signs of shrinkage or decay.

Table 3.1 The number of trees, mean wood density (in kg m^{-3}) and standard error of the mean of 18 species cut in 1991 and measured in 1993.

Species	No. trees	Mean density	SE
Alchornea latifolia	1	0.274	.
Hedyosmum arborescens	1	0.435	.
Psychotria corymbosa	2	0.529	0.0277
Psychotria sloanei	2	0.575	0.0095
Cinnamomum montanum	1	0.608	.
Podocarpus urbanii	1	0.636	.
Turpinia occidentalis	1	0.636	.
Citharexylum caudatum	1	0.664	.
All native species	23	0.703	0.0362
Dendropanax species	1	0.704	.
Clethra occidentalis	2	0.742	0.0245
Vaccinium meridionale	1	0.756	.
Ilex macfadyenii	2	0.775	0.0566
P. undulatum	4	0.782	0.0099
Guarea glabra	1	0.791	.
Critonia parviflora	1	0.818	.
Xylosma nitida	1	0.899	.
Eugenia virgultosa	3	0.902	0.0061
Eugenia monticola	1	1.004	.

P. undulatum had the sixth densest wood, 11.3 % denser than the mean for native species. Its wood was 2.85 times as dense as that of *Alchornea latifolia, (*a common moderately long-lived native species with a similar ability to compete successfully in gaps), and correspondingly has a much higher value as a source of fuel and building materials.

It is difficult to derive a price for *P. undulatum* firewood in the Cinchona area, as there is no organised market, but based on local estimates, a delivered price of about US$27 m^{-3} (J$800 m^{-3}) seems reasonable.

3.2.2 Charcoal

P. undulatum makes excellent charcoal (L. Stamp, pers. comm., 1992) and, given the inaccessibility of much of the invaded forest, charcoal production would seem to be a more feasible option than fuelwood collection for wide-scale economic harvesting. To assess the economics of charcoal production, we obtained independent estimates of the time and amount of wood needed and the value of the resulting charcoal from two local men, Mr

Lloyd Stamp and Mr Albert Hall, shown in Table 3.2. Their estimates were broadly similar but rather approximate, largely because of the suspected imprecision with which they estimated the amount of wood required.

Table 3.2 Economics of charcoal production from *P. undulatum* based on one "skill" (stack) of charcoal produced in accessible heavily invaded forest. Values in US$.

Amount of wood needed to make a skill (m^3)	2.5
Man-days needed to produce one skill	5
Yield of charcoal (kgs)	875
Value per kg	0.15
Total value per skill	130
"Appropriate" weekly wage for such work	25
Surplus m^{-3} delivered of wood	41.33

Thus, with a value of *P. undulatum* wood as charcoal of about US$ 40 m^{-3}, charcoal production would appear to be a more economic use of wood from the species than for firewood, and could be a substantial source of revenue for the park. However, the cutting of small almost stick size stems for charcoal production is locally thought to be desirable, as it helps plug the gaps left between larger logs. Therefore, the cutting of *P. undulatum* trees for charcoal could lead to an even greater opening up of heavily invaded forest than for firewood, which could have serious environmental consequences (see chapter 4).

3.2.3 Building materials

P. undulatum trees are usually straight-trunked and cylindrical so may have potential for use as a timber species. The wood is whitish, attractive, small-pored, and takes stain readily (P. Lowe, pers. comm., 1994), as well as being quite dense. One of its popular names in Australia is "Engravers wood" and it was the preferred wood for golf clubs in Australia (Maiden 1920). It has been used for carving in Kingston (R. Bryan, pers. comm., 1992). However, it would be quite a difficult species to manage silviculturally as it would produce persistent branches very readily if not grown at high densities or pruned. It does occur at such high densities in heavily invaded forest but these densities would not be compatible with maintaining a diverse native understorey.

To get a better idea about its qualities as a timber tree, and the likely value of its wood, we felled and took a 51 cm DBH *P. undulatum* tree to the Forest Industries Development Company sawmill near Kingston. The tree had a straight branch-free length of 4 m, giving a log volume of about 0.82 m^3. The sawmillers liked the wood (despite the fact that the sawdust "burn nose") and converted one log from the tree into bed lathes and the other into a table. They reckoned it would fetch J$28-30 (US$1) a board foot delivered to the sawmill (P. Lowe, pers. comm., 1994), equivalent to US$410 m^{-3}, giving a value to the tree of US$335. Therefore timber is about ten times as valuable as charcoal volume for volume wood equivalent. Trees need to have an overbark DBH of at least about 20 cm before they can be used for sawn timber.

3.2.4 Other products and uses

Ornamental value. *P. undulatum* was introduced to Jamaica and most other countries because of its ornamental value, with its perfumed creamy flowers, orange capsules and fast-growing leafy canopy. Jenman (1883, p. 40) described it as "one of the handsomest flowering trees of Australia. The rich perfume of its white blossoms extends over a considerable area". Other *Pittosporum* species are grown as ornamentals in Jamaica (Adams 1972).

Shelter. *P. undulatum* was introduced to the Azores in the early 19th century to provide wind breaks for orange groves against Atlantic gales (Bean 1976) because of the density of its canopy.

Bio-engineering. A potential use of *P. undulatum* is for "living barriers". These are poles cut from living trees that are placed along the contour on steep farmland and then take root, thereby having the potential to stabilise slopes. A trial using *P. undulatum* and native species was established in 1993 near Cinchona (M.A. McDonald, pers. comm., 1994). However, despite indications from an earlier trial (T. Goodland, unpublished data) no buried stems of *P. undulatum* (or any native species) survived. The dryness of the soil was probably the cause.

Source of chemical compounds. The cultivation of *P. undulatum* on a commercial scale for its essential oils was once advocated in Australia, as has its potential as a source of gum resins (Maiden 1889). *P. undulatum* was one of three "oil-bearing" tree species investigated for their potential as a source of renewable energy in California (Calvin 1985a). The fruit of *P. undulatum* was of particular interest as a source of terpenes for use as a fuel oil (Calvin 1985b). So far as we know, the commercial utilisation of the species has not attempted for any of these uses.

Honey. We do not know the details of enough honey "finds" to say if *P. undulatum* trees are preferred by bees as a place to live, but *P. undulatum* flowers are much visited by bees and are a major source of nectar. However, the flowers last for only about 1-2(-5) months a year, so obviously limiting the species' potential for honey production.

Fodder. In St Helena, *P. undulatum* is pollarded and fed to penned goats and cattle. It is said to make good fodder (Q. Cronk, pers. comm., 1986), which is rather surprising considering its rather coriaceous leaves, with high levels of secondary compounds, including sapogenins (Higuchi *et al.* 1983).

3.3   Valuing *Pittosporum undulatum* as a wood resource

We have made an approximate assessment of the present volume of *P. undulatum* timber in heavily invaded more accessible forest, followed by an examination of some of the practical issues involved with harvesting the species.

3.3.1 Methods

Fourteen *P. undulatum* trees 3.6-14.0 cm DBH were selected for volume determination. The forest from which they came was moderately exposed and heavily invaded with *P. undulatum*. Each tree was cut down at 0.3 m above ground level and its total height measured. The girth at 0.3 m and at 1 m intervals to the tip of the tree were measured. The volume of each 1 m section was calculated separately and summed for each tree. From the data a regression equation of tree volume *versus* DBH was estimated. The volume of every *P. undulatum* tree more than 3 cm DBH in 67 plots in heavily invaded forest (the HIFE, 1989, M.A. McDonald, S. Morrin plots and the most heavily invaded plot of P.J. Bellingham) was calculated from the regression equation, and then summed for each plot.

3.3.2 Results

A cubic regression equation of the form: $Y = b_0 + b_1t + b_2t^2 + b_3t^3$

where Y = tree volume (m^{-3})
b_0 = a constant
b_n = regression coefficient
t = DBH (cms)

most closely fitted the relationship between the DBH and volume of the 14 *P. undulatum* trees. The values are:

Individual tree volume (m^3) = (-31109 + (14198×DBH) - (1598×DBH^2) + (85.8×DBH^3))/1000000

Although the relationship is strong (with an r^2 value of 0.93), if the harvesting of *P. undulatum* trees is seen to be a desirable policy by the national park, similar volume determinations should be carried out on trees of larger size to increase the general applicability of the relationship. The mean volume of *P. undulatum* ≥3 cm DBH in the 67 plots was 28.9 m^3 (SE 24.8) per hectare. The area of forest that we have classified as having over 100 *P. undulatum* trees per hectare is 300 hectares, and multiplying this by the mean volume gives a very approximate estimate of about 8,700 m^3 of *P. undulatum* within "accessible" forest. Using the value of *P. undulatum* timber of US$410 m^{-3} the total value is therefore about US$3.5 million. This is the theoretical maximum value, from this would need to be subtracted the cutting, extraction and transport costs, as well as potential negative environment costs. We have not attempted a more precise valuation because:

- Large uncertainties are caused by the question of how to cost labour, as extraction would be a very labour intensive operation. Given the high unemployment in the area there are strong reasons for discounting the cost of labour, that is to say using a shadow price of labour.
- The marketability and value of *P. undulatum* timber is uncertain.
- The size distribution of *P. undulatum* in heavily invaded forest, with its preponderance of small trees and saplings, means that the number of large *P. undulatum* trees will increase rapidly during the next few years. Therefore, given the higher value of timber from larger trees, any such valuation would soon be outdated.
- There are good practical and environmental impact reasons to lead to a policy of gradual harvesting of *P. undulatum,* rather than a sudden and large scale clearance (see later).

3.3.3 Practicalities of harvesting

Vehicular access is limited and in a national park it would not be desirable to markedly improve or extend the existing road network. Donkey or mules could be important for the extraction of firewood and charcoal but not so useful for the extraction of timber or planks. Although chainsaws are commonly used for cutting larger trees in the area, most smaller trees are cut down with machetes. It would be possible to cut down the largest *P. undulatum* trees with hand tools, by using a combination of an axe and crosscut saw. Within the last four years at least two "Alaskan mills", a specially designed frame that attaches to a chainsaw, have been used in the area to saw up logs into planks. These so far have been used to cut pine and *Juniperus lucayana* but could be used to harvest even the largest *P. undulatum* trees. However, mobile bandsaws would be much preferable as a way of planking large trees *in situ.* There is less wastage (the saw blade is about 3 mm thick compared with about 8 mm for chainsaws), they are safer in difficult terrain, but they cost about US$2,000 (chainsaws would still be needed of course) and the sharpening of bandsaw blades is a more specialised job than that of chainsaws (N. Geddes, pers. comm., 1994).

3.4   Role of *Pittosporum undulatum* outside the forest

We gave comparatively little attention at the start of the project to the role of *Pittosporum undulatum* outside the forest, but it is a subject that we have now considered in some depth. This is partly due to a greater appreciation of

the need to consider *P. undulatum* wherever it occurs in the Blue Mountains, because of the potential of long range dispersal by birds, (as shown by the widely scattered nature of *P. undulatum* trees through primary forest); and partly a specific request from the management of the national park (R. Kerr, pers. comm., 1991) for more information on *P. undulatum*'s potential as a woodlot species outside the forest, as the productive and sustainable use of the park's natural resources is now receiving high priority (Kerr *et al.* 1993). The park includes much underused deforested land.

3.4.1 Suitability of *P. undulatum* for planting outside the forest

P. undulatum is common and locally dominant in gullies quite isolated from the forest, but it is not common on grass or herb covered hillsides. The lowest that we have found the species is in a gully at 850 m altitude. The species also occurs as scattered trees in the coffee plantations and farmed land below Cinchona. Where *P. undulatum* trees do occur outside the forest they often seem not to thrive, with thin crowns and yellowish leaves, such trees are rarely seen in the forest. It has the reputation amongst local farmers for needing cool shady places, whereas much of the land on the south slopes is affected by periods of drought and maximum temperatures of over 30^0 C (Tanner 1980).

Response to cutting

The ability to produce sprouts in response to cutting is an important requirement for firewood species so we have collected data on the coppicing ability of *P. undulatum* and native species. These data also provide evidence on the effectiveness of cutting alone as a way to kill *P. undulatum* of different sizes and in different environments.

Methods. The trees and saplings cut as part of the Heavily Invaded Forest Experiment provide much data on coppicing ability *P. undulatum* and native species. The stems were cut in September-October 1991 and the regrowth was cut and measured in March 1992, January 1993 and December 1993 and cut but not measured in September 1994. A maximum of 15 stumps of *P. undulatum* per 12×12 m plot in the **Remove all *P. undulatum*** and **Remove *P. undulatum* trees** treatments, and all native stumps in the **Remove equivalent native trees** treatment plots, were selected. The following parameters were measured.

- Number of sprouts longer than 1 mm
- Length of the three longest sprouts
- Dry weight biomass of all harvested stems, after oven drying at the University of the West Indies
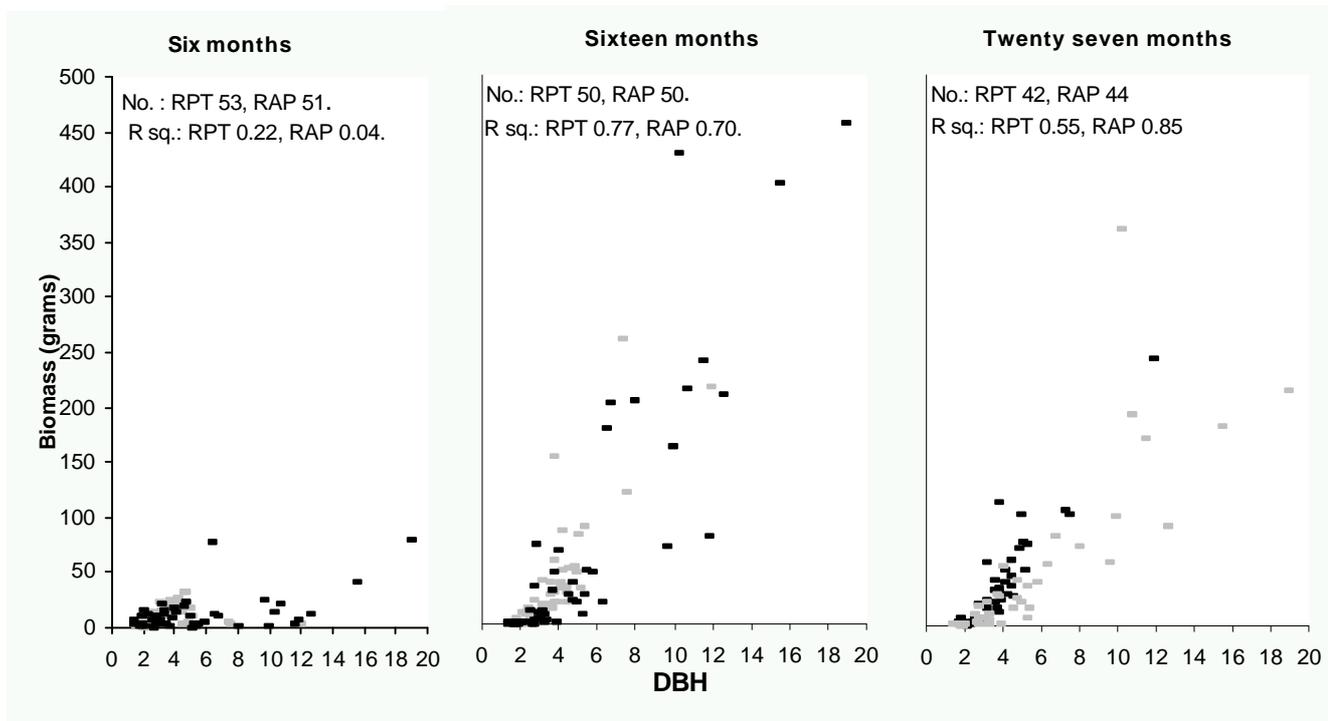
Figure 3.1. Dry weight biomass of *P. undulatum* resprouts in the HIFE. The biomass from each stump is plotted against its DBH at three times after the original trees had been cut. The stumps in the **Remove all *P. undulatum*** treatment are marked black, those in the **Remove *P. undulatum* trees** are grey. The number of stumps, and the r^2 value for each treatment, are shown for each time.

Results. The significantly greater biomass at the second harvest is partly a result of the longer time period since cutting (ten *versus* six months). The biomass was significantly lower at the third harvest, 11 months after cutting, but the sprouts were still vigorous on most stumps at the fourth cutting. Mortality was low in both treatments. None of the stumps appeared to be dead at sixteen months though some had not produced any sprouts at all. By 27 months

14% of the sprouts, all in **Remove *P. undulatum* trees**, had certainly died, none of them having ever produced sprouts. At 39 months, 34% had died in **Remove all *P. undulatum*** and 41% in **Remove *P. undulatum* trees**, a non-significant difference. *P. undulatum* stumps produced a much greater biomass of resprouts than native species, about ten times the mean of all native species combined. But a comparison between the coppicing ability of *P. undulatum* and native trees is not strictly valid, as the light levels near ground level following the removal of *P. undulatum* trees was significantly higher than in the **Remove equivalent native trees** treatment. One native species, *Ilex macfadyenii*, produced a significantly larger biomass of sprouts than any other native species, nearly as much as *P. undulatum* at 27 months. Interestingly, trees of *I. macfadyenii* are common in secondary forests and are nearly always multi-stemmed.

A trial set up by M.A. MacDonald comparing the coppicing ability of *P. undulatum* and three other tree species (*Acacia mearnsii*, *Paraserianthes lophantha* and *Clethra occidentalis*, the first two are exotic leguminous species) provides interesting comparative data (Healey *et al.* 1995). *P. undulatum* survival, together with that of *A. mearnsii,* was nearly 100%, but biomass production was much less than that of the two leguminous species, though still four times that of the only native species, *C. occidentalis.*

Possible limiting factors outside the forest

There are several factors that may be limiting the distribution and health of *P. undulatum* outside the forest.

Lack of water. Rainfall decreases markedly to the south away from the Grand Ridge of the Blue Mountains (Tanner 1980) and *P. undulatum* trees in more open habitats would be exposed to the sun and wind. In its native habitat *P. undulatum* is usually confined to moist and sheltered habitats such as gullies (Cooper 1956).

Fire. Perhaps the greatest barrier to the establishment of trees on the southern slopes of the Blue Mountains is the prevalence of fire, and *P. undulatum*, like the majority of native species, is fairly susceptible to fire. A fire suppression policy may be essential for the re-establishment of forest cover.

Strong winds. Data (Bellingham 1993; T. Goodland, unpublished data) and observations clearly indicate that within the forest *P. undulatum* is more readily blown down than most native species. *P. undulatum* grown for fuelwood would not need to be allowed to grow above about 6 or 7 m (the height at which stems of the species start to become difficult to cut by hand and carry far), but *P. undulatum* trees in exposed places of this height could be vulnerable to windthrow. Coppice stools would be less likely to uproot, but shoots on coppiced stools are known to be weak (compared with those of seedling origin) in some species (P. Denne, pers. comm., 1992), making snap during high winds more likely.

Weeds. A major barrier to the re-establishment of forest cover after fire suppression is likely to be a proliferation of mostly alien weeds. Therefore, the competitive ability of the chosen tree species is of great importance, and there is much evidence to indicate that *P. undulatum* would be more likely to establish than most native species and perhaps would be more likely to then suppress weeds than introduced light canopied N-fixing trees.

3.4.2 Constraints to the use of *P. undulatum*

There are several reasons why the harvesting of *P. undulatum* in the forest or its planting outside the forest should be considered carefully before proceeding, apart from the fact that the species may not thrive outside the forest.

- The threat that continuing seed production poses to the remaining uninvaded forest. This would not be a serious constraint in the Cinchona area, as most forest within about four kilometres of Cinchona has already been invaded, but it is a major reason against planting the species at other places around the Blue Mountains.

- The effect cutting *P. undulatum* may have on the cutting of native species. The operation would need to be closely monitored to ensure that the timber fellers did not cut native trees. *P. undulatum* is sufficiently distinctive that with a little training, identification of it should not be a problem.

- The effect disturbance may have on the recruitment of other mostly alien weeds.

- Too great a reliance on a single species may be unwise, especially an alien species that has already been subject of a lethal (unidentified) agent in another country, South Africa (see later).

3.4.3 Sterile lines

A possible way of using the species in the Blue Mountains without threatening uninvaded forest is to use sterile lines. Sterile lines of *Leucaena* have been developed to minimise problems of weediness in Hawaii (J.L. Brewbaker, pers. comm., 1994), most being triploids produced by 2n x 4n crosses of species native in the Americas. Brewbaker considers that the development of sterile lines for genera such as *Pittosporum* would quite likely be successful.

There has already been some interest in *P. resiniferum* but no formal research. Seedless mutants are often mentioned as options, but induced mutants are so often associated with deletions and loss of vigor/yield that the approach does not appear to be as promising as triploidy (J.L. Brewbaker, pers. comm.). It is not unusual to produce rather sterile hybrids from interspecies crosses of whatever chromosome number. Results with *Leucaena* (about 70% crossability of species) and that of other tropical genera lead Brewbaker to suggest that one would find *P. undulatum* crossing (with some ease) other species in the genus. About 20% of *P. undulatum* trees in the Cinchona area that produced flowers in May 1995, had only flowers that appeared to be female (T. Goodland, unpublished data). But these would not be suitable for exclusive planting outside the forest as there may be significant pollen flow from hermaphrodite trees elsewhere in the area. More research would be needed anyway to ensure that female plants could not revert to hermaphrodicity. An important precondition to the use of sterile lines or a single sex is that *P. undulatum* would need to be capable of vegetative propagation. We have not tested this formally, but there is some evidence that it would be feasible. Cut *P. undulatum* stems planted back into the soil commonly survive and produce new roots and leaves.

3.5 Effect of *Pittosporum undulatum* on tourism

Although the beaches of Jamaica are still much the most important destination for tourists in Jamaica, more people are visiting the Blue Mountains and the number is expected to rise. Most tourists to the Blue Mountains hike, and the overwhelming majority of these people (about 20-30,000 people a year (P. Parchment, pers. comm., 1995)) walk up Blue Mountain Peak, a very popular destination for Jamaican as well as overseas visitors. Large numbers also visit Hardwar Gap each year, with much smaller numbers visiting other parts of the park. Although off-trail hiking is rare, difficult and discouraged by park authorities, it could eventually become a lot harder following heavy invasion by *P. undulatum* and the impact of a major hurricane. The resulting forest is likely to be a chaotic mass of fallen and leaning trees and dense regeneration of *P. undulatum* seedlings and vertical sprouts.

In the Blue Mountains *P. undulatum* is dissimilar to native tree species in its regular growth habit, thick crown and heavy flowering. During August to December orange capsules can be so abundant on *P. undulatum* trees that they can be seen from over two kilometres away, and can colour whole hillsides. The characteristic colours and textures of the natural forest are lost. Most people who walk up to Blue Mountain Peak are probably attracted as much by the prospect of long-distance views, than by the appearance of trail side forest. There seems to be little awareness in Jamaica of the phenomenum of alien weeds overwhelming native vegetation, so many Jamaican visitors may not become aware of, or care about, the *P. undulatum* invasion. Awareness is likely to be higher "ecotourists".

3.6 Conclusions

There will almost certainly be an increasing shortage of fuel and building materials in the Blue Mountain area over the coming years, therefore there is a great need to develop new sources of these essential goods. *Pittosporum undulatum* certainly has potential, particularly as a timber tree and source of charcoal. There is already a substantial volume of *P. undulatum* wood in forests within about two kilometres of Cinchona and this volume is increasing rapidly. The much higher value of sawtimber than firewood or charcoal means that any tree large enough (>20 cm DBH) and of satisfactory form should be used for that purpose, if harvesting proceeds. In heavily invaded forest the native plant diversity is generally very low and may not fall any further even with further increases in the dominance of *P. undulatum*. As only a very small proportion of the *P. undulatum* trees have reached sawlog size there is a strong economic argument for leaving these smaller trees to grow, as increasing size will mean that it will become less costly to clear *P. undulatum* trees from an area but with a much greater yield of larger-sized timber. The main argument against is the likely inability of native species to colonise these areas following harvesting, because the cost of removing the species (as opposed to the trees alone) could rise steeply. Outside the forest, *P. undulatum* might be a successful fuelwood species if grown in larger, wood-lot stands or in more sheltered sites, such as gullies. In comparison with most native species, it seems to be able to compete with and suppress herbaceous weeds. A major constraint to the promotion of the species outside the forest is the threat that it could continue to "infect" natural forest, therefore careful further investigation would be needed before the cultivation of *P. undulatum* was carried out or recommended to local communities or agricultural extension organisations. There has been a very limited reforestation effort so far in the Blue Mountains and little information is available on the suitability of other alien or native species.

4.

Control and management of *Pittosporum undulatum*

This chapter is in five parts.

1. We examine ways in which the distribution and density of *P. undulatum,* and other weeds, can be assessed more accurately, (our best current information on its distribution is given in 2.2).

2. We examine possible manual methods for the killing of individual plants, the term manual control meaning chemical and/or physical control. The desirable qualities of manual control are that it should be cost-effective, with minimal impact on the environment and workers, therefore we have explored every possibility of killing the species by physical means alone, as well as using chemical methods.

3. We assess the control of *P. undulatum* in lightly invaded forest. The objectives for the management of the species in lightly invaded forest should be straightforward, as the only reasonable alternatives are to slow or stop the invasion, or leave it to invade.

4. We examine *P. undulatum*'s management in heavily invaded forest. Its eventual removal from such forest is necessary to prevent further infection of other forest. Its management there is considerably more complex, for two reasons. The greater accessibility and density of *P. undulatum* means that production of wood and timber inevitably becomes a more feasible objective. Secondly, other weeds threaten to invade forest closer to Cinchona following any harvesting or killing of *P. undulatum*.

5. As it may not be possible to control *P. undulatum* manually, we have made a full investigation of the potential of biological control.

4.1   Assessing the distribution of *Pittosporum undulatum*

4.1.1 Visibility of *P. undulatum*

Pittosporum undulatum trees have an architecture that is more regular than perhaps any tree species native to the Blue Mountains. A straight bole, regular whorls of branches and a dense crown give the species a distinctive shape and texture (somewhat resembling a temperate *Picea* from a distance). There is a pronounced seasonality to the appearance of *P. undulatum* trees also. Leaves flush early in the year and for a few weeks are a light green, but gradually through the year they darken, so by August/September their crowns are noticeably darker than those of most native trees, (this is accentuated by the density of *P. undulatum* foliage, which masks more of the lighter coloured trunk and branches). *P. undulatum* trees tend to be found in certain types of location, near landslides and in secondary forest most obviously, but more usefully for identification purposes, in certain topographic locations, particularly on ridges or breaks in slopes.

We compared the usefulness of normal colour film (Kodachrome 24, with a neutral density filter) and colour infra-red film (Kodak colour infra-red film with a Wratten No 12 filter) for recording the presence of *P. undulatum* and other weeds. Colour infra-red photographs have been used successfully in the identification of tree species, mostly in temperate countries where they are particularly useful for distinguishing conifers from broadleaved species, and for the detection of diseased trees. The characteristics of colour infra-red photography are that green healthy foliage (with a high concentration of chlorophylls) shows up as a vivid red, and haze is penetrated to a greater degree than normal film. On the other hand, different greens of different species can become more similar in colour infra-red photography (Kodak 1987). However, only a small proportion of the colour infra-red frames produced useful photos. This was probably because of the highly sensitive nature of infra-red film to the correct light exposure and the difficulties of keeping the film cold enough in tropical conditions (recommended storage temperatures are -18 to -21°C). *P. undulatum* trees were usually clearly visible on the few good colour infra-red photos, though only when lit from behind the camera; if side lit, the reflections of the leaves of all trees seemed to mask the appearance of *P. undulatum*. The resolution was not quite as good as on the normal colour transparencies. Both techniques show promise when a systematic attempt is made to map the distribution of the species*,* from the ground or the air.

4.1.2 Potential for aerial photographs

It was hoped that *P. undulatum*'s distribution might have been discernible on aerial photographs, but this has not yet been possible. Two sets have been taken, the first, funded by The Nature Conservancy, was taken in April 1992 and the second, funded by the Canadian International Development Agency, in August 1992. The Nature Conservancy aerial photographs were at the nominal scale of 1:22,500 and were generally cloudless, except for the western part of the range (mostly between John Crow Peak and High Peak), the area largely invaded by *P. undulatum*. The Canadian International Development Agency aerial photographs were at 1:18,500 nominal scale and showed the whole range cloud-free. However, the scale was not large enough for individual *P. undulatum* trees to be identified, although clumps of trees that were known previously from fieldwork could be distinguished. No new sets of aerial photographs have been taken since 1992, but the park management have indicated that they may soon commission a set at a larger scale, partly to map the distribution of *P. undulatum* and partly to detect illegal logging within the park (D. Lee, pers. comm., 1994). Jamaica is well endowed with air-photo cover, with eight series of aerial

photographs (in addition to the two mentioned already) taken since 1941. Unfortunately, the largest scale used was only 1:10,000 (M. Rothery, pers. comm., 1994).

Detecting different tree species on aerial photographs is a well-developed practise in temperate forests, and guidelines given by Sayn-Wittgenstein (1978) provide a good idea of the scale that would be necessary to detect *P. undulatum* trees. At 1:500 most species can be recognised almost entirely by their morphological characteristics because twig structure and leaf arrangement can be seen; at 1:2,500 small and medium branches are still visible; at 1:8,000 individual trees can still be separated, except when growing in dense stands, but it is not always possible to discern crown shape. Therefore we estimate that the minimum scale that would be necessary to detect small *P. undulatum* trees would be about 1:5,000. Larger scales would allow for more certain identification of trees partially obscured by taller trees. Sub-canopy trees are almost undetectable on aerial photographs (J. Williams, pers. comm., 1994).

4.1.3 Ground survey

The systematic mapping of *P. undulatum*'s distribution by ground survey has not been attempted for a number of reasons. Ground survey's most useful role will be to make accurate assessments of the density of *P. undulatum* and other weeds in areas where they are known to occur. There are more extensive areas where alien weeds *could* occur, being within a plausible dispersal distance from known populations, but where the density is likely to be so low simple ground survey is likely to be unproductive, time consuming and rather hazardous. Some evidence of alien plants from remote sensing (from the air or vantage points) would be needed to justify speculative searching. There is a good case for using vantage points on the ground as an alternative (or complement) to aerial photographs. They could be taken when *P. undulatum* is most visible, (during September/October, which co-incides with the season when cloud cover is generally most frequent), and optimum time of day (when the sun is directly behind the observer). The slopes are sufficiently steep and disturbance of the canopy (particularly landslides and large treefall gaps caused by Hurricane Gilbert) sufficiently common for a network of vantage points to be set up covering most or all of the slopes threatened by *P. undulatum*, with the exception of some of the slopes on the north slopes. Some of the gaps are becoming less useful as vegetation fills them. Photographs taken from the ground would however be less useful than aerial photographs, as they could not be taken stereoscopically, so making the plotting of detected trees onto a map or Geographical Information System more difficult.

4.1.4 Finding *P. undulatum* during clearance operations

Park management would need more detailed information on the location of isolated *P. undulatum* trees and populations than we have at present. Observation of hillsides being invaded by *P. undulatum* suggests that it would often be difficult to detect *P. undulatum* trees before they reach the canopy, by which time they may have already started producing seeds. There are two steps in using spatial data.

1. Obtain accurate locations for the trees or population foci. The best way to do this would be to use stereo aerial photographs.
2. Use this distributional data to find these population foci in the forest. It can be surprisingly difficult to find *P. undulatum* trees in the forest. However, a team of experienced people, at least one of whom can read a compass accurately, and walking apart in parallel lines, should be able to find nearly all *P. undulatum* trees in even the most difficult terrain. A longer term option is to use the Global Positioning System, together with accurate remotely derived distributional information, but from our experience with the system in the forests of the Blue Mountains, its usefulness beneath the forest canopy is limited.

4.2   Killing individual *Pittosporum undulatum* plants

We have tried the following combinations of physical and chemical methods of killing *P. undulatum*.

Uprooting. Seedlings up to a couple of metres high are readily uprootable as they usually have a shallow root system and a strong stem which rarely snaps.

Cutting (with and without herbicide). Cutting is quick, especially for smaller trees, and allows harvesting.

Girdling (with and without herbicide). For large trees, girdling can be quicker than cutting, and it does not open up the canopy as quickly as cutting does.

Injecting herbicide. The injection of herbicides into holes bored into boles has been found to be an effective method of killing some species, in Australian for example.

Bark stripping. Stripping the bark off the base of the invasive tree *Maesopsis emenii* in Tanzania is an effective way of killing that species without the use of herbicides (N. Geddes, pers. comm., 1993).

Herbicides. We thought that it was too early to test a large number of herbicides at this stage in control so we tested only one, Tordon, the one we thought most likely to kill *P. undulatum*. It contains the active ingredients 2,4-D (39.6%)

and picloram (10.2%). An experiment carried out in Puerto Rico into the potency of different herbicides in the late 1960s found that picloram was much the most powerful of those tested (Dowler & Tschirley 1970).

We found out how effective each method or combination of methods is against different sizes of tree by use of a main experiment and three smaller investigations.

4.2.1 Methods

We set up a main experiment with different methods of treating *P. undulatum* trees and saplings in October 1992. The treatments were:-

NT   Control (no treatment).
C   Cut stems at 0.8 m above ground level.
CH   Cut stems at 0.8 m above ground level and apply Tordon at a mean application rate of 0.1 cm^3 cm^{-2}.
G   Girdle stems between 0.8 m - 1.0 m above ground level.
GH   Girdle stems between 0.8 m - 1.0 m above ground level and apply Tordon also 0.1 cm^3 cm^{-2}.
CG   Cut stem at 0.8 m and girdle between 0.3 - 0.5 m.

Four blocks were used, all moderately to quite heavily invaded by *P. undulatum,* (trees in lightly invaded forest are too widely dispersed and too difficult to relocate for inclusion in such an experiment). Not all treaments would be suitable against all sizes of *P. undulatum* (it would not be practical to girdle saplings or cut - by hand - the largest trees). Therefore, the number of replicates of each treatment in each size class are indicated below.

			Treatment			
Size class	NT	C	CH	G	GH	CG
>3 m-3 cm DBH (Saplings)	2	2	2	0	0	0
>3-15 cm DBH (Small trees)	4	4	4	4	4	4
>15 cm DBH (Large trees)	1	0	0	1	1	0

This gave 33 saplings/trees in each block, 132 altogether. Small trees were selected so that their DBH was evenly distributed through the range 3-15 cm, not randomly (small trees were much more numerous than larger trees). Each individual had to be at least 5 m from any other treated tree. Girdles were frilled to increase the amount of herbicide absorbed. Measurement of the regrowth was carried out in mid-March 1993, in December 1993 and in September 1994, i.e. 5 , 14 and 23 months after treatment.

In a second, smaller, control experiment, in January 1994 the bark was stripped off the base of 24 trees. There were two treatments, "Cut at 0.3 m and strip all the bark off to soil level below the cut", and "Girdle from 0.3 m to 0.5 m, and also strip all the bark off to soil level". Three trees of each treatment were selected at each of four locations near of Newhaven Gap.

In a third experiment 5 cm deep holes at two frequencies (either every 10 or 20 cm of stem GBH) were drilled into 40 *P. undulatum* trees at breast height. Undiluted glyphosate was poured in.

Lastly a simple experiment to test the effectiveness of uprooting was carried out in four forest types in September 1992. *P. undulatum* seedlings were pulled up forcefully until 40 had been pulled up at each location that had at least one snapped root of one mm diameter. The exact location of the detached root was marked with a flagged pin. The height and diameter at 30 cm of all seedlings and the diameter of all snapped roots at the point of breakage was measured. 264 seedlings, ranging in height from 20-186 cm were uprooted, of which 60.6% had snapped roots.

4.2.1.1   Results

Main control experiment. Results on survivorship at September 1994 in the main experiment are given below. Initial analysis of the data found no indication of any block effect, (and none was likely as all sites were in rather similar forest), so it were not included in the analysis. Analysis of variance showed that treatment had a highly significant ($p<0.001$) effect, with girdling treatments causing significantly greater mortality than cutting. Size of tree had no significant effect.

Table 4.1. Survivorship of stems by treatment 23 months after treatment.

Treatment	% Survival
No Treatment	100[a]
Cut + Girdle	82[a]
Cut	76[a]
Cut + Herbicide	55[b]
Girdle	56[b]
Girdle + Herbicide	11[c]

Those treatments with the same superscripts are not different at the 5% level (using Scheffe's test)

Those trees treated with herbicide and still alive in September 1994 produced significantly fewer and shorter sprouts than those trees not treated with herbicide ($p<0.001$). However, the production of any sprouts by these herbicide-treated trees is surprising, Tordon is generally thought to be one of the most effective herbicides against woody plants and it was applied at the recommended dose rate. Sprouts from stumps treated with Tordon invariably had a distinctive appearance, with mottled, usually flat, leaves with mucronulate tips. The sprouts produced after cutting *P.*

undulatum (without herbicide application) in HIFE sometimes have the same distinctive appearance, suggesting that the symptoms are indicators of general stress or ill-health in *P. undulatum*. Repeated cutting of *P. undulatum* eventually kills the species, though evidence from HIFE suggests that death occurs more quickly when the stumps are in the understorey than in more open areas. There may be a role for repeated cutting (without herbicide) along trails where cutting can be frequent and herbicide use would be less acceptable.

Girdling by just peeling the bark of *P. undulatum* is not effective, as tissue readily regrows over the exposed surface, bridging the girdle. The xylem itself must be cut into. Cutting into the xylem hastened crown loss and therefore probably death of *Maesopsis emenii* in Tanzania, but at the cost of stimulating sprouting (N. Geddes, pers. comm., 1993). The comprehensive review of Noel (no date) makes it clear that most species produce shoots from just below the girdle, the part of the tree above the girdle always dies eventually (often taking a few years), and that in most cases (though it wasn't clear if in every case) these shoots eventually die also, i.e. the individual dies. Initial analysis of the data on the seed production of girdled and control *P. undulatum* trees indicated that treated trees (either with or without herbicide) did not produce a large crop of capsules, although those that were produced were more visible because of leaf loss, so may have been more visited by seed-eating birds.

Bark stripping experiment. In the bark stripping experiment three of the cut trees and five of the girdled trees produced sprouts but these were clearly only coming from bark near soil level that had been left. Where all the bark was successfully taken off there were no sprouts. The bark of *P. undulatum* trees can be removed readily except near the base where the roots start to splay out and the stem becomes convoluted.

Stem injection. The result of the experiment has not been monitored yet. The potential of the "Hypohatchet", cost about US$300, could be tried, but apparently it is not that tough or reliable (R. Neal, pers. comm., 1994) and spare parts would be unavailable in Jamaica.

Uprooting. Despite about 60% of the roots of uprooted seedlings snapping none of these detached roots resprouted. About half the uprooted seedlings left on the soil surface survived, so they would need to collected and disposed of, which would add time to the operation (and would perhaps be unlikely to be carried out diligently anyway). But the systematic uprooting of seedlings as a management technique may be useful in limited areas with a high priority for clearance. Such areas would need to be monitored to see if alien weeds establish in the pockets of bare soil so created.

4.2.2 Other studies of the herbicidal treatment of *P. undulatum*

Clearly Tordon can kill *P. undulatum*. Since starting the project we have found out that Tordon was used to kill *P. undulatum* in Hawaii (Tunison 1992) and, more interestingly, that two other herbicides have been effective.

- In the Dandenong Ranges in Victoria, Australia, *P. undulatum* is killed by drilling stems and filling with undiluted glyphosate using a Velpar gun. The holes should be 8-15 mm diameter, about 50 mm deep, 4-6 cms apart and 20-50 cm above ground level and below the lowest living branch. It is essential that holes are placed vertically below any main branches, otherwise they may not be killed (Gillespie 1991).
- Concentrated Roundup (glyphosate 360 g l^{-1}) has been effective when applied to cut stumps in Australia (I.K. Stephenson, quoted in Narayan (1993)).
- Also in Australia, to preserve the sclerophyll forest, "bush regenerators" are controlling *P. undulatum* by cutting trees and painting the stumps with glyphosate, either undiluted or 1:3, or injecting glyphosate into sapwood at the same concentrations (R. Buchanan, pers. comm, 1994).
- A mixture of 2,4,5-T and diesel applied to stumps cut just above ground level was very effective at Jonkershoek in South Africa (Richardson & Brink 1985).

Woody plants are commonly treated by cutting then spraying the regrowth a few months later, thereby getting larger quantities of the translocatable herbicide Glyphosate into the plant, as with *Rhododendron ponticum* in the UK (Forestry Commission 1990). This method has limited applicability in the Blue Mountains as many of the cut stumps would be widely scattered throughout remote forest and therefore would be very hard to find again, although in very heavily invaded forest regrowth spraying could be a useful technique.

4.2.3 Time needed to eradicate plants

The time taken to treat various sizes of *P. undulatum* in different ways is shown below. The times do **not** include any time taken to locate the plants. If treatment times in any subsequent real control operation are found to diverge significantly from these estimates, these can easily be modified.

Treatment and size	Size of plant	Estimated time	Comments
Uproot seedlings	<2 m tall	10 to 30 secs	Uprooted seedlings should be either kept off the ground surface or collected and piled.
Cut	>2 m tall	20 to 80 secs	Using machete
Girdle	> 6 m tall	30 to 150 secs	Using machete, frilling as well
Application of herbicide	Any cut or girdled surface	30 to 70 secs	Time for transporting and handling herbicide not included.
Strip bark from 30 cm to soil	Over 3 m tall	50 to 150 secs	All bark must be removed

4.2.4 Environmental impacts and safety of manual control

There are serious worries about the environmental impact and persistence of Tordon. Picloram, a principal active ingredient of Tordon, was still present in soil twelve months after application in a tropical rain forest in Puerto Rico (reaching a maximum of 0.05 ppm at 12-30 cm deep) (Dowler & Tschirley 1970). As glyphosate appears to be effective, and has a very low toxicity to humans (Grossbard & Atkinson 1984), there seems to be little advantage in using Tordon, apart from its low cost. In Mauritius there is a large ongoing programme using herbicides to control many invasive species, and herbicide safety has become an important issue. Because of the strenuous work in humid heat it has not been possible to ensure that workers wear their protective clothing all the time, therefore glyphosate has found a greater role, in preference to more hazardous herbicides (W. Strahm, pers. comm., 1993). Some men in the Blue Mountains are familiar with the application of herbicides, because of their extensive use on the coffee plantations, and realise the potential danger to human health. We think that although herbicides may have significant environmental impact, the alternative of letting the invasion proceed could have considerably greater impact on the environment. The use of herbicides to kill woody plants is common in areas protected for nature conservation, such as national parks in the USA (OTA 1993).

Another potential drawback to the use of Glyphosate is that apparently at high temperatures it can evaporate and affect neighbour plants (Jeremy Williams, pers. comm., 1996), though such affects are not evident in the literature.

There is the possibility that cutting rather than girdling *P. undulatum* trees could increase the probability of alien weeds establishing, but this much less likely in primary forest than in heavily invaded secondary forest. In primary forest *P. undulatum* trees tend to be smaller and to have a dense layer of native seedlings and saplings beneath, in contrast to heavily invaded forest. Additionally, the alien seed input is likely to be lower because of the distance from the highly invaded and disturbed land on the south slopes.

4.3 Eradication of *Pittosporum undulatum* from lightly invaded forest

If the prime objective is to slow or stop an invasion, modelling highlights the importance of eradicating small outlying populations before starting to eradicate the main original population, even if it is expanding slightly faster than the satellite populations (Moody & Mack 1988). Therefore the possibility of *P. undulatum*'s removal from lightly invaded forest should receive top priority. We have had to make a number of simplifying assumptions in order to calculate the cost of doing this:

- age/size at which *P. undulatum* starts to produce seed (a typical height of 3 to 5 m, age 5-10 years). Given the inaccessibility of much of the forest there is a clear case for removing *P. undulatum* plants significantly shorter than this. In follow-up clearance operations the target should be to ensure that all *P. undulatum* individuals are removed before they get to 3 m.
- persistence of the *P. undulatum* seedling bank.
- time period over which *P. undulatum* seedlings are recruited from any seedbank.
- maximum growth rate of *P. undulatum* seedlings.
- time needed to find and then kill different sized plants. As *P. undulatum* grows more rapidly in gaps, greater control effort will inevitably be needed there. These disturbed areas are localised, often small, and workers should be alerted to the importance of checking for *P. undulatum* in any gaps they encounter.

Our assessment of the cost of clearing *P. undulatum* from the 255 four hectare areas ("cells") with an estimated density of <100 trees per hectare, (an area of 1020 ha) is in the range of 110,000 and US$145,000. It would be highly desirable for *Hedychium gardneranum* and *Polygonum chinense* to be removed at the same time but we can estimate the cost of doing this with much less confidence. It could easily cost as much again in labour and material costs, but with little increase in fixed costs. Therefore the total costs of removing the three main weeds from lightly invaded forest would likely be in the range of 220,000 to US$290,000.

The Blue Mountains are different from almost all other areas where much effort has been expended on the control of alien plants in that there are many people who live locally who would be willing and able to work on a manual control programme. Elsewhere labour has either been expensive (such as Hawaii or New Zealand) or scarce (such as the Galapagos Islands or Mauritius, where labour is only available when the sugar cane is not being harvested (W. Strahm, pers. comm., 1993)). A manual control programme would be labour intensive and therefore clearly beneficial in an area with high unemployment and under-employment (R. Kerr, pers. comm., 1993). Rather than hiring local people on a casual basis as labourers, it would be preferable to build up a committed team of people and train them rather like forest rangers, so that they will be able to take a wide responsibility for protecting the forest and promoting it amongst local people.

4.4   Management of *Pittosporum undulatum* in heavily invaded forest

Although it is less urgent to remove *P. undulatum* from heavily invaded forest than from lightly invaded forest, it should still be a long term objective, not least because of the value of *P. undulatum* wood. However, it is very important to discover ways of removing the species without leading to the forest becoming even more heavily invaded by alien weeds. The very high recruitment of *P. undulatum* and the generally low recruitment of other species following the removal of *P. undulatum* trees has been noted in 2.3.1. Other important aspects are mentioned below.

4.4.1 Soil seed bank

Two aspects of the *P. undulatum* soil seed bank are crucial. Firstly, the density of *P. undulatum* recruitment from the soil seed bank following canopy disturbance and secondly the period of time over which *P. undulatum* recruits emerge. If there were no *P. undulatum* soil seed bank then *P. undulatum* could be eradicated from an area in the time it takes for a recently germinated *P. undulatum* to reach a removable size. We carried out an investigation of the soil seed bank in Heavily invaded forest experiment, details of which will be published elsewhere. Briefly, the soil seed bank density of *P. undulatum* ranged from 424 to 11,565 seeds m^{-2} per plot, reaching a maximum for any one sample of 35,264 seeds m^{-2}, (taken from beneath a large *P. undulatum* tree). The highest density any native species (*Clethra occidentalis*) reached was 2611 seeds m^{-2}. *P. undulatum* emergence started after 42 days, reached a peak after 65 days and had ceased by 93 days. Without the artificial disturbance created in the seed trays seeds can remain viable in the soil for much longer, at least 2 years (see 2.2.1.3).

4.4.2 Effects of *P. undulatum* management on other invasive plants

Hedychium gardneranum. We have found *H. gardneranum* surviving, if not growing, beneath the densest canopy of *P. undulatum*, an environment where very little else can grow. The species also benefits from canopy opening, by increasing its rate of vegetative spread, as displayed in natural gaps and man-created clearings in the upper Clyde valley. It would unwise to remove *P. undulatum* without removing *H. gardneranum* if the latter were beneath it.

Polygonum chinense. The potential effect of *P. undulatum* on *Polygonum chinense* is greater than that on *H. gardneranum*. *P. chinense* does not occur beneath dense tree canopies, as some disturbance is necessary for its establishment. A significant finding of HIFE in July 1995 was the presence of at least twelve healthy newly-established *P. chinense* plants, up to four metres long, in the **Remove all *P. undulatum*** treatment in a heavily invaded block (Plot 3 in Block 1). In that plot, *P. undulatum* had comprised only 9.5% of the total basal area (of stems over 3 m tall) so it appears that *P. chinense* is able to colonise sites after the removal of relatively minor amounts of *P. undulatum*. *P. chinense* can also get established in gaps created by the fall of large *P. undulatum* trees. The understorey of heavily invaded forest is sparser than lightly invaded forest, benefitting species that are recruited as a result of gap creation. On the other hand *P. undulatum* is a more aggressive coloniser of gaps than most native tree species, (not just in terms of height but also crown growth), so there is probably a greater chance that the *P. chinense* would be suppressed in gaps where *P. undulatum* regeneration is dominant. It is pertinent whether *P. chinense* will invade forest that is presently heavily invaded by *P. undulatum* faster if large *P. undulatum* trees are left to grow and eventually fall, creating gaps, or if they are killed standing and allowed to disintegrate without such pronounced gap formation.

4.4.3 Managing *Pittosporum undulatum*

Both *H. gardneranum* and *P. chinense* have already spread into non-*P. undulatum* invaded primary forest, though the latter only in a few places. Therefore, their management in heavily *P. undulatum* invaded forest would make little difference to their rate of invasion throughout the Blue Mountains. The key issue in heavily *P. undulatum* invaded forest is the effect of *P. undulatum* management on the *degree* of invasion of other weeds. Their proximity to inhabited areas (and therefore weed seed sources and human disturbance) and potential disturbance caused by *P.*

undulatum harvesting, could mean that the areas become heavily invaded within a few decades, mostly by *H. gardneranum* and *P. chinense,* which could lead to a loss of tree cover within several decades.

It might be possible to eliminate *P. undulatum* from heavily invaded forest by the gradual removal of trees taking care not to cause so much disturbance that *P. chinense* or *P. undulatum* are recruited. As data from HIFE indicates the paucity of native seedling recruitment following *P. undulatum* removal from heavily invaded secondary forest, these areas could be replanted or resown with native species. There are likely to be major problems with the raising of large numbers of tree seedlings of some species in nursery conditions, and probably with the survival of planted seedlings. Lack of resources may mean that the re-planting of such areas would have to be an alternative to the planting, (of mostly different species), outside the forest to try to restore tree cover to the denuded slopes. As some forest is so heavily invaded it could even be sensible to convert the less steep and most accessible land to non-forest tree cover, such as agroforestry or forest plantations. After the cutting and uprooting of all *P. undulatum,* the timber could be extracted, the abundant foliage left on the soil surface to provide some protection and help to suppress weed regeneration, care taken to prevent fire, then plant either native species or exotic N-fixing species.

4.4.4 The use of fire as a way of controlling *P. undulatum* in the Blue Mountains

Controlled burning is being tried at present as a potentially cost effective way of controlling invading *P. undulatum* in Australia (Narayan 1993; R. Gleadow, pers. comm., 1993). For example, in New South Wales, a combination of fire and insect attack is used, where intense and moderate fire (100% leaf scorch) kills *P. undulatum*. If the fire is less intense, damage to the trunk is still severe and then borers (presumably longicorn) attack, killing the plant within a few months (R. Buchanan, pers. comm, 1994). Following a controlled fire in Victoria at a temperature of about 200-250^{0}C twenty percent of monitored *P. undulatum* individuals resprouted from basal buds, but all *P. undulatum* in areas suffering from higher temperatures were killed (Narayan 1993). Its bark is thin and resinous, reaching 6.5 mm thick on a trunk 30 cm DBH in its native range (Gleadow & Ashton 1981). It is more resistant to fire than many marginal rainforest tree species, but is still susceptible compared with fire resistant species such as *Eucalyptus* (R. Gleadow, pers. comm., 1993). A low temperature fire of 120^{0}C led to a net recruitment of *P. undulatum* seedlings (Narayan 1993). Twenty percent of two and a half year old *P. undulatum* seedlings exposed to 200^{0}C in laboratory conditions resprouted (Narayan 1993).

Fire is frequent in the Blue Mountains, especially on hillsides covered by *Melinis minutiflora* (Aldrich 1993). It is unlikely that heavily *P. undulatum* dominated forest would burn without disturbance even during an exceptionally dry period, partly because of the thin litter layer and small quantities of woody debris characteristic of heavily invaded forest, (largely due to the young age of many of these secondary forests). However, a scrubby area (which probably had more fuel due to a dense layer of shrubs and ferns as well as abundant *P. undulatum*) did burn in the dry dry season of 1989 and *P. undulatum* trees had been killed by a fire about 5-10 years ago in another area. This evidence, and what is known about *P. undulatum* in Australia, does suggest that *P. undulatum* would be partially or totally eliminated from an area suffering an intense fire. However, although there are some native species that can readily colonise burnt areas (such as a shrub *Dodonaea viscosa* and bracken *Pteridium aquilinum*) the main beneficiaries would probably be introduced weeds, particularly *M. minutiflora*. In addition, fires result in soil erosion and loss of soil fertility through oxidation and volatilisation, hence the use of fire as a cheap method of controlling *P. undulatum* in the Blue Mountains would be highly undesirable.

4.5 Biological control of *Pittosporum undulatum*

The second main way in which *Pittosporum undulatum* may be controlled is biologically. The Jamaican authorities have no policy directly related to the introduction and release of biological control agents, and park managers have expressed caution over its use (D. Lee, pers. comm., 1994). Nevertheless we have made a full review of the potential usefulness and risks of biological control, not least because many people think it will have wider applicability in natural vegetation in the future (D. Gardner, pers. comm., 1994). We are fully aware of the need for biological control programs to take all necessary precautions to ensure that any introduced agent will not attack or otherwise affect native or other economic species, and of the international protocols for screening potential biological control agents that have been developed to minimise these risks. Useful information has come from Hawaii, where there is the only active programme to control weeds in forests biologically, a programme that may have great relevance to the Blue Mountains (Markin and Gardner 1993; D. Gardner, pers. comm., 1994). The programme was started in 1982, has targeted over 10 weed species and has active programs underway for five of them, carried out by a team of seven full-time research scientists.

The principal objectives of this section are to provide the scientific information necessary to assess the potential effectiveness of biocontrol, targeted at different stages of the *P. undulatum* life cycle, and to consider the more general factors and issues that may affect its success. The most effective biological means of reducing the population of weeds such as *P. undulatum* is usually to target the reproductive stages in the life cycle of the species, from the initiation of flower buds to the time the capsules dehisce and expose the seeds. Most biological control efforts against plants have been directed towards this reproductive stage, especially against perennial plants (S.

Neser, pers. comm., 1992), but the possibility of biological control against the vegetative stage of *P. undulatum* (by a predator or disease that harms or kills by eating or damaging leaves, stem or root) is examined first.

4.5.1 Biological control against vegetative growth of *P. undulatum*

Effects of defoliation

Small *P. undulatum* seedlings are able to resprout following decapitation (Gleadow 1982). To determine if larger plants are resistant to defoliation we carried out a simple test in April 1992. Thirty *P. undulatum* seedlings (from 20 to 300 cm tall) at each of two sites (heavily and lightly invaded forest) were completely defoliated by hand, about two months after most *P. undulatum* plants had produced new leaves, (i.e. the new buds were small). In January 1993 all these seedlings had produced new flushes of leaves from terminal buds, with only very rare shoot dieback (T. Goodland, unpublished data). The ability of cut *P. undulatum* trees to resprout after being cut four times in HIFE indicates that large reserves in the stem or roots of the plant is probably a reason for its ability to produce new leaves following defoliation.

Predation in Australia

The following is a list of all the agents known to attack *P. undulatum* (or the genus more generally), none of them attacking flowers or fruit. Sources, in addition to the publications, are personal communications from Robin Buchanan, Peter Myerscough and Stefan Neser.

- *P. undulatum* is attacked by the Pittosporum longicorn larvae *(Strongylurus thoracicus)* if in a weakened condition. The larvae bore round tunnels in the sapwood just below the bark, usually concentrating on small to medium-sized branches. Individual branches are occasionally killed but otherwise the health of the plant remains unaffected.

- The larvae of a small (3 mm) fly, the Pittosporum leafminer *Phytobia pittosporphylli* (family Agromyzidae), commonly attack *P. undulatum* in certain areas, so that it is rare to see a fully expanded leaf undamaged. The larvae concentrate their feeding around the midrib causing discoloured sunken blotches about 1-3 mm across. Although most leaves on a single tree can be attacked, the plant is not seriously affected (Hering 1962; Jones and Elliot 1986; McMaugh 1985). *Phytobia pittospocaulis* sp. nov. lives in twig galls (Hering 1962). Together with S. *thoracicus,* the Pittosporum leafminer is the most common pest of *P. undulatum*.

- Scales, including soft scales such as the Cottony cushion scale (*Icerya purchasi),* the Pink wax scale (*Ceroplastes rubens)*, an alien, and the Chinese wax scale (*Ceroplastes sinensis),* another alien, can be common on *Pittosporum* species (McMaugh 1985; Jones and Elliot 1986).

- Gall thrips (*Teuchothrips* species, mainly *T. pittosporiicola)* commonly infest the leaves of *Pittosporum* species (especially *P. revolutum*) (McMaugh 1985), sometimes building up huge populations (Jones and Elliot 1986).

- Various "bugs" suck sap, including the Pittosporum bug (*Pseudapines geminata*) (oval in outline, about 8 mm long, and black with light markings) (Jones and Elliot 1986).

- The Pittosporum chermid or psyllid (*Trioza vitreoradiata*) causes small lumps on the leaves (Jones and Elliot 1986).

- The Pittosporum beetle (*Lamprolina aeneipennis)* can cause serious damage to leaves of *Pittosporum* species (particularly *P. hirtissimum* and *P. venulosum*) (Jones and Elliot 1986).

In the Sydney area these agents do not prevent trees being able to produce large numbers of seeds and, unless these agents are themselves controlled in the area, they do not hold much promise for the biological control of *P. undulatum* (P. Myerscough, pers. comm., 1994).

Anti-*P. undulatum* agents outside Australia

During the past twelve years a disease has been decimating the *P. undulatum* population in the Cape Town/Stellenbosch area of South Africa (I.A.W. MacDonald, pers. comm., 1993). The "blight" has been devastating the species in gardens, affecting hedges in particular. It is generally referred to as a viral disease but may be caused by a mycoplasm (M. Morris, pers. comm., 1993). It has not been investigated in depth and has apparently been welcomed by conservationists in South Africa (S. Neser, pers. comm., 1993). The disease may have originated on *P. viridiflorum* which is native to Southern Africa (I.A.W. MacDonald, pers. comm., 1993).

P. undulatum does suffer some herbivory in the Blue Mountains. We have compiled notes on the characteristics and distribution of all diseases and pest damage of *P. undulatum* that we have found in the area. So far we have identified seven different patterns of damage that we suspect are caused by seven distinct agents. Descriptions will not be given here as they are rather incomplete (it has proved very difficult to find any of these agents, despite examining hundreds of leaves), and a team of four Oxford University undergraduates, working in collaboration with us, investigated the subject in depth in mid-1995. The most common type of damage appears to be caused by a leaf miner. This type of damage is common, and may be more common where *P. undulatum* is growing in lightly invaded forest and/or in gaps (T. Goodland, pers. obs., 1993). We do not yet know the identity of the causal agent for this or any of the other types of damage. This "leaf miner" damage is long-standing, as we have found a herbarium specimen from 1947 in the herbarium of the Institute of Jamaica, Kingston of *P. undulatum* with apparently the same

kind of damage. Three of the types of damage were very localised (several square metres) in which all *P. undulatum* individuals were damaged, suggesting the possibility of future spread. All the responsible species are most likely to be local "generalist" species and none have lead to such extensive defoliation that death seems likely.

One potentially fruitful direction to follow in terms of biological control of *P. undulatum* could be a specific rust fungus (if one exists in its native range) (H. Evans, pers. comm., 1993). However, there is no existing knowledge of the rust fungus flora of *Pittosporum* species.

4.5.2 Biological control against reproduction of *Pittosporum undulatum*

Several aspects of the reproductive biology of *P. undulatum* will be dealt with here, not all directly related to biological control, though some of them provide necessary background information.

Phenology

According to Adams (1972) in Jamaica *P. undulatum* flowers between February to July and fruits between March to August. It has not done so during the last four years though. Based on observations (by T. Goodland, P.J. Bellingham, J.R. Healey and E.V.J. Tanner), and monitoring of labelled trees since 1990, flowering starts in mid-November to December and is mostly over by late February to early March, though in 1995 about 10% of trees were producing flowers in May. Capsules start to mature (become orangy-brown) during July to October. Compared with the timing of the onset of flowering, there is considerable variation in the rate of capsule maturing with habitat and light environment. *P. undulatum* in Mor Ridge forest (or transitions between it and other forest types) and/or in gaps or on gap edges mature earlier than other plants (T. Goodland, unpublished data). Capsules start to dehisce in late October to early mid-November with some capsules dehiscing as late as January. There was a high degree of synchronicity within an individual, with a tendency for those more heavily shaded to be less advanced. Strong winds can blow off a large number of capsules well before dehiscence, for example there was a significant capsule fall on the 22nd October in 1992 because of wind.

Monitoring of fecundity

Objectives. We collected detailed information on the fecundity of *P. undulatum* for the following reasons.

- As the high fecundity of *P. undulatum* is probably a key factor in its success it is useful to quantify it, so that it can be compared with the fecundity of native species, and with *P. undulatum* seedling recruitment.

- We wanted to see how fecundity varied in different environments, as observations suggested that degree of canopy openness may be an important influence on fecundity and phenology. The success of introduced agents in locating *P. undulatum* flowers or seeds may depend on their visibility, which could be lower in the understorey.

- The annual variability in seed production could affect the likelihood of a seed-attacking agent establishing.

- It could be important to know how fecundity varies with tree size, so increasing the precision with which managers could prioritise clearance (i.e. clearance of reproductive trees below a certain size could be delayed if their seed production was very low). A significant proportion of *P. undulatum* trees bear no seeds at all each year. If these individuals consistently fail to do so year after year, eradication of these could become a lesser priority, and the only way to investigate this is to label trees and monitor their fecundity for a number of years.

Methods. Thirty gaps and nearby understorey locations (i.e. 30 sites) in four areas were selected. The gaps in three of the areas were caused by Hurricane Gilbert in September 1988. The gaps in the fourth area were created in August 1992 by the selective harvesting of large *Juniperus lucayana* trees. The variability between gaps (in terms of forest type and canopy height, which had a strong bearing on gap size and openness) within each of the first three areas is considerable, greater than that between areas, so they cannot be treated as blocks. There is less variability between the Juniper gaps. Only *P. undulatum* trees less than 25 cm GBH were chosen, as larger trees would have had a greater proportion of their crowns in the canopy, so reducing the likelihood that gap formation would have an effect. With larger trees it also becomes much more difficult to accurately assess the number of capsules, because of the denseness of their crowns, especially when the trees are in undisturbed forest. The fecundity and phenology was assessed each year 1992-95, in August to October as this was the ideal time as the capsules were orange and highly visible but they had not started to dehisce.

All *P. undulatum* plants >2 m tall but <25 cm GBH either in, with at least half their crown in, each gap were tagged and flagged for later relocation. For each of these, a *P. undulatum* in the understorey was selected. Understorey *P. undulatum* trees tend to have a different growth form, as well as a slower growth rate, from those growing in gaps, so are only comparable in a loose sense. They were selected by, firstly, subjectively choosing a nearby point in the understorey (one that had a sufficient density of *P. undulatum*, and a similar aspect and slope to the gap); and then selecting, for each of the *P. undulatum* trees in the gap, the nearest *P. undulatum* to that point within +/- 20% of its GBH (Gilbert gaps) or +/- 10% of its GBH (Juniper gaps).

For each individual the following were measured:

1. GBH (1.3 m).

2. Number of cymes, if any.
3. Number of capsules on ten cymes randomly chosen from throughout the crown.
4. Predominant state of maturity of capsules on the ten cymes.

Results. Six capsules were collected from 23 of the *P. undulatum* trees being monitored for fecundity (11 from gap trees and 12 from understorey trees). The number of seeds in each capsule was counted and the presence of eaten or diseased seeds noted, the results given below.

	No. capsules	Mean	SEM
Gap	66	33.1	0.71
Understorey	72	30.3	0.80
All	138	31.7	0.75

There was no significant difference in the number of seeds per capsule between trees in the understorey and in gaps. There was no sign of any insects having bored into capsules or any other loss due to biotic factors, though capsules have been found in other locations that have been gnawed on the ground (probably by rodents) and on trees (probably by birds).

Figure 4.2 shows the fecundity of *P. undulatum* trees in the gap and understorey, with the data for the Gilbert and Juniper gaps combined.
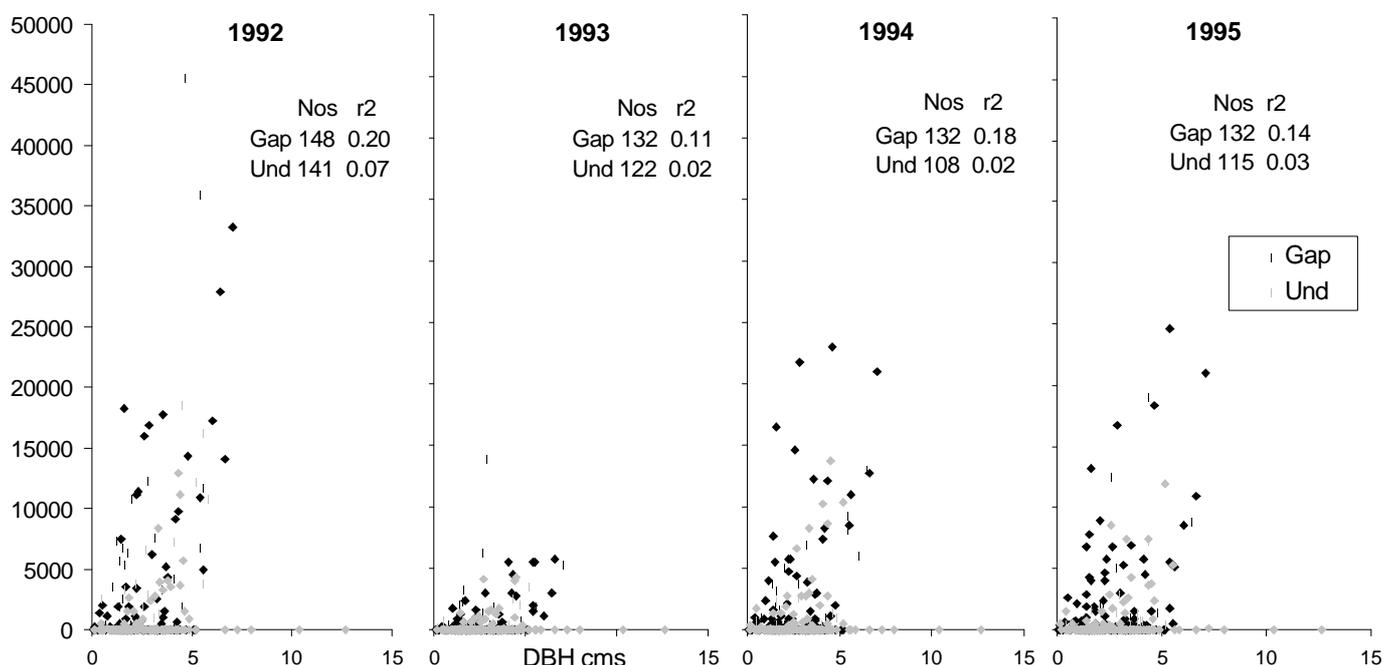
Figure 4.2 The relationship between the DBH of *P. undulatum* trees and their fecundity (number of seeds, on the y-axis) for four successive years, 1992-95, and the effect of gap creation on fecundity. Gap trees black and understorey trees grey.

P. undulatum seed production was high, the regression relationship between DBH and fecundity giving a mean of 37,500 seeds for a 8 cm DBH tree in 1992, though there was very large variability in fecundity amongst trees of a similar size. It varied substantially through the four year period, so that in 1993 only one tree produced more than 7,000 seeds, there being no discernible relationship between DBH and fecundity. The seed production of trees in the generally more open Gilbert gaps was slightly but not significantly greater than in the Juniper gaps, likewise in the understorey of both areas. For the Gilbert sites, understorey trees were more likely to bear few or no cymes, compared with gap individuals. Individuals at the Juniper site started producing cymes at a larger size than those at the Gilbert sites. This may be because before August 1992 the forest was relatively tall and undisturbed, and therefore the smaller *P. undulatum* trees would have been growing in a relatively low light environment. Additional data collected in 1995 (not yet analysed) showed that the fecundity usually falls to very low levels on large *P. undulatum* trees.
Flower and fruit development

To assess the number of flowers that survive to produce capsules, in February 1993 we counted the number of flowers on six flower bearing cymes on 39 *P. undulatum* trees <25 cm GBH (20 in a gap and 19 in the understorey). The trees were revisited in early December 1993 and the number of capsules in each marked cyme was recorded. In summary, overall 17.0% of the flowers produced capsules, with the mean number of flowers surviving slightly higher in gaps than the understorey. Twenty one of the trees produced no capsules at all. Observations made of two branches on separate trees during January 1994 showed that *P. undulatum* flowers took between 2 and 3 weeks

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between flower opening and "withering" after fertilisation. Some of the flowers had already been eaten, probably by birds; the Greater Antillean Bullfinch, *Loxigilla violacea*, has been seen eating *P. undulatum* flowers (T. Goodland, pers. obs.).

4.5.3 Effectiveness of biological control

Type of biological control

Because of the widely scattered nature of *P. undulatum* in lightly invaded forest classical biological control is most likely to be effective. "Augmentation" - involving direct manipulation of established populations of natural enemies through mass production or colonisation, or "Conservation" - involving habitat manipulation to encourage populations of natural enemies which kill the invader or seriously reduce its competitive ability, are not feasible in this instance.

As for the use of anti-reproductive or anti-vegetative control, the former has three advantages over anti-vegetative control. Firstly, it would lead to a much more gradual opening up of heavily invaded forest, thereby avoiding the risks associated with the sudden death of most trees in large areas; secondly, it would not affect the potential for the use of *P. undulatum* as a woodlot fuelwood species by local people, provided that an effective method for vegetative propagation was developed; thirdly, anti-vegetative control may be effective at reducing the vigour of *P. undulatum*, but is less likely to reduce population levels. However an anti-vegetative biological control agent would have advantages: seedlings are already widespread and locally common in primary forest so the development of an effective anti-reproductive biological control agent would allow many *P. undulatum* plants to grow and have a significant effect on parts of the forest for many decades; secondly, in lightly invaded forest rapid control is desirable, and is more likely with an agent that kills trees quickly; lastly it could be simpler and more predictable - assessing the effect a reduction in seed production would have on the population dynamics of the species is very difficult. A critical issue would be the reaction of the bird populations to such a change - if the fecundity of *P. undulatum* were reduced it does not necessarily follow that bird dispersal would decline proportionately, if at all (J. Wunderlie, pers. comm., 1993). Bird behaviour is of course a very complex issue which we have not had time to investigate.

Cost effectiveness

There are several reasons (given below) for thinking that biological control agents would be less than wholly effective. They may be able to reduce the rate of *P. undulatum* spread, and its final density, but would be very unlikely to eliminate *P. undulatum* from the Blue Mountains altogether. On the other hand, *P. undulatum* is likely to have a narrow genetic range in Jamaica, and this might mean that it is poor at adapting to new predators.

- Although pests of forest trees can search for their host trees in a forest, where the target species is widely scattered, as *P. undulatum* is in the Blue Mountains, there may be an appreciable lag before the plant is found, by which time seed production could have already started.
- As the number of *P. undulatum* individuals (or seeds) is reduced, it becomes harder for the agent(s) to find its target and for its population to be supported in a fluctuating environment (S. Neser, pers. comm., 1993).
- The large annual variation in *P. undulatum* seed production may make it more difficult to establish and maintain a seed-attacking agent and would very probably mean that in years of high seed production a large proportion of the seeds would survive.
- The difference in the climate between the Blue Mountains and *P. undulatum's* native range is appreciable (Healey *et al.* 1992b) and so may present an added difficulty in establishing a classical biological control agent.
- *P. undulatum's* spread in Australia, often in areas close to or contiguous with its "natural" 19th century limits, suggests that its population level was held in check by factors other than native predators, factors such as lack of native dispersers and fire.

The likely cost of a biological control programme against *P. undulatum* would probably be at least US$1,000,000, and take about 7-10 years. Because of the fixed costs of building and equipping facilities in Jamaica the costs of attempting biological control against *P. chinense* or *H. gardneranum* could be considerably less than that against *P. undulatum* alone.

Environmental aspects

Introducing an exotic agent to Jamaica inevitably involves some risk to native species, though release of biological control agents is nowadays undertaken only after a rigorous screening programme. There are no native members of the Pittosporaceae in Jamaica which may mean that the risks of any introduced agent attacking native plants is reduced (S. Neser, pers. comm., 1993). However, if introduced agents did kill all *P. undulatum* it is quite likely that the agent(s) would attack native species, especially as the population of *P. undulatum* declined. Evidence from South Africa indicates that the disease which is attacking *P. undulatum* there does not affect other species, however this would clearly need to be fully tested in Jamaica before any introduction of it to Jamaica.

A major problem with the intentional or accidental release of an effective lethal agent in Jamaica is that the sudden death of all *P. undulatum* in heavily invaded forest could have serious ecological consequences. The light levels on the forest floor would be increased substantially, almost certainly enhancing the growth of *Hedychium gardneranum* and probably facilitating the germination and growth of *Polygonum chinense.* The large quantity of dead timber could, during an exceptionally dry summer, leave these areas vulnerable to fire, either accidentally started, or by people deliberately taking advantage of the opportunity to clear forest for farming with the minimum of effort. The consequences would be very serious, with fire resistant weeds, such as *Melinis minutiflora,* benefitting. Perhaps a lethal, vegetative blight could be introduced eventually, once the dominance (and fuel load) of *P. undulatum* in heavily invaded forest had been reduced through management. The areas cleared of *P. undulatum* would also need to be monitored to ensure that *P. chinense* and *H. gardneranum* did not dominate.

4.6   Control options

As long distance dispersal of *P. undulatum* by birds is so prevalent it is essential that an integrated approach to its management in the Blue Mountains be adopted. This means that the management options for lightly invaded and heavily invaded forest and land outside the forest has to co-ordinated. For example, looked at on its own, it would seem to make little sense to try to eliminate *P. undulatum* outside the forest (where it probably does little harm and may even be beneficial), but if the ultimate objective were to remove all risk of the species re-invading the forest, its removal outside the forest may be essential. The various options as to which areas could be cleared of invasive plants are discussed below, in order of increasing ambitiousness. They are not strict alternatives, as easier options could be steps towards a more complete control of the species in Jamaica.

Trails. The invasion of all three main weeds may be hastened by the more open environment along trails. As it would be relatively easy to clear *P. undulatum* from them, this would be a cost effective way of reducing the invasion rate where they pass through little invaded forest. It would be essential if this were done to control *P. chinense* and *H. gardneranum* as well, otherwise they could be the main beneficiaries. Their rampant growth that can quickly make trails impassable.

Restricted areas of forest. Areas to be kept weed free could include:

1. Areas near or highly visible from trails, especially the Blue Mountain trail. To do this would decrease the rate of trail re-invasion and save remnants of natural uninvaded forest for visitors.
2. Sites (small areas of less than a hectare) of particular scientific/archeological interest or those with research uses. The act of removal itself is a disturbance, so for some of these areas it may be better to leave the invasion to proceed undisturbed.
3. Representative areas of natural habitat. These could be modelled on "Special Ecological Areas", a concept originating from the Hawaii Volcanoes National Park in Hawaii (Tunison & Stone 1992). These are areas of a hundred to a few thousand hectares that make the removal and exclusion of alien plants particularly worthwhile. Areas are delimited because of such attributes as high conservation value, a low degree of invasion and easy accessibility. Such areas could either be incorporated into one of the existing Blue and John Crow Mountains National Park management zones, such as the Special Conservation Zones or could be established separately. Two obvious areas are the forest in the vicinity of Portland Gap and the Blue Mountain Peak trail (a more extensive area than in 1. above), and the forest along and close to the Grand Ridge of the Blue Mountains between Morces Gap and John Crow Peak, because of its great diversity of forest types, research interest and accessibility.

All primary forest. Almost all primary forest is much less invaded than secondary forest and generally has a higher conservation value. But there is about three times as great an area of it then heavily invaded forest, and it is mostly remote and inaccessible. A major problem with trying to remove weeds from all primary forest is that some of the slopes in the Blue Mountains are so steep and craggy - places in which *P. undulatum* thrives - that they are only accessible with rope, dramatically increasing the risk and cost of control.

Heavily invaded forest. As the harvesting of *P. undulatum* from heavily invaded forest has the potential to be self-financing it could be a policy that could be self-justifying and could be pursued independently of action in lightly invaded forest. However, if the overriding objective is to leave the slopes of the Blue Mountains forested there is the real possibility that not clearing *P. undulatum* from heavily invaded forest would be the best policy.

Eradicate entirely from the Blue Mountains area. This would involve not only removing *P. undulatum* (and other weeds) from the heavily invaded forest around the less invaded core but also control of them outside the forest. If their eradication were achieved there would be little chance of either of the three main weeds ever reaching the area again, as we have no evidence that any of them occur in Jamaica outside the western Blue Mountains area. We have not attempted to cost this policy as it would be very expensive, unlikely to succeed, and of questionable merit, though it may be feasible in the long term.

4.7   Conclusion

P. undulatum trees would be readily seen on aerial photographs at the scale of 1:5,000, so given the complex topography of the Blue Mountains, such a set of photographs would offer the best chance of mapping its distribution. Further work on manual control needs to optimise the use and application rate of glyphosate to kill *P. undulatum*, and further test bark stripping as a non-chemical method.

Eradication from lightly invaded forest seems to be feasible, so control should start, at a small scale, as soon as possible. Removal from heavily invaded forest is less urgent, but harvesting could start also on a limited scale. Because of the very high recruitment of *P. undulatum* following canopy disturbance and the commoness of other serious weeds in more accessible forest, great care needs to be taken in any control or harvesting. It would be very difficult to manage forest for timber production which is threatened by three alien weeds with such different characteristics.

Biological control is an option that would involve a large commitment of money and would take several years of work, with no guarantee of any success at all. Past biological control efforts have almost always been justified by a weed or pest causing immediate and quantifiable damage to a commercial crop. In addition, it may well be politically unacceptable to introduce a disease that kills *P. undulatum,* as any proposal that may lead to the loss of forest cover would be hard to justify in a country that has lost so much. The use of biological rather than manual control favours scientists and technicians, many from richer countries, instead of local people who would be employed to carry out manual eradication. Therefore, careful political and socio-economic analysis of this option is required and it is crucial that the decision is taken at an appropriate local level.

5.

Synthesis

5.1   Costing the impact of *Pittosporum undulatum*

Costing the impact of *P. undulatum* involves addressing three questions. First, what are the values, present and expected, of the products and services of the forests of the Blue Mountains; secondly, what effect does *P. undulatum* have on these values; and thirdly what positive redeeming values does the species have. The main problem is not in estimating the actual value of the forests (though this is hard enough, especially when taking non-material goods into account), but of estimating the effect *P. undulatum* has on these values. This first part considers *P. undulatum* in the forest, both lightly and heavily invaded. We consider the potential of the species outside the forest separately afterwards.

5.1.1 Value of the natural forests of the Blue Mountains

Biodiversity. In a USAID report the estimated value of the maintenance of biological diversity in the Blue and John Crow Mountains and the Montego Bay National Parks was US$10,000 a year (USAID 1989), (no reasons were given for the derivation of this value). Two thirds of this can arbitrarily be attributed to the former national park (it is many times the area of the latter). We can then increase this US$6,666 to US$10,000 because of inflation over the past six years. Using a 5% discount rate this gives a net present worth of the biodiversity of the park of US$200,000. There is little economic justification for using 5% or any positive discount rate at all (C. Price, pers. comm., 1995), and without discounting the net present worth is of course infinite.

Pig hunting is the only significant use of the non-tree products of the range. Based on the level of hunting in the Cinchona area we estimate that 50 wild pigs, with a mean weight of 50 kg, are caught each year in the Blue Mountains. The value of pig meat is about US$3 kg^{-1}, giving a total value of US$7,500. Also, we estimate that US$5,000 worth of other game and non-tree products are harvested each year (excluding clearly illegally gained goods). Therefore the total is about US$12,500 year^{-1}, equivalent to about US$0.30 hectare^{-1}, equivalent to a net present worth of US$250,000 (using a 5% discount rate) for the whole Blue Mountains. A review of 24 studies of the value of informally collected ("foraged") non-tree products from tropical forests found a mean[2] value of US$69 hectares^{-1} year^{-1} (Godoy *et al.* 1993). Many of the studies were thought to over-estimate the value, sometimes appreciably (Godoy *et al.* 1993). Most of the surveys were carried out in forest that was heavily used by people and many where hunting of wild game for meat was important. In comparison local people do not use the forest resources of the Blue Mountains to such a degree. Reasons for this include the relatively recent arrival of the present people of Jamaica from the Old World, and the influence of Kingston and its fairly modern health care and industry.

Another potentially important value of the biodiversity of the Blue Mountains is the option value for future pharmaceuticals. A recent review of the option value of rainforest plants for medicinal use found a very wide range of values, from 0.2 to 90 US$ hectares^{-1} year^{-1}, most having been done in lowland tropical rain forests (Brown 1994). There are reasons for thinking that option values would be rather less for the Blue Mountains. Firstly, the species richness is less than is usual for lowland tropical rain forest (Latham & Ricklefs 1993). Secondly, a larger proportion of the plant species of the Blue Mountains probably rely on tannins and phenols for their defense, instead of more specific chemicals such as alkaloids common in most lowland tropical rain forests, lowering the likelihood of species having pharmaceutically useful constituents. The flowering plant and fern flora of the range is well known, with lower plants less well known. There may be only a few flowering plant species which have not been described (P.J. Bellingham, pers. comm., 1993). Researchers from University of the West Indies, Mona, Kingston have researched the pharmaceutical properties of many Jamaica plants, although few plants from the Blue Mountains have been tested. Several species show some promise as ornamental plants, such as *Meriania purpurea* and *Begonia acutifolia*. Although timber cutting is not permissible within the national park there are some species that are endemic to the range and have potential for use as a source of timber or for reforestation purposes, for example the large trees *Haenianthus incrassatus* and *Solanum punctulatum.*

Environmental services. The value of the forest in controlling such environmental services such as water yield and sediment loss is not known in the Blue Mountains, (and has not been calculated precisely for any catchment area in the tropics). Over 70% of water for the Kingston area comes from the range, so their importance is obviously very large. The effect of forest cover on water yield is not clear, but data from the experiment of M.A. McDonald indicates that removing forest cover increases runoff but increases sediment loss much more, findings consistent with catchment studies in other tropical countries (Bruinjzeel 1990). As very turbid water cannot be used, the overall effect in the Blue Mountains seems to be that forest loss decreases the volume of water clear enough to be used.

Tourism. We are aware of only one study of the value of the Blue Mountains to tourism. The annual value (1989 prices) of the national park to tourism was assessed as US$55,200 in the Kingston area and the southern slopes of the range, and US$20,500 in the Port Antonio area and the northern half of the park (USAID 1989). These figures

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[2] Where a range of values was given, the mean was used.

are the increases in income to communities in the area attributable to the establishment of the national park, which is likely to be less than the value of the mountains and forests themselves.

5.1.2 Quantifying the predicted effects of *Pittosporum undulatum*

Effect on native plant biodiversity. Although our analyses have indicated that the populations of the great majority of species would be reduced by *P. undulatum*, a few of those presently used by local people are common in heavily *P. undulatum*-invaded forest (for example, *Smilax balbisiana* and *Eugenia* species). But it could be a mistake to assume that these species would always maintain their populations if *P. undulatum* were allowed to maintain or increase its domination. Most important, there maybe irreversible losses, such as the extinction of native species, although there have not been any confirmed cases of an invasive alien plant causing the extinction of another plant species (V. Heywood, pers. comm., 1993). For a given area of forest we will be able to assess (with more data from lightly invaded forest) the decrease in density of a particular species endemic to the Blue Mountains or even predict its elimination. But without data on that species' distribution it is not possible to say how threatened the species is, and there is very little data on forest composition from further east in the range.

Environmental services. We have not been able to gain much hard information on the effect *P. undulatum* may be having on environmental processes. Although the effects on water yield are difficult to determine, any significant negative effect could be very serious; a 10% fall in water yield would be more costly to Kingston than a 10% increase would be beneficial. We think that there is strong evidence that hillsides dominated by *P. undulatum* are likely to have higher rates of erosion and sediment loss than uninvaded forest. Forest that becomes dominated by a single species (alien or native) becomes vulnerable to the same sorts of risk that any monoculture is vulnerable to. The high diversity of the forests ensures greater stability in the sense that a single or few species could probably be lost without major ecosystem consequences. If the invasions were left to "run their course" with the result that two or three alien species dominated the entire range, a single pathogen could have devastating effects on, for example, catchment hydrology.

Tourism. Estimating the impact of *P. undulatum* on tourism is very difficult. If even the whole of the range were denuded of all tree cover people would still visit the range, to climb Blue Mountain Peak for example. On the other hand, if alien weeds were present deep in undisturbed forest but were having no adverse effects at all, to some people knowledgeable about natural history, the value of the forest and the experience it can provide would be reduced.

5.1.3 Positive effects of *P. undulatum*

The principal benefit of *P. undulatum* in the forest has to be its potential as a source of wood and timber. The greater potential productive value of heavily invaded forest above uninvaded forest could be very great, but the cost of harvesting rises rapidly away from the road network. Additionally, the dominance of *P. undulatum* in heavily invaded forest, together with the high density of its wood, suggests that the biomass of such forest can be higher than uninvaded forest. If this is the case, the invasion tends to lead to a higher storage of the main greenhouse gas CO_2.

5.1.4 *P. undulatum* outside the forest

The main negative aspect of *P. undulatum* outside the forest is course the threat of re-invasion of forest. But outside the forest few native plants grow, because of alien weeds as well as disturbance, therefore *P. undulatum* has little impact on biodiversity there. Therefore, the overall balance of effects outside the forest are probably beneficial. More generally, the rapid and dense colonisation by alien weeds of abandoned land, in comparison with native pioneers, is likely to be more effective at reducing soil erosion and restoring soil fertility, increasing the sustainability of the type of shifting cultivation prevalent in the area. Although local farmers often find it harder to clear and keep alien weeds at low levels, some, such as *Polygonum chinense,* are thought to be soil improvers and areas dominated by them are sometimes sought out for clearance (E. Edwardes, pers. comm., 1993).

5.2   Control of invasive plants

Management considerations

Although manual control would be a major and continuing commitment for the national park its positive aspects should be recognised it is a way to provide employment for some of the people who live closest to the forests and thereby strengthen the relationship between local people and the objectives of the national park. However, eradication will require long term financial support and commitment from managers for control operations, and a further period of regular monitoring. A few years of enthusiasm for eradicating *P. undulatum,* during which trails are opened up, disturbance created and many local people made more familiar with the forest, followed by a discontinuation in support, could actually exacerbate problems of the Blue Mountains caused by unrestricted human

access, whilst leaving alien weed species as widely distributed as ever. A more serious decline in support for park management could lead to areas of heavily invaded forest from which *P. undulatum* has been removed being cleared of tree cover completely for crop production by landless farmers. Minimising forest disturbance is critical, even more so because of alien weeds. *Hedychium gardneranum* is spreading and increasing in density because of *P. undulatum* cutting for poles on Cinchona Hill (T. Goodland, pers. obs., 1994). Forest cleared and abandoned shortly afterwards in the Cinchona area is almost invariably dominated by alien weeds, possibly permanently preventing tree regeneration.

Preventing future problems

It is important to identify which alien species are potentially invasive at an early stage. *Syzygium jambos* occurs at two locations (at least) in the western end of the mountains and is regenerating freely (T. Goodland, pers. obs., 1994) but would probably not be thought of as a serious threat because of its localised distribution. However, in Hawaii, *S. jambos* is an even more aggressive invader (in terms of its ability to dominate forest stands) than the notorious *Psidium cattlenianum* (Loope *et al.* 1992), so is certainly a threat in the Blue Mountains. But only a very small proportion of introduced plants do become invasive. Garden records show that at least 300 plant species were introduced to the Cinchona Botanic Gardens (possibly many more) but only about half a dozen appear to be threatening the forests. Examples from the Cinchona area alone highlight the difficulty in deciding which species are a real threat and which are not. For example, *Schinus teribinthifolia*, an aggressive invader in several countries, was growing "freely" close to Cinchona in 1888 (Harris 1888), but we have not noticed it in the area for the several years that we have been working there. (It is now a problem in parts of lowland Jamaica, apparently a result of more recent introductions (C. Delgado, pers. comm., 1994)). *Ipomoea purga* ("jalap") is another weed that by 1881 had become locally dominant and "almost impossible to eradicate" (Morris 1881), but is now uncommon.

Jamaica posesses a system of notifiable noxious weeds, but the system is inactive and no species has been added to it for many years (L. Jefferson, pers. comm., 1994). If the law could be re-activated and new species added to it we recommend that *P. undulatum*, *H. gardneranum* and *P. chinense* should be added to it.

5.3   Conclusion

The director of the PARC project said that he thought a tradeoff between the negative and positive effects of any alien weed was unavoidable (D. Lee, pers. comm., 1994). Park management also believe that it should not be an overriding objective of the park to eliminate all "significant" exotic weeds, a strategy sometimes advocated by conservationists for protected areas in temperate countries. What is needed is complex tradeoff between three factors, the desire to minimise the negative effects of alien weeds, to do this at minimum cost, whilst maximising the positive benefits that alien plants (particularly *P. undulatum*) may bring. Our research has shown that the negative effects of *P. undulatum* on native biodiversity is likely to be very serious and on environmental processes potentially serious; *P. undulatum* could be eradicated from lightly invaded forest at quite a modest cost; *P. undulatum* has the potential to be a major benefit to local people, by providing them with tree products and employment in its control. Our attempt at a Cost-Benefit type analysis has been only partially effective, because of our inability to quantify and cost the negative consequences of *P. undulatum*.

Therefore our main recommendations are:

1. Get better data on the distribution of *P. undulatum*
2. Perfect methods of killing *P. undulatum*, with and without herbicides
3. Start eradication from lightly invaded forest on a small scale, so that more information can be gained about the costs involved
4. Small scale harvesting of the species from heavily invaded forest should also be allowed, carried out under strict supervision and accompanied by the monitoring of effects, and perhaps means of augmenting the establishment of native species
5. Develop markets for *P. undulatum* timber
6. Continue to monitor the permanent sample plots, so that we can gain much better information on the effect of *P. undulatum* on the native forests of the Blue Mountains
7. Initiate research into other serious weed species of the Blue Mountains, particularly their control and the nature of competition between them.

It has to be realised that even if those plants now considered as weeds are successfully controlled, other alien plants will probably always keep on invading the natural communities of the Blue Mountains, either those species already present at a low population level or new introductions to the area. The apparent invasibility of the forests, like montane insular tropical forests worldwide, means that park managers will always have to control alien plants.

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Appendix 1. Taxonomic information on woody plant species in the forests of the western Blue Mountains

The family and biogeographic status of all woody or semi-woody plant species mentioned in the report, and/or present in at least one of the permanent sample plots established during the project, are given in this appendix.

Full species name	Family	Status
Acacia mearnsii, De Wild.	Mimosaceae	Alien
Acalypha virgata, L.	Euphorbiaceae	Jamaican endemic
Alchornea latifolia, Sw.	Euphorbiaceae	Native
Besleria lutea, L.	Gesneriaceae	Native
Bidens shrevei, Britton	Asteraceae	Jamaican endemic
Blakea trinerva, L.	Melastomataceae	Native
Bocconia frutescens, L.	Papaveraceae	Native
Boehmeria caudata, Sw.	Urticaceae	Native
Brunellia comocladiifolia, Humb. & Bonpl.	Brunelliaceae	Native
Brunfelsia jamaicensis, (Benth.) Griseb.	Solanaceae	Blue Mts endemic
Calliandra calothyrsus	Mimosaceae	Alien
Callicarpa ferruginea, Sw.	Verbenaceae	Native
Calyptranthes rigida, Sw.	Myrtaceae	Native
Cassia viminea, L.	Caesalpiniaceae	Jamaican endemic
Cestrum hirtum, Sw.	Solanaceae	Native
Chaetocarpus globosus, (Sw.) Fawcett & Rendle	Euphorbiaceae	Native
Chusquea abietifolia, Griseb.	Gramineae	Native
Cinchona officinalis, L.	Rubiaceae	Alien
Cinchona pubescens, Vahl.	Rubiaceae	Alien
Cinnamomum montanum, (Sw.) Bercht.& Presl.	Lauraceae	Native
Cionosicys pomiformis, Griseb.	Cucurbitaceae	Jamaican endemic
Cissampelos pareira, L.	Menispermaceae	Native
Citharexylum caudatum, L.	Verbenaceae	Native
Clethra alexandra, Griseb.	Clethraceae	Blue Mts endemic
Clethra occidentalis, (L.) Kuntze	Clethraceae	Native
Cleyera theaoides, (Sw.) Choisy	Theaceae	Native
Clibadium terebinthinaceum, (Sw.) DC.	Asteraceae	Native
Clusia havetiodes, (Griseb.) Planch. & Triana	Guttiferae	Jamaican endemic
Conostegia montana, (Sw.) DC.	Melastomataceae	Blue Mts endemic
Critonia parviflora, DC.	Asteraceae	Jamaican endemic
Cyathea concinna, (Baker ex Jenman) Jenman	Cyatheaceae	Blue Mts endemic
Cyathea furfuracea, Baker	Cyatheaceae	Native
Cyathea pubescens, Mettenius ex Kuhn	Cyatheaceae	Blue Mts endemic
Cyathea woodwardioides, Kaulf.	Cyatheaceae	Native
Cyrilla racemiflora, L.	Cyrillaceae	Native
Daphnopsis americana, (Mill.) J.R. Johnston	Thymelaeaceae	Native
Dendropanax arboreus, (L.) Decne & Planch.	Araliaceae	Native
Dendropanax nutans, (Sw.) Decne & Planch.	Araliaceae	Blue Mts endemic
Dendropanax pendulus, (Sw.) Decne & Planch.	Araliaceae	Jamaican endemic
Duranta erecta, L.	Verbenaceae	Native
Eugenia alpina, (Sw.) Willd.	Myrtaceae	Blue Mts endemic
Eugenia brachythrix, Urban	Myrtaceae	Blue Mts endemic
Eugenia harrisii, Krug & Urban	Myrtaceae	Jamaican endemic
Eugenia marchiana, Griseb.	Myrtaceae	Jamaican endemic
Eugenia monticola, (Sw.) DC	Myrtaceae	Native
Eugenia virgultosa, (Sw.) DC	Myrtaceae	Jamaican endemic
Garrya fadyenii, Hook.	Garryaceae	Native
Gesneria alpina, (Urb.) Urb	Gesneriaceae	Blue Mts endemic
Gonolobus jamaicensis, Rendle	Asclepiadaceae	Blue Mts endemic
Gonolobus stapelioides, Desv.	Asclepiadaceae	Jamaican endemic
Gordonia haematoxylum, Swartz	Theaceae	Jamaican endemic
Guarea glabra, Vahl	Meliaceae	Jamaican endemic
Haenianthus incrassatus, (Sw.) Griseb	Oleaceae	Blue Mts endemic
Hedyosmum arborescens, Sw.	Chloranthaceae	Native
Hedyosmum nutans, Sw.	Chloranthaceae	Native
Heterotrichum umbellatum, (Mill) Urb.	Melastomataceae	Native
Ilex harrisii, Loes.	Aquilfoliaceae	Jamaican endemic

Ilex macfadyenii, (Walp.) Rehder	Aquilfoliaceae	Native
Ilex nitida, (Vahl) Maxim	Aquilfoliaceae	Native
Ilex obcordata, Sw.	Aquilfoliaceae	Blue Mts endemic
Ilex sideroxyloides, (Sw.) Griseb.	Aquilfoliaceae	Native
Ilex vaccinoides, Loes.	Aquilfoliaceae	Blue Mts endemic
Juniperus lucayana, Britton	Cuppressaceae	Native
Koanophyllon hardwarense, (Proctor ex C.Adams) R.King & H.Robinson	Asteraceae	Blue Mts endemic
Lantana camara, L.	Verbenaceae	Native
Lobelia assurgens, L.	Campanulaceae	Native
Lobelia martagon, (Griseb.) Hitchc.	Campanulaceae	Blue Mts endemic
Lyonia jamaicensis, (Sw.) D.Don	Ericaceae	Native
Lyonia octandra, (Sw.) Griseb.	Ericaceae	Jamaican endemic
Malvaviscus arboreus, Cav.	Malvaceae	Native
Mannetia lygistum, (L.) Sw.	Rubiaceae	Blue Mts endemic
Marcgravia brownei, (Triana & Planch.) Krug & Urban	Marcgraviaceae	Jamaican endemic
Maytenus jamaicensis, Krug & Urban	Celastraceae	Native
Mecranium purpurascens, (Sw.) Triana	Melastomataceae	Jamaican endemic
Meriania leucantha, (Sw.) Sw.	Melastomataceae	Jamaican endemic
Meriania purpurea, (Sw.) Sw.	Melastomataceae	Native
Metastelma atrorubens, Schltr.	Asclepiadaceae	Native
Metastelma harrisii, Schltr.	Asclepiadaceae	Blue Mts endemic
Miconia dodecandra, (Desr.) Cogn.	Melastomataceae	Native
Miconia quadrangularis, (Sw.) Naud.	Melastomataceae	Native
Miconia rigida, (Sw.) Triana	Melastomataceae	Native
Miconia theaezans, (Bonpl.) Cogn.	Melastomataceae	Native
Mikania maxonii, Proctor	Asteraceae	Blue Mts endemic
Myrcianthes fragrans, (Sw.) McVaugh	Myrtaceae	Native
Myrica cerifera, L.	Myricaceae	Native
Myrsine acrantha, Krug & Urban	Myrsinaceae	Native
Myrsine coriacea, (Sw.) R.Br. ex Roem. & Schult.	Myrsinaceae	Native
Ocotea patens, (Sw.) Nees	Lauraceae	Native
Odontocline fadyenii, (Griseb.) B.Nord.	Asteraceae	Jamaican endemic
Ossaea asperifolia, (Naud.) Triana	Melastomataceae	Native
Palicourea alpina, (Sw.) DC.	Rubiaceae	Native
Passiflora penduliflora, Bert. ex DC.	Passifloraceae	Native
Persea alpigena, (Sw.) Spreng.	Lauraceae	Jamaican endemic
Phenax hirtus, (Sw.) Wedd.	Urticaceae	Native
Phyllanthus arbuscula, (Sw.) J.F. Gmelin	Euphorbiaceae	Native
Picramnia antidesma, Sw.	Simaroubaceae	Native
Pilea grandifolia, (L.) Blume	Urticaceae	Jamaican endemic
Piper arboreum, Aublet	Piperaceae	Native
Piper fadyenii, C.DC.	Piperaceae	Jamaican endemic
Pittosporum undulatum, Vent.	Pittosporaceae	Alien
Pittosporum viridiflorum, Sims vel.aff.	Pittosporaceae	Alien
Podocarpus urbanii, Pilger	Podocarpaceae	Native
Polygonum chinense, L.	Polygonaceae	Alien
Prunus occidentalis, Sw.	Rosaceae	Native
Psychotria corymbosa, Sw.	Rubiaceae	Jamaican endemic
Psychotria sloanei, Urban	Rubiaceae	Blue Mts endemic
Relbunium hypocarpium, (L.) Hemsl.	Rubiaceae	Native
Rhamnus sphaerospermus, Sw.	Rhamnaceae	Native
Rubus ellipticus, Sm.	Rosaceae	Alien
Salmea scandens, (L.) DC.	Asteraceae	Native
Sapium harrisii, Urban ex Pax	Euphorbiaceae	Jamaican endemic
Schefflera sciadophyllum, (Sw.) Harms	Araliaceae	Jamaican endemic
Schradera involucrata, (Sw.) K.Schum.	Rubiaceae	Jamaican endemic
Sideroxylum montanum, (Swartz) Pennington	Sapotaceae	Jamaican endemic
Smilax balbisiana, Kunth	Smilacaceae	Native
Smilax domingensis, Willd.	Smilacaceae	Native
Solanum punctulatum, Dunal	Solanaceae	Blue Mts endemic
Symplocos octopetala, Sw.	Symplocaceae	Jamaican endemic
Tournefortia glabra, L.	Boraginaceae	Native

Trema floridanum, Britton	Ulmaceae	Native
Turpinia occidentalis, (Sw.) G.Don	Staphyleaceae	Native
Urbananthus critoniformis, (Urban) R.King	Asteraceae	Blue Mts endemic
Urera elata, (Sw.) Griseb.	Urticaceae	Jamaican endemic
Vaccinium meridionale, Sw.	Ericaceae	Native
Vernonia pluvialis, Gleason	Asteraceae	Blue Mts endemic
Viburnum alpinum, Macf. ex Britton	Caprifoliaceae	Native
Viburnum villosum, Sw.	Caprifoliaceae	Native
Wallenia calyptrata, Urban	Myrsinaceae	Blue Mts endemic
Wallenia crassifolia, Mez	Myrsinaceae	Blue Mts endemic
Wallenia fawcettii, Mez	Myrsinaceae	Blue Mts endemic
Weinmannia pinnata, L.	Cunoniaceae	Native
Xylosma nitida, (Hellenius) A.Gray ex Griseb.	Flacourtiaceae	Jamaican endemic

Appendix 2. Alien plants in the forests of the western Blue Mountains

Information on all alien plant species that we have found in the western end of the Blue Mountains, within about three kilometres of Cinchona is given here. The table below does not include alien species only present outside the forest or below 1450 m in altitude (1300 m in the primary forests in the Mabess River Valley). Modified areas within the forest such as roads were searched less assiduously than more natural environments. A sizeable proportion of the species are herbs restricted to paths and clearings. We have also tried to describe the distribution and pattern of "invasion" of each species by using two terms, frequency and dominance. Frequency is our assessment of how common the species is within the area and is categorised as either Common, Occasional or Rare. Density is our assessment of the maximum density that each species has attained within the area, and three categories (High, Medium and Low) have been used[3]. This list is based on collections and observations made primarily by P.J. Bellingham, T. Goodland and J.R. Healey, and identification by C.D. Adams of the Natural History Museum, London.

Full name	Family	Lifeform	Frequency	Density	Native range
Acacia mearnsii De Wild.	Mimosaceae	Tree	Common	High	Australia
Agapanthus africanus (L.) Hoffmgg.	Amaryllidaceae	Herb	Localised	High	S. Africa
Ageratum houstonianum Mill.	Compositae	Herb	Localised	Medium	Mexico
Athoxanthum odoratum L.	Gramineae	Graminoid	Localised	High	Euroasia
Arthraxon quartinianus (A.Rich.) Nash	Gramineae	Graminoid	Localised	Low	Ethiopia
Aristea gerrardii Weimarck	Iridaceae	Herb	Common	High	S. Africa
Briza maxima L.	Gramineae	Graminoid	Localised	Low	Temperate old world
Briza minor L.	Gramineae	Graminoid	Localised	Medium	Temperate old world
Brugmansia suaveolens Bercht.& J.S.Presl.	Solanaceae	Shrub	Occasional	Low	Brazil
Calceolaria chelidoniodes Kunth	Scrophulariaceae	Herb	Localised	Medium	Mexico to Ecuador
Cerastium triviale Link	Caryophyllaceae	Herb	Occasional	Low	N. Hemisphere temperate
Cinchona officinalis L.	Rubiaceae	Tree	Localised	High	Peru
Cinchona pubescens Vahl.	Rubiaceae	Tree	Occasional	Low	Peru
Conyza candensis (L.) Cronquist	Compositae	Herb	Occasional	Low	Temperate N. America
Eucalyptus species L'Herit.	Myrtaceae	Tree	Occasional	Medium	Australia
Fuchsia boliviana Carr.	Onagraceae	Shrub	Occasional	Low	Guatemala to Bolivia
Geranium carolinianum L.	Geraniaceae	Herb	Localised	Low	S. Canada to Mexico
Gladiolus undulatus L.	Iridaceae	Herb	Localised	Low	S. Africa
Grevillea robusta A. Cunn.	Proteaceae	Tree	Occasional	Low	Australia
Hedychium gardneranum Sheppard	Zingiberaceae	Herb	Common	High	Himalayas
Lapsana communis L.	Compositae	Herb	Common	Low	Eurasia
Lophospermum erubescens D.Don	Scrophulariaceae	Scrambler	Occasional	Low	Mexico
Macrothelypteris torresiana (Gaudich.) Ching	Polypodiaceae	Fern	Occasional	Low	Tropical SE Asia, W Pacific
Melinis minutiflora Beauv.	Gramineae	Graminoid	Common	High	Africa
Oxalis corniculata L.	Oxalidaceae	Herb	Occasional	Low	Cosmopolitan
Physalis peruviana L.	Solanaceae	Herb	Occasional	Low	Tropical South America
Pittosporum undulatum Vent.	Pittosporaceae	Tree	Common	High	S.E. Australia
Pittosporum viridiflorum Sims vel.aff.	Pittosporaceae	Tree	Localised	High	Southern Africa
Plantago lanceolata L.	Plantaginaceae	Herb	Localised	Medium	Europe
Plantago major L.	Plantaginaceae	Herb	Localised	Medium	Old World
Polygonum chinense L.	Polygonaceae	Scrambler	Common	High	E. Indies to Japan
Rhododendron arboreum Sm.	Ericaceae	Tree	Localised	Medium	Himalayas
Rubus ellipticus Sm.	Rosaceae	Shrub	Common	High	Montane subtropical Asia
Rubus racemosus Roxb.	Rosaceae	Shrub	Occasional	Low	Montane South India
Rubus rosifolius Sm.	Rosaceae	Shrub	Localised	Medium	S.E. Asia
Selaginella kraussiana (Kunze) A.Braun	Selaginellaceae	Club moss	Occasional	Low	Southern Africa
Setaria palmifolia (J.Konig) Staff.	Gramineae	Graminoid	Localised	Medium	S.E. Asia
Shuteria vestita Wight & Arn.	Papilionaceae	Scrambler	Common	High	Montane S. Asia
Sonchus asper (L.) Hill	Compositae	Herb	Occasional	Low	Eurasia
Sonchus oleraceus L.	Compositae	Herb	Occasional	Medium	Eurasia and North Africa
Stachys arvensis L.	Labiatae	Herb	Localised	Medium	Europe
Syzygium jambos (L.) Alston	Myrtaceae	Tree	Localised	High	Indo-Malaysia, Pacific
Taraxacum offinale Weber	Compositae	Herb	Localised	Low	North temperate regions
Thelypteris dentata (Forssk.) E.P.St.John	Polypodiaceae	Fern	Occasional	Medium	Tropical Old World
Trifolium dubium Sibth.	Fabaceae	Herb	Localised	Low	Eurasia
Trifolium repens L.	Fabaceae	Herb	Occasional	Low	Eurasia
Ulex europaeus L.	Fabaceae	Shrub	Localised	Medium	W. Europe
Veronica serpyllifolia L.	Scrophulariaceae	Herb	Occasional	Low	Macaronesia, temperate Eurasia
Viola patrinii D.C.	Violaceae	Herb	Occasional	Medium	C. Asia

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[3] An allowance has been made for size class, so that an alien tree which occurs at a density of one stem m^{-2} would be put into the high category together with a herb growing at 100 stems m^{-2}.

Appendix 3. Description of other main weeds

Hedychium gardneranum **Sheppard,** (Wild ginger), is a rhizomatous herb with leaves reaching two metres high. The species is able to survive and spread vegetatively in deep shade, but does not often flower in the forest understorey. Its distinctive yellow flowers are often visited by hummingbirds and its fruits are bird dispersed. It is cold tolerant, occurring up to 2500 m in the Himalayas (Graf 1978). When growing in more open areas, light levels beneath it may be sufficient for seedling establishment of other species but we have no data to confirm this. At least three other *Hedychium* species have naturalised in the hills and mountains of eastern Jamaica, though they frequently hybridise so sometimes identification is difficult (Adams 1972).

H. gardneranum was almost certainly introduced to the Cinchona Botanic Gardens, as the herbarium at the University of the West Indies, Mona, Kingston has specimens of *H. gardneranum* collected from Cinchona in 1913, but we have not been able to find records of its introduction. Herbarium specimens of two other *Hedychium* species (*H. flavum* and *H. coccineum*) were collected at the same time but neither are now common either in the garden or nearby. In 1947 *H. gardneranum* was said to have "spread very rapidly in recent years" in the Cinchona area and had "displaced native ferns in Fern Gully" (between St Helens Gap and Morces Gap) (Porten 1947). It is now concentrated in the mostly *P. undulatum*-invaded secondary forests close to Cinchona but is more widely distributed than *P. undulatum*. We have seen isolated clumps of a *Hedychium* species fringing old landslides far down the north slopes of the Blue Mountains (T. Goodland, pers. obs., 1992). It also occurs, together with *H. flavum*, as a dense almost continuous layer covering many hectares beneath secondary forest and Blue Mahoe plantations (*Hibiscus elatior*) between Section and Hardwar Gap, severely threatening the long term tree cover of the area. The genus is a serious weed elsewhere in Jamaica and may be widespread on the fringes of the Blue and John Crow Mountains National Park. *H. gardneranum* is a serious problem in many other countries. It is a "massive problem" in Mauritius (Q. Cronk, pers. comm., 1994). It is also a problem in New Zealand (P.J. Bellingham, pers. comm., 1994). On the island of Hawaii, *H. gardneranum* is considered one of the most threatening alien plants in the Hawaii Volcanoes National Park and has heavily invaded about 500 ha of forest (Santos *et al.* 1992). It is one of the three most serious weeds in the Kipahulu valley on the island of Maui, Hawaii (Anderson *et al.* 1992).

The eradication of *H. gardneranum* would be difficult because of the very high capacity of uprooted rhizomes to stay alive and resprout on contact with soil (a rhizome uprooted in August 1993 and placed in the crux of a tree was alive in November 1993, 15 months later), and the difficulty of removing all rhizomes from the soil. In addition to the problem of disposing of the uprooted plants, much soil disturbance would be caused. In New Zealand it is being controlled by a combination of physical (uprooting) and chemical means (P.J. Bellingham, pers. comm., 1994). In Hawaii thorough tests have been made of the chemical and physical control of the species (Santos *et al.* 1992).

Polygonum chinense **L.** (Redbush) is a scrambling herb that can form dense mats in disturbed areas two to three metres thick. Shoots of *P. chinense* can extend along the ground into the forest understorey from gaps for at least 7 m and can scramble up trees to about 10 m (T. Goodland, pers. obs., 1993). It can only establish following disturbance but once it has established it can be moderately shade tolerant, surviving (but not thriving) beneath light canopies. It seems to prevent the establishment of other plant species for perhaps an indefinite period because of the denseness and thickness of the mat it forms. A natural gap dominated by the species in the early 1970s (E.V.J. Tanner, pers. comm., 1991) is still dominated by it. A quick search under two of these dense mats failed to find any seedlings of any species. What may determine whether *P. chinense* will spread out from these areas (probably following tree fall at the edges) is whether seedlings of any tree species are able to grow in the sometimes narrow zone around the *P. chinense* mat where it is beneath bordering trees, and therefore thinner, but where the light levels are adequate for seedling growth.

P. chinense was probably introduced to Cinchona as it was collected from near there in 1905, by W. Harris, who described it as "rapidly spreading and becoming quite naturalised" (information from Institute of Jamaica herbarium). Its fruits are bird dispersed. By 1947 it was locally dominant in Clydesdale (Porten 1947). Its present distribution is poorly known but it is certainly more restricted in its distribution within the forest than *P. undulatum* or *H. gardneranum* as it is confined to disturbed forest around the fringes of the less human-disturbed forest core and a few large natural gaps deep in undisturbed forest. It occurs about 3 km west and east of Cinchona, on farmland on the north slopes of the mountains, near Claverty Cottage, as well as further east in St Thomas (Adams 1972). A herbarium specimen was collected from near the Hope River, near Kingston between 0-500 ft in altitude in 1950 (Institute of Jamaica). We have only found one other reference to it having naturalised elsewhere, on the island of Hawaii, where it was first collected in 1966 (Wester 1992). Apparently, it is not a problem there, as it was not otherwise mentioned in Stone *et al.* (1992). It is occasional near the sea in southern Japan (P.J. Bellingham, pers. comm., 1994) and is a gap coloniser in Papua New Guinea, though is not a problem in either of those two countries (P.J. Grubb, pers. comm., 1988).

Eradication of *Polygonum chinense* would be difficult. It is regularly cut and treated with the herbicide paraquat in coffee plantations, but this operation needs to be carried out every six months, as it grows back vigorously. Local opinion in the Blue Mountains is that the plant would have to be dug up by the roots when growing in the forest, where herbicides cannot be used because of its intimate association with native plants. Its distribution is confined to disturbed areas, so finding the species would not be so difficult. Biocontrol of *P. chinense* and *H. gardneranum* using an anti-reproductive biological control agent is likely to be a less effective means of eradication than for *P. undulatum,* as vegetative reproduction and spread of these two herbaceous species is important.

***Melinis minutiflora* Beauv.** (molasses or Wynne grass) is a stoloniferous grass that also readily establishes by seed. It only establishes in open areas, particularly after fire, so it is not as direct a threat to the forest as the other weeds are. Once it has established, it forms a highly flammable blanket that burns every few years and prevents tree establishment. It can also establish in woody areas that burn, increasing the likelihood that they will burn again. Fire may not be a requirement for its spread. *M. minutiflora* greatly increased in cover in areas that did and did not burn between 1971 and 1984 in the Hawaii Volcanoes National Park (Stone *et al.* 1992).

M. minutiflora was introduced into Jamaica about 1925 (Hitchcock 1936). We do not know if it was introduced to the Cinchona Botanic Gardens (we have found no records to suggest it was, and herbarium records have not been checked). However, the spread of *M. minutiflora* in the Cinchona area between 1943 and 1947 was described as "amazing", "choking out flowers once so common" (Porten 1947), so it may have been introduced to the area, rather than the gardens themselves. The Cinchona area has been a centre of innovation ever since the establishment of the original Cinchona Botanic Gardens in the 1860s. *M. minutiflora* was first collected in Hawaii in 1913, and is now found on all islands in the main chain except Ni'ihau (Wester 1992). In the Hakeakala National Park it spread "explosively" throughout formerly goat-browsed areas (Loope *et al.* 1992). In the Kamakou Preserve on the island of Moloka'i in the Hawaiian chain *M. minutiflora* was described as the perhaps the principal concern in the preservation and restoration of dry and mesic shrublands and forests below about 1125 m, a critical management issue (Holt 1992). It is also a serious threat in other reserves in the island chain (Loope *et al.* 1992).

Melinis minutiflora was one of the most important weeds in the Hawaii Volcanoes National Park that Santos *et al.* (1992) recommended as a subject for chemical control research. In the same park it has been targeted for biological control research, but problems are foreseen in the importation of an agent to control a grass species (Tunison 1992). No biological control program has ever been attempted against one because of fears that the agent would attack commercially important species (Pemberton 1980). Experimental chemical control with glyphosate was initiated in the Haleakala National Park in 1990 (Loope *et al.* 1992), but final results are not known.

***Pittosporum viridiflorum* Sims.** is a second invasive *Pittosporum* species in the Blue Mountains, one that we have discovered since the project was initiated. It was first collected by P.J. Bellingham in 1990 and its identity confirmed by C.D. Adams of the Natural History Museum, London. *P. viridiflorum* is an extremely widespread and polymorphic species distributed throughout most of southern Africa, with a number of informally named "entities", some perhaps comparable with populations (Friis 1986). It is mostly resticted to altitudes below 2500 m and occurs in a wide range of forest and evergreen bushland, including riverine forest and humid and dry montane forest. It occurs as a tree up to 20 m tall and 50 cm DBH. Its bark is pale to dark grey, or brownish, smooth (Friis 1986), compared with the more lenticel-pitted bark of *P. undulatum*.

P. viridiflorum was introduced to the Cinchona Botanic Gardens in 1883 (Morris 1883). We have found it about two kilometres from Cinchona in relatively undisturbed forest. There are only two trees of it in all of the permanent sample plots in the Blue Mountains (both in the Heavily Invaded Forest Experiment). One of these trees, which was probably too small to set seed (there were no seedlings in its vicinity), had between the first and second enumerations, the highest relative girth increment of any of the trees in the experiment, increasing from 3.7 to 8.5 cm DBH in 27 months, a 129.3% increase; (though the stem was rather elliptical in cross section so exaggerating its girth). As well as being fast growing as a tree, seedlings of *P. viridiflorum* appear to be similar in shade tolerance to *P. undulatum*, and locally can be as dense. *P. viridiflorum* can become a medium-sized tree, the largest so far found being 29.2 cm DBH.

P. viridiflorum is noticeably localised in distribution, seedlings occurring away from adult trees only occasionally. Its capsules are smaller than those of *P. undulatum* and contain a maximum of four seeds, each about the same size as those of *P. undulatum*. Therefore it seems possible that its localised distribution may be due to its low attractiveness to birds, as *P. undulatum* (with a much higher number of seeds per capsule, about 30) is common in all the places that *P. viridiflorum* has been found. *P. viridiflorum* flowered in July (in 1994), and capsules dehisce between March to May (T. Goodland, pers. obs., 1993). *P. viridiflorum* is from southern Africa so any predators introduced to control *P. undulatum* would not be a natural predator of *P. viridiflorum* though the vegetative and reproductive morphology of the two species appears to be quite similar.

***Rubus ellipticus* Sm.** is locally dominant in areas outside the forest, a species that was described as "spreading rapidly " in 1905 (Fawcett 1905).

***Setaria palmifolia* (J. K™nig) Staff.** is a shade tolerant grass now common along trails in the area, forming dense patches.

***Shurteria vestita* Wight & Arn**, (Sampson wisp) is a twiner that is very common within about a kilometre of Cinchona, often covering other weeds and climbing high into trees. This is only other alien plant that we suspect of being a major threat to the forest.

***Syzygium jambos* (L.) Alston**, (Rose apple) is a tree species spreading slowly from plantings.