AN ABSTRACT OF THE MASTER OF FORESTRY PAPER OF

Carolina Hooper for the degree of Master of Forestry in

Forest Science presented on May 29, 1992.

Title: The Ecology and Management of Invasive Plant Species on Tropical Islands.

Abstract approved: ______________________________
Dr. Michael Newton

Tropical island ecosystems have proven to be inordinately vulnerable to invasions by exotic plants and animals. Today these islands only contain small remnant populations of the original flora and fauna, and these populations are facing increasing pressure from invasive plants. This paper attempts to answer four important questions whose solutions will help explain the phenomena of greater invadeability on islands. These questions are 1) Why is the vegetation on tropical islands so vulnerable to invasions? 2) What are the attributes of the woody plants which invade islands? 3) What are the impacts of the invasions on the natural plant communities? 4) What are the methods of control and their costs and benefits? Section two contains a management plan for the control of the invasive species *Psidium cattleianum* in Ranomafana National Park, Madagascar.

The vulnerability to invasion of vegetation on tropical islands was attributed to evolution with different selective forces, the history of human disturbance on islands, disharmony, and an inherent vulnerability of island flora.

The attributes of invasive woody plants which promoted invasions was

explained in terms of safe site occupancy, demography, physiology, dispersal characteristics, and seed and fruit characteristics.

The impacts of invasive plants on tropical islands can cause long-term changes in ecosystems. These changes include; a change in community structure, a change in fire distribution patterns, a change in nutrient status, a change in the soil-water regimes and hydrological processes, and changes in diversity.

Ecological control methods in natural systems were examined as were their costs and benefits. The objectives of the control operation must be clearly understood before an integrated weed control program can be implemented. Mechanical, chemical, biological and legislative control options are testable, and such tests are recommended. The target plant should be understood on both an individual and population level. The economic, sociological and ecological implications of the control operations must be understood before an operation can begin. Often two or more control strategies may be used simultaneously to optimize the effectiveness of control. Once the desired level of control is achieved, native species for the site must be established to insure a maximum degree of ecological restoration.

The objectives of the management plan in Ranomafana National Park, Madagascar are to control and greatly reduce the extent and dominance of strawberry guava within the boundaries of the park, and to occupy the current guava infestation sites with native species. The management plan will be implemented in two phases. The first phase will be for research and development, during which the baseline information regarding extent of infestation, control methods, and nursery and outplanting technology will be developed. The second phase will be the integrated management program, in which the control measures and the restoration efforts will be accomplished. An estimated minimum budget for an eight-year management plan without biological control is $ 214,740.

The Ecology and Management of Invasive Plant Species on Tropical Islands

by

Carolina Hooper

A MASTER OF FORESTRY PAPER

submitted to

Department of Forest Science
Oregon State University

in partial fulfillment of
the requirements for the
degree of
Master of Forestry

Completed May 29, 1992

Commencement June 1993

APPROVED:

Professor of Forest Science in charge of major

Head of department of Forest Science

Dean of graduate school

Date Master of Forestry paper is presented May 29, 1992

ACKNOWLEDGEMENTS

I acknowledge and thank Dr. Robert Kellison and the Man and the Biosphere program for supporting my work in Madagascar, and Dr. Michael Newton for his assistance and guidance during the last two years.

Table of contents

LIST OF FIGURES

THE ECOLOGY AND MANAGEMENT OF INVASIVE PLANT SPECIES ON TROPICAL ISLANDS

INTRODUCTION

Isolated tropical islands have long fascinated explorers and scientists with their unique flora and fauna. These islands have evolved without the same major selective forces that have shaped continental organisms. This prolonged isolation has resulted in high levels of speciation and subspeciation with its associated endemism (Bramwell, 1979). The resulting species composition is often significantly different from that of continents. The disharmonic nature of tropical island biota has been long noticed, and inspired Charles Darwin to lay down the basis for modern evolutionary theory. Today these plants and animals are some of the most unique, least understood and most highly threatened life forms in the world (Brockie et al., 1988).

In the last 10,000 years there has been a dramatic acceleration in the rate of change in the configuration of tropical island ecosystems because of the increasing influence of human beings, and the organisms they took with them. Many of these changes have been purposeful and have allowed for the survival and development of human populations on tropical islands. Inevitably, when humans disperse, they carry with them plants, animals and agricultural practices from their place of origin (Jennings, 1979). These three factors can cause system-level disturbances which promote biological invasions.

Historical evidence shows that early human migrants caused many extinctions and produced fundamental changes in prevailing disturbance patterns (Kirch, 1982). The changing of the disturbance patterns set the frame work for many of the plant invasions which occurred with the arrival of the European migrants (di Castri, 1989). The Europeans carried with them an aggressive agricultural technology, plants which were well adapted to grazing and fire, and animals whose foraging chronically and selectively removed certain species from the plant community.

Plant Invasions

Tropical island ecosystems have proven to be inordinately vulnerable to invasions by exotic plants and animals. Today, only small remnant populations of the original flora and fauna exist on most tropical islands, and many extinctions have already occurred (Bramwell, 1979). Partly because of the extinctions and the presence of alien plants, underlying ecological processes on which native plant communities are structured have changed, in some cases irreversibly. Ongoing biological invasions have been identified as being the most serious threat to endemic island biota (Brockie et al. 1988).

The subject of plant invasions on tropical islands is complex, and widely variable depending on specific site conditions. The complexity of the problem poses many questions that ecologists would like to be able to answer. Of fundamental concern is why some plants become invasive while others do not. Plants which are invasive are of the most concern to managers, because they require little to no disturbance to enter plant communities. For example, more then 4600 different plants have been introduced to Hawaii, but only 86 (less than 2%) have become serious pests. Of these 86 species, twelve (32%) are considered invasive. These twelve invasive species are capable of invading plant communities without requiring human induced disturbances, and changing the structure and function of these ecosystems (Smith, 1985). (In this sense undisturbed is synonymous with naturally disturbed, or minimally disturbed, where there are no obvious disturbances such as cultivation, forest harvesting, forest clearing. Some disturbance, which may not even be perceptible, exists in all plant communities.) Worldwide, sixty species from 40 families have been identified as having the ability to invade natural or semi-natural ecosystems (Rejmanek, 1989). This paper is about this select group of species, and their invasions into conservation areas, or national parks.

Recognizing the importance of the problem of biological invasions, a world-wide effort to understand and halt these invasions has been initiated by the United Nations Scientific Committee on the Problems of the Environment (SCOPE) in 1982. This effort has resulted in research being conducted in South Africa, Australia, The United States, The United Kingdom and the Netherlands. One of their important conclusions is that tropical island ecosystems are the most vulnerable and the most invaded ecosystems in the world, and therefore they should be given top priority for both management and research. In an effort to increase management effectiveness, SCOPE recommended that the earlier the control operation is initiated, the higher the likelihood for success; that managers should aim to reduce system-level disturbances; and that it is important to have a clearly defined set of goals and plan of management (Usher, 1988).

Tropical islands around the world are establishing national parks (Usher, 1988), to preserve their remaining unique flora and fauna, and provide a source of revenue through tourism at the state and local level. However, the creation of the parks is often riddled with problems for both sociological and ecological reasons (Vitousek, 1988). Sociologically, the people living on islands are often very poor and have no choice but to exploit the land within the park. Ecologically the parks are often incomplete vestiges of the original biota, and already contain alien, invasive species. Because National Parks are set up for reasons of tourism, conservation, and research, it is often difficult to alleviate these problems. It is the challenge to both the land manager and the researcher to overcome these problems.

Psidium cattleianum in Madagascar

During the summer of 1991 I had an opportunity to inventory parts of Ranomafana National Park in eastern Madagascar in coordination with the United Nation's Man and the Biosphere initiative. The park was created in May of 1991 to preserve habitat for the endangered Golden bamboo lemur, and contains 45,000 hectares of tropical, cloud rain forest. While completing the inventory, I observed that large populations of strawberry guava, *Psidium cattleianum*, existed on land which had been previously treated with slash and burn agriculture, within the boundaries of the park. Strawberry guava is a highly successful introduced, aggressive invader on tropical islands world-wide (Smith, 1985; Huenneke and Vitousek, 1990; Lorence and Sussman, 1986). It was obvious that the *Psidium* was strongly out-competing native vegetation in both the overstory and in regeneration, in disturbed areas. Since it has the potential to spread to undisturbed rain forest, I became interested in what could be done to stop the spread of this plant to other areas of the park, and limit the infestations which have already occurred.

Psidium cattleianum, a native of Brazil, now has a pandemic distribution, and has emerged as a serious threat to indigenous, tropical island flora world-wide. Its red, fleshy fruits are easily dispersed by both birds and mammals, it has vigorous vegetative growth, is tolerant of shade, and it is capable of sprouting after cutting or burning and forming dense thickets under which very few other trees can grow (Smith, 1985; Huenneke and Vitousek, 1990). The hard shiny leaves do not appear to be palatable to domestic livestock in Madagascar (Hooper, personal observation). The strawberry guava has invaded areas of Hawaii, the Galapagos, Mauritius, Tahiti, Madagascar and undoubted many other islands. In Hawaii, it is regarded as the most serious threat to the integrity of native Hawaiian ecosystems (Smith, 1985). From personal observation, I predict that from the large quantities of strawberry guava I saw in Madagascar, and the poor native species regeneration below it, that it could transform large tracts of diverse rain forest to homogenous, single-species stands.

Because of the serious threat posed to the rain forests of Madagascar, a management plan for the control of *Psidium cattleianum* is of immediate concern and urgency. This plan would help preserve the indigenous rain forest of Madagascar from the encroachment of an aggressive invader which has been shown to change ecosystem structure and function in other parts of the world.

To answer the larger questions addressed by a management plan, information should be gathered from the experiences of plants which are able to invade native ecosystems without requiring disturbance, particularly *Psidium cattleianum*, in other parts of the tropics. These plants represent the most dangerous threat to the integrity of native, moist, tropical ecosystems worldwide, and the more information gathered about them the better land managers will be able to combat this problem. In addition to a management plan for *Psidium cattleianum,* this report will include information on the attributes of the alien plants which encourage their invasion without disturbance; the characteristics of the islands which have been invaded; the impacts of the invasive plants on natural systems and the methods of control being used in other areas.

Terminology

The following five terms will be used throughout this paper to describe plant species which are not native to an area (Smith, 1985; Macdonald, 1984).

Colonizers - Those species which enter unoccupied or sparsely occupied habitats.

Invaders - Those species which enter relatively intact vegetation and strongly dominate it or even displace it altogether. These plant require no disturbance, natural disturbance, or minimal evident disturbance to enter an area.

Immigrants - Those species which do not displace or markedly depress the resident population and become integrated into the communities.

Alien - Those non-native species which are economically or aesthetically undesirable. This category may include all classes of weeds, immigrants and colonizers.

Weeds - Plant pests of agriculture and other managed ecosystems, regardless of their geographic origin.

Objectives

In general terms the objective of this paper is to provide a source of information for a management plan for the control of the invasive species *Psidium cattleianum* in Ranomafana National Park, Madagascar.

Specifically this paper has five objectives. The first four are to answer the questions:

1) Why is vegetation on tropical islands so vulnerable to invasion?

2) What are the attributes of the woody plants which invade islands?

3) What are the impacts of the plant invasions on the natural plant communities?

4) What are the methods of control currently being used, and their costs and benefits?

The fifth objective is to write a management plan for the control of *Psidium cattleianum* in Ranomafana national park, Madagascar.

THE VULNERABILITY OF VEGETATION ON TROPICAL ISLANDS

The greater invadeability of islands relative to continents has long been recognized (Elton, 1958), although the reasons for this are still being debated. Darwin (1859) attributed the vulnerability to invasions of island flora to the low numbers of indigenous species. Since the time of Darwin there has been a great deal of work done on island flora. The broad patterns which have emerged from this research can be grouped into four general categories which help explain the island's greater invadeability. These categories are: 1) evolution with different selective forces; 2) History of human disturbance on islands; 3) disharmony; and 4) an increased vulnerability. This section will summarize the relevant literature in an attempt to explain why islands are so vulnerable to invasions.

Several characteristics of island vegetation have been reviewed by MacArthur and Wilson (1967); Carlquist, (1974); and Williamson, (1981), therefore only the characteristics which relate to invasions will be examined here. Tropical islands are found within 35 degrees latitude of the equator, and are primarily in the Pacific and Indian oceans. They are relatively small, although there are notable exceptions such as Madagascar and Indonesia, and surrounded by water, but otherwise exhibit remarkable individuality. Most smaller tropical islands are geologically young in relation to continents, and many are low lying atolls, although substantial topographic, geologic and environmental diversity does exist. Islands near continents tend to have more continental species and lower endemism than more isolated islands. Older islands, distant from continents, especially those with great environmental diversity, exhibit higher endemism than younger islands. The severity of island invasions increases in general with increasing isolation from continents (Bramwell, 1979; Brockie et al., 1988).

Evolution with different selective factors

Many ecologists believe that the fundamental reason for islands vulnerability to invasive species is that the plant communities on islands have evolved in response to disturbance regimes which are much more restricted in scope than those on continents. In contrast to continental systems, the predominant natural (pre-human settlement) disturbances on islands are volcanism, wind throw and landslides. The native biota are well adapted to these disturbances (Mueller-Dombois, 1981). Major disturbance patterns on continents include glaciation, trampling, browsing and selective foraging by ungulates, anthropogenic devegetation, frequent and intense fires, and a variety of predator-prey relationships. On continents, due to these disturbance regimes, opportunistic species have evolved adapted to persistence, dispersal, and colonization of both occupied and unoccupied areas (di Castri, 1989). Island flora are extremely vulnerable when confronted with the types of disturbances most common on continents.

Fire was not an important type of disturbance on most islands so very few island plants possess adaptations to it. Lightning is relatively uncommon on islands because their small land mass is not conductive to convective build-up of thunderheads. Even though vulcanism is common on many islands, most island ecosystems lack adequate dry fuel to carry fires once they have been ignited (Loope and Mueller-Dombois, 1989). The introduction of fire and plants which increase the level of ground cover by humans has caused the extinction of many native plants (Smith, 1985). In Hawaii, fires were traditionally small in area because of the natural firebreaks in the vegetation mosaic. However, exotic grasses have now invaded many of these areas, creating a continuous cover between islands of native scrub forest. This has resulted in a much larger area burned by fires in lowland ecosystems (Smith, 1985).

Evolution in the absence of large mammalian herbivores, (which consume large quantities of vegetation, and cause disturbance through trampling, selective foraging and digging), has resulted in high vulnerability to damage by introduced ungulates (di Castri, 1989). Island plants are relatively non-poisonous and have not evolved characteristics to deter animals such as chemical or physical defenses (Carlquist, 1974). Many of them do not have reproductive strategies which would allow them to maintain viable populations under heavy removal pressure (Loope and Mueller-Dombois, 1989).

History of human disturbance on islands

It is impossible to understand the current predicament of island ecosystems without having a clear idea of the history and effects of human interference. In contrast to continents where man was present long ago, on many tropical islands humans have only been present for the last 2-3000 years. In many ways the history of biological invasions on tropical islands is the history of man's influence during this period. This discussion will be broken down into three time periods: 2-3000 years ago to 1500 AD, 1500 AD. to the early 1900's, and the early 1900's to the present. These periods are chosen because they each can be typified by human forces which have promoted biological invasions (di Castri, 1989).

Between 2 and 3,000 years ago, the first Polynesians dispersals began in the Pacific and westward to India and Madagascar. It appears that their dispersal was triggered by large population pressure (Jennings, 1979). It is believed that in Hawaii they cultivated most of the land below 600 meters between the 13th and 18th century. Through the examination of bones found in caves, Olsen and James, (1982) estimate that the introduction of agriculture and hunting, led to the elimination of half the species of endemic birds on Hawaii.

Many Polynesian islands have relatively good archeological evidence for a human population decline. This is presumably due to attainment of a population level exceeding carrying capacity and accompanying resource deterioration (Kirsh, 1985). The effects of prehistoric man in altering island environments may have been more consistently severe because of the small sizes of islands which results in a lack of alternative land for exploitation. When the land becomes degraded, and species become threatened or extinct, it creates conditions which are favorable for biological invasions.

In South Africa, Deacon (1986) has studied the archaeological evidence for alien invasion of plants and animals. He states that until the arrival of Europeans in 1652, "there is no evidence that alien plants were replacing indigenous ones." The impact on the land seems to have been in clearing and redistribution rather then replacement. This conclusion may have been reached because the plants may have come from a source region close enough to South Africa that they were not really 'alien' plants, and did not have strong invasive properties.

The second period represents an historical turning point regarding biological invasions. This occurred around 1500 AD with the European exploration, discovery and colonization of new territories. It marks the beginning of the breakdown of natural biogeographical realms and their barriers, the cornerstone of the overall problem of biological invasions (di Castri, 1989). This period represents an enormous increase of communications and trades (and opportunities for biological invasions) in a relatively short period (Crosby, 1986).

The spread of Europeans throughout the world, with their domestic livestock and forage species, greatly enhanced the introduction of old-world species. The introduction of many new crop and weed species was facilitated by the adoption of the old world agricultural practices. This usually involved a more aggressive technology than had previously existed and facilitated colonization and naturalization processes of the alien plants.

In addition to agricultural technology, several other factors contributed to the spread of old-world species. It was often easier for the plant to establish if it left behind it's own pathogens, parasites, predators and competitors (Dean et al., 1986). Open spaces and spare resources were available in newly man-disturbed environments. Also, European man tended to choose climatic regions similar to Europe and establish similar cultural conditions there.

The third period is from the 1900's to the present. A marked and widespread impact by European settlers has taken place within the last 100 to 150 years, which has also led to many island extinctions (di Castri, 1989). In the last 100 years there has been a shift from the Euro-centric focus to a complete multi-focal globalization of human-governed forces that promote biological invasions. Even at present, development schemes guided by another country (or by an international organization), often carry technologies and biological introductions which could promote the establishment of invasive species. During this period there have been vast improvements in transport systems, including the opening of inner oceanic canals and airport transport. Human populations have become more mobile than ever. Multinational companies and governments began to exploit previously untouched resources. The result of increased transportation, technology, human population pressures and subsistence agriculture, has been the deforestation of tropical islands and a decreasing in the ecosystems resistance to invasive plants.

In summary, all environments existing today have been modified to one degree or another by man (Usher, 1988). On most islands the human induced environmental changes first occurred within the last 5,000 years, often much more recently. The periods of first human settlement are associated with high extinction rates as are the periods of first European settlement. Conditions changed dramatically around 1500 AD when the barriers of the biogeographical realms were broken owing to new transportation systems. However, the greatest changes on islands have occurred within the last 150 years, primarily attributed to chronic disturbances by subsistence agricultural practices, increased human pressures and greatly improved transportation.

Disharmony

The greater invasibility of tropical island forests is often attributed to low species numbers in certain groups in relation to their relative proportions on continental areas (Elton, 1958). This phenomenon is known as "disharmony" and denotes a taxonomic balance that differs from continental norms (Harper, 1977). Because of their isolation from the rest of the world, the flora on islands depends on the size, structure of the island, distance from the mainland and whether it originated in isolation from volcanic activity or separated from an established continent. Large islands such as Madagascar, which has been isolated since the Triassic period, contain a high degree of endemics, especially on the family and generic level (Bramwell, 1979). For islands located a long distance from the mainland, whose flora evolved in relative isolation, dispersal must have been accomplished by either wind, ocean currents, animal migration (mainly birds) or transport by man. Because of the difficulties in dispersing over vast ocean areas, the first three dispersal mechanisms produce a rate of introduction which is very low (Moulton and Pimm, 1986).

Islands of volcanic origin like Hawaii, which started in isolation without primary producers, have different types of disharmony than islands such as Madagascar, which were fully vegetated at the time of separation. Once a species arrived on a volcanic island, subsequent speciation (the evolution of similar closely related species from a parent population) occurred to produce a variety of unique and fascinating attributes. It is estimated that in Hawaii, 272 colonizing ancestors evolved into a flora of 1729 species and varieties of flowering plants, 95% of which are native and unique to Hawaii (Fosberg, 1948). The pre-human settlement figures are similar for Madagascar in which 90% of the estimated 10,000 plant species are native and unique to Madagascar. However, many of these plants have evolved from ancestors, or close relatives which still exist in southern India and Sri Lanka (Bramwell, 1979). The relative degree of endemism is fairly similar between isolated islands, but the initial amounts of plant cover dictated the relative success of the new arrivals.

Isolated islands are full of examples of disharmony. On many islands groups of organisms such as ants, rodents, and mammalian carnivores were completely missing. Madagascar has eight species of Baobab, while all of mainland Africa has only one. 95% of the world's chameleon species live on Madagascar. On the Hawaiian islands there are only four native species of orchid, the largest angiosperm family worldwide and only one mammal, a small bat. It also lacks gymnosperms, major plant families such as Fagaceae and Betulaceae, reptiles and amphibians. On the other hand, certain groups have undergone spectacular adaptive radiation, resulting in a diverse assemblage of closely related species occupying a wide range of habitats. The Drosophilidea of Hawaii, comprise about 600 species, a large percentage of the worldwide total (Loope and Mueller-Dombois, 1989).

Some of the same factors which have promoted adaptive radiation (evolutionary divergence of members of a single phylogenetic line into a variety of different adaptive forms (Futuyma, 1986)) over evolutionary time appear to promote vulnerability to invasion in modern time (Sussman and Raven, 1978). The arrival of exotic species, especially ants, mammals and certain aggressive plants has often caused large changes in the indigenous flora and fauna with only minimal resistance. Certain introduced ants are voracious predators and have obliterated native arthropod fauna in Hawaii (Vitousek, 1988). Rodents eat great quantities of seed and have prevented certain plant species from reproducing if they are without mechanisms to protect even a fraction of their seeds. Rodents have also proven disastrous to the reproductive success of endemic birds which have independently evolved flightlessness on different islands (Atkinson, 1985). A common pattern is the creation and exploitation of new resources through multiple invasions. This has happened in Hawaii with the introduced tree *Myrica faya*, which is dependent on the introduced nitrogen-fixing symbiont *Frankia* and introduced birds for seed dispersal (Vitousek and Walker, 1989).

The demise of co-evolved organisms such as birds and other potential obligate pollinators has led to the extinction of several plant species in Hawaii (Olsen and James, 1982). In Mauritius, the extinction of the Dodo has been related to the lack of regeneration of the principal rain forest tree, *Calvaria major*, which has had no recruitment for the last 300 years (Temple, 1977). In contrast, successful alien plant vectors often utilize introduced dispersal vectors (Smith, 1985).

The total number of species per unit area is smaller on islands than on continents, even though some families may be better represented as a result of disharmony (Elton, 1958). For isolated islands, MacArthur (1972) states this in the principle that "no island has nearly the number of species it would have if it were part of the mainland". The central idea in MacArthur and Wilson's theory of island biogeography is a balance between immigration, speciation and extinction rates. However, man has changed this pattern in recent time, and Williamson (1981) estimates that the immigration rates have been accelerated from one introduction in 50,000 years to 20 or more per year. On the other hand, there is little current evidence that recent introductions by themselves have much impact on current extinction rates. If they did the native Hawaiian flora would have been completely gone over 100 years ago.

Increased Vulnerability

There is some information in the literature supporting the idea of an inherent vulnerability to extinction by island biota even under optimal environmental conditions. E.O. Wilson (1961) first proposed a theory called the 'taxon cycle'. The taxon cycle concept involves the increasing habitat specialization and increasing vulnerability to extinction that a taxonomic group undergoes in the progressive invasion of an archipelago. Extinctions are due both to invasions by later arriving forms and to over-specialization in small habitats, hence a reduction in available niche space (Loope and Mueller-Dombois, 1989).

In both Hawaii and New Zealand the taxon cycle theory has been generally supported among island flowering plants. Loope and Mueller-Dombois (1989) have shown that the higher the taxonomic level of endemism (eg. endemic family versus endemic subspecies) the more chance of that plant becoming extinct or endangered with extinction. For example, a flowering plant in an endemic Hawaiian genus is 1.3 times as likely to be extinct or endangered as one endemic at the species level, and ten times as likely as a non-endemic plant (Loope and Mueller-Dombois, 1989).

The genetic basis of the phenomenon of inherent vulnerability is virtually unexplored. However, Carsen (1981, p.474) states:

> "The isolated nature of most island demes may be conducive to the evolution of restrictive specializations, whereas continental conditions are capable of giving rise to the genetic basis of a generalism, wherein the organism is homeostatic. This difference may underlie the observed failure among island organisms to evolve aggressive weedy organisms that have genotypes adapted to general purposes".

Of course, natural selection does operate on the island populations and they achieve a high level of speciation and adaptation to a range of environments, but in many instances may not be so well adapted to a variety of environmental conditions as the 'general purpose genotypes' of invasive species. Thus, when an invader arrives and modifies the habitat slightly, the native species are at a disadvantage.

Conclusions

It has long been recognized that tropical islands are more vulnerable to invasions and contain more invasive plants than continents. Possible explanations for this include evolution in the absence of major continental disturbance forces such as grazing and fire. Island flora have evolved with very few adaptations to these and are poorly prepared to effectively compete against continental species when invaders and unusual disturbances arrive together. Humans and their accompanying livestock and agricultural practices have led to the extinction of many native plants and animals, both with the arrival of the first Polynesians and with European man. A major result of evolutionary isolation is a taxonomic balance which differs substantially from that of continents. Because of this "disharmony" certain introductions such as rodents and ants have caused a drastic decline in indigenous flora and fauna with only minimal resistance. The competition within island flora does not seem to be as intense as on continents, but this has not been satisfactorily explained. Much what is known about plant invasions on tropical islands is theory, and could be applied to invasions on disturbed or undisturbed sites.

THE ATTRIBUTES OF INVASIVE WOODY PLANTS

Many introduced species are preadapted to their new environment by coming from source regions which have similar climates and substrates, but only a few of them have become invasive. Thus, bio-climatic matching may aid in successful introductions, but it does not necessarily result in invasive spread. Poynton (1984) has shown that in South Africa there is a poor correlation between the number of zones to which species are suited and the number in which they are classified as invasive ($r^2=0.177$, $p=.189$). Some species with broad habitat tolerances invade widely whereas others, with similar life histories and tolerances, invade only a few regions. Because of the poor predictive models available today it is important to study the attributes of woody plants which are known to be invasive, and do not normally require disturbance to enter undisturbed native plant communities. This section will synthesize the available literature in an attempt to determine the attributes of successful invaders in undisturbed areas.

While it seems to be possible to make some generalizations about successful invaders in disturbed and successionally young communities (Radosevich and Holt, 1984; Baker, 1965), there is apparently nothing unifying for invaders which do not require disturbance in natural communities. A list of 60 species from 40 families which are known to invade natural communities is given in Rejmanek (1989). The species on this list represent plants from many life forms, with a variety of different survival strategies. Some like *Pinus radiata* and *Hakea sericea* form new strata. Some apparently do find 'open niches", like the climbing fern *Lygodium japonicum* in bottomland hardwoods from Louisiana to florida. Others, like *Impatiens parviflora*, occupy disturbed habitats for some time and invade undisturbed areas only after some time. In general these plants have the ability to utilize resources more effectively, or in a different manner then the native plants. It may also be that areas which appear to be undisturbed actually contain subtle, system-level disturbances which are giving these plants a competitive advantage.

Despite the size of the task, ecologists and land managers need a predictive capability for management and prevention of plant invasions. In addition, interest in preserving and maintaining remnant tropical forests has escalated in recent years, and the invasion by pandemic plant species is directly counter to these efforts. For these reasons research is continuing to identify biological characteristics and relationships which are similar between many invasive alien plants. Although the best approach to do this has not been agreed upon. Noble (1989) suggests it is to look for 'syndromes of invasiveness', and that lists of ideal properties and broadly based plant strategies do not seem to be sufficient. This section will synthesize information on the important characteristics associated with the attributes of invasive woody plants.

Regardless of the conditions of disturbance in an area, the abundance of a particular species is governed by the frequency of occurrence of a particular microenvironment that suits plant recruitment. These areas known as "safe sites"

(Harper, 1977), are zones in which a seed is provided with the resources and conditions required to germinate and the stimuli required for breakage of seed dormancy. In addition, specific hazards such as predators, competitors, toxic substrates and pathogens must be absent. In summary, the number of seedlings appearing can be thought of as a function of the number of 'safe sites' offered by the environment (Harper, 1977). These microenvironments either are not being used by the indigenous flora or the invasive uses them more efficiently. Whether a plant can do this depends on the environmental characteristics and the plant characteristics discussed in this chapter, such as demography, resource allocation, physiology, dispersal capabilities and seed characteristics for soft-fruited plants. In reality, the factors which will be discussed are all related, and it is multiple characteristics of the plant and the environment which make it able to occupy the 'safe sites'.

Physiology

High reproductive potential and growth rate have supported the rapid spread of alien vegetation in many areas, especially those that are heavily disturbed. Rutherford et al. (1986) has compared growth rates of invasive alien plants (those which are able to invade natural areas) and assessed these rates to those of indigenous species. They determined that many invasive aliens in South Africa possess high growth rates, but that is was unlikely to be the sole cause of their success. The invasive plants they studied, mostly woody plants, were also well adapted to high moisture conditions and tended toward the avoidance of low water conditions. Many of the species have long-lived seeds and bird dispersed seeds. The plants often exhibit efficient nutrient uptake at low nutrient levels and are often evergreen or have a relatively long leaf duration. Many of the plants are capable of opportunistic growth behavior, contain high nutrient levels in the seed and have high nitrogen fixing capacities under favorable environmental conditions. It appears that for many plants which invade undisturbed environments, flexibility of response to changing environmental circumstances may be the most common life history design (Bazzaz, 1986).

The hypothesis that the invasive success of aliens can be accounted for in terms of the processes of photosynthesis and carbon allocation has been tested by Whiting et al. (1986). They determined that of the major invasive plant species in South Africa, (6% of the total terrestrial alien flora) 20 possessed the C_3 photosynthetic pathway, 3 the CAM pathway and none the C_4 pathway. Of the C_3 plants 16 are trees, three are shrubs and one is a grass; they are all perennials. They suggest that life-form, and hence carbon allocation is of more value to performance than mode of carbon assimilation. This differs from the situation in agronomy where many of the world's worst weeds possess the C_4 photosynthetic pathway. The reason for this phenomena was unexplained, and it is also unknown if this is true for countries other then South Africa.

A similar relationship between successful exotics and the C_3 pathway has been observed in India. The C_3 pathway is one of two principal differences between early successional natives and exotics (Ramakrishnan, 1984). The biomass of C_3 species is higher than that of C_4 in early succession outside of cultivation. The success of C_3

exotics was attributed to the temporal separation in peak photosynthetic production between the two groups. Biomass accumulation by C_3 species was more pronounced during the latter part of the growing season when solar radiation and temperature were both less; a situation where the C_3 photosynthetic pathway should be superior to the C_4.

Demographic Factors

Demographic factors are helpful in describing the process by which aliens become invasive. These plants become a problem when populations spread and grow from the initial introduction to the extent that they cause some direct or indirect cost to society. This involves population changes arising from the more favorable birth and death rates in the successful invader, relative to the native species.

It has been hypothesized that invaders have larger seed crops in the new environment than they had at their place of origin, and this difference is one reason for their invasion. Gill and Neser (1984) confirm this by showing there are larger seed crops of the alien, invasive *Hakea servicea* in South Africa than in its native Australia, but the same was not true for *Acacia cyclops*. Kruger et al. (1986) argues that high fecundity is important for the establishment of new populations, but favorable soil moisture conditions are more important for seedling establishment if the sites are not fully occupied. Efficient seed vectors and to a lesser degree large seed numbers are important in expanding populations because the range of dispersal is affected. Harper (1977 p.458) emphasizes the importance of seed numbers when he writes: "A reduction in the seed crop of 50% will halve the numbers of seeds reaching any point in the dispersal range. It will also alter the distance from the seed crop at which any particular density of seed will land." In addition, the greater the number of seeds, the greater the chance that a successful new genotype will be added to the population (Harper, 1977). Harper (1977) also states that differences in seed number are not necessarily important in the maintenance of established populations, where factors governing relative survival rate, such as time till reproduction, and availability of 'safe sites' become more important than absolute seed numbers.

The real strength of many successful invasives may lie in the fact that crowding of seedling populations causes little increase in mortality. This is certainly the case with exotic invasions in the wet forest remnants of Mauritius. Lorence and Sussman (1986) sampled two stands of mature evergreen wet forest in Mauritius to assess the extent of invasion by weedy exotic plants. The area with the higher percentage of exotics contained twice the density of seedlings than the less invaded stand. Virtually all of the seedlings counted in the heavily invaded stand were exotics. *Psidium cattleianum* is typified by having surviving stem densities ranging from 3 to 9 stems/m^2 at five elevations studied in Hawaii. These stems were measured from all age classes in thickets occurring along the roadside (Huenneke and Vitousek, 1990). Taylor and Walker (1984) found that population density and recruitment rate of *Cereus peruvians* positively correlates, there was no negative feedback on new seedling establishment, and populations increased rapidly to very high densities, in excess of 20,000 stems/ha by the age of five, in dry, desert-like communities. Other South African woody invasives, such as *Hakea servicea*, *Pinus* *pinaster*, and *Chromolaena odorata*, can sustain similar seedling densities and also invade seemingly undisturbed areas (Kruger et al., 1986). It may be that species with these characteristics can invade natural communities with less need for initial disturbance then invaders with variable mortality at the seedling stage, adequate if safe sites exist for them to do so.

The age at which a plant first reaches reproductive age is important for a successful invasion. Most invasive trees mature early, at ages between two and six years (Kruger et al. 1986) This is not necessarily different from native species, but it can preclude invasion by introduced species that have a late attainment of reproduction age. This is especially true if there is a recurring disturbance regime with a periodicity less than the time it takes for a plant to reproduce, and seed viability is short.

Dispersal

There is strong selective pressure for the separation of the offspring and parent in most plants, if nothing else is present to prevent establishment. A seed germinating remote from the parent plant may escape severe infestation by host-specific pathogens, predators and parasites and avoid competition with the established plant for light and water (Harper, 1977). Janzen (1970) calls this "distance-responsive predation", by which predation will decrease at an increasing distance from a parent tree, hence probability that the seed will mature to produce a seedling will increase. This section will focus on the nature of dispersal in alien plants and explore where it differs from natives.

A model to describe the process of invasion of a new habitat by a plant species has three principal elements (Roughgarden and Diamond, 1985). First the colonizing species must overcome the barriers to dispersal, such as vast distances and lack of dispersal mechanisms, between its native habitat and the new; second it must withstand the rigors of the new habitat; and thirdly it must grow and reproduce. The absence of an inherent long distance (trans-oceanic) dispersal mechanism is not a hinderance as humans are now the main vector; therefore this discussion will deal with dispersal after a successful introduction.

Rejmanek (1989) noticed that introduced species sometimes persist in small colonies, often in cultivation, for several generations before invasions are noticed. This phenomenon has been called "infection pressure". The theory behind infection pressure is that introduced species would have to reproduce sufficient generations locally for the population to reach a condition which allows spread from the point of introduction (Baker, 1986). The hypothesis has been tested in South Africa on species of *Acacia*, *Eucalyptus* and *Pinus*. A weighted regression analysis of the percentage of invasive species on period of introduction is highly significant (r^2 = .835, F=15.1, P=.03) with a negative coefficient, indicating a real increase in the density of infestation with time since introduction (Poynton, 1984). This provides support for the hypothesis of infection pressure. In addition to estimating density of establishment, the relative spread across the landscape by individuals should be assessed. This can present difficulty in sampling if the individuals are widely scattered, however, it may be a better indicator of population abundance and distribution.

A discussion in dispersal is complicated because it is often difficult to isolate the effects of unaided dispersal from human-aided effects. In general, successful alien plants in different environments are characterized by different modes of dispersal which are all important. Dispersal by animals is of cardinal importance in several of the South African biomes (Kruger et al., 1986), birds being the principal agent in the forest and savanna, and mammals in the karoo and desert. Wind is the primary disperser in the grasslands and fynbos. Riverine habitats host many alien plants which can disperse in water.

While it is generally true with wind dispersal that few seeds are dispersed over distances greater then a few times the height of the parent tree, the few carried in turbulent air are crucial in the colonization of new habitats (Harper, 1977). When talking about dispersal distances Harper (1977 p. 459) writes: "Wind-dispersed seeds (and probably most seeds) fall close to the parent plant. Where seed is shed densely, other causes of mortality such as the many effects of overcrowding bring plant density down to the supportable population size." This has been observed in South Africa where hakea and pine species have been shown to develop dense populations over several generations, originating from one or a few seeds.

Dispersal of species with fleshy fruits follow a typical pattern. Establishment of second generation individuals is generally restricted to the area around bushes or trees that bear fruit attractive to birds, or are foci for birds at the end of feeding. The bush clumps which arise serve as areas for further bird activity and thus seed deposition. The distribution, size and complexity of these clumps increases with age and with the number of native and alien plant species involved (Dean et al. 1986).

Seed and Fruit Characteristics of Species with Soft Fruits

Seeds which are embedded in soft fruits are usually, but not exclusively, from plants which are restricted to mesic, often fertile soils. They are usually dispersed by birds and mammals which may transport the seeds long distances before scarification and deposit. Plants with soft fruits are often long-lived and occur in areas where fire is both rare and common. Throughout the world these plants have been introduced as ornamentals and hedges (Knight, 1986). In South Africa, Knight (1986) did a comparative analysis of fleshy fruit displays in alien and indigenous plants. He concluded that seeds in soft fruited exotics differ from natives in two primary ways: most invasives have dormant, long-lived seeds while the indigenous have short-lived seeds; and the majority of soft- fruited invasive species have multi-seeded soft fruits, while the indigenous are mostly single seeded (Knight, 1986).

Other reproductive characteristics which seem to be important for invasive plants include a means of vegetative regeneration, dormancy mechanisms in the seed, species specific seed predators or pathogens in the place of origin, and disturbance regimes for which germination and establishment of the species are adapted (Dean et al., 1986).

An interesting study was performed by Knight (1986) in which he determined that the alien invasives have larger and more conspicuous fruit displays then many of the indigenous species. He concludes that the visible evidence of spread and infection of invasive alien plants might be related to its adaptations for vertebrate dispersal, but these adaptations are not critical for invasive success.

Conclusion

In conclusion, for an invasive plant to be successful it must be more able to find and utilize 'safe sites' than the indigenous flora. This can occur when seeds are adapted to develop in a wide range of microenvironments, or if the dispersal of the plant facilitates the arrival of the seed and subsequent seedling in a 'safe site'.

Research on invasives has produced some generalities which can help to explain their success in invading natural environments. While the generalities are not applicable to all invasives plants, they do appear to generally hold true. Invasives often have a high reproductive potential and growth rate; they exhibit efficient nutrient uptake at low nutrient levels and are often evergreen or have long leaf duration; carbon and other resource allocation may not be optimally fixed; many of the worst invaders are trees with the C_3 pathway, however grasses and other lifeforms are also major invaders; they have efficient seed vectors and often benefit from positive feedback loops with other exotics; crowding of seedlings causes little mortality; reproductive characteristics often include a means of vegetative regeneration; and have dormancy mechanisms in the seed and have species specific seed predators. Undoubted, many of the variables touched on in this section can be important depending on the specific site conditions. In all likelihood it is more than simply the attributes of the invading plant and the site characteristics which are important. Invasions are also enhanced based on what processes, such as pollination, herbivory, seed dispersal and seed predation are disrupted.

Attributes of *Psidium Cattleianum*

This section will review the literature regarding the specific characteristics of *Psidium cattleianum*, also known by the common names of strawberry guava and Chinese guava, to determine why it is able to invade environments without requiring disturbance. Strawberry guava is a member of the Myrtaceae family and originates from the tropical forests of Brazil (Huenneke and Vitousek, 1990). *Psidium* is strictly an American genus, which contains about 100 species (Baijnath et al. 1982). The majority of *Psidium cattleianum* introductions on islands have occurred between two and three hundred years ago. In Hawaii, the strawberry guava was first introduced by a Spaniard, Don Francisco de Paulo Marian, in 1791, as both an edible fruit and an ornamental (Shigeura and Bullock, 1983) and has been widely used by both native Hawaiians and immigrants. In South Africa the first introduction was made by a Dutch gardener, around 1659, and has subsequently invaded through the Cape province.

The general characteristics of *Psidium cattleianum* are prolific fruit production (10-12 seeds per fruit), high germination rates and broad environmental tolerances. It is a medium-small sized tree which is capable of forming dense thickets by producing clonal suckers originating from the roots, under which very few other trees can grow. These suckers can obtain the same height and diameter as the parent tree (Lorence and Sussman, 1986; Smith, 1985; Huenneke and Vitousek, 1990).

In Hawaii *Psidium cattleianum* has primarily been studied by Huenneke and Vitousek (1990). There it is a small tree which presents management problems in national parks and other natural areas. It is primarily of concern because, as Huenneke and Vitousek (1990, p.200) write: "While strawberry guava forms dense thickets along roadsides and in other disturbed areas, it is also successful in seemingly undisturbed intact forest." The guava forms a dense layer of vegetation at 4 to 5 meters height, below a native canopy. Very few native trees are able to regenerate under the guava, so when the canopy trees die, the guava becomes dominate. They show that leaf flushing and reproduction are independent of elevation, but that slightly higher reproduction rates occurred from 460 meters to 610 meters. Some stems were able to flower all year, but peak flowering occurred from June through October, while the lowest occurred in January through March. They recorded peak leaf flushing from June through August.

Strawberry guava can reproduce with seedlings or clonally produced suckers (Huenneke and Vitousek, 1990). The percentages of each varied considerably between two sites studied; one area had 16% seedlings and 84% suckers, while another site had 94% seedlings and 6% suckers (Huenneke and Vitousek, 1990). Seedling mortality was low on five study sites on an elevational gradient west of Hilo, in the range of 3-8%, during the first year. However, clonal reproduction had more successful growth and higher leaf area, probably because of the resources obtained from the parent. Huenneke and Vitousek (1990 p.208) made the observation that:

> "Seedling establishment in the field appears to be independent of soil disturbance, such as that provided by the rooting activities of feral pigs. Few

germinants established in experimental soil disturbances, despite high seed inputs; naturally occurring seedlings were found disproportionately often on bryophyte mats and other undisturbed substrates".

Even though soil disturbance by feral pigs is not critical to the development of strawberry guava, the fruit is very attractive to the pigs, as well as native and exotic birds, who widely distribute them throughout the forest (Smith, 1985). Soil characteristics are not a consideration, as *Psidium* is found as a weed in every soil type (Shigeura and Bullock, 1983).

Because of the presence of guava in remote, seemingly undisturbed areas of intact forest, it must be able to utilize 'safe sites' on the same substrates that support germination of many native plants. Unfortunately, little is known on the possible interaction between guava and native seedlings. However, prolific fruit production and high rates of suckering show that these isolated populations can rapidly spread from even a very small population under low light conditions (Huenneke and Vitousek, 1990). The actual rates of suckering have not as yet been recorded. Smith (1985 p. 228) describes: "stands of the species infest hundreds of hectares of mesic and well-drained rain forest areas between 200 and 1300 meters. Vast areas of mature ohi'a and koa forests have a dense understory of strawberry guava." In light of the present circumstances in Hawaii, *Psidium cattleianum* appears to be very adaptable to areas already occupied by native plants.

Psidium cattleianum has been observed growing on a broad variety of sites. It can grow in areas with 200 inches of rain to areas with less then 10 inches, and can grow with continuous free standing water, for short periods of time (Macdonald, 1983). In South Africa it is typically found on forest edges and mesic riverine habitat, however in Hawaii it exists in both wet and dry areas (Huenneke and Vitousek, 1990).

The seeds have been shown to germinate rapidly under a wide range of environmental conditions, and germination success is not correlated with elevation in Hawaii. Because germination is easily triggered under a broad range of conditions, it is unlikely that there is a large seed bank stored in the soil. However, scarification produced seeds with lower germination rate than unscarified (Huenneke and Vitousek, 1990).

Strawberry guava responds favorably to most known disturbances. It can tolerate wind very well due to its deeply penetrating roots, and requires a tremendous horizontal wind force to uproot (Shigeura and Bullock, 1983). It's roots are an excellent source of propagation and will readily develop shoots when exposed to light or slightly injured by herbicides or ground moving equipment (Macdonald, 1983). It also responds favorably to natural disturbance regimes within the rain forest of Hawaii. It appears to remain uninhibited by the falling of tree fern fronds which have been noticed to cause mortality in native seedlings (Huenneke and Vitousek, 1990). The effects of fire on strawberry guava have not been recorded.

As with many exotic organisms, positive feedback loops between introduced vertebrates and *Psidium* do exist. Of primary importance in Hawaii is the feral pig which relishes the juicy red fruits of *Psidium* and deposits them in soil disturbed by their rooting and digging. Birds are also important vectors (Smith, 1985). However the control of exotic animals alone will not control *Psidium*, it will simply slow down its rate of spread to otherwise unaffected areas. This is primarily because the establishment of *Psidium cattleianum* seedlings is independent of soil disturbance, and it is spread by both indigenous and exotics animals (Huenneke and Vitousek, 1990).

THE IMPACT OF INVASIVE PLANTS ON TROPICAL ISLANDS

Plants which invade natural undisturbed forest communities (invaders) have the ability to alter the conditions of life for all of the organisms in an ecosystem (Ramakrishnan and Vitousek, 1989). The observed consequences of biological invasions have offered the opportunity to demonstrate the effects of individual species on ecosystem properties.

The study of the impacts on natural communities has often been extremely difficult, although the concern about these invasions and the motivation for their control stems almost entirely from their presumed deleterious effects. The available evidence is almost exclusively from description, with very few controlled experiments. One reason for this situation is that it is difficult to quantify an impact without having a suitable control. A controlled experiment must have areas which show a range of the likely densities of the invasive organism and it should be suitably replicated. An important reason to quantify the effects of a plant invasion is to determine the effect removal measures will have, compared with the effects of the invasion itself.

The following discussion of impacts contains examples from both islands and continental systems. The continental examples have been included for three primary reasons; 1) to illustrate a broad range of potential impacts, many of which probably occur on islands but have yet to be documented, 2) Many of the continental examples occur in areas of relative insularity, which have some similarities to islands,

3) some of the best work describing the effects of invasions has been done on continents.

It is important to have a historical perspective when studying the impacts of alien organisms. The impacts can vary according to the age of the invasion. Most of the current, serious invasions on islands are less than 200 years old. Older invasions may well be inseparable from the native biota. Recent evidence shows that the impacts may vary with time. In South Africa there are two examples of post 1960 reduction in alien species due to a build-up of an adapted community of predators and pathogens (Morris, 1982). However, the reverse has been shown in Australia and the U.S. where both relative abundance and absolute abundance of alien weeds were shown to increase from 1900 to 1980 (Forcella and Harvey, 1983). Long-term monitoring of the impacts of both controlled and uncontrolled stands of alien plants is important to better understand conflicting impacts over time.

It is well established that weeds have a number of very different effects on associated plants. These include physical displacement directly or indirectly, depriving associated plants of water and nutrients, allelopathy, and acting as primary or alternate hosts to pests or diseases. All of these impacts can be found in the activity of alien species on native ecological processes. However, aliens which invade undisturbed systems can have additional effects, which can cause long-term changes in ecosystems, which are not commonly attributed to weed activity. These include: a change in community structure; a change in fire disturbance patterns; a change in nutrient status; change in the soil-water regimes and hydrological processes;

changes in diversity and creation of mutually beneficial interactions between exotics (Smith, 1985). Each of these will be discussed separately.

Change in Community Structure

The replacement of a relatively diverse native forest community by monotypic stands of aliens is a serious disruption of the original ecosystem. This has occurred in large tracts of Hawaii (Smith, 1985), Mauritius (Lorence and Sussman, 1986) and Madagascar (Hooper, personal observation), where *Psidium cattleianum*, infests hundreds of hectares of mesic and well-drained rain forest. In Hawaii large areas of the fairly open phi'a and koa forests have dense understories of strawberry guava. Native species regenerate rarely under guava and few are able to survive. The prognosis for native ecosystem re-establishment in guava thickets is poor for several reasons. First the shade is so thick that no seeds germinate, or if they do the seedlings die for lack of sufficient light. In old strawberry guava stands there is no ground cover. Second the fruit is relished by feral pigs, which move into these thickets during the fruiting season. Lastly, there is growing evidence of allelopathic activity in the fallen leaves, although this has not been substantiated (Smith, 1985).

Succession is very unlikely to occur under single-species stands of alien plants, if the light levels are very low, and alien regeneration is already occurring. There are few native species which are able to maintain themselves in heavy infestations of arborescent weeds. Seed production is often severely diminished, weakened or absent. The outcome is that the seedbank of the native species is exhausted and that species is effectively excluded from the area. Re-invasion is the only means of reestablishment. Succession, if it occurs at all tends to be dominated by alien species (Smith, 1985).

Community structure can also be altered if a life form unrepresented in the native flora is introduced. A massive altering of community structure occurred when the floating aquatic weed *Salvinia molesta* was introduced to tropical river systems. Within ten years it had measurably altered productivity in fisheries of flood plain lakes of New Guinea (Mitchell et al., 1980).

Changes in Fire Disturbance Patterns

Changes in the fire characteristics have been documented with the arrival of aggressive invaders. Changes occur in biomass, size and distribution of plant parts, and plant moisture and energy contents. Invasion of the habitat by introduced plants affects fire hazard by changing the structure of the fuel bed. This occurred when invasions of *Hakea servicea* resulted in a 60% increase in fuel loads and reduced moisture contents of the live foliage from 155 to 110%. In South Africa they have reported increases of fire intensities and impact in grassland biomes but a decrease in frequency with invasions of *Hakea servicea* (Van Wilgen and Richardson, 1985).

In Hawaii, Mueller-Dombois (1981) found that carbon-dated charcoal deposits and other evidence suggested very low fire frequency. Human activity and the introduction of weeds, particularly grasses, has changed that. Since 1910, most fires in the national parks have been started by man, but until the early 1970's were small in area because of the natural firebreaks in the vegetation mosaic. In the 1970's, the bush beardgrass and broomsedge (*Andropogon*) invaded the area and a program to rid the park of feral goats began. After most of the goats were removed the grasses were no longer grazed and consequently the two grasses colonized the many ash-filled cracks in lava flows creating a continuous ground cover between islands of native shrub forest. The flowering stalks are persistent, creating excellent fuel and the grasses are fire stimulated, rapidly resprouting from basal sprouts after burning. The total area burned by fires in lowland ecosystems is now 100 times larger than before the invasion of these grasses (Smith, 1985).

Changes in Soil-Water Regimes

Many weeds introduced into oceanic islands came from temperate areas of Europe and America. Their phenology is not always in synchrony with local climatic conditions, and can result in significant changes in the edaphic ecology of the area. This vegetation can also affect other aspects of the hydrological cycle such as rainfall interception, infiltration, erosion, and water yield as stream flow.

Broomsedge, established on heavily disturbed areas of the Hawaiian islands about fifty years ago, has retained the phenological pattern of its native area in the southeastern U.S. (Mueller-Dombois, 1973). The dormant period for broomsedge coincides with the wettest months on the islands, so that water is retained in the soil instead of being depleted through transpiration. Surface run-off results in increased rates of mass failures with occasional slumping on steeper slopes. Under native vegetation, water is transpired rapidly and most of the precipitation percolates through the soil or is held as recharge.

There are no actual data which show that invasives reduce infiltration rates, although water repellancy has been noted under some Acacias. Water repellency may also develop in pine litter accumulating in the soil after fire, due to secondary compounds. Accelerated erosion has been noted to occur in river banks invaded by *Acacia mearnsii* and other Australian acacias. These trees are shallow rooted and are prone to collapse easily from bank erosion by the force of floods (Versfeld and van Wilgen, 1986). It is problematic to make conclusions is this area because of the difficulty in separating the effects of species habits from the confounding influences of grazing impacts.

In general, the best-documented illustration of the alteration of ecosystem properties by invaders involved effects on the hydrologic cycles. A reduction in stream flow following invasions has been well established throughout the world. Some increases have also been noted when trees are replaced by herbs.

Change in the Nutrient Status

Changes in the nutrient status of the soil due to invasions has been credited to increases in both litter and live biomass, and the invasion by nitrogen-fixing plants. In South Africa where grassland was invaded by *Pinus patula*, the soil levels of most nutrient elements declined in the first 30 centimeters. There were some increases in litter and the nutrients contained in litter. Second rotation sites showed an 18% decrease in soil organic carbon compared to grassland, and a 40% decrease (measured by CO_2 evolved) in soil respiration. These differences were explained by more nutrients being contained in above ground biomass (Versfeld and van Wilgen, 1986).

Many tropical islands are volcanically active and do not have mature soils. They have either lava or ash substrate, characterized by low levels of nitrates. Of the 86 plants which are regarded as serious pests on Hawaii, 11 are nitrogen fixers (Smith, 1985). These plants can both enrich and acidify the soil as their litter decomposers. The outcome is often an enriched soil in which other plants may grow, potentially replacing the original occupants of the area.

This has happened in both Australia and Hawaii. In Hawaii the exotic faya tree (*Myrica faya*) has invaded extremely nitrogen deficient sites on young flows, which contain no native symbiotic nitrogen fixers. It has been shown to produce 18 kg/ha/yr of nitrogen, compared to only .2 kg/ha/yr from all native sources (Vitousek and Walker, 1989). The dramatic increase in fixed nitrogen has been attributed to tripling the diameter growth of native trees. Since nitrogen is now available, land managers in Hawaii now fear that a subsequent invasion by *Psidium cattleianum* will occur (Smith, 1985).

In Australia, a similar relationship between low soil fertility and the success of exotic species has been observed. *Pinus radiata* has been spreading from plantations into native eucalypt forests, which showed minimal disturbance from rabbits. Comparative measurements from 1974-1981 showed that the growth rate of the pines was 10 times that of the native trees. Most of the densest invasions occurred at dry sites with shallow soils. This is attributed partly to the pines success in the acquisition of underground resources, and to the pines intolerance to shade found on more mesic sites (Chivers and Burdon, 1983).

In South Africa the success of the Australian acacias is attributed to its nitrogen-fixing ability and the fact they form mycorrhizal associations which may aid in phosphorus uptake (Versfeld and van Wilgen, 1986). Milton and Siegfried (1981) suggest that the *Acacia* may make the environment less suitable for fynbos species, which are adapted to nutrient poor soils, by enriching the soils. The fynbos biome is characterized by the presence of shrubs of the Proteaceae family in the upper story and ericoid shrubs in the understory (Macdonald and Richardson, 1986).

Community-Level Feedbacks

Many island ecosystems are faced with serious threats to their integrity due to an increase in the scale of alteration that occurs with successive invasions. This can occur in the form of mutually beneficial interactions between exotic organisms or the disruption of ongoing ecological processes, especially those involving biotic interactions.

An example of the positive interactions between plants and animals is the exotic *Agropyron desertorim* which produces three to five times more tillers following clipping or defoliation than the native *Agropyron*. Grazing pressure from cattle give the exotic a competitive advantage, if other variables are held constant (Mack, 1989). In Hawaii, Smith (1985) shows that there is a strong positive relationship between feral pigs and a number of important weedy plants, including *Psidium cattleianum*. The pigs consume and disseminate the seeds of the invaders, and they also disturb areas of soil and create excellent conditions for seed germination and seedling growth. Both soil characteristics and a community structure distinctly different from those in native forests are maintained by this interaction.

Ecological processes which are dependent on the interaction between organisms, for example pollination, herbivory, competition and predation are especially prone to disruption by invasion (Vitousek, 1986). Because the degree of specificity alters with each case, the effects of invasives are difficult to predict. South Africa where 16% of the fynbos plants are dispersed by ants (Breytenback, 1986), the change in fire frequency caused by *Hakea sericea* has negatively affected native plant dispersal. The effect of intense hot *Hakea* fires decreases considerably the survival of termite and ant communities.

Diversity

The final topic to consider is the impact of alien plant invaders on biological diversity. As evidenced from the previous discussion the most commonly recorded phenomenon with alien plant invasions is a rise or decline in species diversity. This information is largely based on taxonomic criteria and is expressed as a community statistic such as richness.

Almost without exception tropical oceanic islands have experienced a dramatic increase in species richness, but decline in native species abundance due to exotic species. This is simply due to an often spectacular increase in the number of species present. Hawaii serves as a good case in point. According to Smith (1985) there have been 4600 introduced vascular plants in Hawaii, while only 200 endemic plants species are believed to be extinct, and 800 endangered.

Obviously the formation of a single species stand of a light excluding tree will negatively impact species richness in that specific location. However, most invasions are not this severe, and at a 'forest' level the effect of invasions on diversity are more ambiguous. If alien plant communities are taller, and form denser canopies than the plants from indigenous communities, native light-demanding plant species could be lost locally (Versfeld and van Wilgen, 1986). Most species which disappear do so after canopy closure. Diversity has also decreased in sites in the southern Cape invaded by *Hakea sericea*. Communities with light to moderate *Hakea* invasion supported fewer species than the vigorous non-invaded biomes. 4.5 species per $4m^2$ were recorded in the invaded areas as compared to 8.2 species per 4 m^2 in the non-invaded (Breytenback, 1986).

Much of the work quantifying bio diversity remains to be done. It is necessary to look at diversity on a species level and a community level. An intact assemblage of native island flora may often be less diverse than an assemblage of alien and indigenous species, provided the mixtures are sufficiently compatible to maintain themselves.

Conclusions

Invasive plants are of concern to land managers and researchers world-wide because they have the ability to change the functional environment for all of the organisms in an ecosystem. The largest impact on a plant community will occur if the alien invader has a completely different and dominating structure or life form. In severe cases this has resulted in homogeneous, single species stands which exclude the regeneration of all other species. A common impact in association with grazing is a change in the fire disturbance pattern, which usually creates conditions unfavorable for the growth of the indigenous flora, while promoting fire-adapted exotics. Changes in the nutrient status of the soil due to invasions has been credited to increases in both litter and live biomass and the invasion by nitrogen-fixing plants. The impacts of invasive aliens can be increased when mutually beneficial interactions between exotic organisms occur. Whereas introductions of exotics enriches species diversity temporarily, knowledge of how to keep invaders at endemic, non-threatening levels remains weak.

METHODS OF CONTROL AND THEIR COSTS AND BENEFITS

The control of undesirable plant species has been a problem for as long as humans have been agriculturists. The methods and sophistication of control have changed a great deal through the ages, but the basic aim has always remained the same: to limit the number of plant species competing with crops or occupying work space to a level compatible with human-use activities. Modern control plans for invasive plants go under a variety of names. Groves (1989) defines ecological control as the: "planned use of one or several methods of control when integrating with an understanding of the dynamics of the ecosystem in which the plant occurs". Kluge et al. (1986) defines integrated weed control as the "integration of any number of control options in any way, to achieve the most favorable outcome in terms of predictable economic, sociological and ecological consequences." Ecological control methods in natural systems differ from agricultural systems in that they aim to maintain the native community stability and diversity through interactions of native species habits, climate, soil and disturbance regimes.

The basic outline of an integrated weed control (IWC) program is shown in Figure 1 (Kluge et al., p. 295, 1986). IWC programs have three main components and four basic procedures. The components include knowledge of the control

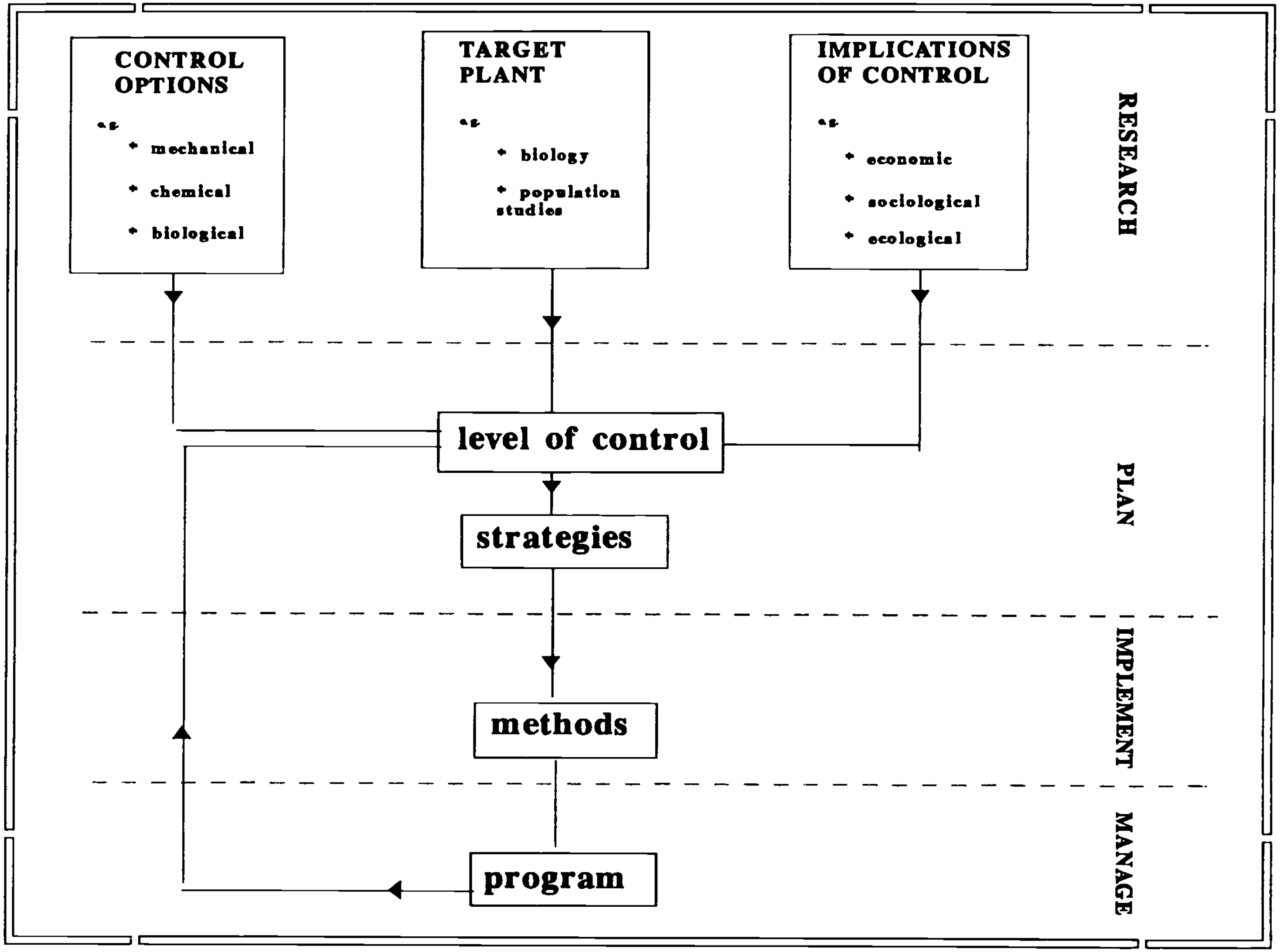

Figure 1: The components and procedures of an integrated weed control program (Allan and Bath, 1980

options, target plants, implications of control. The procedures are research, planning, implementation and management. The shortcomings seen in many existing control programs are that they do not take into consideration all three components, and do not remain committed to the four procedures through time.

Before embarking on a control program the objectives must be clearly understood. The objectives of the plan will determine the way in which the problem is perceived, the required level of control to be used, and therefore the whole approach to control. Deciding the level of control can sometimes be difficult when there is a conflict of interest regarding the user status of the plant. Such is the case with *Prosopis* which is an aggressive invader in South Africa, but also a source of fuel and fodder for animals (Kluge et al., 1986). Because of the difficulty in determining an acceptable level of control of invasives, most programs have poorly defined objectives.

It is extremely difficult to predict the effects of an invasion and only slightly easier the impacts of the control operation. This is because of the inherent complexity and variability of biological systems. However, to implement a control program, some understanding of the nature of the invasion in terms of its rate of spread should be realized. One theory about the rate of spread of an invading species is that the square root of the area occupied by the species increases linearly with time (Skellam, 1951). Elton (1958) examines this theory using the spread of the muskrat in central Europe and found it to be quite robust. This appears to hold true for both plant and animal populations. However, as Atkinson (1985) points out, the relationship between native species and introduced predators are rarely straightforward. He cites the example of the unsuccessful introduction of commensal rats to several tropical islands. Apparently these islands already had large land crabs which were able to burrow and climb trees, and the native avifauna had adapted to the presence of a terrestrial predator.

Once the required level of control is decided, all available information should be collected on the ecological characteristics of the target plant. Ideally an IWC program will attempt to exploit as many vulnerable aspects of the biology of the target plant as possible. This should include an exploration of the positive feedback links which encourage the exotic plant. For the control of *Hakea sericea* in South Africa three aspects of its biology are used in the control plan. These are its complete dependency on seed for reproduction, the absence of a soil stored seed bank, and the vulnerability of the thin-coated seeds to destruction by fire (Kluge et al., 1986). In Australia, a control program has been implemented for *Chrysanthemoides monilifera*, an invasive species imported from South Africa. This species has long-lived dormant seeds which may be stored in the soil and can regenerate after fire. However an appropriate fire regime can control the plant if a prescriptive fire of low intensity follows a wildfire of higher intensity. The second fire should be tried before the commencement of flowering of the seedlings induced to germinate by the first fire (Lane and Shaw, 1978).

An understanding of the target should exist at both the individual and population level. Population parameters which may be addressed in a control program include: the distribution patterns of the plants, the population growth rate, the reproductive strategy, and the size structure of the population (Groves, 1989).

The success of many invasive plants has resulted because of system level disturbances, which usually arrived with the humans. Among others the disturbances may include a change in the fire regime, and/or the introduction of exotic vertebrates, largely livestock. Controlling, or at least understanding the relationship between the disturbances and the success of the target plant will be critical to a successful control program (Groves, 1989). On many islands this can involve eradication programs for feral goats and feral pigs and other potentially injurious exotic vertebrates. To remove the controls on native species, ceasing the practice of maintaining sustained yields of feral animals will reduce the level of chronic disturbance (Smith, 1985; Coblentz, 1978).

Feely (1980) discusses the impact that a return to wilderness conditions would have in South Africa. He argues that an open tree canopy and a luxuriant grass layer is a direct result of the frequent fire regime perpetuated by Iron age man. A realization of the original community structure, a closed or nearly closed tree canopy, will significantly alter the flora and fauna of today's woodland. Avoidance of fire will not only limit the success of the exotics, but will also significantly alter the native's status. However, a full array of native species will be required for some time before the community approximates its original condition without exotics.

Impacts of the Control Program

The impacts of a control program must be considered in the context of the economic, sociological and ecological consequences. While the economic and ecological consequences are often considered (sometimes retrospectively), the sociological implication of control programs are often ignored. The discussion of impacts needs to consider the four following areas a) availability of suitable alternatives among the indigenous or non-invasive alien flora, b) whether manipulation rather then eradication of the alien invader would result in a more successful and less expensive control, c) how alternative land-use practices will maintain the alien in an endemic state, d) whether the land-use changes will influence the crucial economic activities of the people.

When considering economics, many invasives, including those in undisturbed natural systems, have a variety of uses for which they would be suitable (Geldenhuys, 1986). These uses need to be considered in light of a cost-benefit analysis. South African invasive *Acacia melanoxylon* is an example of a tree which has both useful and detrimental effects on native forests, but is of commercial value. Often it is not economically feasible to control exotics on large areas of low productivity ground, especially when the resultant successional processes are unknown, and the economic rewards are small.

Any attempted control should be viewed in the context of the environment, with as complete an understanding of the underlying ecological processes as possible, to avoid negative consequences. For example, the control of *Hakea sericea* is achieved through the application of intense prescribed fires. This has effectively controlled *Hakea*, but has also led to reduced regeneration of indigenous plants

(Versfeld and van Wilgen, 1986). The chemical control of *Lantana* has also had negative consequences for the indigenous flora. When this plant dies it makes way for the even more invasive trifled weed, *Chromolaena odorata.* Control methods should attempt to prevent this unwanted succession. One option would be slow-acting, selective herbicides which allow increased light penetration and encourage the establishment of grasses and later native tree species (Kluge et al. 1986).

The sociological conditions on many tropical islands are far from desirable. Many inhabitants are forced to eke out a meager existence on an already overcrowded land base with depleted resources. The impoverished rural population often finds itself in the paradoxical situation where they cannot afford to save the resources, yet by exploiting their local environment they deplete its capacity to produce, and endanger their local economies. For these reasons the sociological effects of the control of invasive species must be considered very carefully. If the target plant is economically important the control measures will probably never succeed, unless there is a substitution by a more desirable species. In addition certain types of control tools may be culturally inappropriate. For example, without professional oversight, people with little education might not understand proper operating procedures and precautions which need to be implemented when applying certain phytotoxic chemicals, especially near sensitive desirable plants.

Control Options

Five different control approaches will be discussed in this section, along with combinations. These include legislative control, mechanical control, the manipulation of the fire regime, chemical control and biological control. Often two or more strategies are used simultaneously to optimize the effectiveness of control.

Because of the isolated nature of oceanic islands, legislative control is ideally suited for the prevention of new introductions through quarantine inspection. The prevention of further introduction of alien species is imperative if native habitats are to be effectively managed. Smith (1985) suggests imposing a ban on the importation of certain plant groups to Hawaii. He chooses a ban on the melastomes, the genus *Rubus*, members of the Myrtaceae family, as well as species which are known to be problems in other parts of the tropics. This ban would necessarily need to extend to all living plants and propagules.

It is unknown how successful legislative control is in preventing the number of invasives entering a country. This method does not usually keep out "new" species of known taxonomic identity unless they are already regarded as troublesome in other parts of the world. To improve this control option education is essential, and cooperation from the general public should be solicited. The education of tourists and other visitors could be done on the plane prior to arrival. A five minute informational video about the impacts of exotic plants on native plant communities would be beneficial, followed by careful inspection of carry-out luggage. Tourism poses danger for nature reserves, a positive correlation between visitation rate and the number of introduced species has been documented by Usher (1988). When tourists, aircraft or ships arrive from other tropical areas they should receive specially intensive attention.

There are a variety of types of mechanical control methods, but few are used on a large scale in natural areas, and few are effective. The cultivation of land is seldom appropriate in natural areas but has been used for years in agriculture. Hand-pulling of seedling, invasive plants can be effective, especially when used in conjunction with other methods, such as fire or herbicides. For this to work seedlings must be visible before growing to large to pull. However, most mechanical control methods are expensive if they require manpower (Kluge et al., 1986). If a great deal of disturbance under the plant is generated it can stimulate weed seeds to generate. In some instances, hand-pulling and slashing has been completely ineffective such as with the control of *Rhododendron* on British nature reserves (Usher, 1986).

The use of fire represents an inexpensive control method which can conserve natural fire adapted plant communities under limited circumstances, and temporarily control the growth of undesirable species (Christensen and Burrows,1986). Manipulation of fire frequencies, season and intensity are the three main applications. Fires which occur frequently will often favor plant invasion. Frequent fires favor re-sprouting perennials over non-sprouting species and disadvantage plants which rely solely on seed stored in the soil (Vogl, 1977). Many plants on tropical islands are poorly adapted to fire, so any increase in the fire frequency will negatively affect the native flora. In general, legumes species with wind dispersed seeds and bulb and corm reproducing plants will be at a disadvantage on infrequently burnt sites. The effectiveness of using fire depends on the interaction between time since the last burn, the life span of the plant and its propagules and the successional status of the invaded community (Mueller-Dombois, 1981).

Chemical methods have been tried with varying degrees of success in nature reserves, and can be used effectively in limiting newly discovered infestations. Control of invasive species by herbicides is generally short term and directed at individual target plants. This may or may not be effective. MacDonald (1983) writes that *Psidium guajava* has proven almost impossible to kill and all of the herbicides tested so far have failed. Chemical control of already widespread invasive plants is often expensive and rarely effective in the long term, unless integrated with other methods of control (Kluge et al., 1986).

Herbicides can be effective and economical for controlling small outbreaks. When the use of herbicides is optimized to coincide with critical points in the life cycle of the plant, they can be most effective. Use of herbicides has the additional advantage that the soil is left undisturbed, and often the dead plant tissues will form a ground cover that will impede the growth of seedlings. Moreover, a gradual program of spot treatment will favor natives at the expense of the exotics, and permit expanding areas of pest-free plant communities, or nearly so. For sprouting species, systemic herbicides have the special advantage of killing roots.

Much of the literature on conservation of indigenous vegetation on island nature reserves suggests biological control is the preferred alternative, or at least a logical part of any large program (Kluge et al.1986; Neser and Kluge, 1986; Moran et al; 1986; Groves, 1989). This is primarily because the scope of the problem is broad, and a long-term solution is needed. This is not to suggest that biological control is a panacea, when effective it is an ideal method, but it is not always effective, and rarely is it predictable.

One possible drawback to biological control can be its cost, especially when it is implemented in countries such as the United States that have restrictive regulations which force up operating costs. Dr. George Markim, a pest specialist from Hawaii, estimated that the cost of developing appropriate controls and testing them is greater than $2 million per pest per environment. In other countries, with lower operating costs such as South Africa, the Institute of Forest Protection recommends biological control as the most cost-efficient method to control invasive plants, when integrated with other methods (Kluge et al., 1986; Zimmermann et al., 1986).

The earliest cases of biological control of a weed occurred accidentally in India in 1795. Since then in excess of 100 species of weeds have been targets of biological control worldwide. The vast majority (98%) of the control attempts have achieved some level of success (Julien, 1982). Control has been achieved with negligible risks, no known cases of a reduction in the population of a native congener of the invader has occurred (Groves, 1989).

The idea behind biological control is that the alien plants have been removed from their natural enemies when they were transported to islands, and thus became invasive. When released from herbivores or seed predation pressure they become superior competitors. This attribute is unique for introduced plants and is why biological control is seldom used or appropriate on indigenous flora (Moran et al.

1986).

Alien perennial and woody plants that are unrelated taxonomically to native beneficial plants require simpler screening procedures. It is unlikely that the most highly specialized insects which feed on them will attack natives. More care must be taken for biological control of plants which are similar to the natives, but it is still possible (Turner, 1986). Turner (1986) gives the example of the control of *Acacia longifolia* in South Africa by the gall wasp *Trichilogaster acaciaefoliae*. The wasp was extremely effective in the control of the exotic *Acacia* but did not affect the native acacias. He concludes that the genetic closeness of the target plant to indigenous or useful plants dictates the degree of precautions, but should not be used as an argument to preclude biological control.

When using biological control it is important to understand the aggressive attributes of the plant, such as prolific seeding or a large tap root, and use insect herbivores that are aimed at countering the specific aggressive and noxious qualities of the plant. Many of the plants are so robust that they require two or more predator for control. Examples of successful bio-control measures include the control of St. John's wort, *Hypericum peroratum* in south-western Africa and in California by leaf feeding beetles, *Chrysolina quadrigeminal* and a gall midge, *Zeuxidiplosis giardi*; the control of *Eupatorium adenophorum* in Hawaii with the tephritid gall fly, *Procecedochores utilis*; the control of the jointed cactus, *Opuntia aurontiaca* by the insect herbivore *Dactylopius austrinus*. The control is so effective in places that the host plant and the insect are almost completely eliminated (Groves, 1989).

The South Africans have had a long history of successful biological control. They have experienced plant invasions that have affected large tracts of game parks where the unit costs must be kept low. The majority of their work has been with insect herbivores and insect, seed attacking agents. They recommend releasing several species of herbivorous natural enemies for a single problem weed. This allows for a more consistent, widespread and stable attack. Their research suggests that specialist monophagus insects that are closely adapted to the host plants are usually the most effective weed control agents since they are freed on their natural food supplies. (Moran et al. 1986; Neser and Kluge, 1986).

In Hawaii the use of biological control is increasingly rare because of legislative restrictions imposed on the control method (Kepler, 1985). A list of the reasons for not allowing control include:

1) The target agent may be parasitized by insects or fungi already present.

2) The target plant is related to an important agricultural crop.

3) The ecological requirements of the agent may be found only in part of the target source's range.

4) Many pests are not controlled by insects, but rather by succession in its native habitat.

Ecological Restoration

In many natural settings the elimination of an undesirable alien is not enough to ensure that a return to the indigenous flora is achieved. A process of ecological restoration may be needed, which would involve an evaluation of its present and past history. Most of the ecological restoration literature involves growing something (almost anything) on artificial, highly modified landscapes; such as strip mines, waste dumps, or areas reclaimed by the sea. Very little has been written about restoring landscapes, which are unmodified by man, or at least not subject to any major change in land-use. Two important exceptions include Thorhoug and Austin, (1976) and Kio (1981).

Because of the enormous complexity and diversity of tropical ecosystems very little descriptive work has been done on past conditions, so as to permit a full reconstruction of ecosystems. As a practical matter, restoration efforts are likely to be limited to changing the proportion of certain higher plants and vertebrate animals in the community, and restoring as many as possible of the native species.. When planning for ecosystem restoration it is necessary to choose some date in the past to indicate the point at which the system is to be restored to. Since this has often never been adequately quantified, historical records and native vegetation specialists need to be consulted for their "best approximation".

Techniques of ecological restoration on tropical islands would involve one or all of the following list (Lamoureux, 1985):

1) Removal of non-native herbivores and predators and prevention of their re-introduction.

2) Removal or drastic control of all dominant and escaped alien plants, at least those causing significant environmental disturbance.

3) Re-planting and promoting native plants in some ratio of their original abundance, as far as is known.

4) Restocking with native animals where species are absent, after their habitat is largely restored.

5) Preventing or at least slowing, detrimental changes in the plant community before, during and after the restoration process.

Conclusions

In conclusion, the technology to control the spread of invasive plants exists, but often the financial costs are too great. For this reason, very few control efforts have been initiated, even in developed countries, such as the USA. However, the theory, supported by many case studies is in place and can be drawn upon when needed.

To effectively achieve integrated weed control the control options must be known. In addition, the ecology of the target plant in the new environment must be clarified and all of the implications of the control measures must be studied. This should include both ecological and sociological implications. IWC programs must be monitored through time, the initial phase is research, then planning, the implementation, then management. Before beginning an IWC program the objectives must be clearly understood. Traditional control options include legislative control, mechanical control, chemical control and biological control. Constraining any one of the control options will greatly reduce the chances of success. Often two or more strategies are used simultaneously to optimize the effectiveness of control. Once the desired level of control has been achieved the final step should be to restore the environment to a condition as close as possible to its pre-invasion condition to prevent the conditions that make it invadable.

BIBLIOGRAPHY

Abbott, I. 1974. Numbers of plant, insect and land bird species on nineteen remote islands in the Southern Hemisphere. Biological Journal of the Linnean society, 6:143-152

Atkinson, I.A.E. 1985. The spread of commensal species of *Rattus* to oceanic islands and their effects on island avifaunas. In: Moors, P.J. (ed.), Conservation of Islands Birds, pp. 35-81. In H. Council for bird Preservation, Tech. Publication No. 3.

Baijnath, H., Ramcharun, S. and S. Naiker. 1982. *Psidium* (Myrtaceae); very successful weeds. South African journal of Botany. 1:3, 78.

Baker, H.G. 1965. Characteristics and modes of origin of weeds. In: Baker, H.G., Stebbins, G.L. (eds.), Genetics of colonizing species. Academic press. New York, pp. 147-172.

Baker, H.G. 1986. Patterns of plant invasion in North America. In: Mooney, H.A. and Drake, J.A. (eds). Ecology of biological invasions in North America and Hawaii, pp. 44-57. Ecological studies no. 53, Springer-Verlag, New York.

Bazzaz, F.A. 1986. Life history of colonizing plants. In: Mooney, H.A. and Drake, J.A. (eds). Ecology of biological invasions in North America and Hawaii, pp.96-108. Ecological studies no. 53, Springer-Verlag, New York.

Breytenbach, G.J. 1986. The impacts of alien organisms of terrestrial communities with emphasis on communities of the south-west Cape. In: Macdonald, I.A.W., Kruger, F.J. and Ferrar, A.A. (eds.) The ecology and management of biological invasions in southern Africa, pp.229-238. Oxford University Press. Cape Town.

Bramwell, D. 1979. Plants and islands. Academic press. USA. 459 pp.

Brockie, R.E., Loope, L.L., Usher, M.B. and O. Hamann. 1988. Biological invasions of island reserves. Biological conservation, 44:9-36.

Carlquist, S. 1974. Island biology. Columbia University Press. New York. 660pp.

Carson, H.L. 1981. Microevolution in insular ecosystems. In: Mueller-Dombois, D., Bridges, K.W., and H.L. Carson (eds.), Island Ecosystems: Biological organization in selected hawaiian communities. pp. 471-482. Hutchinson-Ross. USA.

Christenson, P.E. and N.D. Burrows. 1986. Fire: an old tool with a new use. In: Groves, R.H. and J.J. Burdon (eds.), Ecology of biological invasions: an Australian perspective, pp. 97-105. Australian academy of science, Canberra.

Chilvers, G.A. and J.J. Burdon. 1983. Further studies on a native Australian eucalypt forest invaded by exotic pines. Oecologia, 59:239-245.

Coblentz, B.E. 1978. The effects of feral goats (*Capra hircus*) on island ecosystems. Biological conservation, 13:279-286.

Crosby, A.W. 1986. Ecological imperialism: the biological expansion of Europe, 900-1900 A.D. Cambridge University Press. 368 pp.

Darwin, C.R. 1859. On the origin of species by means of natural selection, or the preservation of favored races in the struggle for life. John Murray, London. 470 pp.

Deacon, J. 1986. Human settlement in South Africa and archaeological evidence for alien plants and animals In: Macdonald, I.A.W., Kruger, F.J. and Ferrar, A.A. (eds.) The ecology and management of biological invasions in southern Africa, pp.3-20. Oxford University Press. Cape Town.

Dean, S.J., Holmes, P.M. and P.W. Weiss. 1986. Seed biology of invasive alien plants in South Africa and Namibia. In: Macdonald, I.A.W., Kruger, F.J. and A.A. Ferrar (eds.), The ecology and management of biological invasions in southern Africa. pp. 157-170. Oxford University Press, Cape Town.

di Castri, F. 1989. History of biological invasions with special emphasis on the old world. In: Drake et al. (eds) Biological invasions: a global perspective, pp.437-461. John Wiley and Sons, New York.

Elton, C.S. 1958. The ecology of invasions by plants and animals. Methuew and Co. London. 181 pp.

Feely, J.M. 1980. Did iron-age man have a role in the history of Zululand's wilderness landscapes? South African Journal of Science. 76:150-153.

Forcella, F. and S.J. Harvey. 1983. Relative abundance in an alien weed flora. Oecologia. 59:292-295.

Fosberg, F.E. 1948. Derivation of the flora of the Hawaiian islands. In: Zimmerman, E.C. (ed.), Insects of Hawaii, Vol. 1. Introduction, pp. 107-119. University of Hawaii press, Honolulu, Hawaii.

Futuyma, D. 1986. Evolutionary Biology. Second edition. Sinauer associates, inc. Massachusetts. 600 pp.

Geldenhuys, C.J. 1986. Costs and benefits of the Australian Blackwood *Acacia melanoxylon* in South African forestry. In: Macdonald, I.A.W., Kruger, F.J. and A.A. Ferrar (eds.), The ecology and management of biological invasions in southern Africa. pp. 275-283. Oxford University Press, Cape Town.

Gerrish, G. and D. Mueller-Dombois. 1980. Behavior of native and non-native plants in two tropical rainforests on Oahu, Hawaiian islands. Phytocoenologia. 8:237-295.

Gill, A.M. and S. Neser. 1984. *Acacia cyclops* and *Hakea sericiea* at home and abroad. In Dell, B. (ed.) Proceedings of the 4th International Conference on Mediterranean ecosystems. University of Western Australia, Nedlands. pp.57-58.

Groves, R.H. 1989. Ecological control of invasive terrestrial plants. In: Drake et al. (eds) Biological invasions: a global perspective, pp.437-461. John Wiley and Sons, New York.

Harper, J.L. 1977. Population biology of plants. Academic press. London.

Holm et al. 1977. The world's worst weeds: distribution and biology. University Press of Hawaii, Honolulu. 609 pp.

Huenneke, L.F. and P.M. Vitousek. 1990. Seedling and clonal recruitment of the invasive tree *Psidium cattleianum*: implications for management of native Hawaiian forests. Biological conservation, 53:199-211.

Janzen, D.H. 1969. Seed eaters versus seed size, number, toxicity and dispersal. Evolution. 23:1-27.

Janzen, D.H. 1970. Herbivores and the number of tree species in tropical forests. American Naturalist. 104:501-528.

Jennings, J.D. (ed.) 1979. The prehistory of Polynesia. Harvard University Press. Cambridge, Mass. 399 pp.

Julien, M.H. et al. 1984. Biological control of weeds: an evaluation. Protection ecology. 7:3-25.

Lamourex, C.H. 1985. Restoration of native ecosystems. In: Stone, C.P. and J.M. Scott (eds.). Hawaii's terrestrial ecosystems: Preservation and management, pp. 483-492. Cooperative park studies unit. University of Hawaii. Hawaii.

Lane, D.W. and K.Shaw. 1978. The role of fire in boneseed (*Chysanthesmoides monilifers* (L.)) control in bushland. Proceedings of the 1st conference of the council of Australian weed science society, pp.333-335. Melbourne, Victoria.

Loope, L.L. and D. Mueller-Dombois. 1989. Characteristics of invaded islands with special reference to Hawaii. In: Drake et al. (eds) Biological invasions: a global perspective, pp.257-274. John Wiley and Sons, New York.

Lorence, D.H. and R.W. Sussman. 1986. Exotic species invasion into Mauritius wet forest remnants. Journal of Tropical ecology. 2:147-162.

Kepler, C. 1985. Current and future roles of agencies, conservation groups, legislature, and the public in preserving and managing Hawaiian ecosystems: A summary. In: Stone, C.P. and J.M. Scott (eds.). Hawaii's terrestrial ecosystems: Preservation and management, pp. 483-492. Cooperative park studies unit. University of Hawaii. Hawaii.

Kirch, P.V. 1982. The impact of the prehistoric Polynesians on the Hawaiian ecosystem. Pacific Science. 36(1) 1-14.

Kio, P.R.O. 1981. Regeneration methods for tropical high-forests. Environmental conservation. 8(2): 139-147.

Kluge, R.L., Zimmerman, H.G., Cilliers, C.J. and G.B. Harding. 1986. Integrated control for invasive alien weeds. In: Macdonald, I.A.W., Kruger, F.J. and A.A. Ferrar (eds.), The ecology and management of biological invasions in southern Africa, pp. 295-305. Oxford University press. Cape Town.

Knight, R.S. 1986. A comparative analysis of flesh fruit displays in alien and indigenous plants. In: Macdonald, I.A.W., Kruger, F.J. and A.A. Ferrar (eds.), The ecology and management of biological invasions in southern Africa, pp. 171-179. Oxford University press. Cape Town.

Kruger, F.J., Richardson, D.M. and van Wilgen. 1986. Processes of invasion by alien plants, In: Macdonald, I.A.W., Kruger, F.J. and A.A. Ferrar (eds.), The ecology and management of biological invasions in southern Africa, pp. 145-155. Oxford University Press, Cape Town.

MacArthur, R.H. 1972. Geographical ecology: Patterns in the distribution of species. Harper and Row, New York. 269 pp.

MacArthur, R.H. and E.O. Wilson. 1967. The theory of island biogeography. Princeton, Princeton University Press. 203 pp.

Macdonald, I.A.W. 1983. Alien trees, shrubs and creepers invading indigenous vegetation in the Hluhluwe-Umfobzi game reserve complex in Natal. Bothalia, 14 (3 and 4):949-959.

Macdonald, I.A.W. 1984. Is the fynbos biome especially susceptible to invasion by alien plants? A re-analysis of available data. South African Journal of Science. 80:369-376.

Macdonald, I.A.W. and D.M. Richardson. 1986. Alien species in terrestrial ecosystems of the fynbos biome. In: Macdonald, I.A.W., Kruger, F.J. and A.A. Ferrar (eds.), The ecology and management of biological invasions in southern Africa, pp. 261-268. Oxford University Press, Cape Town.

Mack, R.N. 1989. Temperate grasslands vulnerable to plant invasions: characteristics and consequences. In: Drake et al. (eds) Biological invasions: a global perspective, pp. 155-180. John Wiley and Sons, New York.

Milton, S.J. and W.R. Siegfried. 1981. Above-ground biomass of Australian *Acacias* in the southern Cape, South Africa. Journal of South African Botony, 47:701-716.

Mitchell, D.S., Petr, T., Viner A.B. 1980. The water-fern *Salvinia molesta* in the Sepik river, Papua New Guinea. Environmental Conservation. 7:115-122.

Moran, V.C., Neser, S. and J.H. Hoffmann. 1986. The potential of insect herbivores for the biological control of invasive plants in South Africa. In: Macdonald, I.A.W., Kruger, F.J. and A.A. Ferrar (eds.), The ecology and management of biological invasions in southern Africa, pp. 261-268. Oxford University Press, Cape Town.

Morris, M.J. 1982. Gummosis and die-back of *Hakea sericea* in South Africa. Proceedings of the 4th national weed conference of South Africa, Balkema, Cape Town. pp. 51-54.

Moulton, M.P. and S.L. Pimm. 1986. Species introductions to Hawaii. In: Mooney, H.A. and Drake, J.A. (eds). Ecology of biological invasions in North America and Hawaii, pp.231-248. Ecological studies no. 53, Springer-Verlag, New York.

Mueller-Dombois, D. 1973. A non-adapted vegetation interferes with water removal in a tropical rain forest in Hawaii. Tropical Ecology. 14:1-18.

Mueller-Dombois, D. 1981. Fire in tropical ecosystems. In:Mooney, H.A. et al. (eds.), Proceedings of conference on fire regimes and ecosystem properties. pp. 137-76. U.S. Forest service general technical report WO-26. 594 pp.

Neser, S. and R.L. Kluge. 1986. The importance of seed-attacking agents in the biological control of invasive alien plants. In: Macdonald, I.A.W., Kruger, F.J. and A.A. Ferrar (eds.), The ecology and management of biological invasions in southern Africa, pp. 261-268. Oxford University Press, Cape Town.

Noble, I.R. 1989. Attributes of invaders and the invading process: terrestrial and vascular plants. In: Drake et al (eds.), Biological invasions: a global perspective, pp.301-314. John Wiley and Sons, New York.

Noble, I.R. and R.O. Slatyer. 1980. The use of vital attributes to predict successional changes in plant communities subject to recurrent disturbance. Vegetatio. 43:5-21.

Olson, S.L. and James, H.F. 1982. Prodomus of the fossil avifauna of the Hawaiian islands. Smithsonian Contributions to Zoology. 365:1-59.

Poynton, R.J. 1984. Characteristics and uses of selected trees and shrubs cultivated in South Africa. South African Journal of Botony, 3:387-396.

Radosovich, S.R. and J.S. Holt. 1984. Weed ecology: implications for vegetation management. John Wiley and sons. USA 265 pp.

Ramakrishnan, P.S. 1984. The science behind rotational bush fallow agriculture systems. Proceedings Indian academie science (Plant science) 93:379-400.

Ramakrishnan, P.S. and P.M. Vitousek. 1989. Ecosystem-level processes and the consequences of biological invasions. In: Drake et al (eds.), Biological invasions: a global perspective, pp.281-300. John Wiley and Sons, New York.

Rejmanek, M. 1989. Invisibility of plant communities In Drake, J.A. et al (eds), Biological Invasions: a global perspective. pp. 369-388. John Wiley and Sons, New York.

Robinson, T.W. 1969. Introduction, spread and areal extent of salt ceder (*Tamarix*) in the western states. U.S. Department of the Interior Geological Survey Prof. paper 491-A, 12 p.

Roughgarden, J. and J. Diamond. 1985. Overview: the role of species interactions in community ecology. In: Diamond J. and Case T.J. (eds.) Community ecology. Harper and Row, New York. 665 pp.

Rutherford, M.C., Pressinger, F.M., and C.F. Musil. 1986. Standing crops, growth rates and resource use efficiency in alien plant invaded ecosystems. In: Macdonald, I.A.W., Kruger, F.J. and A.A. Ferrar (eds.), The ecology and management of biological invasions in southern Africa, pp. 189-200. Oxford University Press, Cape Town.

Shigewra, G.T. and R.M. Bullock. 1983. Guava (*Psidium guajava L.*) in Hawaii - history and production. Research - Extension Series, Hawaii Institute of Tropical Agriculture and Human Resources. #035. 28pp.

Skellum, J.G. 1951. Random dispersal in theoretical populations. Biometrika, 38: 196-218.

Smith, C. 1985. Impact of alien plants in Hawai'is native biota. In: Stone, C.P. and J.M. Scott (eds.). Hawaii's terrestrial ecosystems: Preservation and management, pp.180-250. Cooperative park studies unit. University of Hawaii. Hawaii.

Sussman, R.W. and Raven, P.H. 1978. Pollination by lemurs and marsupials: an archaic co-evolutionary system. Science, 200:731-736.

Thorhoug, A. and C.B. Austin. 1976. Restoration of seagrasses, with economic analysis. Environmental conservation, 3(4): 259-267.

Taylor, S.E. and Walker, B.H. 1984. Autocology of an invading population of the cactus *Cereus pervianus* (Queen of the night) in the central Transvaal. South African Journal of Botony. 3:387-396.

Temple, S.A. 1977. Plant-animal mutualism; co-evolution with dodo leads to near extinction of plant. Science. 197:885-886.

Turner, C.E. 1986. Biological control, conflicting interests and the issue of risk to non-target plants. Proceedings of the 6th International Symposium on biological control of weeds, pp. 103-113. Vancouver, Canada agriculture, Canada.

Uhl, C. 1987. Factors controlling succession following slash and burn agriculture in Amazonia. Journal of ecology. 75:377-407.

Usher, M.B. 1988. Biological invasions of nature reserves: a search for generalizations. Biological conservation. 44:119-135.

Van Wilgen, B.W. and D.M. Richardson. 1985. The effects of alien shrub invasions on vegetation structure and fire behavior in South African fynbos shrubland: a simulation study. Journal of Applied Ecology, 22: 955-966.

Versfeld D.B. and B.W. van Wilgen. 1986. Impact of woody aliens on ecosystem properties. In: Macdonald, I.A.W., Kruger, F.J. and A.A. Ferrar (eds.), The ecology and management of biological invasions in southern Africa, pp.239-246. Oxford University Press, Cape Town.

Vitousek, P.M. 1986. Biological Invasions and Ecosystem properties. Can species make a difference? In: Mooney, H.A. and Drake, J.A. (eds). Ecology of biological invasions in North America and Hawaii, pp. 163-173. Ecological studies no. 53, Springer-Verlag, New York.

Vitousek, P.M. 1988. Diversity and biological invasions of oceanic islands. pp. 181-192 in E.O. Wilson (ed.). Biodiversity . National Academy Press. USA. 521 pp.

Vitousek, P.M. and L.R. Walker. 1989. Biological invasion by *Myrica faya* in Hawai'i: plant demography, nitrogen fixation, ecosystem effects. Ecological monographs, 59(3): 247-265.

Vivrette, N.J. and C.H. Muller. 1977. Mechanisms of invasion and dominance of coastal grassland by *Mesembryanthemum crystallinum*. Ecological monographs. 47:301-318.

Vogl, R.J. 1977. Fire frequency and site degradation. In: Mooney, H.A. and Conrad, C.E. (eds.). Symposium on the environmental consequences of fire and fuel management in Mediterranean ecosystems, pp. 193-201. USDA Forest service general technical report. WO-3.

Whiting, B.H., Bate, G.C. and D.J. Eramus. 1986. Photosynthesis and carbon allocation in invasive plants of South African biomes. In: Macdonald, I.A.W., Kruger, F.J. and A.A. Ferrar (eds.), The ecology and management of biological invasions in southern Africa, pp. 179-189. Oxford University press. Cape Town.

Williamson, M. 1981. Island populations. Oxford university press. Oxford. 286 pp.

Wilson, E.O. 1961. The nature of the taxon cycle in the Melanesian ant fauna. American naturalists. 95:169-193.

MANAGEMENT PLAN FOR THE CONTROL OF *Psidium cattleianum* IN RANOMAFANA NATIONAL PARK

This section contains a management plan for the control of strawberry guava, *Psidium cattleianum*, infestations in Ranomafana National Park (RNP), in eastern Madagascar. The objectives of this plan are to control and greatly reduce the extent and dominance of strawberry guava within the boundaries of the park, and to occupy the current guava infestation sites with native tree species. The management plan will be implemented in two phases. The first phase will be the research and development phase, during which the baseline information regarding extent of infestation, control methods, and nursery and outplanting technology will be developed. The second phase will be the integrated management program, in which the control measures and the restoration efforts will be accomplished. This period will also include education, and future monitoring. The management plan has been written with an emphasis on economic feasibility and a sensitivity to the social conditions, culture and customs of the Malagasy people. This plan deals with the current guava infestations and their control, and discusses ecological restoration and future monitoring.

Background information

Madagascar, the worlds fourth largest island, is located 400 kilometers from the east African coast, is composed of 587,000 square kilometers, and stretches 1,600 kilometers from northern to southern tip. It is an island of enormous biological diversity, containing rain forests in the eastern escarpment, dry deciduous woodlands in the northwest, and the semi-arid "spiny desert" in the south.

Madagascar has been targeted as a conservation priority by the World Wildlife Fund for nature (WWF), the World Bank, and other international organizations, because of the abundance of endemic species which are severely threatened. According to Jolly (1989), 80% of the flora and fauna are unique to the island. This includes 80% of the flowering plants, 97% of the amphibians, 96% of the reptiles and 97% of the terrestrial mammals. Jolly (1989, p.193) writes that: "there are perhaps more unique species in more immediate danger of extinction than anywhere else".

The decline in Madagascar's native plants is caused mostly by habitat destruction from slash and burn agriculture and the deliberate setting of fire. The rain forest has diminished by 50 per cent in the past thirty years, and between 1950 to 1985, the rate of deforestation averaged 111,000 hectares per year. If this rate continues, all of the forest will be gone in the next thirty to fifty years (Green and Sussman, 1990).

Since 1960 Malagasy income has been declining, and it now rests at $225 per capita per year, making it one of the world's poorest countries. 80% of the work force is in agriculture, rice and coffee being their main products. Rice is the most important crop and has psychological, social and political significance beyond its status as a mere food. The Malagasy are the world's greatest rice consumers, eating

112 kilograms per person per year (Jolly, 1989).

Madagascar is one of the last habitable regions settled by man. The first inhabitants arrived around 1500 to 2000 years ago from Indonesia. They came across the Indian ocean, bringing rice paddy cultivation techniques with them. Later Africans arrived, bringing large zebu cattle, and Arabs settled in the northern regions. Today, UNICEF estimates a total population of 11,208,000 with an annual population growth of 2.9%

Ranomafana National Park is located in the southeastern region of Madagascar between 47·18'- 47·37'E and 21·02'-21·25'S. The park contains 45,000 hectares which straddles the eastern escarpment of the island, having an altitudinal gradient from 1375 meters at Mt. Maharira to about 400 meters on the eastern boundary. The park lies approximately 90 kilometers west of the Indian Ocean, 60 km. southeast of the provincial capital Fianarantsoa, and 265 km. southeast of the capital city of Antananarivo.

Two ethnic groups live within the immediate vicinity of the park. These are the Betsileo, the people of the highlands, and the Tanala, the people of the lowlands. The Betsileo people cultivate paddy rice in the valley bottoms and the Tanala practice slash and burn upland rice agriculture. These people live in 110 small communities surrounding the park with a total population of approximately 25,000 (Kittlinger, 1990).

Due to the wide elevation range there are three principal types of forest within the RNP boundaries. These types are the high plateau forest, the cloud forest and the lowland rain forests. Thirty percent of the park is in the first type, the plateau forest, which occupies those lands above 1,000 meters, and encompasses the headwaters of the Namarona, the major river bisecting RNP. It is a mosaic of primary and secondary dense chaparral type. Forest stature is smaller than in the other zones and appears to be less diverse. The second type, the cloud forest, comprises the majority of the remaining 70% of the park land, and is considered those lands between 500-1000 meters, with mountainous topography, slopes exceeding 120% and granite cliffs. Average rainfall in the cloud forest exceeds 2500 mm. The forest is a mosaic of uncut primary forest, selectively cut primary forest, and naturally regenerated secondary forest of varying age following cultivation. There is very little of the third type, the lowland forest, within the park boundaries. What does exist is found in areas between 400-500m elevation and are considered sacred areas by the natives, where clearing is taboo (Minnick and Kellison, 1989).

It is unknown when strawberry guava first arrived in Madagascar, but a probable introduction date is around the beginning of the 1800's. Two facts support this idea; first in Mauritius, a neighboring island in the Indian ocean, the strawberry guava was reported present by 1822 (Lorence and Sussman, 1986). Second, in Madagascar, strawberry guava is called Chinese guava, and the early 1800's was a period of Chinese immigration.

Summary of current guava management attempts around the world

Even though *Psidium cattleianum* is considered to be one of the worst plant invaders on tropical islands, there has been virtually nothing written about its management and control. Huenneke and Vitousek (1991) have done the most documented research in Hawaii, and made several management suggestions from their study of the life history of guava. They wrote that thinning alone will not reduce the guava population because the population can rejuvenate itself from a single individual. They suggested a complete removal of the guava from each infected area, and an elimination of the feral pigs. They also wrote that the elimination of the stored seed in the soil would help control the invasion. How these objectives would be achieved was not specified.

In Hawaii Volcanoes National Park the guava infestation has extended to even the most remote parts of the rain forest. In an attempt to preserve some of the remaining forest from guava infestation they have designated certain small areas as important for the protection of species diversity. Within these designated conservation areas all of the guava is being removed by cutting stems and pulling seedlings and suckers. These areas are fairly small in size (several hectares), but even maintaining them in a guava free state is quite an effort. It is labor intensive work and is only effective in maintaining small areas free of guava (Tim Tunison, personal communication).

In South Africa, the removal of guava has been tried by cutting and spraying with herbicides. No specifics were written, but they were unable to find an herbicide which would kill *Psidium cattleianum*. They also expressed an interest in finding a biological control agent for the guava.

In Mauritius, WWF has been trying to control exotic plants and animals for the past 10 years. A majority of the vegetation work has centered on the control of *Psidium cattleianum*. The work was initiated by a WWF employee and a crew of laborers. They were able to successfully kill the guava using the herbicides picloram and agent orange, when applied to the stumps of the cut trees. Other herbicides, such as glyphosate were tried with a lesser degree of success. Eventually the herbicide control program was eliminated, because it was impossible to get the workers to follow the manufacturers suggested safety regulations. Now the guava trees are cut and seedlings hand-pulled. This allows them to keep certain, restricted areas free of guava (Wendy Strom, personal communication).

Growth habits of *Psidium cattleianum*

The reproductive biology of *P. cattleianum* has best been documented by Huenneke and Vitousek (1990) in Hawaii. They observed that the peak reproductive activity occurred in June to October, but there were stems flowering all year long. Leaf flushing and vegetative growth peaked during the same time that the flowers were active. They observed mortality of between 3 and 8% for first year seedlings. However these same seedlings grew slowly, gaining .3 cm. in diameter on stems that were less than 1.0 cm. in DBH. Stem densities were between 3 and 9 stems/m^2 at five different elevations studied. They observed germination rates of 76 to 40.5%, and observed that most naturally occurring seedlings grew on undisturbed bryophyte mats. They observed that guava could produce dense population even at low light levels, and sustain these densities to the exclusion of native species.

In Madagascar, I observed guava growing on many hectares of rain forest land which had previously been cultivated with upland rice. In these areas the canopy was between six and eight meters and predominantly composed of guava. These areas often contained from five to ten stems/m^2. I also observed it growing under a native forest canopy, in close proximity to the areas which had been cultivated. In these cases the guava canopy would be from three to five meters tall, below a native canopy which would be from 15 to 25 meters tall. When grown in a closed stand the tree will grow up to 8 meters tall and achieve a maximum diameter of 12 centimeters. The bark of the guava is completely smooth and a shiny dark brown color (Hooper, personal observation).

1991 Inventory Results

It is unknown how much land is covered by strawberry guava in either RNP or in eastern Madagascar. I believe that it may be considerable, since guava is frequent along roadsides throughout the tropical forest region and during an inventory I performed in 1991, large quantities were encountered in the park itself.

The inventory examined three stands of forest, each with a different disturbance history. Guava was found in one of the areas that had been burned and used for agriculture forty years previously. In this area sixty-eight different tree species were recorded, but 41.5% of the overstory and 17.89% of the total basal area was in strawberry guava. Fifty-four per cent of the regeneration was also observed to be in guava. Since the inventory I performed was in a single site, it is unknown how the level of guava I recorded compares to other areas of the park with similar or different disturbance histories. It is also unknown whether guava is an invasive species in RNP, or whether it is restricted to sites with previous human-induced disturbances. It could very possibly be present in small isolated clumps throughout the park because four different species of lemur were observed eating the guava during the period the inventory was performed. These lemurs may be spreading the guava seeds within a several kilometer radius of the inventory site.

Review of the control options

This section briefly discusses the various control options in regard to how they would perform in RNP. It should be noted that every method of control will involve use of crews initially unfamiliar with the method and its tools. Thus the need for training accompanies each approach, but differs in context and who must be trained. The advantages and disadvantages of each option is discussed at the end.

Mechanical control

In terms of operations, mechanical control is one of the least expensive, most widely understood, easily implemented methods of control available to RNP. The strawberry guava is easily cut down by both hand axes and machetes. Both of these tools could be provided by the worker himself. In addition, seedlings could be hand-pulled, probably throughout the first year. However, the cutting of guava will not control its spread for long, since it is an aggressive, vigorous sprouter. Successive cuttings, to ground level, with the aim of eliminating all photosynthetic material and depleting reserves, may make the guava inviable if the control is maintained long enough. A hypothetical time period in which the cutting would need to be maintained could be six months to one-year. This would be an extremely labor intensive operation, at least until all of the guava stands are cut. After this point it would require many fewer individuals to simply cut sprouts and hand-pull seedlings as they appear. An additional control option which could be employed where guava is found under an existing canopy is cutting the stump to ground level and covering it with 1 m^2 of thick, black plastic. The transport and acquisition of the black plastic may be expensive if it is not available in Madagascar. While mechanical control would require a great deal of man-power, the cost will not be very expensive since the cost of labor is very low in Madagascar. An additional advantage of mechanical control is that the wood from the initial cutting of the larger stems could be utilized by the men who are doing the cutting, or the park could give the wood to the villages. Guava makes an excellent fence post, and is very good fire wood.

ADVANTAGES

- Socially acceptable method which would be agreeable to park management
- Locally accessible work materials
- Provides employment to the local population
- Guava can be used by either the workers, the villages, or the Park as either firewood or fence posts.
- Inexpensive method; hiring 30 men for 1 month would cost $US 1200.

DISADVANTAGES

- Periodic cuttings to ground level may not actually kill guava, thus efficacy is based on multiple visits to the same stumps.
- Weeding of sprouts and hand-pulling of seedlings may be required for a long period of time.
- Difficult to coordinate and organize a large crew of workers.
- Risk of personal injury.

Chemical control

Because of the guava's sprouting potential, herbicides may be an essential component of a control program, however their acquisition and application could be expensive when compared to a single use of mechanical control. Herbicide can be sprayed on the sprouts, suckers, young seedlings as well as applied to the stumps after cutting it with a hand axe or machete. Some products may be applied as pellets to the soil at each tree. Chemicals may be also used to control small outbreaks and seedlings when they appear in isolated locations, or as broadcast treatments of larger patches. Once the herbicide is in the park it would be relatively inexpensive to apply it to areas which had been cut. Before any herbicides are sprayed it would be important to do a number of trials, to determine which herbicide rate and method of application would be effective, and at what point in the guava's phenological cycle it should be applied. A broadcast spray to mixed forests would probably not be desirable since that may injure many of the native species as well. If an effective chemical can be found, spot application may provide a means of effectively and immediately controlling outbreaks that would be impossible to monitor or control otherwise. However, special care should be in the education of the workers, who will be completely unfamiliar with the technology.

ADVANTAGES

- May be the only option that will actually kill the guava.
- May be an excellent method to control small start-up populations.
- May be highly effective when used in conjunction with mechanical and biological control.
- Would provide jobs for the local inhabitants
- Could use the guava wood, if the trees were first chopped down and herbicide

applied to the stump., or if the trees were cut down after wilting.
- Small volume of chemical is very easy to transport.

DISADVANTAGES

- May be difficult to train the workers to use the chemicals in an effective manner and to use the recommended precautions to protect non-target plants and avoid personal contamination.
- May be expensive to import and transport chemicals to the park.
- Need for foreign exchange capital to purchase chemicals.

Biological control

If the guava is found throughout the park, biological control may be the only method theoretically able to control, or weaken the plant over large land areas. A decision regarding the use of biological control needs to be made as soon as the extent and amount of guava is determined. If it appears that the current guava levels in and around the park represents a pervasive problem, then the research on biological control should be initiated immediately. However, if guava does not appear to be a pervasive problem then it would be too expensive to initiate and implement a biological control program, and the other control options should be thoroughly explored. If biological control is initiated, then a Malagasy expert should be trained by a consultant during the research phase to direct the control operation, after searching for, importing and validating efficacy and safety. This is an expensive option in the short term, and the expenses may be larger than the park is capable of financing. If this is the case, national attention should be drawn to the problem of guava infestations in the indigenous rain forests, and international donors should be solicited to help with the biological control operation. It would be ideal if the biological control studies could be done in cooperation with the South African Plant

Protection Institute. South African land managers are also very interested in finding a biological control agent for *Psidium cattleianum*, and are experienced, successful and enthusiastic about using biological control, where appropriate, as an integrated part of control programs. They would also have the facilities and personnel capable of understanding such a study.

ADVANTAGES

- May be the only method which offers a long-term solution to control or weaken the plant.

DISADVANTAGES

- May be too expensive if the guava infestation is not severe enough.
- May not work effectively, high risk of failure.
- May have negative effects on the native flora
- Will take at least three to five years to set-up the beginnings of a control program.
- May require multiple introductions of insects/pathogens.

Cultural control

In RNP cultural control methods would include the use of fire and grazing. Currently there are no intentional fires set in the park and the natural fire frequency is probably extremely low due to the high rainfall and brief dry periods. The areas where I saw guava in the park are all areas that have been burned previously. In general very few sprouters are inhibited compared to non-sprouters in a fire regime, and fire may well accelerate the guava to dominance. In addition, serious leaching losses of soil nutrients may occur with successive burnings. Because of the poor chances of success, and probable negative consequences, I am not recommending the use of fire.

Grazing is now permitted in the park by the large zebu cattle. This appears to favor the unpalatable guava at the expense of the native species. Further studies need to be done to confirm this hypothesis. If this is the case, the cows should be excluded from the park, and the villagers must seek a reasonable alternative place to graze them.

ADVANTAGES

- Reduction in grazing may help increase the competitive advantage of native plants over guava.
- Elimination of grazing may help reduce the spread of guava into areas not yet colonized.

DISADVANTAGES

- May cause bad feelings between the villagers and the park if the villagers are excluded from using their traditional grazing areas.
- Fires getting out of control can cause extraordinary damage in a native culture.
- Successive fires may cause severe soil leaching.
- Fire may encourage guava regrowth.

MANAGEMENT PLAN

The writing of the management plan has been made more difficult without having a clear idea of which control options will be effective, and without having studied the control of *Psidium cattleianum* in Madagascar. Given the current state of information, I will outline what I consider the most reasonable approach to the development of a management program for the park's guava infestation. The outline is followed by a narrative which more exactly explains procedures and rationale behind the decisions I made, given the data on hand. A final cost analysis is presented at the end of the narrative.

MANAGEMENT PLAN OUTLINE

Phase 1 - Research and Development

1. Evaluate the severity of the problem
   - A. Use existing aerial photographs to identify guava infestations covering greater than one hectare.
   - B. Visit each area identified having guava on the photos and do an inventory to determine the extent, density and pattern of guava spread into the adjacent forest.
   - C. Conduct a general forest inventory with the intention of identifying forest types/land-use conditions which favor guava, and those that don't.

2. Test and evaluate control measures
   - A. Study life history of guava to learn its weaknesses.
   - B. Evaluate broadcast removal methods in areas that have an overstory of guava.
     - a. grazing
     - b. herbicides
     - c. mechanical
   - C. Patches of guava which are under a native forest canopy.
     - a. apply herbicides to cut stumps, bushes, regeneration
     - b. apply black plastic to cut stumps
   - D. Biological control

3. Test and evaluate nursery culture and reforestation
   - A. Seed collecting and germination trials.
   - B. Establish nurseries with most promising tree species.
   - C. Outplant seedlings and evaluate survival and stocking.

Phase 2 - Integrated Management Program

4. Apply the control methods
   - A. Control guava growing in open areas, using most promising method(s).
   - B. Control guava under native forests
     - a. herbicides
     - b. black plastic cover
   - C. Biological control, if warranted.

5. Begin restoration work
   - A. Make plan to remove guava and undesirable secondary succession from newly planted areas.
   - B. Outplant most favorable seedlings
   - C. Set up enclosures

D. Evaluate results

6. Education, monitoring and future control.
   A. Initiate education at a local level
      a. schools
      b. village councils
      c. tourists
   B. Monitoring and future control

Two primary factors are dominant in this plan: The first is that all of the actual weed control will be done inside a National park, which by definition is established for conservation, tourism and research. Second, the park is surrounded by an impoverished, rural population which could greatly benefit from employment, including that derived from heavy use of the park.

Objectives of the plan:

1) To control and greatly reduce the extent and dominance of strawberry guava within the boundaries of RNP.

2) To occupy the current guava infestation sites with native tree species.

3) To minimize further spread and dissemination of the strawberry guava seeds within the park boundaries.

Phase 1 - Research and Development

Because of the scope of the problem and the number of uncertainties involved, it will be necessary to implement the management plan in two phases. The first phase, lasting for three years, will be the research and development phase. During this period the majority of the exploratory work and testing which is critical to a control operation will be accomplished. Three separate areas will be addressed;

the severity of the guava infestation will be assessed, control measures to eliminate guava will be tested and evaluated, and nursery cultures and reforestation will be tested and evaluated. The results obtained from Phase 1 will be used in the second phase, the integrated management program.

Evaluate the severity of the problem

An estimate of how much land is covered by *Psidium cattleianum* and the characteristics of the spread pattern within the boundaries of the park is the first necessary step of the management plan. How much land is covered will determine the future control measures and the general approach to the problem. If *P. cattleianum* appears to occupy vast land acreages and is invasive, then biological control could be considered as a viable solution, provided the appropriate organisms are available and themselves noninvasive. However, if *P. cattleianum* is only occupying previously disturbed sites, and of limited acreage, and is not spreading into the native forests, then mechanical, chemical and cultural methods may control it effectively, and at a much lower development cost.

Aerial photographs are now in the process of being developed for the park, at a scale of 1:20,000. These photos should be used to identify forest patches, such as the kind I inventoried, in which guava is a major component of the canopy. From the photos the location and approximate acreage, and percent cover of guava for each patch should be recorded.

After identifying each suspect area on the photos, it will be visited on the ground to confirm the information retrieved from the aerial inventory. In addition, an inventory will be done adjacent to the patches to see if the guava is invading the "natural" forest. The objective of the inventory would be to determine if the guava is invasive, and if so, how far into the forest it penetrates and the pattern of its spread. This could be done by either walking the perimeter in successively larger circles around the guava patch, or walking transects in fixed directions from the center of the patch. Percent cover of guava, or number of individuals seen could be recorded for each distance away from the patch. This would be a simple, quick way of estimating how much guava there was and how far away from the patch it extended.

General inventory

While the photo work and reconnaissance is being done a general survey of the park should be completed with the intent of identifying forest types or land-use conditions which favors guava and those that don't. This survey should concentrate, although not exclusively, on areas of previous human disturbance, such as trails and roads both on the interior and adjacent to the forest. Areas throughout the forest should be sampled as well, even if the human and cattle disturbance history has been very low to non-existent. An important result from this work will be a map which identifies the foci and scattered spread into the forest. Such a map will help identify where the control priorities are highest and will give a clearer idea of the threat of re-invasion from foci outside the park.

Test and Evaluate Guava Elimination Measures

While the extent of guava is being determined, testing and experimentation on the control of the plant should begin. The different methods to test include; grazing, herbicides, mechanical methods and if control is to extend outside of the park, biological control. The strategy for control should be different for the guava plants which are located under a native forest canopy, compared to those in relatively homogenous stands of guava, but the same tools may be valuable in both situations. The guava under the native forest canopy will probably be present as isolated individuals or in small clumps. I am recommending that these be treated rapidly to avoid the growth and development of isolated individuals into larger reproducing populations. On the other hand, those areas which have an overstory of guava already contain populations of thousands of individuals, and may require a treatment which covers the entire infested area. These are the seed sources, hence the sources of infection, and the location from which future spread will originate. For these reasons, these areas would be targets for treatment as soon as methods are available, with the intent of eliminating current seed crops, and replacing the tree with competitive useful cover.

Areas that have an overstory of guava should be evaluated for control with grazing, herbicides, hand-cutting or combinations of these.

Testing

The effects of the different control options need to be tested, both individually and in combination of twos and threes, with the objective of finding the most effective control combination that can be managed by the park and also be cost efficient. I recommend that two initial experiments be installed. The first to test the effects of different herbicide treatments, and the second to test the other mechanical methods. For the second study, small 5 by 10 meter plots should be installed within existing guava thickets to test the effects of the eight different control options which are outlined below. A randomized block experiment which is replicated in several sites should be installed. Each site is a block and the control options are replicated four times at each site. This would allow a simple statistical analysis to be performed. Numbers of guava individuals and per cent cover of guava should be recorded for each control option at 1 year and 2 years after treatment. Each control option will need several cycles of study as the techniques are evaluated, refined and upgraded.

The control methods to be tested include:

1) Cutting the trees to ground level and hand-pulling seedlings at 3 months
2) Cutting the trees to ground level and hand-pulling seedlings at 3 months, and 6 months
3) Cutting the trees to ground level and hand-pulling seedlings at 3, 6, and 9 months
4) Cutting the trees to ground level and hand-pulling seedlings at 3, 6, 9, and 12 months
5) Cutting the trees to ground level and applying herbicides to the cut stumps
6) Cutting the trees to ground level and applying 1 m^2 of black plastic to the cut stumps
7) Cutting the trees to ground level and burning the area following resprouting
8) Cutting the trees to ground level, applying herbicides and burning the area

A similar study should be installed that tests the effect of different herbicides, methods of application, timing of application and rate of application.

Mechanical control - All standing biomass of guava should be removed to ground level and weeded as needed to maintain zero stems of guava in the plot for 3 months, 6 months, 9 months, 12 months. These time periods are chosen to determine if there is a minimum time period needed of hand-weeding which would kill the guava. If at the end of the 12 month period there continues to be a recovery of guava, then this method would probably not be suitable, at least not if it is used alone.

Chemical - I recommend testing soil applications of imazapyr, picloram, tebuthiuron, bromacil, and hexazinone for their efficacy on guava. These herbicides are chosen because of their success in controlling herbicide-resistant hardwood species in the United States. The variables to be defined are method of application, timing of application and amount of active ingredient. Imazapyr can be applied with the "hack and squirt" stem injection technique using either 'chopper' or 'arsenal' with several different cutting spacing. (Cole and Newton, 1989). Tebuthiuron, imazapyr, hexazinone and picloram can be dispensed in granular form, which can be transported in a tree planters bag. Broadcast spray techniques can also be assesses using foliage application of picloram, bromacil and imazapyr. It needs to be determined how effectively roots are killed on guava, and effects of herbicide residues on future species composition, particularly the planted hardwoods.

Black plastic - I recommend securing a piece of 4ml. thick, black plastic over the guava stumps immediately after cutting to prevent sprouting. Sizes should be tested in the range of 50 cm x 50 cm, to 1 m^2.

Grazing - Since the effects of grazing on guava are unknown, studies need to be done to determine if guava can be eradicated by intense grazing, and the effect of grazing on guava and the native plants in the natural forest after cessation. The first question can best be answered by clearing the guava from an infestation area and then allowing the cattle to graze the area. The intensity and duration of the grazing after clearing the guava should be monitored. The second question can be determined by setting up 1/16th ha. exclosures and measuring the difference between the frequency and % cover guava, within and outside the enclosures, over a period of 1 and 2 years. The exclosures can be constructed of guava poles, with sharpened tips, that are pointed outward. Park workers often construct this type of exclosure to keep cattle out of their crops and family compounds.

Biological control - If guava appears to be a pervasive problem throughout the park and eastern Madagascar, then using biological control may be essential. This would represent too large an area to be handled by conventional methods. The first step would be to hire an expert scientist as a consultant to oversee the operation, and train a local Malagasy counterpart. This individual could be based in either Madagascar or South Africa, to complete the research and testing phase of the work. Because of the uncertainty associated with the long-term success of biological control, conventional control options should be employed in areas where it is a high priority to hold and eliminate the guava.

If guava does not appear to be a pervasive problem then I do not recommend starting a biological control program, simply because the cost would be too great, and the outcome too uncertain for the park to initiate such a program. If biological control of guava is achieved elsewhere in the world, then this would greatly reduce the start-up costs and it may become more of a feasible alternative.

Study the life history of guava

Concurrently with control tests, a cooperative program with the University at Antananarivo should be established with the objective of studying the ecology and life history of *Psidium cattleianum*. Four graduate students should be funded with a stipend during the period that they are working at the park. Information should be gathered on the life span, persistence under shade, reproduction, dispersal, phenology and response to disturbance of guava.

Test and evaluate the nursery culture and reforestation

When the guava is removed from the infested areas, these areas will require immediate, intensive restorative work in order to ensure a complete canopy of native tree species. This will entail planting fast growing, native species with high leaf area potential which can quickly achieve full site occupancy. Guava trees which are treated under a native forest canopy will not require restoration work, it is presumed that the forest cover will provide seeds to replace the guava, and that canopy spread will cover gaps.

Because virtually nothing is known about the nursery culture of native Malagasy rain forest plants, basic information on growing these trees in a nursery and outplanting them will need to be acquired. A study of the germination rates and a nursery of native seedlings should be established at the beginning of the phase 1, with the intention of developing the technology to grow native trees for outplanting in the treatment areas. It is important that the seedlings grown in the nursery be indigenous to the rain forests of Madagascar and able to achieve full site occupancy quickly. I recommend that seeds be collected from tree species which were either abundant or dominant in the other forested areas I inventoried. In addition to growing seedlings, vegetative propagation should be tested with the native trees. This may represent an easier, relatively inexpensive method to establish fast-growing regeneration. Some legumes should be chosen for mixed plantings, for their ability to fix nitrogen and therefore maintain site quality. Seeds of commercially valuable tree species which had been selectively logged from parts of the forest should also be tried. Table 1 lists some of these species.

Table 1: A list of seedlings to germinate in the nursery.

Genus/species	**Family**
Gambeya madagascariensis	Sapotaceae
Ocotea sp.	Lauraceae
Sloanea rhodantha	Eleacarpaceae
Cannarium madascariensis	Burseraceae
Calliandra alternaus	Leguminaceae
Tambourissa sp.	Leguminaceae
Dalbergia sp.	Leguminaceae
Palisandra sp.	Ebenaceae

Sideroxylon sp.	Sapotaceae

If certain trees prove to be impossible to grow under nursery conditions, the seeds can be taken to the treatment site and planted throughout the area.

Once the ability to grow seedlings in a nursery exists the outplanting survival and performance should be studied. It is essential that the chosen seedling species perform well in the treatment areas. I recommend outplanting the seedlings when they are two feet tall, at 800 to 1000 trees per hectare. Plots should be established where the growth and survival of the seedlings are recorded, and also the types and intensity of competing vegetation. This latter information will help determine what type of weeding program needs to be organized during the actual restoration work. Studies should be done to determine the best way to reduce the weed cover while encouraging the planted hardwoods. This will probable involve either the use of herbicides or mechanical methods.

Phase 2 - The Integrated Management Program

The second phase of the management plan will be the implementation of the most promising methodologies tested in the Research and Development phase. The control operations will be initiated, seedlings will be planted and competing vegetation will be minimized until the planted trees have a clear competitive advantage. It will be important to monitor the successes and failures carefully throughout this period, in order to refine the existing procedures. While the guava is being controlled, educational messages should be prepared and delivered to the villagers and tourists that enter the park. This information should include an explanation of the current control measures and information on how to minimize introductions.

Apply the control measures

Once a control option, or a combination of options has been chosen, work should begin immediately on both the treatment of the large infestations and the isolated individuals. A large crew of workers will need to be hired to complete this work. All of the workers will need to be trained so that they understand the proper procedures and objectives of the work that they are doing. Workers should be evaluated periodically to ensure that the work is being performed properly. In addition, surveys will be needed to evaluate whether control is holding and native species replacing guava.

Ecological restoration

Once the areas have been cleared of guava, the seedlings from the nursery should be planted as quickly as possible, to minimize competition with undesirable secondary succession, or guava seedlings. Nurseries should be established at villages around the border of the forest, or at the treatment sites themselves. This will minimize transportation time to the treatment areas. In most instances the treatment area will only be accessible by foot.

During phase 1 a methodology will have been developed to reduce the undesirable vegetation, present at planting time. This weeding should be done as frequently as is needed to insure that the planted hardwoods achieve dominance.

It is essential that cattle not disturb the areas under restoration, except where grazing is being used for control. Since there are no native ground dwelling herbivores in the rain forest, cattle represent the only mammalian threat to the seedlings. By the time the restoration work is begun, the cattle may have already been excluded from the park if not, the cooperation of the villagers should be enlisted to keep the cattle away from restoration areas. If this is not possible then a guava fence (such as the kind previously described) can be installed around the perimeter of each area. Another less labor expensive option would be a cheap electric fence, or a herdsman.

It will be necessary to evaluate the results of the restoration process to insure that the objectives of the management plan are actually being met. The staff should remain flexible, and incorporate changes, or further research when needed.

Education, monitoring and future control

In order for a long-lasting control to be established, education and cooperation need to be initiated for both the local population and the tourists visiting the park. Inadvertent introductions of exotic plants by tourists can be minimized if an informational brochure is handed out at the time that they buy the park entrance permit. Additional brochures and information about the dangers of exotic plants can be made available at the entrance of the park. This will also serve to help keep down the spread of guava elsewhere in Madagascar.

Local populations can be reached through the school system and presentations given to the village counsels. There is already a group of Malagasy extension agents, who are hired by WWF and live around the outside of the park. These agents could easily provide information to the villages about the dangers that exotic plants pose to the native flora, and describe measures of prevention and eradication.

Once the initial infestations of guava are controlled, it will be necessary to monitor the success of the treatments in the longer-term, and determine if the spread of guava has really been stopped. If populations are re-establishing, then appropriate control measures will be needed to insure that these populations of guava do not provide the basis for recovery. It is important to have an infrastructure in place so that when the guava is located it can be treated immediately.

I recommend a reconnaissance visit every six months for the first two years to each treatment area, and an on-going reporting program by workers throughout the park. All existing guava should be removed during each visit. The surrounding native forest should also be examined to see if guava has re-infested, if so key forest workers will remain equipped to remove specimens located. This type of monitoring should probably continue indefinitely, at least on an annual basis.

Estimated Minimum Budget for 8-year Management plan without biological control

Phase 1 - Research and Development

	Year 1	Year 2	Year 3	Phase 1 total
1. Salaries and Wages				
Project supervisor	$ 5000	$ 5000	$ 5000	$ 15,000
Nursery supervisor	$ 4000	$ 4000	$ 4000	$ 12,000
Technical assistance				
Technicians	$ 6000	$ 6000	$ 6000	$ 18,000
Laborers				
Nursery workers	$ 1920	$ 1920	$ 1920	$ 5,760
Control workers	$ 4800	$ 4800	$ 4800	$ 14,400
Students				
Four graduate students	$ 1400	$ 1400	$ 1400	$ 4,320
Expatriate technical advice	$ 10,000	$ 10,000		$ 20,000
Total:	$ 33,120	$ 33,120	$ 23,120	$ 89,360
2. Materials and supplies				
Control supplies	$ 3000	$ 1000	$ 1000	$ 5,000
Nursery supplies	$ 3000	$ 2000	$ 2000	$ 7,000
Education supplies	$ 500	$ 200	$ 200	$ 900
Total:	$ 6500	$ 3200	$ 3200	$ 12,900
3. Total costs for Phase 1				
Salaries and wages	$ 33,120	$ 33,120	$ 23,120	$ 89,360
Materials and supplies	$ 6,500	$ 3200	$ 3200	$ 12,900
Total:	$ 39,620	$ 36,320	$ 26,320	$ 102,260

Phase 2 - Integrated Management Program

	Year 4	Year 5	Year 6	Year 7	Year 8	Phase 2 total
1. Salaries and wages						
Project supervisor	$ 5000	$ 5000	$ 5000	$ 5000	$ 5000	$ 25,000
Nursery supervisor	$ 4000	$ 4000	$ 4000			$ 12,000
Laborers						
Nursery workers	$ 1920	$ 1920	$ 1920	$ 910	$ 910	$ 7,580
Control workers	$ 14,400	$ 14,400	$ 7,200	$ 3,600	$ 1800	$ 41,400
Expatriate technical advice	$ 10,000					$ 10,000
Total:	$ 35,320	$ 25,320	$18,120	$ 9,510	$ 7,710	$ 95,980

2. Materials and supplies							
Control supplies		$ 5,000	$ 5,000	$ 1000	$ 500	$ 250	$ 11,750
Nursery supplies		$ 1000	$ 1000	$ 1000	$ 250	$ 250	$ 3,500
Education supplies		$ 250	$ 250	$ 250	$ 250	$ 250	$ 1,250
	Total:	$ 6,250	$ 6,250	$ 2,250	$ 1,000	$ 750	$ 16,500
3. Total costs for Phase 2							
Salaries and wages		$ 35,320	$ 25,320	$ 18,120	$ 9510	$ 7,710	$ 95,980
Materials and supplies		$ 6,250	$ 6250	$ 2250	$ 1000	$ 750	$ 16,500
	Total:	$ 41,570	$ 31,570	$ 20,370	$ 10,510	$ 8460	$ 112,480

Phase 1 and Phase 2 total costs

Phase 1 total	$ 102,260
Phase 2 total	$ 112,480

Total project costs: $ 214,740 *

* if biological control is used then the total cost will increase by $ 500,000 to 2 million dollars.

Budget Explanation

Phase 1 - Research and Development

Salaries and Wages

An annual salary for both a Malagasy project supervisor @ $ 5000/yr and a nursery supervisor @ $ 4000/yr is requested. The project supervisor should have a advanced degree and be responsible for overseeing the entire control operation including: the inventory, aerial photo interpretation, the control option testing, the installation of the enclosures, the collection and the analysis of the data. The nursery supervisor will oversee the nursery operations, including: the collection of seeds, germination trials, the start-up and maintenance of the nursery, the growing of the fertilizing of the treatment areas, and the outplanting. Two technicians will be hired at @ $ 3000 a year to do the inventory, and the ground truthing of the aerial photos. The salaries for 10 laborers (10 x $2.00 x 20 days x 12 months = $ 4800) for the first year. The 2nd year 30 laborers will be needed, for the outplanting and control operation, the 3rd year 10 will be needed, the 4th and 5th year 5 laborers will be needed to monitor and control the treatment areas. Four nursery workers will be hired @ $ 160/month for five years. Four graduate students from the university of Madagascar at Antananarivo will be provided a stipend to assist their research on the ecology of *P. cattleianum* @ $ 1440/yr. One or several expatriate technical advisors should be hired to help train the project supervisor and provide the technical assistance on the application of the various control methods.

Materials and Supplies

Control supplies - These will include such things as black plastic, herbicides, applicators, flagging, string, paper.

Nursery supplies - These will include tree planting bags, plastic pots, watering cans, fertilizer.

Educational supplies - These will include felt boards, animation and educational pictures, battery operated slide projector, film development.

Phase 2 - Integrated Management Program

Salaries and Wages

The project supervisor will continue to be employed for five more years @ $ 5000/yr. The nursery supervisor will be hired for three years @ $ 4000 to oversee the production of seedlings to outplant. Four nursery workers @ 160/month will be hired to help with the plant production during the first three years of outplanting, after this point two workers will be retained to continue a much smaller scale operation. Control workers will be hired at the rate of $ 2.00/day. Year 4 and 5, 30 workers will be needed to conduct the control measures. Year 6, 15 workers will be retained, year 7 and year 8 this number will be reduced by half each year. The four workers remaining after year 8 should remain employed by the park, as a surveillance team to clear up any guava which reoccurs.

Materials and Supplies

Control supplies - It is unknown at this time what control strategy will be employed, how much it will cost, and how much guava there is to control. For these reasons, an intermediate sum @ $ 5000/yr was chosen for the control operation during year 4 and 5. The amount was reduced by half every year, assuming that the guava infestation will be much reduced. This may overestimate or underestimate the actual costs.

Nursery supplies - An average cost was chosen @ $ 1000 for years 4,5,6 and reduced by half when the outplanting should no longer be as needed, since the actual number of trees to grow is unknown.

Educational costs - Educational costs @ $ 250/yr were chosen to cover brochures, film and other information extending material.

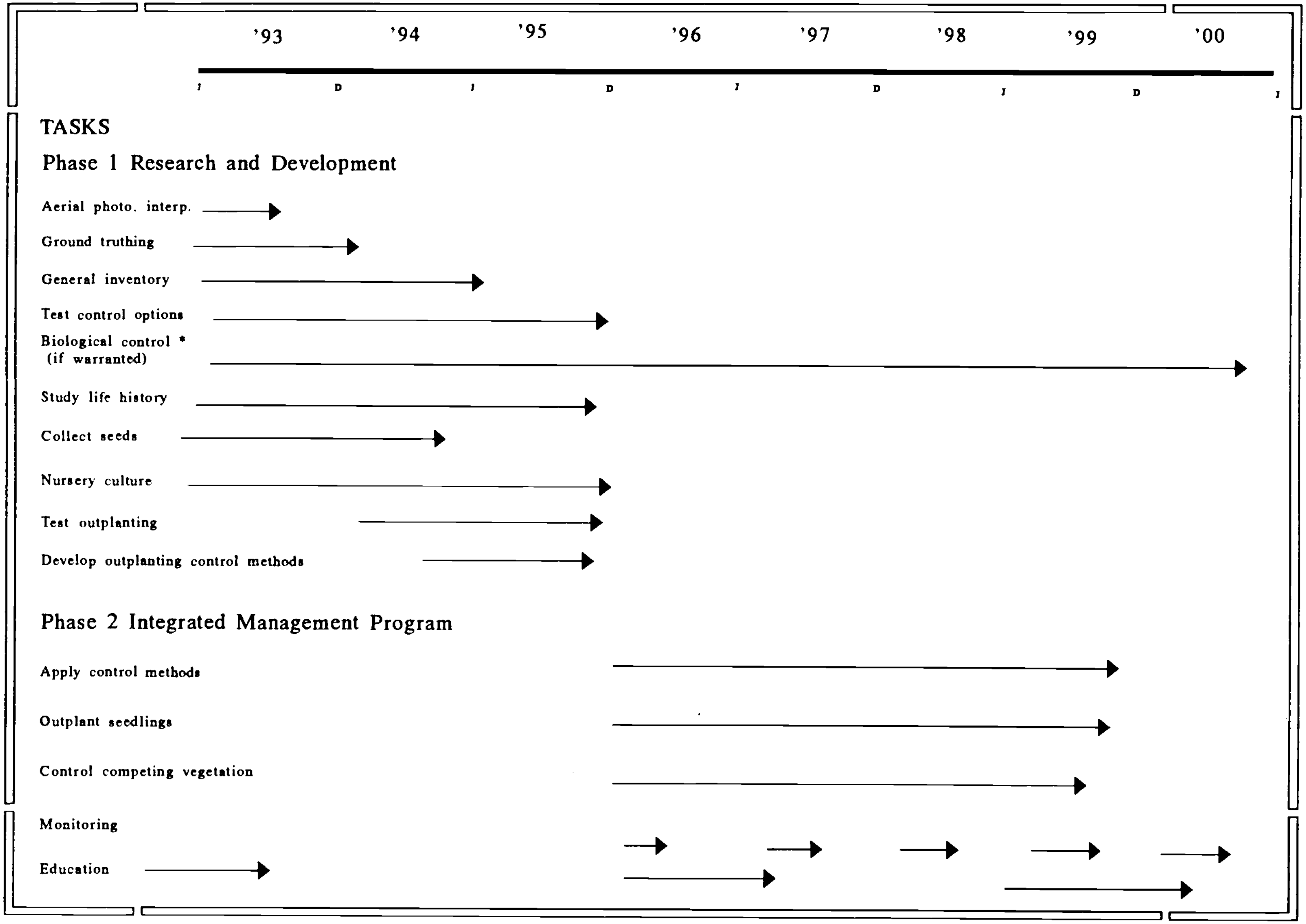

Figure 2: Time schedule for management activities

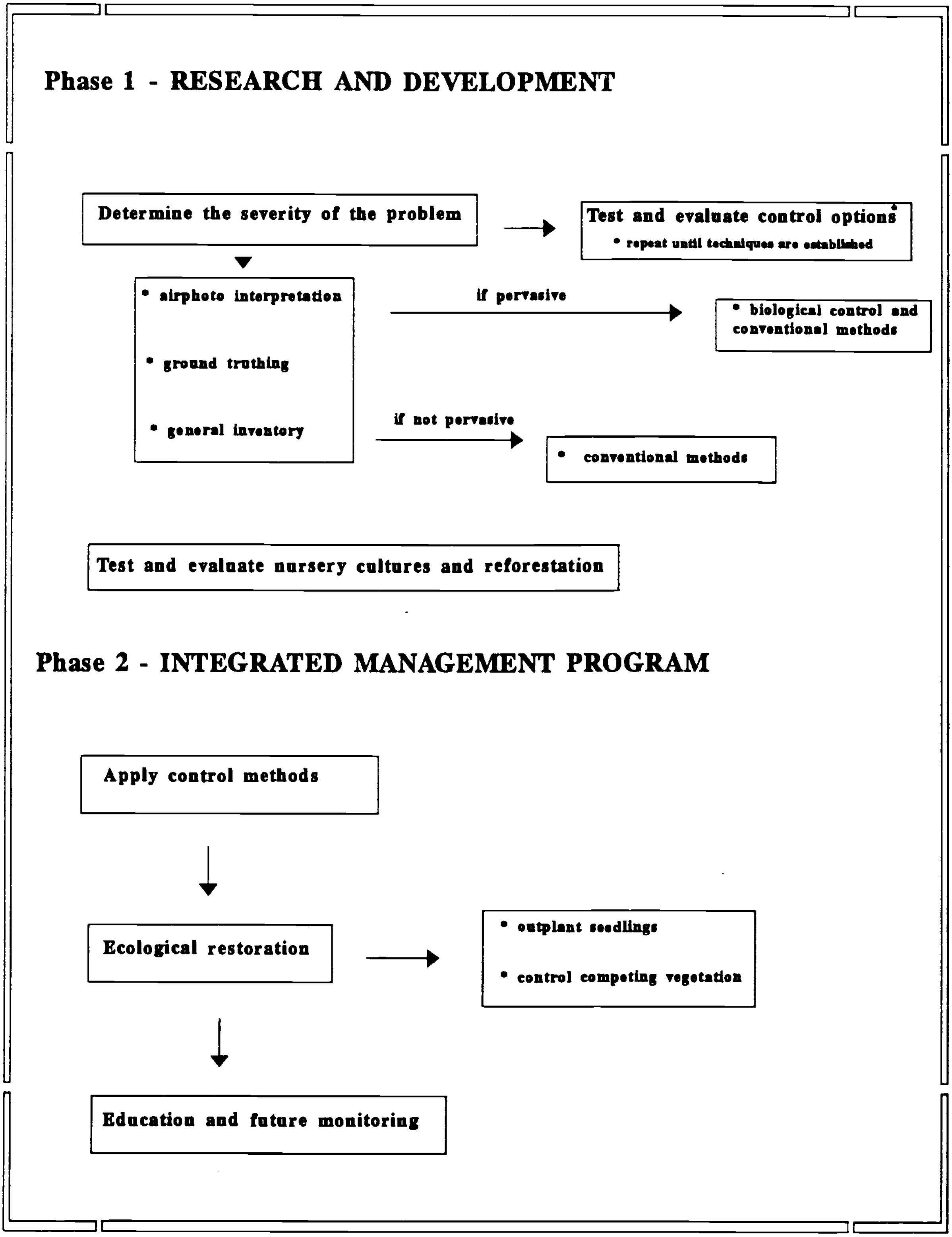

Figure 1: A flow chart for the management of P. cattleianum in Ranomafana National Park.

BIBLIOGRAPHY

Allan, G.E. and J.E. Bath. 1980 The conceptual and institution aspects of integrated pest management. Bioscience 30:658-664.

Cole, E.C. and M. Newton and D. Newton. 1989. Imazapyr stem injection trials. Research progress reports, Western society of weed science. pp 145-146.

Green, G. and R.W. Sussman. 1990. Reforestation history of the eastern rain forests of Madagascar from Satellite Images. Science, 248: 212-215.

Harley, K.L.S., Miller, I.L., Napompeth, B. and S. Thamasara. 1985. An integrated approach to the management of *Mimosa pigra* L. in Australia and Thailand. Proceedings of the 10th conference of the Asian Pacific Weed Science Society, Chiang Mai, Thailand. 1985.

Huenneke, L. and P. Vitousek. 1991. Seedling and clonal recruitment of the invasive tree *Psidium cattleianum*: Implications for management of native Hawaiian forests. Biological conservation, 53: 199-211.

Johnson, T.N. and R.S. Dalen. 1990. Managing individual juniper and pinyon infestations with pelleted tebuthiuron or picloram. Journal of range management, 43(3): 249- 252.

Jolly, A. 1989 The Madagascar challenge - Needs and Fragile Ecosystems. In Leonard, J. (ed.) Environment and the poor: development strategies for a common agenda, 215 pp.

Kittlinger, L. 1990. Health component: Man and the Biosphere initiative. Man and the Malagasy rain forest.

Minnick, G. and R. Kellison. 1989. Description and evaluation of the forestry sector within the Ranomafana National Park project area, Ranomafana, Madagascar. 17pp.

Oxby, Clare. 1985 Forest farmers the tradition of land-use and society in eastern Madagascar. Unasylva 37(148):42-51.

Peters, W.J. and L.F. Newenschwarter. 1988. Slash and burn: farming in the third world forest. University of Idaho Press. Moscow, I.D. 113 pp.

Shipman, R.D. 1989. Controlling hardwood sprouts to release planted larch following clearcutting. Proceedings of the northeastern weed science society, 43:60-61.

Shipman, R.D. 1989. Effects of post-emergent herbicides on survival and growth of planted hardwoods after clearcutting. Proceedings of the northeastern weed science society, 43:66-67.

Van Pelt, N.S. and N.E. West. 1990. Effects of manual application method on application time, thoroughness and tebuthiuron outlays. Journal of range management, 43(1):39-42.