Abstract of the thesis entitled

THE ESTABLISHMENT AND SURVIVAL OF NATIVE TREES ON DEGRADED HILLSIDES IN HONG KONG

submitted by

Hau Chi Hang

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Deforestation and land degradation in the tropics and subtropics are proceeding at an unprecedented rate, threatening a massive loss in global biodiversity, comprising a fifth of the global warming potential, and resulting in losses in goods and services to people living in or near the forest. Most of tropical and subtropical China has been deforested and only tiny remnants of the original forest of Hong Kong remain. Tropical and subtropical reforestation is thus an urgent priority. However, both spontaneous forest succession and active reforestation in degraded grasslands are hindered by a succession of filter-barriers, including poor seed dispersal, high seed predation, low seed germination, and poor survival and growth of established seedlings. These successive barriers were investigated at degraded hillside sites in Hong Kong.

Seed traps placed in grassland for one year caught 12,600 seeds of 35 woody species, with over 94% of these from traps under perches used by birds. The seed rain in the open was less than 0.2 seeds per m^2 per year. All woody seeds trapped were from animal-dispersed species but a survey of spontaneous woody invasion of other grassland sites showed that wind-dispersed species were also important, although more patchily distributed. Seeds of 16 species placed in grassland and shrubland sites suffered 60-day predation rates of 6.5 - 100 %, with 11 of 12 species suffering 100 %

loss at one shrubland site. Trapping and laboratory feeding experiments showed that the major seed predators were two species of rat, although the same rat species also disperse some small seeds. Seed predation was independent of seed size but was lower on species with a tough or thick seed coat. Germination rates for cage-protected seeds at grassland sites were generally much lower than under nursery conditions, suggesting that some factors other than seed viability were affecting seed germination. These three studies together show that seed dispersal, seed predation and germination all limit the rate and diversity of woody plant succession at degraded hillsides in Hong Kong.

A four-way ANOVA experimental design was used to investigate the effects of species, fertiliser, dry-season irrigation and herbicide control of competing plants on tree seedling survival and growth. None of these treatments resulted in a significantly higher seedling survival over two years, suggesting that low soil nutrients, seasonal drought and grass competition are not limiting factors for survival. Irrigation had little or no effect on seedling growth, but fertiliser and herbicide together led to significantly higher growth. An additional 10 species were planted at 3 sites without any treatments. Survival was high for 9 species and growth rates were high for 4. These two experiments suggest that once seedlings are established, the main barriers to forest succession have been overcome. However, direct planting of tree seedlings is labour intensive and expensive. The promotion of natural seed dispersal, direct seeding, and controlling seed predators may provide cheaper alternatives. Where seedlings are planted, careful attention to post-nursery care and planting precautions can greatly reduce initial mortality and make native species a viable alternative to the commonly-planted exotics.

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by

Hau Chi Hang

B.Sc. (Hons); M.Sc. (Environmental Management),
The University of Hong Kong

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Declaration

I declare that this thesis represents my own work, except where due acknowledge is made, and that it has not been previously included in a thesis, dissertation or report submitted to this University or to any other institution for a degree, diploma or other qualification.

Signed ....

Hau Chi Hang

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Chapter 1

Introduction

1. Tropical Deforestation

Tropical deforestation and the degradation of lands that formerly supported forests are proceeding at an alarming rate. An estimated 154 thousand km^2 per year of tropical forests and woodlands were destroyed or seriously degraded mainly through agricultural expansion, livestock ranching, logging, and fuel-wood collection in the 1980s (Brown & Lugo, 1994; Mastrantonio & Francis, 1997; Parrotta *et al.*, 1997). This is already a very worrying figure but Nepstad *et al.* (1999) report that present estimates of annual deforestation for the Brazilian Amazonia capture less than half of the forest area that is impoverished each year, and even less during years of severe drought. In addition, large areas of standing forests that are burnt by fires are not normally documented. They also find that logging crews severely damage 10,000 to 15,000 km^2 per year of forests that are not included in deforestation mapping programmes. Among the different types of forest in the tropics, tropical dry forests are the most endangered, even more so than tropical rainforests (Janzen, 1988).

In China, despite a reported gradual increase in forest cover over the last two decades (from 12% in 1977-1981 to 13.9% in 1989-1993; Zhang *et al.*, 1999), annual deforestation was 4,400 km^2 between 1989 and 1993 (Huang *et al.*, 1997). The increase in forest cover is due to an increase in plantation forests: China has had the world's largest plantation area since two decades ago (Zhang *et al.*, 1999). The total extent of plantations in 1996 was 318,000 km^2 (World Resources Institute, 1999). In South China, the subtropical forests of the three main provinces, Guangdong, Guangxi and Hainan, had decreased from the original 352,988 km^2 to 75,026 km^2 by 1992, a 78.7% decrease (MacKinnon *et al.*, 1996). Tropical and subtropical China has over 460,000 km^2 of degraded land that formerly supported forests (Sun *et al.*, 1995). The tropical forest cover in China in 1996 was only 1,090 km^2 (World Resources Institute, 1999). Clearance for cultivation, replacing the origin forest with tropical economic crops such as rubber in Hainan, logging, cutting for firewood and hill fires are the main current causes for deforestation in tropical and subtropical China (MacKinnon *et al.*, 1996; Peng, 1996a; Wang, 1997). Hong Kong has also lost almost all of its original forest cover for similar reasons (see Section 4 below).

2. Reforestation in the Tropics

2.1 Rationale

Tropical forests regulate water-flow and protect watersheds for farmers who grow food for over a billion people. Tropical forests also regulate climate, provide hardwood timber and fuelwood, and harbour untapped genetic resources that are invaluable to humankind (Lean & Hinrichsen, 1994; Mastrantonio and Francis, 1997). People who live in and near forests depend on them for much of their food, medicine, clothing and timber (Russell, 1993). Large-scale deforestation leads to the loss or significant reduction in all of these services and materials provided by the forests. The lives of those people who live nearby are the first ones to be affected. They can be deprived of vital resources of food, medicine, building materials, fuel and water. Reforestation to restore the biological productivity of forests can support people and reduce pressures for degradation of additional tropical forestlands (Lovejoy, 1985; Matyas, 1996). Sustainable forestry practices could ensure the continued survival of the people living near the forests and maintain their livelihood. Therefore, Principle 10 of the Forest Stewardship Council's Principles and Criteria states that "while plantations can provide an array of social and economic benefits, and can contribute to satisfying the world's needs for forest products, they should complement the management of, reduce pressures on, and promote the restoration and conservation of natural forests" (www.fscoax.org/html/noframes/1-2.html).

Tropical forests are rich in biodiversity. They cover only about 7 % of the earth's surface, but harbour at least half of the total number of species on earth; most of them are yet undiscovered (Lean & Hinrichsen, 1994). A typical, 1000 ha patch of tropical moist forest contains as many as 1,500 species of flowering plants, up to 750 species of trees, 400 bird species, 150 kinds of butterflies, 100 different types of reptile and 60 species of amphibians. The insects are too numerous to count. Over-exploitation, and habitat destruction, especially of tropical forests, coral reefs and wetlands, are the main causes of the current mass species extinction (Lean & Hinrichsen, 1994). Species extinction does not simply mean the loss of species, it also means the permanent loss of genetic materials that may be highly useful to the humankind, e.g. they may be the raw materials for new pharmaceutical products and wild stocks for improving food crops. To protect the existing forests in the tropics from further destruction and to restore destroyed tropical forests could slow or even stop the increasing rate of species extinction. Article 8f of the Convention on Biological Diversity states that "contracting parties should rehabilitate and restore degraded ecosystems and promote the recovery of threatened species, inter alias, through the development and implementation of plans and management strategies" (www.biodiv.org/chm/conv/art8.htm).

Carbon dioxide is the major greenhouse gas contributing to global warming. The potential impacts of global warming on soil moisture, agricultural production, sea level and biodiversity are well known (e.g. Lean & Hinrichsen, 1994; Markham *et al.*, 1993) and are not discussed here. Apart from the burning of fossil fuel, deforestation also contributes to the increase of atmospheric carbon dioxide and other greenhouse gases such as methane and nitrous oxide. The Large Scale Biosphere-Atmosphere Experiment in Amazonia has shown that 240 million tonnes of carbon were released annually to the atmosphere due to deforestation in tropical America between 1979 and 1989 (www-eosdis.ornl.gov/lba_cptec/conciseplan/cpi08.htm).

Tropical forest is currently also a very important carbon sink. A study on the changes in biomass of several hundred plots of mature tropical trees around the world between 1958 to 1996 has suggested that the average forest biomass of the surviving forest for the tropics as a whole increased substantially over the study period (Phillips *et al.*, 1998). In the Neotropics alone, the increase amounted to approximately 40 % of the terrestrial carbon sink of the whole world. Intact forests are likely helping to buffer the rate of increase in atmospheric carbon dioxide, thereby reducing the impacts of global warming.

The effects of tropical deforestation extend well beyond increasing carbon dioxide emissions. Recycling of atmospheric carbon through the biota and soil is the major control of the lifetime of carbon dioxide in the atmosphere. Decreased productivity due to deforestation causes added carbon dioxide to remain longer, reach a higher level in the atmosphere, and absorb more heat than would be the case without deforestation (Goreau, 1990). Thus, in addition to conserving energy and halting deforestation so as to control carbon emissions, reforestation is necessary to increase the rate at which carbon dioxide is removed by photosynthesis and stored in biomass and soils, so as to regulate the already seriously disrupted carbon cycle (Goreau, 1990; 1991; 1992). Article 4, 2a of the United Nations Framework Convention on Climate Change (UNFCCC) states that "each party to the Convention shall adopt national policies and take corresponding measures on the mitigation of climate change, by limiting its anthroprogenic emissions of greenhouse gases and protecting and enhancing its greenhouse gas sinks and reservoirs". In addition, Article 2 of the Kyoto Protocol of the UNFCCC calls on Parties "to take into account their commitments under relevant international environmental agreements, including promotion of sustainable forest management practices, afforestation and reforestation" (www.unfcc.de/resource/docs/1998/sbsta/inf01.htm).

Many reforestation projects for carbon sequestration have already been commissioned (e.g. www.unfcc,de/program/aij/aijact/crinor01.html; Trexler *et al.*, 1989). While it is still early to say if any of these projects are successful, scientists generally believe that the combined effort of controlling carbon dioxide emission by conserving energy, controlling deforestation and implementing large scale reforestation is the most logical strategy (Myers & Goreau, 1991; Nilsson & Schopfhauser, 1995; Trexler *et al.*, 1989).

2.2 Approach

Interest in reforestation in the tropics for environmental reasons has been growing over the last ten years (Holl & Kappelle, 1999). Three broad approaches to overcoming tropical deforestation and hastening the recovery process have been identified (Brown & Lugo, 1994; Lamb, 1994; Lamb *et al.*, 1997). Restoration refers to the attempt to recreate the original forest ecosystem by re-assembling the original complement of plants and animals that once occupied the site. Rehabilitation is the return of the converted forest ecosystem, damaged or degraded, to a fully functional ecosystem, irrespective of its original or desired final state. Reclamation means using one or more exotic species to achieve stability and productivity at derelict or very degraded forestlands without the attempt to restore any of the original biodiversity at the site.

However, despite great advances in our understanding of tropical forest ecology and management in recent years, the area of well-managed tropical forests and the rate of reforestation are insignificant in comparison with the rate of tropical deforestation (Gomez-Pompa & Bambridge, 1995). Jordan (1997) points out that restoration ecology is an immature discipline and the success of restorationists' efforts varies widely. Continued research in reforestation is needed. In fact, reforestation research in the tropics is flourishing. So far, it seems generally agreed that reforestation for the purposes of restoring the productivity of tropical forests and for biodiversity conservation will often require the use of native species (Lesica & Allendorf, 1999; Miyawaki, 1992; 1993). To promote natural regeneration by creating foster ecosystems using native plant species is considered the most cost-effective approach (Alias *et al.*, 1998; Elliott *et al.*, 1995; Forest Restoration Research Unit, 1998; Goosem & Tucker, 1995; Gomez-Pompa & Bambridge, 1995; Kartawinata, 1994; Kolb, 1993; Moline, 1999; Pone, 1997). Some studies have shown that some native tree species can be competitive with fast-growing exotic species (Butterfield & Espinoza C., 1995; Butterfield & Fisher, 1994; Islam *et al.*, 1999). Native species also have high potential for carbon sequestration (Sautu, 1997). A recent study in Costa Rica has already shown that the annual carbon sequestration values of mixed native tree species plantation are comparable to plantations of exotic species commonly grown in the tropics (Montagnini & Porras, 1998). However, on derelict tropical lands where the use of exotic tree species is unavoidable, some studies have shown that exotic plantations, if managed properly, could act as nurse crops for native species invasion (Lamb, 1997; 1998; Lamb & Lawrence, 1993; Lugo, 1992; Otsamo *et al.*, 1997; Parrotta, 1992, 1993, 1995; Parrotta & Knowles, 1999).

Degraded tropical forestlands, if left undisturbed by human activities, are subject to a range of natural filters that delay or even stop forest succession, depending on the history of disturbance and site characteristics. These natural filters to forest succession include low propagule availability, seed predation, poor germination conditions, seedling predation, seasonal drought, weed competition, and poor soil conditions (Aide & Cavelier, 1994; Holl, 1998a, 1998b, 1999; Kolb, 1993; Montagut, 1996; Nepstad, 1989; Nepstad *et al.*, 1990). Understanding the importance of these natural filters in different parts of the tropics is vital. Active reforestation using native or mixed native and exotic species, or creating foster ecosystems to accelerate natural forest regeneration, are likely to fail if the natural filters of the specific site are not well understood.

3. Reforestation in Tropical and Subtropical China

Apart from a long history of deforestation, China also has a long history of reforestation. However, forestry for environmental causes is a relatively recent theme (Li *et al.*, 1999; Zhang *et al.*, 1999). The two recent driving forces of environmental reforestation in China have been biodiversity conservation, and the control of soil erosion and flood prevention.

3.1 Conservation of biological diversity

China is one of the world's richest countries in terms of biodiversity. It is estimated that there are over 27000 species of higher plants belonging to 353 families and 3184 genera of which 190 genera are endemic (MacKinnon *et al.*, 1996). China participated at an early stage in the Biodiversity and Climate Change Conventions and became a Party when these two Conventions came out of the Earth Summit in Rio de Janeiro in 1992 (MacKinnon *et al.*, 1996). In 1994, the Chinese Government published three seminal documents concerning the environment and sustainable development. The China Conservation Strategy 1991-2000 outlined targets for a range of environmental sectors (MacKinnon *et al.*, 1996). Secondly, a National Biodiversity Action Plan was published, only two years after the Convention on Biological Diversity was signed (Anon., 1994). With respect to reforestation, Action 2 of the Fifth Goal of the Action Plan states that forestry practices that are beneficial to biodiversity conservation should be adopted. These practices include a logging ban on fragmented primary forests; promoting natural forest regeneration in reforestation; using more native species in active reforestation and planting mixed forest (Anon., 1994). On the research side, studies on restoration and rehabilitation of degraded habitats, including forest are stressed. Thirdly, China was the first developing country to issue a national Agenda 21 to guide the reconciliation of environmental protection and economic development (MacKinnon *et al.*, 1996). The priority programmes of China's Agenda 21 include 69 programmes in nine distinct groups (http://sedac.ciesin.org/china/policy/acca21/21desc.html). One of the priority programmes (Group 9-5) is on the conservation and restoration of tropical rain forest in southern China. The long-term objective of this programme is to establish networks for monitoring tropical rainforests in China and to develop models for restoring degraded ecosystems and balancing conservation and sustainable development of forests. One of the research projects, on measures to prevent soil erosion and restore damaged ecosystems in tropical rain forest in the Xishuangbanna area in Yunnan Province and Jianfengling area in Hainan Province, focuses on restoration technology and to establish pilot projects (http://sedac.ciesin.org/china/policy/acca21/219-5.html).

A lot of the goals and actions set in the Biodiversity Action Plan actually originated from the Biodiversity Working Group (BWG) of the China Council for International Cooperation on Environment and Development (CCICED). CCICED was established in 1992 to further strengthen cooperation and exchanges between China and the international community in the field of environment and development. It is a high-level, non-government consultative forum, which advises the Chinese Government at the highest level. The BWG comprises both national and international members (Xie, 1997). In all the annual reports of the BWG, better protection of the remaining forests and reforestation with diverse tree species has always been a key recommendation made to the Chinese Government (Anon., 1999; Xie, 1997).

3.2 Flood and soil erosion control

It has been estimated that as much as 50 % of the flooding along most parts of the Yangtze River in 1998 can be attributed to deforestation (Moad, 1999). This led to a direct economic loss of over 200 billion RMB (or 26 billion US dollars), which led the State Council to pass an urgent order to ban the logging of all natural forests and the change of forest lands into agricultural lands throughout the country (State Environmental Protection Administration, 1999). In addition, two Provinces in the upper Yangtze catchment, Sichuan and Yunnan, launched massive reforestation schemes in late 1998. Sichuan targets to close 6900 km^2 of forests from forestry and other types of exploitation to promote natural regeneration and establish 4200 km^2 of ecological forests between 1998-2000. Yunnan aims to plant an average of 630 km^2 of forest per year until 2010, which will make up to 8200 km^2 (State Environmental Protection Administration, 1999). Although the national ban on logging natural forests has already been put in place, logging of natural forests in some remote mountain areas in South China is still continuing (Hau *et al.*, 1999). This is attributed to the serious poverty of the mountain areas. This indicates that successful forest protection and reforestation requires continued efforts and must involve community participation. Repairing deforested and degraded forest lands is important so that their biological productivity can support people and reduce pressures for degradation of additional tropical forest lands (Lovejoy, 1985).

3.3 Approach

Silvicultural technology is well developed in China (Anon., 1983; Huang and Shen, 1993). However, forestry in China has been criticized for concentrating too much on either native or exotic monocultures (Anon., 1999; Xie, 1997). In South China, the native China Fir *Cunninghamia lanceolata* and Masson Pine *Pinus massoniana* are the two most widely planted species, often in single species plantations (Anon., 1983; Huang & Shen, 1993). Declines in soil fertility and productivity after a few rotations have been reported (Liang *et al.*, 1996). Masson Pine plantations have also been subject to severe pest problems (Wilson, 1993). In view of the problems of monoculture and the increasing concern for environmental forestry, experimental trials on mixed native and exotic species are increasing (Hao & Wei, 1999; Liang *et al.*, 1996; Peng, 1994). On the other hand, China has a long history of closing degraded forestlands from further disturbance so as to allow natural regeneration. Since 1950, closing forest to promote natural regeneration has become an important and standard forestry practice (Huang and Shen, 1993). However, this method is only suitable for sites that have not been subject to prolonged disturbance or are surrounded by mature natural forests. Such sites are becoming scarce. In recent years, a new strategy for reforesting degraded tropical forestlands in South China has been proposed. It involves firstly the establishment of a pioneer crop, such as Masson Pine, by planting seedlings or direct seeding. Once the plantation is established, the pioneer crop will be thinned and then native broadleaf species will be planted. The process will be completed by closing the forest to allow forest succession to proceed (Peng, 1996a; Wang & Peng, 1989; Xu & Liu, 1996).

It appears that scientists in China have also found promoting natural forest regeneration a cost-effective reforestation strategy on degraded tropical and sub-tropical forestlands. However, on the most degraded sites, active planting of exotic tree species is still considered necessary (Peng, 1996a; Yang *et al.*, 1995; 1999; Zhou, 1995).

4. Hong Kong

Between 1842 and 1997, Hong Kong was a self-administered British Dependent Territory. Since 1 July 1997, Hong Kong has become a Special Administrative Region of the People's Republic of China but maintains a high degree of autonomy. The legal, political and economic systems also remained more or less the same after the change of sovereignty. Since the mid 19^{th} century, the population has grown from around 3000 people on Hong Kong Island to about 6.8 million in the whole Territory today, making Hong Kong one of the most densely-populated cities in the world.

Hong Kong lies between 22°09' - 22°37'N and 113°52' - 114°30'E. It consists of a large, irregularly-shaped peninsula extending from the south-eastern coast of Guangdong Province into the South China Sea and approximately 230 offshore islands (Styles & Hansen, 1989). Large marine embayments exist on either side of the Territory, with the Pearl River estuary to the west, and Mirs Bay to the east (Figure 1.1). The total land area of Hong Kong is about 1053 km^2, excluding over 37 km^2 of land reclaimed from the sea (Styles & Hansen, 1989). The land area of the mainland (Kowloon and the New Territories) is about 788 km^2. The largest two islands are Lantau Island (145 km^2) and Hong Kong Island (78 km^2).

The topography of Hong Kong is rugged (Figure 1.2). Over 800 km^2 of the land area is considered hilly (Chiu & So, 1986). The majority of the population resides on these relatively flat lands and lands reclaimed from the sea. Substantial lowland plains are confined to the northern parts of the New Territories. Hong Kong has 30 peaks that are over 480 m above Principal Datum (Styles & Hansen, 1989). The highest peak is Tai Mo Shan (957 m) in the central new Territories, followed by four peaks ranging from 751 – 934 m on Lantau Island. The highest peak on Hong Kong Island is the Victoria Peak (554 m).

Most of Hong Kong is underlain by volcanic and volcaniclastic rocks (429 km^2 or 39.3 %) and intrusive igneous rocks (238 km^2 or 21.8 %; Styles & Hansen, 1989). In general, the bedrock materials have been subjected to severe weathering over much of the Territory (Dudgeon & Corlett, 1994). Large areas of alluvium are present in the northwest and northern New Territories (148 km^2 or 13.6%; Styles & Hansen, 1989). Small areas also occur in valley floors throughout the Territory. Dudgeon and Corlett (1994) point out that alluvial deposition has undoubtedly been accelerated in the past millennium by deforestation of the uplands and the resulting increase in erosion. Colluvial materials of different parent rock types occur over 157 km^2 or 15 % of the Territory, mainly in hill footslopes (Styles & Hansen, 1989). Dudgeon and Corlett (1994) suggest that some of the more recent colluvial deposits may be the result of landslides triggered by deforestation.

Much of the natural drainage pattern in the Territory is changed by human activities, especially in or near the urban areas (Styles & Hansen, 1989). Even outside the urban areas, channelisation of some major streams, the construction of catchwaters, dams and reservoirs, and the provision of additional drainage systems to some unstable slopes have significantly altered the natural drainage. Much of the lowland rivers and streams have been grossly polluted by livestock waste and household discharges in this century. Upland streams are often small and slow flowing but could become torrential after heavy rainstorm. Many of these streams are seasonal and dry up, except in pools, in the dry season.

The soils of Hong Kong are broadly divided into alluvial soils and hill soils. Hill soils are classified mainly as Red-Yellow Podsolic on granitic with Krasnosem on volcanic parent materials (Grant, 1983). The Red-Yellow Podsolic was later referred to as Ultisol in the nomenclature developed by the United States Department of Agriculture Soil Taxonomy in the mid 70s while the Krasnosem became Oxisols (Dudgeon & Corlett, 1994). However, Dudgeon and Corlett (1994) regard classification at this high level of generalization in the hierarchy as a great simplification of the true picture and are doubtful if any ecological significance can be attached to it. They propose that the most useful divisions should be first into granite-derived and volcanic-derived soils, and then on the basis of soil depth and, perhaps, organic content. Many granitic areas are badly eroded and support only sparse vegetation while most volcanic areas have a continuous vegetation cover. In general, hill soils are described as thin, acidic, low in organic matter and nutrient poor, with low to very low levels of nitrogen, phosphorus and calcium irrespective of the types of parent materials (Grant, 1983; Dudgeon & Corlett, 1994).

Hong Kong is more than 100 km south of the Tropic of Cancer but does not have a typical tropical climate. The absolute minimum and maximum air temperatures at the Hong Kong Observatory in Kowloon since records began were 0 and 36.1 °C (Figure 1.3). The mean annual temperature is 23 °C (1961-1990). January has the lowest mean temperature (15.8 °C) and July has the highest (28.8 °C; Figure 1.3). At least a few days every winter have temperatures below 10 °C, which is in the range known to cause chilling damage to sensitive plant species (Dudgeon & Corlett, 1994). A sea-level frost (0 °C) has only been recorded once on Hong Kong Island but frosts occur more frequently at higher altitudes and in the northern New Territories. Due to this low temperatures in the winter, local ecologists regard Hong Kong's climate as subtropical monsoon (Corlett, 1999; Dudgeon & Corlett, 1994). The mean annual precipitation at the Hong Kong Observatory is 2214 mm (1961-90) and it is highly seasonal; monthly precipitation from November through February averages < 50 mm (accounting for only 6 % of the annual total). Over 77% of the annual total rainfall falls between May and September with the highest in August (18%). The difference in rainfall across Hong Kong is great, with mean annual rainfall ranging from less than 1600 mm in the periphery to over 2400 mm on Tai Mo Shan. Typhoons regularly affect Hong Kong in the summer months, bringing strong winds and heavy rains.

5. Hong Kong Vegetation

The potential climax vegetation in Hong Kong has been suggested to be tropical semi-evergreen or evergreen broad-leaved monsoon forest, or subtropical evergreen broad-leaved monsoon forest (Thrower, 1975; Zhuang, 1993). None of the original forest exists today except tiny forest remnants on some remote and steep ravines that may have escaped or recovered from a long history of human destruction. Today, the most common vegetation types are secondary shrublands (396 km^2 or 37 % of the total land area), grasslands (177 km^2 or 16.5 %), and forests (86 km^2 or 8 %; Ashworth *et al.*, 1993). Though the date at which major deforestation occurred is uncertain, Corlett (1997) suggests that it was most likely in the period from 1300 to 1600 AD since the Chinese and European accounts of the region in the 17th century described Hong Kong's landscape as largely barren. However, it could have been centuries earlier. The earliest villages in the northern New Territories were established by the Hau clan starting from the Song Dynasty (960 – 1279 AD) in the area around Long Valley because of good alluvial plain and forested hills (Anon., 1985). The Hau and the other four major clans, Tang, Pang, Man and Liu subsequently took over the whole of the northern New Territories. The Song Dynasty probably was the beginning of a sizeable inland population that could significantly affect the natural environment in Hong Kong. This agrees with Corlett's (1997) proposal on the approximate time of major deforestation.

Clearance for cultivation, cutting for firewood and charcoal, and hill fires are believed to be the main cause of deforestation in Hong Kong (Daley, 1975; Jim, 1986; Dudgeon and Corlett, 1994; Thrower, 1975). With the rapid economic advancement and the gradual decline in agriculture in the last two decades, only hill fire remains an important threat. In the last eight years, there were 1,083 hill fires in Country Parks burning over 0.7 million trees (www.info.gov.hk/afd/afdparks/zcpstat.htm). In recent years, urban development in the rural areas became an additional threat in the lowlands (Webb, 1993; Territory Development Department, 1995; Chong, 1996).

6. Reforestation in Hong Kong

Corlett (1999) reviewed the forestry history in Hong Kong from 1871 to 1996. This review points out that Hong Kong may have been the first to start afforestation for purely protective reasons in the tropics but that afforestation in Hong Kong does not appear to be a great success. Tens of millions of trees have been planted but most of the existing forest cover consists of spontaneous secondary forests (about 8% of the land area) that have developed after 1945. Most of the Hong Kong vegetation was cut for fuel during the Japanese Occupation between 1942 and 1945 (Zhuang & Corlett, 1997). These secondary forests are dominated by *Machilus* spp., which have not been planted in significant numbers until very recently (Corlett, 1999). Today, only about 62 km^2 or 5.8 % of the land area is plantation woodlands (Ashworth *et al.*, 1993). Moreover, most plantations in Hong Kong are monocultures and the invasion of native woody species, especially in young plantations, is significantly impaired by management practices such as weeding (Zhuang, 1997).

6.1 A review of Hong Kong forestry history with respect to afforestation methods

The review of Corlett (1999) concentrates on the changes in planting policies and species choice. Here, I review the changes in afforestation methods in the Hong Kong forestry history. The Colonial Government started afforestation on Hong Kong Island in 1872 (Flippance, 1940). Soon after the New Territories were added to the Colony in 1899, afforestation gradually extended to the New Territories (Appendix 1.1). However, over the period between 1871 and 1940 before World War II, a much larger afforestation effort had been put on Hong Kong Island than Kowloon and the New Territories.

a. 1871-1880

In this period, afforestation was accomplished by planting bare-rooted seedlings of both native and exotic species raised in nurseries (Ford, 1880). Seeds were sown in prepared ground in two nurseries, one in Kowloon and one on Hong Kong Island. Seedlings were allowed to grow for a year after germination. They were then lifted from the nursery ground, taken to the planting sites and planted in prepared pits. Ford (1880) says that afforestation in these 10 years was not very successful. Seventy five percent of the trees planted in the afforested areas died. Most of the surviving trees appeared very sick and had little promise of surviving. The failure was attributed to poor nursery practices, post-nursery care and planting skills. Seedlings were not carefully lifted from the nursery ground so that the roots were severely injured. The roots were not properly protected during transportation and transplanting, so that most seedlings died soon after being planted. Some planting holes were not filled-in with enough soil and some seedlings were planted too deep.

b. 1880-1940

Direct seeding experiments were initiated in the late 1870s involving mainly the native pine *Pinus massoniana*, although a few other native and exotic species were also tried (Ford, 1883). The results were satisfactory for pine on sites with good soil and the scale of direct seeding was gradually enlarged. All the suitable sites on the northern side of Hong Kong Island were earmarked for direct seeding in 1883. However, Ford (1883) noted that on south-facing slopes, direct seeding appeared less promising due to the stronger drying influence of the sun. In addition, on steep slopes heavy rains tended to wash away the loose soil together with the seeds. Direct-seeding was first done by spot sowing in prepared pits but in 1883 and 1885, experiments were conducted on broadcast sowing pine seeds on hillside grasslands that had no ground preparation (Ford, 1887). The results were successful and the scale of this method was gradually enlarged in subsequent years (Ford 1889). Throughout this period, direct seeding of *P. massoniana* seeds was the main afforestation method (Appendix 1.1). Flippance (1939; 1940) noted that although germination was a little variable, much better results were obtained from broadcast than from spot sowings.

Pit planting of bare-rooted seedlings of both native and exotic tree species was used on poorer sites where direct seeding was not appropriate (Ford, 1883). However, Flippance (1939) noted that direct seeding of *Pinus massoniana* was gradually found more effective than planting bare-rooted pine seedlings raised in the nursery. Thus, the latter method was discarded for pine trees but retained for broad-leaved tree species, which, in contrast, were generally not successful by direct seeding (see also Appendix 1.1).

Ford (1892) noted that after many years of afforestation, the most suitable lands on Hong Kong Island (i.e. with good soil, water and shelter) for tree growth had been filled up and the difficulty of carrying on planting was much greater. The planting scale on Hong Kong Island was gradually reduced from the early 1890s. On the other hand, natural regeneration was reported to be making considerable progress. However, natural regeneration was not mentioned again in subsequent forestry reports.

In 1907, an experiment was conducted with a small number of *Castanopsis fissa* seedlings that had been raised from seeds sown in pots, i.e. planting container-grown seedlings. The seedlings were planted out in spring like bare-rooted *P. massoniana* seedlings on open ground in Pokfulam Road. The results were negative and the method was said to be unsuitable (Dunn, 1908). Except in 1908, this planting method was no longer mentioned in subsequent forestry reports in this period. On the contrary, it appeared in various forestry reports throughout this period that only bare-rooted seedlings were used in afforestation. Daley (1975) also indicated that until the 1950s, planting bare-rooted seedlings was the usual afforestation method.

A forestry licence system was developed in 1904 in the New Territories to allow village farmers to reforest their hillsides (Dunn, 1905). This indicated that Chinese farmers also planted trees but the planting method and number were not mentioned in any of the forestry reports. However, Flippance (1939) noted that the licensed forest lots in the New Territories generally produced a sparse cover of stunted pine trees due to the local custom of removing side branches for fuel.

Starting from 1883, exotic and native species seedlings were planted in some of the older pine plantations to increase the species diversity (Ford, 1884).

c. 1945-1999

Afforestation restarted immediately after the war mainly by broadcast sowing as nursery stock for planting was not yet available (Daley, 1975). Planting container-grown seedlings were soon introduced and quickly became standard practice due to higher survival rates and lesser dependence on weather conditions (Corlett, 1999; Daley, 1975). The decline in reliance on *Pinus massoniana*, due to its susceptibility to fire damage and the occurrence of two serious new pests (Corlett, 1999) contributed to the disappearance of direct seeding as an afforestation method in Hong Kong.

Experiments with chemical fertilizers were started soon after the war and their use, along with chemical pesticides became routine (Corlett, 1999). However, weeding was increasingly done by mechanical rather than chemical means in recent years due to rising environmental concerns (personal observation). Ever since afforestation started, light pruning, adding fertilizers and replacement planting were provided in the first few years after seedlings were planted. Since the 1960s, the annual number of trees planted has increased from around 300,000 to over a million in the 1990s (Corlett, 1999).

d. Site selection and nursery location

Site selection was not specifically mentioned in any of the forestry reports. However, for the period before World War II, it was obvious that the aim of afforestation was to cover all the barren hillsides on Hong Kong Island. Although not explicitly stated in the forestry reports, it is reasonable to assume that the Colonial Government regarded Hong Kong Island as more important than Kowloon and the

New Territories during that period. Though catchments were avoided before 1900, sites in the water catchments of reservoirs and burnt hillsides were also selected as the targets of afforestation both before and after the war. Since the 1980s, degraded lands such as quarries for fill materials (locally called 'borrow areas'), old rock quarries and closed landfills, have also become the focus of afforestation projects.

During the pre-war period, the provision of tree nurseries near the major afforestation areas was a normal practice. This had the advantage of reducing the cost of transportation and the stress to seedlings during transplantation. However, with the increasing reliance on planting container-grown seedlings after the war, a centralised nursery was formed. Today, most afforestation projects in Hong Kong rely on the supply from the nursery managed by the Agriculture and Fisheries Department, which is able to produce up to 1 million seedlings per year. Commercial nurseries also serve as a major source of exotic tree seedlings but have, so far, played a minor role in supplying native tree species in local afforestation.

e. Contractor involvement

Up to 1882, afforestation was organized by less than 10 forestry staff. Seed collection and transplanting was done by "coolies" (i.e. casual labourers) on a daily basis. With the increase in afforestation scale, the government started contracting out seedling supply and planting work in the early 1880s (Ford, 1883). In 1886, the planting operations were carried out by five different contracts: seed supply, nursery production, making tree pits, planting tree seedlings and direct seeding (Ford, 1887). The frequent failures of contractors to fulfill the planting contracts forced the Botanical and Forestry Department to take over the large forestry operations previously done by contracts in 1907 (Dunn, 1908). In the transition forestry year 1906-1907, the part of the forestry programme carried out by contractors largely failed while that done by the department was most satisfactory. None of the forestry reports between 1908 and 1940 mentioned forestry contracts. Apparently, the Botanical and Forestry Department employed large numbers of temporary workers to accomplish afforestation after 1908. For example, the average number of daily temporary employees was 58 in 1938 and 116 in 1939 (Flippance 1939; 1940).

Contractor involvement in afforestation resumed after the war. Since the 1980s, the majority of the afforestation projects, especially those managed by the Territory Development Department, have been carried out by contractors. Normally, an afforestation contract will include the supply of seedlings (if seedlings are not to be supplied by the Government nursery), the delivery of seedlings to the planting site, site preparation, seedling transplant, and post-planting maintenance, which lasts for 1-3 years (Webb, 1993; personal observation). The performances of contractors nowadays are also questioned.

As a result of my involvement with the Kadoorie Farm and Botanic Garden in Hong Kong, I was able to follow a planting contract from May to July 1999. The contract involved planting 40,000 exotic trees (20,000 *Acacia confusa*, 10,000 *Eucalyptus robusta* and 10,000 *Lophostemon confertus*) and 10,000 native trees of 17 different species at a hillside site on South Lantau Island that had been burnt in early 1999. All exotic trees were obtained from a commercial nursery in China and all native trees were obtained from the native tree nursery of the Kadoorie Farm and Botanic Garden. The contractor is one of the approved government contractors.

The contract has very detailed clauses with respect to seedling specification, post nursery care, the planting method and post-planting maintenance (Appendix 1.2). The contractor must attain a 75% survival rate after a year, except for damage that the contractor is not able to prevent. In reality, a lot of the clauses were not met. Since the contract had been open for tender and the lowest tender was picked, the contractor had to minimize the operation cost in order to make a profit. For example, in seedling delivery, in order to maximize the number of seedlings in each truckload so as to reduce the number of trips between the nursery and the planting site, seedlings were put in baskets in several layers and lots of seedlings were damaged or killed during transportation. The daily temporary workers employed by the contractors failed to carry seedlings in the correct way. They lifted and carried seedlings by the stems, leading to damage to stems and roots. Seedlings were delivered to the planting site in two equal batches. The second batch was delivered after the first batch had been planted. Due to the absence of water sources and vehicle access at the planting site, seedlings that were delivered to the planting site received no water prior to being transplanted. The second batch of seedlings, especially the native species, suffered from severe dehydration since they were left under the sun for two dry days.

A lot of the seedlings were not planted properly as specified in the contract conditions. Some were planted not deep enough while some were planted too deep. Around 100 seedlings were found not planted, just left in the planting hole and died. Some of the seedlings that were killed or damaged during transportation were still planted. This was again attributed to the need to minimise costs. Ten workers worked for about 8 days to finish the planting (seedling delivery excluded). For 50000 seedlings, each worker planted 625 seedlings per day, including pit preparation and adding fertilisers. For working 8 hours a day, each worker thus planted 1.3 seedlings per minute, non-stop in these 8 hours. Under such a tight schedule, it was almost certain that seedlings were not planted properly. Despite the fact that the contractor has to replace the dead seedlings caused by poor craftsmanship (see Appendix 1.2), no replacement planting has been carried out.

One thousand native tree seedlings in four quadrats were tagged immediately after transplantation for long term monitoring. Survival in one month after planting was 96.5% for the first batch and 88.3% for the second batch. However, defoliated seedlings and seedlings with broken stems or yellowing leaves were recorded as sick and this was up to 35% for both batches. More seedling mortality is envisaged in the first winter dry season.

Although this single event cannot represent all afforestation contracts in Hong Kong, it does highlight some of the problems associated with the involvement of contractors in afforestation projects, especially with the tender system in Hong Kong whereby contracts will always be awarded to the lowest tender. Corlett (1999) points out that a striking feature of Hong Kong's forestry experience has been the small number of tree species that has proved amenable to mass planting on degraded hill slopes. This may be attributed to the fact that many tree species simply are not able to get through the planting shock. Many tree species, if planted properly, may be proved amenable.

f. Conclusion

An obvious hole in the reforestation history in Hong Kong is the lack of scientific research. Very few studies are available in the literature concerning forest restoration ecology in Hong Kong. Most planting trials in the past were not adequately documented (Corlett, 1999). Despite the trials of different afforestation methods, none of them seem to be very cost-effective under present economic conditions, i.e. in a developed, high-wage economy. Creating foster ecosystems to accelerate natural forest regeneration has been put forward as a cost-effective method in the tropics as well as in inland China (see Section 2 & 3 above) but this is yet not been considered in Hong Kong. A comprehensive reforestation strategy, combining fire prevention, active reforestation by various means, and practices to accelerate natural forest regeneration, is currently lacking. This may be partly attributed to the fact that reforestation in Hong Kong involves more than two different Government Departments (Agriculture and Fisheries Department, Territory Development Department and to a lesser extent, Highways Department and Civil Engineering Department) belonging to different Bureau.

7. Forest Succession in Hong Kong

Until the mid 1970s, the harvesting of biomass (mostly grasses, ferns and small shrubs) for domestic fuel was a major factor preventing forest succession on uncultivated hillsides in Hong Kong (Chen, 1993; Chen, Corlett & Hill, 1996). This has now stopped in Hong Kong but still produces "shaven", treeless hillsides over large areas of rural Guangdong. The cessation of biomass harvesting has probably increased the impact of hill fires, both by increasing the fuel load and by reducing the incentive for rural people to prevent fires. A study of the relationship between fire and vegetation in Hong Kong found that grassland persists largely in areas which have been burned in the last 10 years, and is replaced by shrubland in the absence of fire (Chau, 1994). The most frequently-burnt areas are occupied by grasslands and grasslands are the most frequently-burnt. On favourable sites, *Machilus*-dominated secondary forest, 10-15 m in height, can develop in a further 30-50 years (Zhuang, 1993; Zhuang & Corlett, 1997), but forest succession is apparently much slower on sites which are remote from tree seed sources or where the soil has been highly degraded (Wong, 1999; Zhuang, 1997). The only older forests in Hong Kong, apart from some tiny patches in topographically-protected montane sites, are fung shui woods, behind active or abandoned villages (Zhuang & Corlett, 1997; Chu & Xing, 1997). However, these are not simply the product of forest succession, and include a variable proportion of planted species. *Machilus*-dominated secondary forest supports both a much higher bird density and a higher diversity of forest-dependent bird species than monoculture plantations of *Lophostemon confertus*, the most widely planted tree species (Kwok & Corlett, in press). It also typically contains 16-45 tree species in a 400 m^2 plot (Zhuang & Corlett, 1997). However, the majority of the c. 400 native tree species in Hong Kong are confined to upland sites which were, apparently, never completely cleared, and have not succeeded in invading secondary forests (Zhuang & Corlett, 1996).

8. Research Objectives of this Study

Despite massive planting effort over more than 120 years, the forest cover in Hong Kong remains low and most existing forest has arisen by secondary succession rather than planting. This is also true for the tropics as a whole, where plantation cover was only 0.8 % of the natural forests in 1990 (World Resources Institute, 1999). Natural succession is a cost-effective approach to forest restoration in Hong Kong (Zhuang, 1997), inland China and other parts of the tropics. It often results in forests with a more diverse flora and fauna than in planted forests. However, relying solely on forest succession to reforest Hong Kong and the rest of the tropics is not the answer.

Firstly, the rate at which woody vegetation develops on cleared sites is related to the duration and intensity of disturbance (Nepstad *et al.*, 1996; Hughes *et al.*, 1999). On highly degraded sites, such as most non-forest areas in Hong Kong, succession can be very slow. In grassland areas, this means that there is a long period of vulnerability to fire (over 10 years in Hong Kong), before the fire-promoting grasses are suppressed by a closed, woody canopy. Secondly, most of the various biotic and abiotic processes, which control the rate of succession, are highly selective. As a result, only a subset of the mature forest flora occurs in secondary forests even after several decades of succession (Finegan, 1996; Turner *et al.*, 1997; Ferreira & Prance, 1999). This is particularly apparent in Hong Kong, where most of the original forest remnants are confined to remote upland ravines and the dispersal agents of forest dominant species, such as the Fagaceae, are apparently lacking (Dudgeon & Corlett, 1994). Forest succession is thus more successful at restoring forest biomass than floristic diversity, and more successful on less degraded sites.

It seems likely that we will have to rely on succession as the main means of forest restoration in the tropics, so it is important that we understand the factors which limit its rate and control the species composition of the resulting secondary forest. If these barriers – or, more accurately, filters - to succession can be understood, it may be possible to accelerate and diversify succession over large areas at a much smaller cost than that required for artificial planting. Even if this does prove possible, understanding the factors that control natural succession will help formulate strategies for the use of native tree species in active afforestation.

This project is aimed at investigating the limitations to natural forest succession in hillside grassland of Hong Kong. Nepstad *et al.* (1990) illustrate the pathways by which trees can establish in the abandoned pastures in the Amazon (Figure 1.4) and the same basic principles must apply in Hong Kong. However, the duration of disturbance in degraded hillside sites in Hong Kong is much longer than in the abandoned pastures in the Amazon. As a result, most grassland sites in Hong Kong have long ago lost the tree roots and soil seed bank present in young pastures. Seeds dispersed from nearby forests, if there are any, will be the only source of tree propagules. Seed trapping experiments were thus conducted to assess seed dispersal into degraded hillside sites in Hong Kong. Birds are usually the most important seed dispersers (Stiles, 1989; 1992). It is also true in Hong Kong since most of the native trees, shrubs, and climbers rely on birds for seed dispersal (Dudgeon & Corlett, 1994; Corlett, 1992a). More than 85% of the trees, shrubs and climbers of hillside shrublands in Hong Kong bear fleshy fruits that are of the right sizes and colours for dispersal by birds (Corlett, 1992b). Isolated trees in abandoned pastures in Brazil play a significant role in attracting bird dispersers (Kolb, 1993), so the role of tree islands in seed dispersal in degraded hillsides in Hong Kong was also studied (see Chapter 2).

Seeds are concentrated packets of food and are consumed by many animals, especially ants, rodents and birds (Nepstad, 1989). Deforested lands are usually overgrown by invasive grass and shrubs, which in turn provide excellent habitats for seed-predating ants and rodents (Pone, 1997). Seeds dispersed into hillside grasslands *in Hong Kong may be subject to seed predation. Seeds of different tree species differ* in terms of size, texture, and nutritional contents etc., and thus vulnerability to seed predators. Tree seed predation experiments and animal trapping were conducted to test if this is a significant factor in delaying forest succession on degraded hillside sites and to identify the seed predators. The experiments also attempted to test what types of seeds were more vulnerable (see Chapter 3).

Despite the fact that the Barking Deer is common in Hong Kong, and considered a threat to forest regeneration elsewhere, seedling predation is apparently not an important factor on degraded hillsides in Hong Kong. No widespread seedling damage has been noted in Hong Kong Forestry Reports. However, domestic cows, which have been allowed to run wild since the decline in agriculture in the last decade, were reported to destroy some tree seedlings planted by the government. No cows occurred at any of my study sites. Rats were also reported as damaging tree seedling planted on Lantau Island.

Those tree seeds that have escaped from seed predators may not be able to germinate due to the extreme physical environment of the degraded grasslands. However, different tree species are likely to have.different ability to germinate in such conditions. Seed germination experiments were carried out at degraded hillside sites to test the performances of selected common native tree species (see Chapter 4).

The survival and growth of germinated seedlings before tree establishment is complete is the longest stage in forest succession. Immediately after germination, tree seedlings will be faced with new filter-barriers to survival. Seasonal drought, poor soil nutrients and above and below ground competition are the factors affecting tree seedling survival and growth in the tropical grasslands (Holl, 1999; Nepstad, 1989). These factors may well be important in Hong Kong. A seedling transplant experiment was thus conducted in a typical degraded grassland site to study the effects of grass competition, seasonal drought and poor soil nutrients on the initial survival and subsequent growth of four common native tree species (see Chapter 6). A wider range of species was used in an additional field transplant experiment to test the survival and growth of these species in degraded grasslands and shrublands in Hong Kong (Chapter 7).

Finally, a survey of woody species in typical hillside shrublands in Hong Kong was conducted to identify the initial successful woody invaders of degraded grassland sites (Chapter 5).

Figure 1.1

Hong Kong in its regional setting.

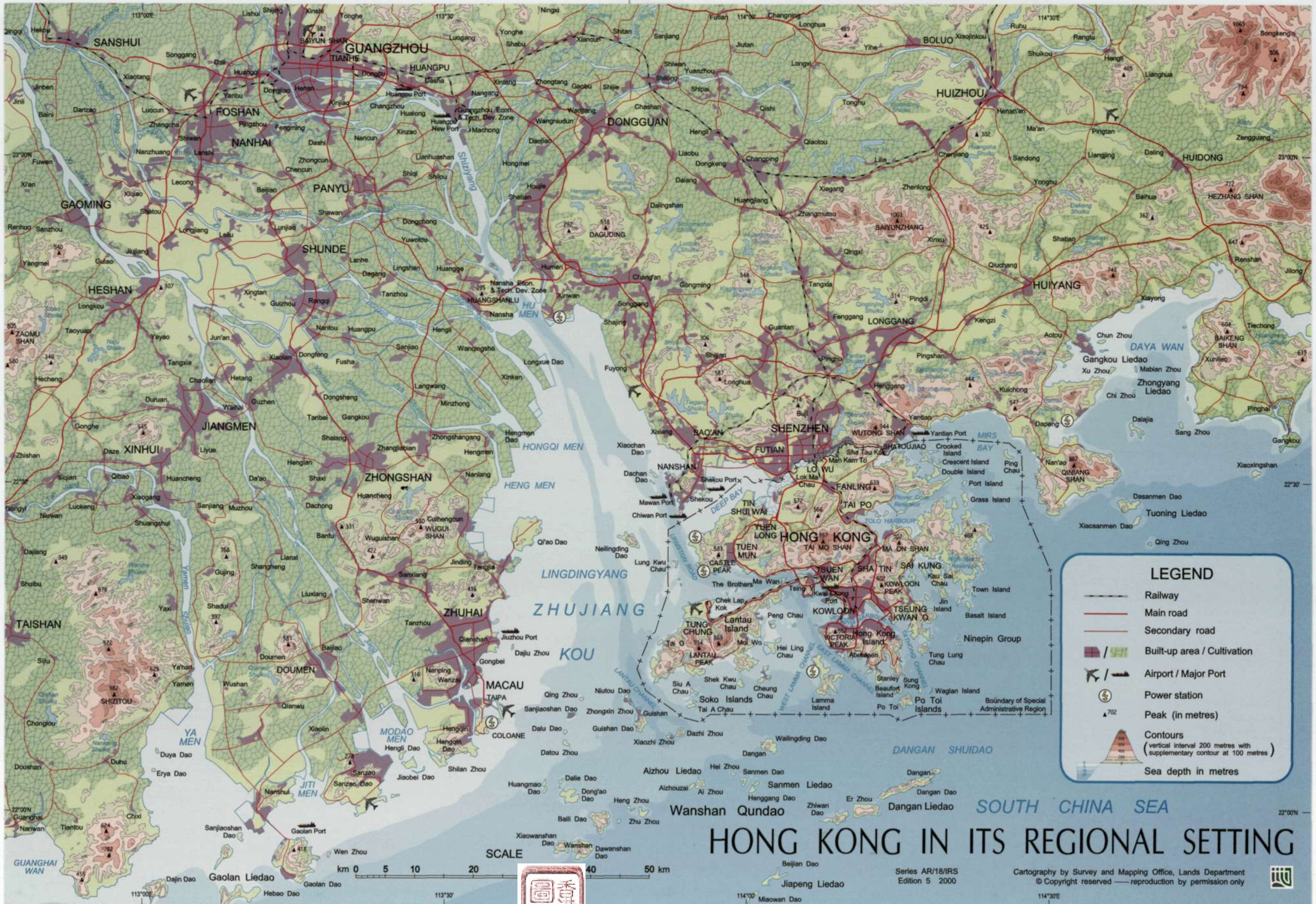


Figure 1.2

Hong Kong Special Administrative Region.

Figure 1.3

Mean monthly temperature (lines) and total rainfall (bars) of 1961 – 1990 at the Hong Kong Observatory, Kowloon.

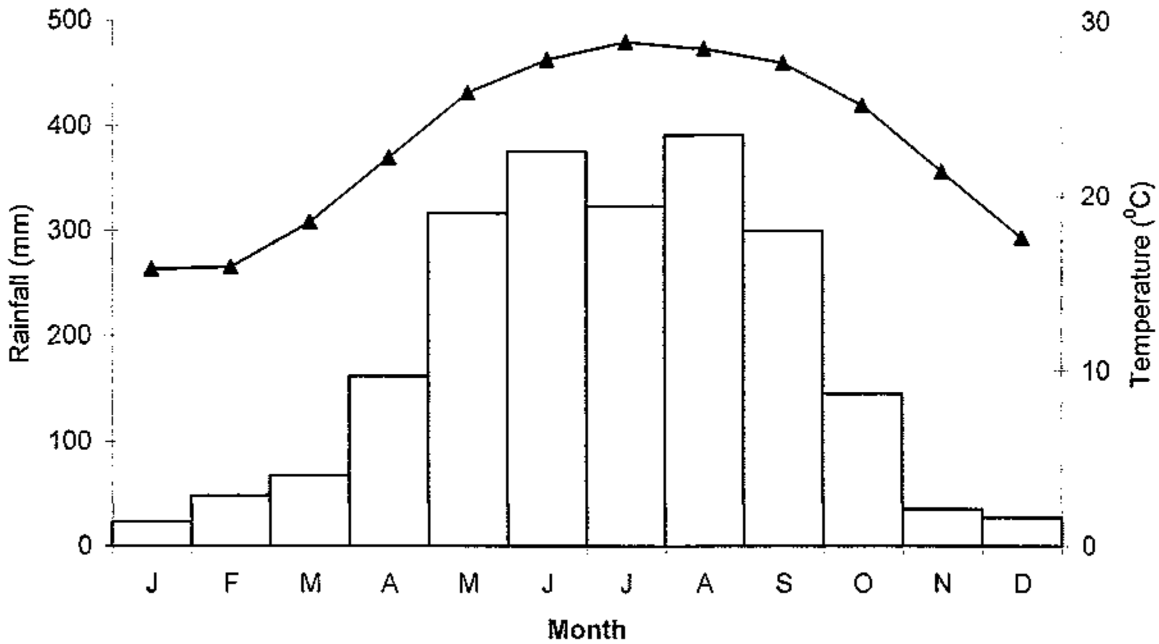
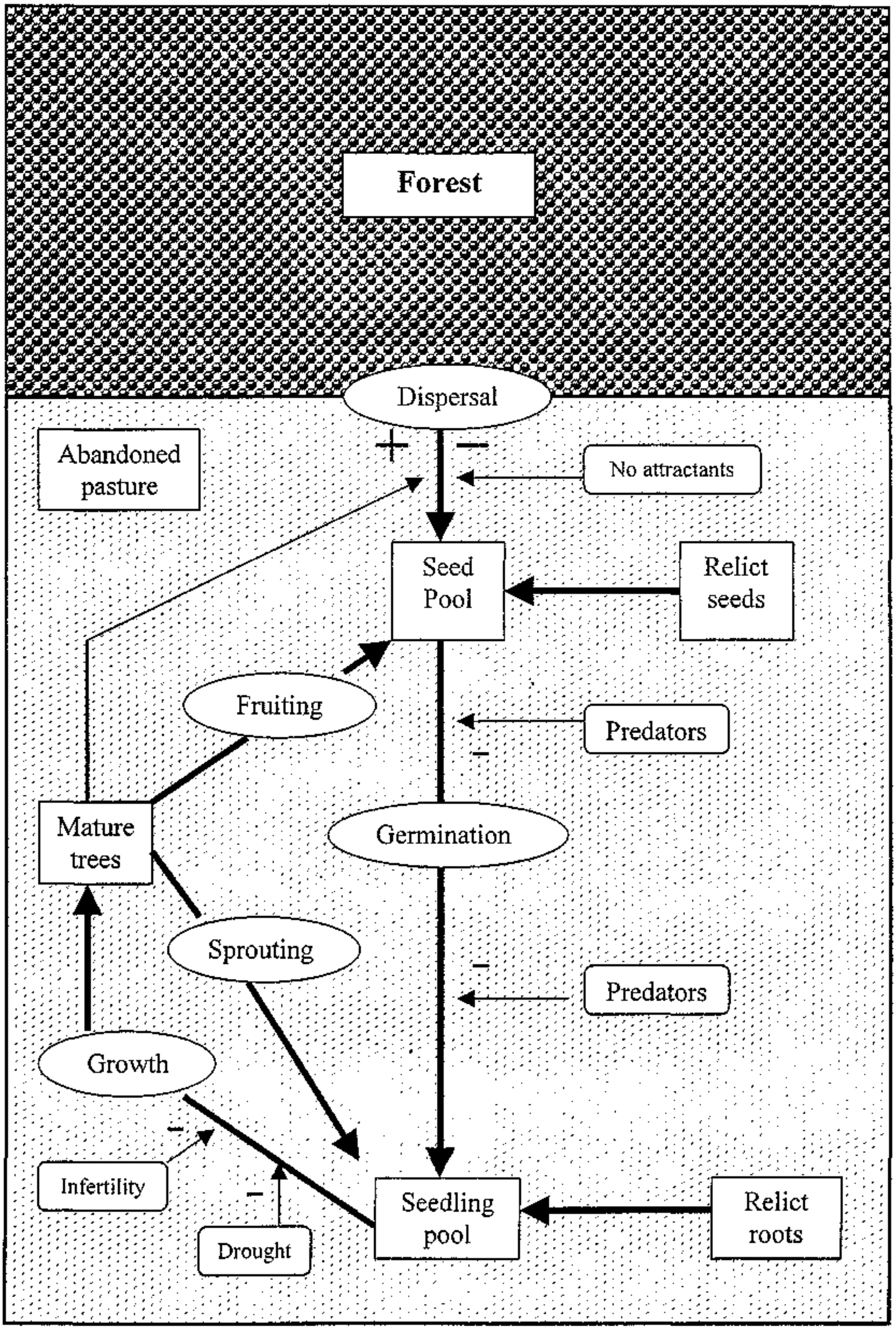


Figure 1.4

Mechanisms of tree establishment in an abandoned, grass-dominated pasture, Paragominas, Brazil.
Source: Nepstad *et al.*, 1990



Chapter 2

Tree Seed Dispersal on Degraded Hillsides in Hong Kong

Abstract

A seed trapping study was conducted in a hillside grassland and shrubland in Hong Kong for one year. Seed traps were set in the open and under treelets at both sites. A total of 2,417 seeds of 17 tree species, 10,097 seeds of 14 shrub species, 132 seeds of 5 species of climbers and 78 seeds of 5 herbaceous species were collected. All woody seed species collected are native to Hong Kong except *Lantana camera*, which is a naturalised exotic. Far more seeds (>94%) were collected by seed traps set under treelets at both sites. This result agrees with many studies in other parts of tropics that the availability of perches would make a site more attractive to seed dispersers, especially frugivorous birds. Surprisingly, two rat species were also found to disperse seeds in this study. Sladen's Rats, *Rattus koratensis*, were responsible for 56.6% of the seed collected from shaded traps on the grassland site. The Chestnut Spiny Rats, *Niviventer fulvescens*, were responsible for 0.35 % of the seeds trapped at the shrubland site. All seeds dispersed by rats were small in size and only one, *Ficus fistulosa*, out of the 6 woody species was a tree. Predation on seeds in the traps was observed three times at the shrubland site only.

1. Introduction

There are a number of ways that trees could establish on degraded hillsides in Hong Kong. Tree roots that persist in the soil following deforestation can give rise to new stems by sprouting. Tree seed in the soil seed bank is another possible source of seedlings. However, like some abandoned pastures in the Amazon region (Nepstad *et al.*, 1990), most of the grasslands in Hong Kong have been deprived of relict roots and seeds in the soil after repeated fires over a long period of time. As a result, forest succession at these very degraded sites has to rely on seeds dispersed from nearby forests to start. Holl (1999) shows that the lack of seed dispersal is the most important barrier to natural forest regeneration in a recently abandoned rainforest pasture in southern Costa Rica. This study aims to assess tree seed dispersal on degraded hillsides in Hong Kong and evaluate if it is an important barrier to natural forest succession.

Seeds can be dispersed naturally to a site in many ways (Ridley, 1930; van der Pijl, 1982). However, the usual means of seed dispersal are by wind, water and animals (Wunderle, 1997). Wind is an effective mean of seed dispersal in the temperate regions. Wind-dispersed plants normally fruit when wind is most available and predictable. For example, the maples (*Acer* spp.) of eastern North America disperse seeds either immediately before the leaves have fully expanded in Spring or after leaf fall in the Autumn (Stiles, 1989). However, wind is less predictable in the tropics and in the understories of most plant communities (Stiles, 1989). For most of the tropical woody species, wind dispersal is inefficient due to large seed sizes. Few of the winged diaspores could travel more than 15 m from the parent plants. The wind-dispersed shrubs and trees in Hong Kong, such as *Gordonia axillaris*, tend to have highly-clumped distributions where they occur (e.g. Victoria Peak, Chung Hom Kok and Ma On Shan) and are absent from large areas of apparently suitable habitat elsewhere (Dudgeon & Corlett, 1994). However, wind-dispersed species with very small seed size such as, *Rhododendron* spp., *Itea chinensis*, *Liquidambar formosana* and *Cratoxylum lingustrinum* could be dispersed over a much longer distance. Nevertheless, wind dispersal is not important in natural forest regeneration at those sites that are not close to the seed source.

Water is an effective vector only in coastal areas where tides and currents move seeds far away from the parent plants. However, freshwater could also be an effective vector under certain circumstances, such as tropical rivers that predictably flood, and seeds may well be dispersed within the flood plain (Stiles, 1989). For the degraded hillsides in Hong Kong, water is not at all an effective vector for tree seed dispersal.

Animal seed dispersal is the predominant form of dispersal in the tropics (Wunderle, 1997). Primary animal seed dispersers are ants and vertebrates. Birds and then mammals are the most important vertebrate seed dispersers in the wet tropics (Stiles, 1989, 1992; Wunderle, 1997). Apart from dispersing seeds from the parent plants, certain animal seed dispersers could enhance the germination probabilities of the dispersed seeds. However, this is not a widespread phenomenon and most animal-dispersed seeds apparently receive no or very little advantage in seed germination (Wunderle, 1997). Fruit availability is highly seasonal in Hong Kong. Fruit diversity and abundance reaches a maximum in December/January (Corlett, 1993). Twenty-seven percent of the native angiosperm flora of Hong Kong bears fleshy, presumably vertebrate-dispersed fruits. These include 76% of the 377 tree and shrub species and 70% of the 103 climber species (Corlett, 1996). However, Hong Kong's potential disperser fauna has been greatly reduced by deforestation and hunting (Dudgeon & Corlett, 1994).

Birds are usually the most important vertebrate seed dispersers (Stiles, 1989, 1992). In Hong Kong, most of the native trees, shrubs, and climbers rely on birds for seed dispersal (Dudgeon & Corlett, 1994; Corlett, 1992a). More than 85% of the trees, shrubs and climbers of hillside shrublands in Hong Kong bear fleshy fruits that are of the right sizes and colours for dispersal by birds (Corlett, 1992b). Fruits consist of seeds and pulp, and frugivorous birds handle them in a variety of ways (Herrera, 1985). Frugivorous birds refer to those that consume the fruit pulp and/or the seeds for food. Some species, like certain finches, concentrate on seeds and discard the pulp. Others ingest the pulp and seeds together, cracking the seeds in the bill or gizzard (e.g. parrots, most pigeons, and finches). Some others, such as tits, sometimes consume the pulp but discard the seeds. The last and the most important group of frugivorous birds as dispersal agents are species that swallow whole fruits, later defecating or regurgitating the seeds intact. These are the genuine frugivores as they maintain a mutualistic relationship with their food plants and have been ultimately responsible for the evolution of fruit producing plants (Herrera, 1985).

In Hong Kong, at least 35 species of resident birds and 40 species of migrants probably eat at least some fruits (Corlett, 1996). The commonest resident frugivorous birds on Hong Kong hillsides are the Light-vented Bulbul (*Pycnonotus sinensis*), Red-whiskered Bulbul (*P. jocosus*), and Japanese White-eye (*Zosterops japonicus*). Other common frugivorous birds include the Hwamei (*Garrulax canorus*), and various laughingthrushes (*Garrulax* spp.) as well as a number of winter visitors that eat some fruits (thrushes, *Turdus* spp., and robins) (Corlett, 1992b, 1998a). Some flycatchers (*Ficedula mugimaki* and *F. zanthopygia*) and the Chinese bush warbler (*Cettia canturians*) also consume a small amount of fruit.

The availability of perch sites, fleshy fruits and the structural complexity of the vegetation tend to affect the attractiveness of a site to animal seed dispersers, especially, avian seed dispersers (Holl, 1998c; Stiles, 1992; Wunderle, 1997). Many studies have demonstrated that seed rain beneath perches is significantly higher than in nearby sites without perches (Debusche & Isenmann, 1994; Guevara *et al.*, 1992; Kolb, 1993; McClanahan & Wolfe, 1993; Nepstad *et al.*, 1991; Willson & Crome, 1989). The abundant seed input under perches can be related to the observation that most seed defecation and regurgitation by avian frugivores occurs when birds perch or immediately after they take off, rather than during flight (Guevara & Laborde, 1993).

The presence of fleshy fruits in a site tends to attract more avian seed dispersers, which in turn bring in other seed species. Seeds carried into the abandoned pasture in the Amazon are concentrated beneath fruiting treelets (Nepstad, 1989). The observations of Nepstad *et al.* (1990) show that seeds are concentrated beneath shrub-like lianas and trees that produce fruits throughout the year. Levey (1988) found that the abundance of fruit-eating birds in lowland Costa Rican rain forest followed the same general patterns of spatial and temporal variation in fleshy fruit abundance. Kolb (1993) found that seed flow into abandoned pastures in tropical America was positively correlated with fruit availability on island vegetation and negatively correlated with spatial isolation. However, Toh *et al.* (1999) show that whether or not a tree offers a fruit reward appears less important than its structure and suitability as a bird perch. In a bird perching study in abandoned pasture in Costa Rica, artificial perches baited with banana did not increase either bird visitation rates or seed rain (Holl, 1998c).

Structurally complex vegetation has been demonstrated to be attractive to avian seed dispersers in studies of old field succession (Wunderle, 1997). Structurally complex vegetation cover would have either or both of the above two site traits that attracts avian seed dispersers. In addition, it provides more refuges to avian seed dispersers from predators. Parrotta (1992) attributed the much higher species richness of woody seedlings and vines in a 4.5-year-old *Albizia lebbeck* plantation than the adjacent control area with no tree cover in Puerto Rico to the fact that canopy development in the plantation indirectly increased propagule availability by providing roosting and nesting sites for a variety of birds. Kollmann (1995) found that there was an increasing gradient of seed rain with progressive shrub development and successional time in a dry grassland with interspersed shrubs in central Europe. Holl (1998) showed that branch perches had significantly higher bird visitation rates and seed rains than crossbar perches placed in abandoned pasture in Costa Rica.

Mammal seed dispersers are functionally divided into four groups: bats, primates, rodents and others (Stiles, 1989). In the tropics, frugivorous bats rival birds as seed dispersers. They have easy access to ripening fruits, minimizing the time ripe fruits remain undispersed and maximizing the movement of seeds away from the parent plants (Stiles, 1989). Fruit bats are widespread and diverse in the tropics (Corlett, 1998b; Ridley, 1930; Stiles, 1989). However, the ecology of most species and their roles as seed dispersers are not well studied (Corlett, 1998b).

The fate of seeds dispersed by bats depends on bat, fruit and seed characteristics (Corlett, 1998b). Fruit bats squeeze the fruit between the tongue and the palate, consume the juice and discard all but the smallest seeds and pulp as a fibrous wad (Corlett, 1998b; Phua & Corlett, 1989; van der Pijl, 1982; Stiles, 1989). Larger bat species e.g. *Pteropus*, tends to process fruit in the fruiting tree. Smaller species e.g. *Cynopterus*, will bite off pieces of the flesh of large fruits in situ or carry either a single or several fruits (depending on fruit sizes) to a feeding roost (Corlett, 1998b). Large seeds are usually dropped under feeding roosts. Small seeds will be dropped under the feeding roost in the fibrous wad or swallowed and defecated in flight or in a roosting site. Fruits with semi-fluid interiors or very slippery seeds tend to have a higher proportion of seeds swallowed by bats. Fruit bats can move seeds for long distances but the majority of seeds are not carried far. Fruit bats usually disperse seeds within the range of 20 to 300 m from the fruiting tree (Corlett, 1998b; Phua & Corlett, 1989; Ridley, 1930; van der Pijl, 1982). By and large, fruit bats usually disperse relatively large seeds to those sites not suitable for tree establishment and growth e.g. under the fruiting tree or another tree. A higher proportion of small seeds (e.g. <2.4 mm diameter for 35 g *Cynopterus* in Singapore, Phua & Corlett, 1989) will be swallowed and defecated in flight and thus widely dispersed (Dudgeon & Corlett, 1994). In degraded sites, the presence of roost sites e.g. tree islands or even exotic tree plantations, would likely attract fruit bats (as well as frugivorous birds) and increase seed rain (Guevara & Laborde, 1993; Guevara *et al.*, 1992; Nepstad, 1989; Parrotta, 1995).

In Hong Kong, the two species of fruit bats, *Cynopterus sphinx* (Vahl) and *Rousettus leschenaulti* (Desmarest), may be important seed dispersal agents but their role has not yet been studied in any detail (Dudgeon & Corlett, 1994). However, five species of figs (*Ficus fistulosa*, *F. hispida*, *F. microcarpa*, *F. superba* and *F. variegata*) are known to be commonly consumed by them during the summer-wet season when very few fleshy fruits are availability (Corlett, 1996). They may be the sole dispersal agents for *F. fistulosa, F. hispida* and *F. variegata* (Dudgeon & Corlett, 1994). The role of these fruit bats in dispersing small-seeded tree species in Hong Kong demands more detailed studies.

The only primates that occur in Hong Kong are the macaques (*Macaca* spp.). Monkeys are generally frugivorous but they also eat leaves, leaf buds and flowers. Their importance in seed dispersal is still unclear (Corlett, 1998b; Howe, 1986; van der Pijl, 1982). A study in a Panamanian rainforest found that the Howler Monkey, *Aloutta palliate*, was responsible for 74% of the seeds removed from *Tetragastris panamensis* trees (Howe, 1980). However, only about 11% had a small chance of establishment. Most seeds removed were either dropped under the fruiting trees or concentrated in faecal clumps where germination and establishment are not favourable. A detailed study in Singapore revealed that the Long-tailed Macaque, *Macaca fascicularis*, would swallow small seeds (<2.3 mm in diameter), spit or drop large seeds, and destroy seeds housed in dry fruits (Lucas & Corlett, 1998). Of these, only seed spitting could lead to effective seed dispersal if seeds are spat away from the parent plant (Corlett, 1998b; Lucas & Corlett, 1998).

A study on macaque populations in Hong Kong estimated that 50% were *Macaca mulatta*, which is native to the region. The rest were hybrids of *M. mulatta*, *M. fascicularis* and *M. fuscata* (Fellowes, 1992). Hong Kong macaques are confined to the Kowloon Hills, Shing Mun Country Park and Tai Po Kau Nature Reserve, all in the Central New Territories (Fellowes, 1992; Wong, 1994). Although they rely heavily on human-supplied food, especially at the Kowloon Hills, a total of 163 natural food plants were identified (Fellowes, 1992; Wong, 1994). Of these, sixty-nine species had their fruits consumed. Unfortunately, no detailed study about seed dispersal by macaques in Hong Kong has ever been conducted and their importance as seed dispersers remains unclear. However, Dudgeon & Corlett (1994) observed that three plants with large fruits (*Garcinia oblongifolia*, *Melodinus suaveolens* and *Artocarpus hypargyreus*) were dispersed by macaques through seed spitting in the Central New Territories.

Corlett (1998b) and Lucas & Corlett (1998) pointed out that in areas with an intact vertebrate fauna, fruit taxa consumed by macaques are also consumed by other frugivores that provide higher quality seed dispersal than seed-spitting. In such case, macaques become less important in seed dispersal. However, in degraded and fragmented landscapes where the other vertebrate fauna is depleted, macaques could become important seed dispersers (Corlett, 1998b).

Rodents, especially rats and mice, are considered more as seed predators than seed dispersers (Corlett, 1998b; van der Pijl, 1982; Stiles, 1989). The common hillside rat species in Hong Kong (*Niviventer fulvescens* and *Rattus sikkimensis*) were reported to eat some fruits but seeds were rarely found in the faeces (Corlett, 1996). Intact seeds are more frequent in the faeces of the larger but rare bandicoot rat, *Bandicota indica*.

Squirrels could potentially disperse seeds by defecation or caching but the currently available information is insufficient to assess the role of squirrels in seed dispersal in the Oriental Region (Corlett, 1998b). Dudgeon & Corlett (1994) suspect that the large and dry fruits of the Fagaceae, Camellia and Styrax in Hong Kong were dispersed by squirrels and large rodents that are already extinct. The two squirrel sub-species in Hong Kong (*Callosciurus erythraeus*) were introduced at least twenty years ago (Goodyear, 1992; Ho, 1993). Their role in seed dispersal in Hong Kong is unknown. However, causal field observations found that *C. erythraeus* was a definite seed predator in Hong Kong. It was observed to eat the seeds (but not the pulp) of *Cinnamomum camphora*, *Sapium discolor* and many other species (pers. observations).

Other fruit-eating mammals in Hong Kong include civets (*Paguma larvata*, *Viverricula indica*), Small Asian Mongoose (*Herpestes javanicus*), Ferret Badger

(*Melogale moschata*), and Yellow-throated Marten (*Martes flavigula*). All these species disperse intact seed by defecation (Corlett, 1998b). Among these species, the civets are the most consistently frugivorous (especially in the winter) and are probably the most important as seed dispersers. They can harvest fruits on trees and vines and swallow large fruits whole. They travel long distance and defecate in open areas (Corlett, 1998b). They appear to be particularly important to those large-seeded species like *Choerospondias axillaris* and *Gnetum luofuense* that are not dispersed by birds and bats. However, relatively few studies have been conducted on civets and other medium-sized mammals in Hong Kong and little is known about their ecology in the local degraded landscape. Civets are widespread in Hong Kong (Reels, 1996), probably a result of low hunting pressure.

Barking deer (*Muntiacus* sp.) also consumes fruits. However, they may also destroy seeds in the mouth or spit them out near the parent tree (Corlett, 1998b). Their role in seed dispersal in Hong Kong is unclear.

Myrmecochory occurs in many habitats throughout the world. Diverse genera of ants collect seeds to their nests, consume the oily elaiosome and discard the rest at the refuse piles near the nest, or bury them in nest tunnels, where seeds can germinate and grow (van der Pijl, 1982; Passos & Ferreira, 1996). A lot of ant genera are also known as seed predators but no ants were found eating seeds in a local seed predation experiment (Hau, 1997). Little is known about the role of ants in seed predation and dispersal in Hong Kong but they were seen carrying seeds of *Litsea rotundifolia* and *Aquilaria sinensis* back to their nest in the field (personal observation).

In view of the relatively higher importance of birds in increasing seed rain on degraded hillsides, this study was designed to assess the role of birds but not the other potential seed dispersal fauna, like frugivorous bats and civets.

2. Materials and Methods

2.1 Study sites

The study was conducted at one grassland and one shrubland site in the northern New Territories of Hong Kong (Figure 2.1). The Kadoorie Farm and Botanic Garden (KFBG) grassland is on a 40º slope at 550 m and is dominated by *Arundinella* sp., *Ischaemum* sp., *Eulalia* sp., *Eragrostis* sp., *Cymbopogon* sp. and *Miscanthus sinensis*. This site is divided by a 2 m wide firebreak made of *Acacia confusa*. There are some small trees and shrubs (up to 2 m. in height) scattered across this grassland north of the firebreak. They include *Machilus chekiangensis*, *Litsea cubeba*, *Zanthoxylum avicennae* and *Melastoma sanguineum*. However, there are no trees and shrubs on the southern side of the firebreak. In a ravine about 300m to the south of this grassland is a secondary woodland which is dominated by *Machilus* spp.

The shrubland site is a 30 m x 30 m gap created by bird-ringers in a young secondary forest at the Kadoorie Agricultural Research Centre (KARC). It is on a 25° slope at 200 m. The dominant shrubs include *Rhodomyrtus tomentosa*, *Litsea rotundifolia* and *Rhaphiolepis indica*, all less then 1 m in height. Some young trees between 2 - 3 m in height remained intact across this gap. They include *Schefflera octophylla*, *Cratoxylum cochinchinense*, *Sapium discolor*, and *Mallotus paniculatus*.

2.2 Experimental design

Twenty 0.5 m x 1 m x 0.1 m, 0.5 mm mesh size rectangular plastic seed traps were erected 0.5 m above ground level using bamboo sticks (0.5 cm in diameter) at each site. Setting seeds traps at this height allowed easy sampling and avoided seeds being washed away during heavy rainstorms but was unable to stop rodent species from climbing up to the seed traps. Ten of the 20 traps were haphazardly placed at each site and were not shaded by any vegetation. The other ten traps were placed under the canopies of the small trees at each site. At the KFBG grassland, all ten traps were placed under *Machilus chekiangensis*. At the KARC shrubland, all ten traps were placed under *Schefflera octophylla*. Intact fruits of these two species found in the traps underneath them were not counted.

All seed traps were set on 1 October 1996 and checked weekly in the winter dry season, when the majority of tree species produce fruits (September to the end of March), and monthly in the wet season until the end of September 1997. Seeds found in the seed traps were first examined to identify the dispersal agents. Seeds were then identified and counted in the laboratory. Any evidence of potential seed predators disturbing the seed traps was also noted (e.g. the occurrence of rodent faecal pellets or broken seed coats).

Soon after the seed traps were set, rodent faecal pellets containing intact seeds were found in the seed traps. A rodent trapping study was therefore conducted to determine what rodent species was responsible and the faecal pellets of the trapped rodents were examined for intact seeds. Five 10 x 10 x 30 cm wire cage traps were haphazardly placed at both sites using bread as a bait for five consecutive nights from 06/01/97 to 11/01/97. Rodent traps were checked in the early morning every day. Any rodents trapped were identified, measured and had the tip of their tail marked with green paint. All faecal pellets left by the trapped rodent were collected and examined under the microscope for intact seeds.

Seeds from nine of the seventeen tree species and six of the fourteen shrub species were tested for viability. The rest of the tree and shrub species were not tested because there were either too few for a germination test or they had been contaminated by fungus at the time of collection. The seeds were sown in sandy loam soil in the nursery under a 50% shade cloth and irrigated daily. Seeds were allowed to germinate for 6 months.

3. Results

A total of 2,417 seeds of 17 tree species, 10,097 seeds of 14 shrub species, 132 seeds of 5 species of climbers and 78 seeds of 5 herbaceous species were collected (Table 2.1). All seed species are native to Hong Kong except *Lantana camera*, which is a naturalized exotic species. *Schefflera octophylla* made up 71.1% of the total number of tree seeds collected. Other relatively common tree species included *Rhus* spp. (either *R. chinensis* or *R. hypoleuca* or both), *Bridelia tomentosa*, *Litsea rotundifolia, Mallotus paniculatus* and *Sapium discolor.* For shrubs, *Melastoma candidum* made up 63.9% of the total number collected. Other relatively common shrubs included *Ilex asprella, Melastoma sanguineum,* and *Rhodomyrtus tomentosa.*

At both sites, far more seeds were collected by seed traps set under a tree canopy (Table 2.2). Shaded traps accounted for 94.4% of the seeds collected at the KARC shrubland and 99.8% of the seeds collected at the KFBG grassland. Only one woody seed species (*Rhodomyrtus tomentosa*) was collected by the open traps in the KFBG grassland. Of the tree seeds collected, 87.6% were from the shrubland. All tree species collected are pioneers and none of the woody species collected are dispersed by wind. All except *Glochidion eriocarpum, G. puberum, Aralia armata, Memecylon* sp. and *Symplocos* sp. are known to be dispersed by birds (Table 2.3) and these species probably are since they all have small, fleshy fruits and appeared as discrete seeds in the seed traps. No macaque, civet and bat faeces were found in the seed traps. Intact seeds of one tree (*Ficus fistulosa*), five shrubs (*Eurya chinensis*, *Ilex pubescens*, *Melastoma candidum*, *Melastoma sanguineum* and *Rhodomyrtus tomentosa*) and two herb species (*Carex baccans* and *Dianella ensifolia*) were recovered from the faecal pellets of rats in the grassland seed traps (Appendix 2.1). They comprised 56.6% of the seeds trapped at this site. In the shrubland site, intact seeds of only two shrubs species (*Lantana camara* and *Rhodomyrtus tomentosa*) were recovered from rat faecal pellets in the traps, representing only 0.35% of the seeds trapped at this site.

One rat species was trapped at each site. Sladen's Rat, *Rattus koratensis,* was trapped in the part of the KFBG grassland scattered with small trees and shrubs, and intact seeds of *Melastoma candidum*, *M. sanguineum* and *Rhodomyrtus tomentosa* were found in its faecal pellets (Table 2.4). Its faecal pellets were similar in size and shape to those found in the seed traps at this site. The Chestnut Spiny Rat, *Niviventer fulvescens,* trapped at the KARC shrubland, had intact *Melastoma candidum* and *Mussaenda pubescens* seeds in its faecal pellets (Table 2.4). Its faecal pellets were smaller than those of Sladen's Rat and similar in size and shape to those found in the seed traps at this site. However, Sladen's Rat appeared to disperse a larger quantity of seeds than the Chestnut Spiny Rat (Table 2.4).

No signs of seed predation were observed in the seed traps in the grassland site but 2 *Mallotus paniculatus* seeds, 17 *Ilex asprella* seeds and 13 *Aporosa dioica* seeds were destroyed by seed predators in the seed traps at the KARC shrubland.

All seed species tested were viable with germination rates ranging from 11% (*M. sanguineum*) to 90% (*S. octophylla*) (Table 2.5). Seeds recovered from both bird droppings and rat faecal pellets were viable.

4. Discussion

The total of 12,516 seeds of woody plants collected on degraded hillsides of Hong Kong in this study is comparable to the 13,700 woody seeds collected on a degraded site in the Amazon with similar seed trapping effort (*Nepstad, 1989*). The results of this study also agree with similar studies elsewhere (Debusche & Isenmann, 1994; Guevara *et al.*, 1992; Kolb, 1993; McClanahan & Wolfe, 1993; Nepstad et al., 1991; Otero-Arnaiz *et al.*, 1999; Toh *et al.*, 1999; Willson & Crome, 1989). Over 94% of the seeds were collected under treelets at both sites (excluding the rat-dispersed seeds) and most of these were bird-dispersed (except five species where the dispersal agents are unknown, see Table 2.2). Only one *R. tomentosa* seed was collected in the part of the KFBG grassland where there were no perches. Taking out the rat-dispersed seeds from this study (see Table 2.1), the number of woody seeds collected from the KARC shrubland (5,208) is about 1.7 times that collected from the KFBG grassland (3,138). This is not a big difference given the general patchness of seed dispersal. Remembering that the shrubland site is relatively small and is enclosed by young secondary forests, the frugivorous bird density is thus expected to be higher than the much more open grassland site. The higher seed rain in the shrubland is consistent with studies elsewhere which showed that seed flow is positively correlated with the availability of perches and negatively correlated with spatial isolation (Duncan & Chapman, 1999; Kolb, 1993).

Similar studies in the tropics have documented rodents as seed predators (Nepstad, 1989; Nepstad *et al.*, 1991) but not as seed dispersers. However, certain rat species were known to disperse seeds elsewhere in the world (van der Pijl, 1982). Sladen's Rat and Chestnut Spiny Rats were found dispersing seeds in this study. They are the most common wild rat species in Hong Kong (Rao, 1994). The Chestnut Spiny Rat is present at all habitats in Hong Kong except lowland grassland and abandoned cultivation (Rao, 1994). Sladen's Rat is also widespread in Hong Kong but slightly less common than the Chestnut Spiny Rat (Rao, 1994). There were no significant differences between the diet and behaviour of Sladen's Rat and the Chestnut Spiny Rat, and they might have overlapping territories (Goodyear, 1992). However, these two species were not trapped in the same site in this study.

The Chestnut Spiny Rat accounted for only 0.35 % of seeds collected in the KARC shrubland and intact seeds of only two shrubs species were recovered from the faecal pellets (Table 2.1). This rat was found to be a major seed predator in this shrubland (Hau, 1997). However, seed predation was observed only three times, all in Summer, in the seed traps of this shrubland over the period of the study. This may be attributed to the fact that fruit availability was low during the summer and the rats were more desperate in searching for food so that they climbed up to the seed traps that were set 0.5 m above ground level. In comparison with the amount of seeds dispersed by avian seed dispersers at this site, this rat is not a significant seed disperser.

All of the shrub seeds dispersed by Sladen's Rats were collected from the traps set under treelets in the KFBG grassland. It is unknown why this rat defecated in seed traps 0.5 m above the ground. No signs of seed predation were observed in this site. Sladen's Rats accounted for 56.6% of the seeds trapped at this site. The intact seeds found in the Sladen's Rat faecal pellets were among the smallest seeds trapped in this study: the seed diameters of *Rhodomyrtus tomentosa* and *Melastoma* spp. are 2.4 mm and 0.5 mm respectively. At least five different individuals of this rat species were recorded from this study (Table 2.3). However, during a seed predation experiment in the treeless part of this grassland no Sladen's Rats were trapped (Hau, 1997). This suggests that Sladen's Rat may not even venture into open grassland in the immediate neighbourhood. The results of this study suggest that the Sladen's Rat could at most be an important seed disperser in grasslands with low shrubs for woody species with small seed size. On the other hand, the role of the Sladen's Rat as seed predators is yet unknown and should be studied. No Sladen's Rats were found during seed predation experiments on hillside grasslands and shrublands in Hong Kong (Hau, 1997).

In summary, avian seed dispersers, especially birds, appear to be the most important seed dispersal agent on degraded hillsides in Hong Kong. The woody seed species collected are mostly small-seeded, fleshy-fruited species that are common in shrubland and young secondary forest, suggesting that the current woody species composition of hillside shrubland and young secondary forest may be the long-term result of this selective seed dispersal. This study is insufficient in assessing the role of fruit bats in seed dispersal on degraded hillsides. Bats defecate in flight and are unlikely to use small shrubs and trees as feeding roosts. The small number of seeds in the open traps may be due to the small total trapping areas. In general, the low input of seeds into open grasslands (1 woody seed per 5 m^2 per year) suggests that it is a rather significant barrier to natural succession at such sites. The presence of perches would make a site more attractive to avian seed dispersers. Degraded hillsides with isolated patches of trees would certainly increase seed input, and thus the chances of natural forest regeneration. However, the increase in seed rain alone may not be sufficient to accelerate forest succession if seed predation is serious and seed germination remains low (These two factors will be covered in the next two chapters). A further study to survey woody species established under small isolated patches of trees in comparison with the open grassland will give more hints on the role of vegetation island in accelerating forest succession in Hong Kong. Sladen's Rat was also found to be dispersing large quantity of small-sized seeds under isolated trees in the grassland in this study, more thorough and detailed studies are required to assess its roles in seed dispersal and predation on degraded hillsides in Hong Kong.

Figure 2.1

Locations of the two study sites

a. KFBG grassland

b. KARC shrubland

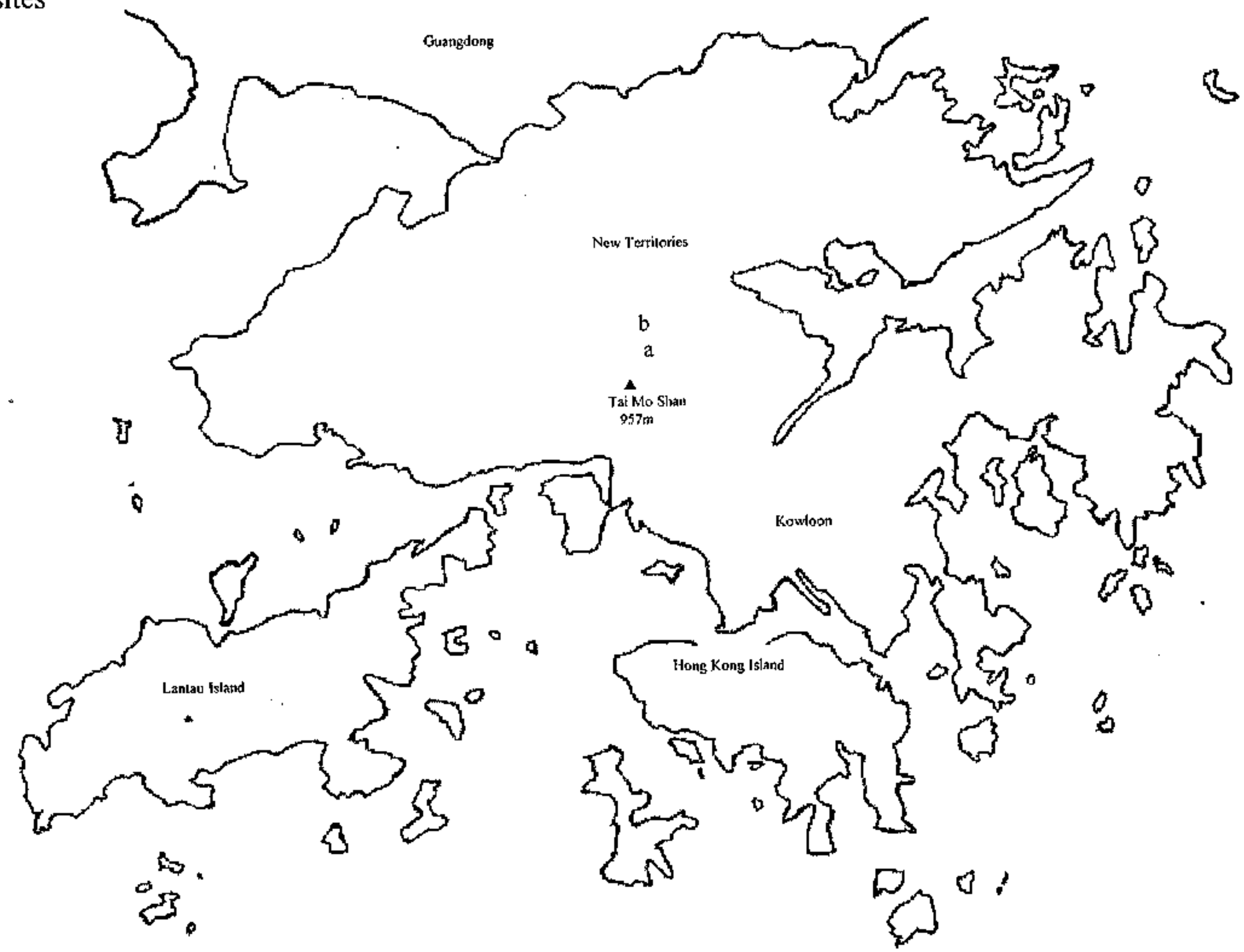


Table 2.1

A list of seed species and numbers collected from each site. The underlined numbers indicate that those seeds were found from rat faecal pellets rather than bird droppings.

Seeds species	No. collected at KARC Shrubland		No. collected at KFBG grassland		Total
	Open traps	Shaded traps	Open traps	Shaded traps	
Herbaceous species:					
Carex baccans				6	
Carex cruciata	7	1		1	
Cassytha filiformis		2			
Dianella ensifolia		4	14	1	
Rubus reflexus		4		38	
Total:	**7**	**11**	**14**	**46**	**78**
Climbers:					
Embelia laeta	1	17			
Embelia ribes	1	102		4	
Psychotria serpens		2			
Smilax china		2			
Tetracera asiatica		3			
Total:	**2**	**126**		**4**	**132**
Shrubs:					
Eurya chinensis				69+5	
Glochidion eriocarpum		25			
Glochidion puberum		1			
Ilex asprella	3	194			
Ilex pubescens		43		3+1	
Lantana camera		11+1		3	
Ligustrum sinense		5			
Melastoma candidum		1021		2824+2612	
Melastoma sanguineum		349		100+796	
Psychotria rubra		57			
Rhodomyrtus tomentosa	259+1	1114+18	1	98+477	
Sarcandra glabra		1			
Viburnum sempervirens		5			
Total	**263**	**2845**	**1**	**6988**	**10097**
Trees*:					
Aporosa dioica	2	30			
Aralia armata		6			
Bridelia tomentosa		49			
Ficus fistulosa				259	
Ficus variolosa		7			
Litsea cubeba	1	7		17	
Litsea rotundifolia	5	50			
Mallotus paniculatus		40			
Memecylon sp.		5			
Microcos paniculata		15			
Machilus chekiangensis		1		2	
Rhus sp.	3	103		1	
Rhus succedanea	8	31			
Sapium discolor	1	45			
Schefflera octophylla	8	1699		19	
Symplocos sp.		1		1	
Syzygium levinei		1			
Total	**28**	**2090**	**0**	**299**	**2417**
Total (all species)	**300**	**5072**	**15**	**7337**	**12[illegible]**

Table 2.2

Two sample T-Test results on the mean number of seeds collected in shaded traps and open traps over one year at KARC shrubland and KFBG grassland.

Site	Mean number of seeds collected		T-test
	Shaded traps	Open traps	
KARC	423	26	$T = 6.01, p < 0.001$
KFBG	615	1	$T = 5.20, p < 0.001$

Table 2.3

Known dispersal agents of the woody species collected

Shrubs:	Dispersal agent	Trees:	Dispersal agent
Eurya chinensis	Bird [1,2]	*Aporosa dioica*	Bird [1]
Glochidion eriocarpum	?	*Aralia armata*	Bird [3]
Glochidion puberum	?	*Bridelia tomentosa*	Bird [1,2]
Ilex asprella	Bird [1]	*Ficus fistulosa*	Fruit Bat [1], Bird [3]
Ilex pubescens	Bird [1,2]	*Ficus variolosa*	Bird [1]
Lantana camera	Bird [1,3]	*Litsea cubeba*	Bird [1,2,3]
Ligustrum sinense	Bird [1]	*Litsea rotundifolia*	Bird [1,2,3]
Melastoma candidum	Bird [2], Rat [3]	*Mallotus paniculatus*	Bird [1,2,3]
Melastoma sanguineum	Bird [1,2], Rat [3]	*Memecylon sp.*	?
Psychotria rubra	Bird [1,2,3]	*Microcos paniculata*	Bird [1,2,3], Civet [3]
Psychotria serpens	Bird [1,2]	*Machilus chekiangensis*	Bird [1,2,3]
Rhodomyrtus tomentosa	Bird [1,2,3], Civet [1], Macaque [1], Rat [3]	*Rhus sp. (R. chinensis and/ or R. hypoleuca)*	Bird [1,2,3]
Sarcandra glabra	Bird [1]	*Rhus succedanea*	Bird [3]
Viburnum sempervirens	Bird [1]	*Sapium discolor*	Bird [1,2,3]
		Schefflera octophylla	Bird [1,2,3]
		Symplocos sp.	?
		Syzygium levinei	Bird [1]

Notes: [1] According to Corlett, 1996.
[2] According to Corlett, 1992a.
[3] My observations.

Table 2.4

Rodents trapped at the study sites and the intact seed species recovered from their faecal pellets.

			Measurement (cm)					Intact seeds found in faecal pellets		
Date	Cage No.	Rat Species	Sex	Head	Body	Tail	No. of faecal pellets	Species	No.	Notes
KFBG Grassland										
07/01	2	*Rattus koratensis*	-	-	-	-	3	No seeds found	-	**1**
07/01	3	*Rattus koratensis*	-	-	-	-	14	*Melastoma candidum*	163	**1**
07/01	5	*Rattus koratensis*	-	-	-	-	4	*M. candidum*	179	
08/01	3	*Rattus koratensis*	F	4	18	21	23	*1. M. candidum* *2. M. sanguineum*	56 297	
08/01	5	*Rattus koratensis*	F	4	15	20	25	*M. candidum*	217	
09/01	3	*Rattus koratensis*	F	4.5	18	18	21	*1. M. candidum* *2. M. sanguineum*	406 14	
10/01	2	*Rattus koratensis*	-	-	-	-	7	*M. candidum*	52	**1**
10/01	3	*Rattus koratensis*	F	5	9	21	20	*1. M. candidum* *2. M. sanguineum*	82 58	
11/01	2	*Rattus koratensis*	-	-	-	-	2	*M. candidum*	20	**1**
11/01	3	*Rattus koratensis*	-	-	-	-	18	*1. M. candidum* *2. Rhodomyrtus tomentosa*	42 1	**1**
11/01	5	*Rattus koratensis*	F	4	18	19	25	*1. M. candidum* *2. M. sanguineum*	467 416	

Table 2.4 Cont.

Date	Cage No.	Rat Species	Measurement (cm) Sex	Head	Body	Tail	No. of faecal pellets	Seed found in faecal pellets Species	No.	Notes
KARC Shrubland										
07/01	2	*Niviventer fulvescens*	-	-	-	-	9	*Mussaenda pubescens*	19	**2**
07/01	4	*Niviventer fulvescens*	F	3	10.5	12.5	38	*M. candidum*	5	
08/01	1	*Niviventer fulvescens*	F	3.5	13	15	44	*M. candidum*	21	
08/01	2	*Niviventer fulvescens*	-	-	-	-	5	No seeds found	-	**2**
08/01	3	*Niviventer fulvescens*	F	3.5	13	15	44	No seeds found	-	
08/01	4	*Niviventer fulvescens*	F	3	11	12	28	No seeds found	-	
09/01	2	*Niviventer fulvescens*	-	-	-	-	12	No seeds found	-	**2**
09/01	3	*Niviventer fulvescens*	F	3	10.5	12	26	No seeds found	-	
09/01	4	*Niviventer fulvescens*	F	3	11	12	42	No seeds found	-	Recapture of Cage 4, 08/01
10/01	1	*Niviventer fulvescens*	F	3.5	13	15	8	No seeds found	-	Recapture of Cage 1, 08/01
10/01	3	*Niviventer fulvescens*	F	3	13.5	15	40	No seeds found	-	
10/01	4	*Niviventer fulvescens*	F	3	10	12	32	No seeds found	-	
11/01	2	*Niviventer fulvescens*	F	3.5	13	15	2	No seeds found	-	Recapture of Cage 1, 08/01
11/01	3	*Niviventer fulvescens*	F	3.5	13	15	53	No seeds found	-	
11/01	4	*Niviventer fulvescens*	F	3	10	12.5	36	No seeds found	-	

Note: **1** Bait was taken but no rat was trapped. The shape, size & color of faecal pellets resembled that of *R. koratensis*.
2 Bait was taken but no rat was trapped. The shape, size & color of faecal pellets resembled that of *N. fulvescens*.

Table 2.5

Seed viability of some of the woody species collected from the seed traps.

Species	From droppings of	No. sown	Date sown	Total No. germinated in 6 months	% germination
Tree:					
Aporosa dioica	Bird	20	15/10/97	8	40
Bridelia tomentosa	Bird	10	20/02/97	5	50
Litsea rotundifolia	Bird	45	15/10/97	22	49
Mallotus paniculatus	Bird	20	08/01/97	11	55
Microcos paniculata	Bird	10	08/01/97	4	40
Rhus sp.	Bird	20	20/02/98	9	45
R. succedanea	Bird	29	15/10/97	10	35
Sapium discolor	Bird	10	08/01/97	5	50
Schefflera octophylla	Bird	100	20/02/97	90	90
Shrub:					
Melastoma candidum	Rat	100	20/02/97	15	15
M. sanguineum	Rat	100	20/02/97	11	11
Psychotria rubra	Bird	10	20/02/97	8	80
P. rubra	Bird	18	15/10/97	11	61
Rhodomyrtus tomentosa	Rat	50	08/01/97	10	20
R. tomentosa	Bird	500	15/10/97	137	27

Chapter 3

Tree Seed Predation on Degraded Hillsides in Hong Kong

Notes:
This chapter consists of a paper with the same title published in 1997 in Forest Ecology and Management 99:215-221 (Appendix 3.1) and an additional field experiment with tagged seeds totally buried in the soil. Additional literature is cited.

Abstract

The aim of this study was to determine whether seed predation was a barrier to natural forest regeneration on degraded hillsides. Removal of seeds of eight tree species in the winter of 1995 and 12 in 1996 at four Hong Kong hillside sites was monitored. Seed removal at the two shrubland sites was higher than at the two grassland sites in both 1995 and 1996. Most seeds placed in the shrubland sites in 1996 were removed: 11 of 12 species were totally removed from one shrubland site within 60 days, while only one of 12 species was totally removed in one grassland site. Rats were found to be the major seed predator. They included *Niviventer fulvescens* and *Rattus rattus flavipectus*. The tough and thick-coated seeds of *Choerospondias axillaris* and *Elaeocarpus sylvestris* had the lowest mean percentage removal. A seed-tagging test showed that rodent seed predators are able to detect seeds buried underground at low density. The results of this study suggest that seed predation is a significant factor in reducing the amount of seeds available for seedling recruitment on degraded hillside in Hong Kong. Currently, all reforestation efforts in Hong Kong use container-grown seedlings, which is expensive even on accessible sites, and impractical in remote areas. The results of this study suggest that direct seeding may be possible if species with tough and thick-coated seeds are used.

1. Introduction

Seed predation is a significant factor in causing demographic and evolutionary changes in plant populations (Blate *et al.*, 1998; Hulme, 1993; Janzen, 1971; Louda 1989). Nepstad *et al.* (1991) have identified seed predation as one of the biotic barriers to natural forest regeneration in abandoned Amazon pasture derived from rainforest. Small mammals and ants are typical seed predators (Hulme, 1993; Janzen, 1971; Louda 1989; Sánchez-Cordero & Martínez-Gallardo, 1998; Santos & Tellería, 1998). Many studies have shown that seed size is related to the vulnerability of a seed species to seed predators (Louda, 1989; Osunkoya, 1994; Reader, 1993). Nepstad *et al.* (1991) show that small seeds (< 0.02 g) are more vulnerable to fire ants (*Solenopsis* sp.), cutter ants (*Atta sexdens*) and harvester ants (*Pheidole* sp.) than larger seeds (> 5 g). Apart from seed size, Blate *et al.* (1998) also show that predation rates are negatively associated with the thickness and hardness of seed coat in South Asian rainforest. Mack (1998) shows that large seed size could enable tolerance of seed predation.

In addition to the seed dispersal filter, the difference in attractiveness between seed species to seed predators on degraded hillside sites may also act as a filter on the composition and diversity of the resulting forest formed on the degraded hillside sites.

This is the first study in Hong Kong examining seed predation in degraded hillsides. It aims to determine if post-dispersal seed predation is a filter-barrier to natural forest regeneration in hillside grassland and shrubland.

2. Materials and Methods

2.1 Study sites

The study was conducted at two grasslands and two shrubland sites. The Kadoorie Farm and Botanic Garden (KFBG) grassland and the KARC shrubland were the same sites as described in Chapter 2 but on different patches. The Ho Sheung Heung (HSH) grassland and shrubland are next to each other on a 15º slope at 20 m. The grassland is dominated by *Arundinella* sp., *Eulalia sp.* and *Miscanthus sp.*. The dominant shrubs are *Baeckea frutescens* and *Rhodomyrtus tomentosa.* These sites are immediately next to a secondary woodland which is dominated by *Microcos paniculata*, *Acronychia pedunculata*, *Schefflera octophylla* and *Machilus chinensis*. A few young trees of *Schefflera octophylla* and *Cratoxylum cochinchinense* occur in the shrubland.

2.2 Experimental design

Since the majority of tree species in Hong Kong produce fruits in winter, this study was conducted in winter and only seeds from the same winter were used to maximise viability. In addition, all floating seeds were discarded. A total of 16 seed species of different sizes were tested in this study. Ten species were used in the winter of 1995 and 12 in 1996 (see Table 3.3.1). The difference in the number of species used at each site in each year was due to seed availability. For example, *Acronychia pedunculata* and *Choerospondias axillaris* were used in only one site in 1995 because there were not enough seeds for all sites in both years.

For the 1995 trial, ten 25 x 30 x 3 cm plastic seed trays were evenly spaced along two 50 m transects, 10 m apart at each site. All seed trays were shaded by vegetation at all sites to avoid direct exposure of seeds to the sun. The seed trays were lined with a piece of hemp cloth for the ease of sampling. Ten seeds of each species were placed in each seed tray at the same time. In 1995, seeds were simply placed on seed trays. In 1996, the seed tray was covered with a layer of nursery soil and the seeds were half-buried. Trays were checked daily in the first week, twice a week in the next 3 weeks, and once a week afterwards for a total of 60 days. The numbers of missing seeds were counted and any evidence of seed predator occurrence (e.g. the presence of broken seed coats and faecal pellets) was also noted.

Five 10 x 10 x 30 cm cage-traps were set al.ong a 50 m transect at each site for 7 consecutive nights after there had been evidence of rodent seed predators visiting the seed trays. Rodent trapping was conducted only at the HSH shrubland in 1995 using bread as bait. *Cyclobalanopsis myrsinifolia* seeds, which had high rate of removal, were used as bait at all sites in 1996. Traps were checked every morning and all rodents trapped were identified, measured and one individual of each species was kept in the laboratory. They were fed with the same seed species to see how readily they would consume the seeds and what evidence they left behind. They were released back to where they were trapped at the end of the trapping study.

The experimental set up described above was questioned by some researchers because the high concentration of seeds in the seed tray was seen as an obvious attraction to seed predators. The experiment was therefore repeated in November 1996 at the KARC shrubland with tagged seeds completely buried underground. The seeds of two native tree species were used: *Microcos paniculata* and *Sapium discolor*. A 5 cm cotton thread, 2 mm in diameter, was glued on to the surface of each seed by Aron Alpha® instant glue as the tag. A 5 cm cotton thread with a small knot at one end and a drop of glue but no seed was used as the control. Five tagged seeds of each species and five seedless tags were buried in three rows (20 cm between each tag/seed and between each row) on the shrubland floor. Each tagged seed and seedless tag was buried 2 cm below the soil surface and covered by leaf litter. Ten replicates of this set-up were evenly spaced along two 50 m transects, 10 m apart (the transects were set at different locations from those in the previous experiment). Data were collected for 30 days in the same manner. The percentage seed removal in 30 days (arcsine transformed) between the two species and the control was compared by one-way ANOVA using Minitab Release 12. The Newman-Keuls multiple range (SNK) test was used for multiple comparison.

3. Results

Nonparametric sign tests (Ryan *et al.*, 1976) show that the mean percentage seed removal over 60 days at shrubland sites was higher than at grassland sites in both 1995 ($P<0.05$) and 1996 ($P<0.001$) (Table 3.1). The pattern of seed removal at all sites was very similar in both years. The shrubland sites almost always had higher initial and total percentage seed removal than grassland sites (Figure 3.1). In 1996, most seeds at shrubland sites were removed in the first week (Figure 3.1). Eleven of 12 species were totally removed in 1996 from the KARC shrubland but there was only one (*Melicope pteleifolia*) totally removed in one grassland site (Table 3.2). In 1996, *Mallotus paniculatus*, *Machilus breviflora* and all four species of Fagaceae (*Cyclobalanopsis championii*, *Cyclobalanopsis myrsinifolia, Lithocarpus glaber* and *Castanopsis fissa*) were totally removed at both shrubland sites (Table 3.2).

Choerospondias axillaris, which has tough and thick coated seeds, had the lowest mean percentage removal at shrubland sites in 1995 (14.5%). *Elaeocarpus sylvestris,* which also has tough and thick-coated seeds, had the lowest mean percentage removal at both grassland (28.4%) and shrubland (64%) sites in 1996 (Table 3.3.1). For both species, a lot of the intact seeds left on the seed trays had clear rodent teeth marks. The mean percentage seed removal was not significantly related to seed size (Simple linear regression: $R^2 = 0.1165$, $P > 0.05$ for shrubland and $R^2 = 0.0188$, $P > 0.05$ for grassland).

Two rat species, *Rattus rattus flavipectus* (body length 16 cm, tail length 20 cm) and *Niviventer fulvescens* (body length 10.5 - 13 cm, tail length 11 - 16 cm) were the only seed predators caught at all sites. Both species readily ate the same seed species in the laboratory and left the same kinds of remains as observed in the field. The overall trapping success in shrubland was higher than in grassland. Fewer rats were trapped in HSH shrubland than HSH grassland in 1996 but ten baits in the traps at the former site were taken without triggering the traps. No rats were trapped at the KFBG grassland even after two further seven-night trappings with double number of cages (Table 3.3). However, another rat species, the Sladen's Rat *Rattus koratensis* were trapped in the other part of this grassland where islands of trees or shrubs occur, in the seed trapping study (Chapter 2). In 1996, trapping success was higher at KARC shrubland where mean percentage seed removal was also the highest (all except *E. sylvestris* were totally removed).

In the tagging experiment, most tagged seeds (74 % for *Microcos*, 62 % for *Sapium*) were removed within one night while only 18 % of seedless tags were removed after 30 days (Figure 3.2). The difference between tagged seeds and seedless tags was significant but the difference between seed species was not (Table 3.4).

4. Discussion

The results of this study indicate that seed predation by small rodents at hillside grassland and shrubland could significantly reduce the availability of tree seeds for regeneration. However, it was not clear why the percentage seed removal of *Microcos paniculata*, *Machilus breviflora* and *Lithocarpus glaber* at KFBG grassland were exceptionally low in 1995. Both pioneer species (*Melicope pteleifolia*, *Mallotus paniculatus* and *Sapium discolor*) and potential climax species (the Fagaceae) and the secondary forest dominant species, *Machilus breviflora*, were heavily consumed by seed predators. A study of small mammal distribution in Hong Kong has shown that rodent densities in degraded habitats (i.e. hillside shrubland and grassland) were relatively high in comparison with other countries (Rao, 1994). It was estimated that small rodent density could reach as high as 60 ha^{-1} in lowland grassland and 30 ha^{-1} in shrubland. However, the overall trapping success in shrubland was higher than in grassland in this study. High rodent densities in Hong Kong may be due to high ground cover of shrubland and grassland. This may have significant implications for similar degraded habitats elsewhere since seed predation by small mammals would likely become a more important factor in affecting the species composition and limiting natural forest regeneration.

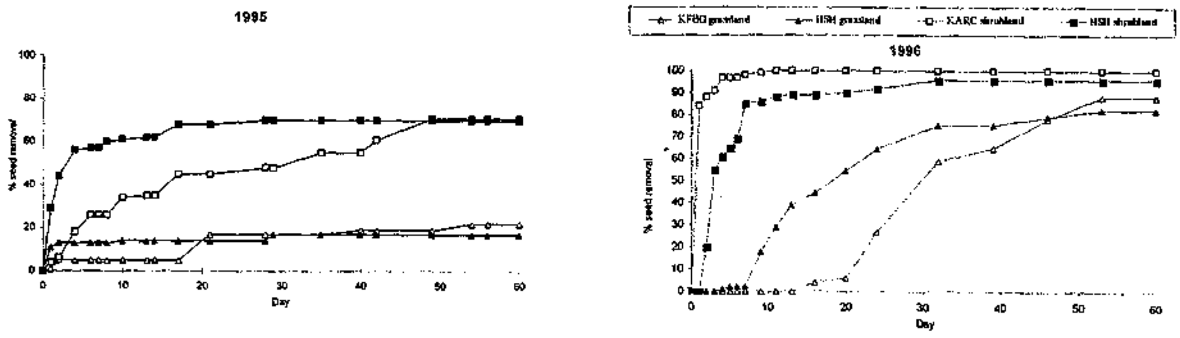
Unlike similar seed predation studies elsewhere (Louda, 1989; Nepstad *et al.*, 1991; Osunkoya, 1994; Reader, 1993), the probability of a seed being eaten by seed predators was not predicted by seed mass. Ants were found to be a major seed predator for very small seeds (e.g. <0.01g) in the tropics (Reader, 1993; Nepstad *et al.*, 1991; Andersen, 1987) but no ants were observed carrying seeds away in this study.

All species, except *C. axillaris* and *E. sylvestris*, had broken seed coats found in the seed trays at all sites. The missing *C. axillaris* and *E. sylvestris* seeds were probably carried away whole by rodents as their tough seed coats would take them a significantly longer time to handle. The low seed removal rates of these two species have significant implications for reforestation in Hong Kong. At present, the Hong Kong Government uses only container-grown seedlings from its tree nurseries for reforestation. Since 1980, about 330 ha of borrow areas and erosion control sites have been reforested using nearly 1.5 million container-grown seedlings produced in government tree nurseries (Webb, 1993). Since it is expensive to produce container-grown seedlings, the Government has been striving to reduce production cost as well as improve productivity. In recent years, nursery trials in sowing seeds directly into polythene seedling containers had been proven successful in producing more cost-effective seedlings (Agriculture and Fisheries Department, 1995). Direct seeding using tough/thick coated species at target reforestation sites, if proved possible, would further reduce the cost of reforestation and allow it to be conducted at remote areas and on steep slopes (see Chapter 4).

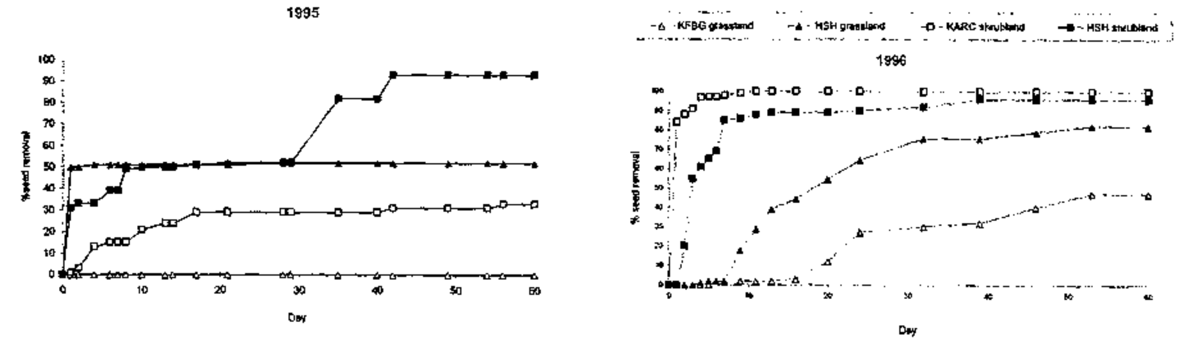
The higher percentage seed removal at all sites for all species in 1996 showed that half-burying the seeds did not reduce the chance of them being eaten. The tagging experiment further showed that rodent seed predators were able to detect seeds buried below the soil surface and leaf litter at low density. In contrast, Cintra (1997) has shown that seed removal was significantly lower for the microsites with leaf litter cover than for those with bare soil in the forest floor in the Amazonian forest. These results suggest that if species with a soft seed coat are used in direct seeding in Hong Kong, rodent trapping and seed predation experiments should be conducted at the target site to determine the seriousness of seed predation prior to sowing seeds. If seed predation pressure at the target site is high, seeds should be treated with rodent repellent such as arsenate and endrin (Pancel, 1993).

In summary, this study indicates that seed predation is a significant barrier to natural forest regeneration in degraded hillsides in Hong Kong. Seed predation is probably also affecting the species composition of natural forest regeneration. The vulnerability of a seed species to seed predators was not related to its size but seeds with tough and thick seed coats were less attractive to seed predators.

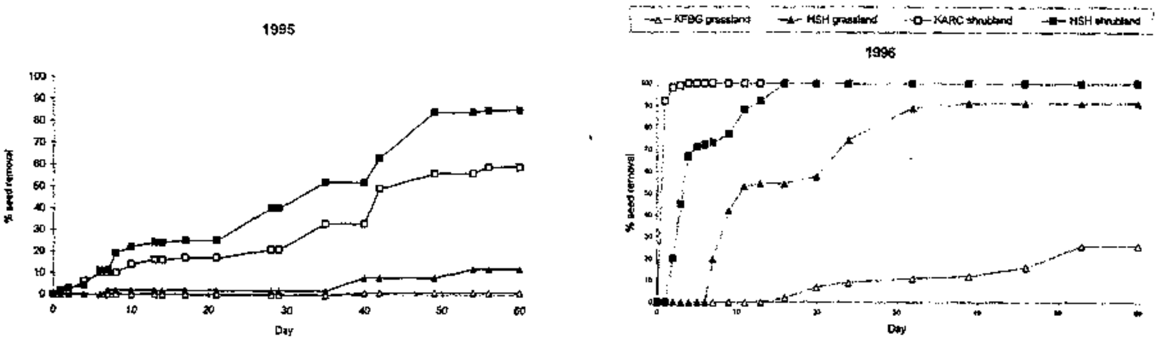
% removal of *Gordonia axillaris* seeds over time



(a) % removal of *Microcos paniculata* seeds over time



(b) % removal of *Machilus breviflora* seeds over time



(c) % removal of *Lithocarpus glaber* seeds over time

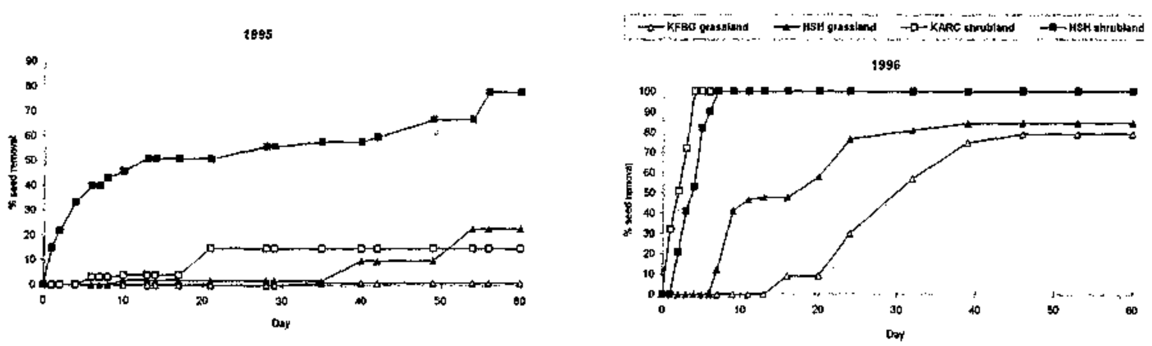


Figure 3.1

Percentage seed removal over time for (a) *Gordinia axillaris*, (b) *Microcos paniculata*, (c) *Machilus breviflora*, (d) *Lithocarpus glaber* at all sites in both 1995 and 1996.

Figure 3.2

Percentage removal over time for tagged seeds of *Microcos paniculata*, *Sapium discolor* and seedless tags (control) in the KARC shrubland in 1996.

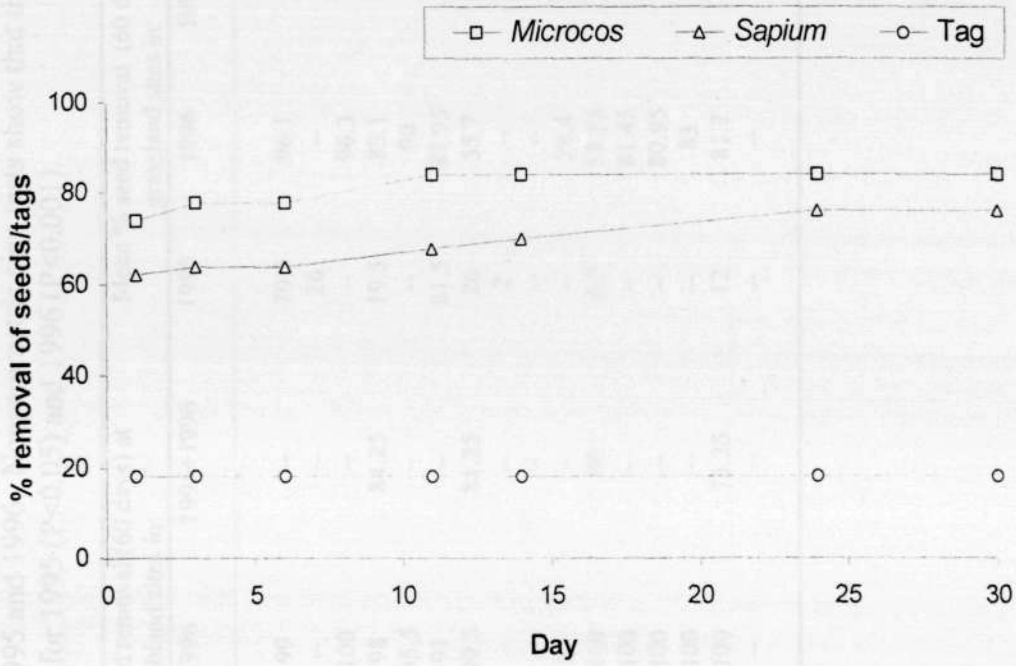


Table 3.1

Mean percentage seed removal at two shrubland and two grassland sites in 1995 and 1996. Nonparametric sign tests show that the mean percentage seed removal at shrubland sites was higher than at grassland sites for 1995 ($P<0.05$) and 1996 ($P<0.001$).

				Mean % seed removal (60 days) at shrubland sites in:			Mean % seed removal (60 days) at grassland sites in:		
No.	Species	Family	Mean seed mass (g)[1]	1995	1996	1995+1996	1995	1996	1995+1996
1	*Melicope pteleifolia*	Rutaceae	0.0066	--[2]	99	--	70.5	96.1	83.3
2	*Bischofia javanica*	Euphorbiaceae	0.0101	99	--	--	26	--	--
3	*Mallotus paniculatus*	Euphorbiaceae	0.0148	--	100	--	--	96.3	--
4	*Gordonia axillaris*	Theaceae	0.0158	70.5	98	84.25	19.5	85.1	52.3
5	*Reevesia thyrsoidea*	Sterculiaceae	0.0311	--	95.5	--	--	90	--
6	*Sapium discolor*	Euphorbiaceae	0.0466	--	91	--	81.5	87.95	84.73
7	*Microcos paniculata*	Tiliaceae	0.1276	63	99.5	81.25	26	55.7	40.85
8	*Diospyros morrisiana*	Ebenaceae	0.1383	45	--	--	2	--	--
9	*Acronychia pedunculata*	Rutaceae	0.1611	31	--	--	--	--	--
10	*Elaeocarpus sylvestris*	Tiliaceae	0.2900	--	64	--	--	28.4	--
11	*Machilus breviflora*	Lauraceae	0.3390	72	100	86	6.5	58.55	32.53
12	*Cyclobalanopsis championii*	Fagaceae	0.8068	--	100	--	--	81.45	--
13	*Cyclobalanopsis myrsinifolia*	Fagaceae	0.8100	--	100	--	--	80.95	--
14	*Castanopsis fissa*	Fagaceae	1.6300	--	100	--	--	83	--
15	*Lithocarpus glaber*	Fagaceae	1.6624	46.5	100	73.25	12	81.7	46.85
16	*Choerospondias axillaris*	Anacardiaceae	2.4648	14.5	--	--	--	--	--

[1] Mean seed mass was the average weights of 50 seeds.
[2] "--" indicates that the species was not tested at that site in that year.

Table 3.2

Percentage seed removal at two shrubland and two grassland sites in 1995 and 1996.

		Percentage seed removal (60 days)							
No.	Species	KARC Shrubland		HSH Shrubland		KFBG Grassland		HSH Grassland	
		1995	1996	1995	1996	1995	1996	1995	1996
1	*Melicope pteleifolia*	--[1]	100	--	98	71	100	70	92.2
2	*Bischofia javanica*	98	--	100	--	25	--	27	--
3	*Mallotus paniculatus*	--	100	--	100	--	97	--	95.6
4	*Gordonia axillaris*	71	100	70	96	22	88	17	82.2
5	*Reevesia thyrsoidea*	--	100	--	91	--	90	--	90
6	*Sapium discolor*	--	100	--	82	64	97	99	78.9
7	*Microcos paniculata*	33	100	93	99	0	47	52	64.4
8	*Diospyros morrisiana*	38	--	52	--	1	--	3	--
9	*Acronychia pedunculata*	10	--	52	--	--	--	--	--
10	*Elaeocarpus sylvestris*	--	94	--	34	--	49	--	7.8
11	*Machilus breviflora*	59	100	85	100	1	26	12	9.1
12	*Cyclobalanopsis championii*	--	100	--	100	--	74	--	88.9
13	*Cyclobalanopsis myrsinifolia*	--	100	--	100	--	73	--	88.9
14	*Castanopsis fissa*	--	100	--	100	--	76	--	90
15	*Lithocarpus glaber*	15	100	78	100	1	79	23	84.4
16	*Choerospondias axillaris*	13	--	16	--	--	--	--	--

[1] "--" indicates that the species was not tested at that site in that year.

Table 3.3

Rodent trapping results. Trapping was carried out at Ho Sheung Heung (HSH) shrubland only in 1995.

	Number of rodents trapped							
Site	KARC shrubland		HSH shrubland		KFBG grassland		HSH grassland	
Species	1995	1996	1995	1996	1995	1996	1995	1996
Rattus rattus flavipectus	--[1]	0	2	1	--	0	--	0
Niviventer fulvescens	--	9	4	2	--	0	--	5
Total	--	9	6	3	--	0	--	5

[1] "--" indicates that trapping was not conducted.

Table 3.4

One-way ANOVA to compare the mean percentage removal of seedless tags, *Microcos paniculata* and *Sapium discolor* seeds. Mean percentage removal data were arcsine transformed. Significant differences further analyzed using SNK test.

ANOVA

Factor	df	MS	F	p
Site	2	10031	16.69	0.000
Error	27	601		
Total	29			

SNK tests

Microcos = *Sapium* > seedless tag

Level	Mean	s.d.
Microcos	75.92	24.33
Sapium	69.47	27.71
Seedless tag	18.12	21.05

Chapter 4

Tree Seed Germination on Degraded Hillsides in Hong Kong.

Abstract

Seed germination of 12 native tree species sown in hillside grassland was compared with germination of the same species sown in the nursery in May 1996. The importance of proper short-term seed storage was overlooked so that many of these species showed abnormally low germination even in the nursery. The experiment was repeated in March 1997, two months earlier than the first trial, to reduce the seed storage period. The germination of 6 of the 12 species sown in hillside grasslands and shrubland was compared with that of those sown in the nursery. Germination of all six species in the nursery was 4 - 15 times higher than in the three grassland and shrubland sites. This suggests that tree seed germination in degraded hillside in Hong Kong is limited by some factor other than seed viability and a likely barrier to natural forest succession. Studies to identify the attributes that cause low germination in degraded hillside are recommended. Such studies would be useful in improving the success of reforestation by direct seeding, which is generally less costly than planting seedlings and more practical at remote and steep sites.

1. Introduction

Apart from seed predation, the lack of increase in seedlings in degraded tropical forest lands despite elevated seed dispersal is likely due to a number of other factors that may inhibit seedling establishment, the first one being low seed germination rate (Holl, 1998c). Seeds in tropical pastures commonly show low rates of germination (Aide & Caveliar, 1994; Holl, 1998c; 1999; Nepstad, 1989). If most tree or shrub seeds that are dispersed into degraded forest land and have escaped from predation cannot germinate, natural forest regeneration will remain very slow.

Tropical rainforest trees are notable for having seeds whose viability lasts, in some cases, only for a few days (Fenner, 1985). In general, germination in most tropical trees occur soon after dispersal (Montagut, 1996). In a survey in Malaysian rainforest, 118 out of 180 species germinated all their viable seeds within 12 weeks. Even amongst the pioneer gap-colonizing species, only 25 % exhibited prolonged germination (Fenner, 1985). A study on tree seed dormancy in Hong Kong found that most of the 26 native tree species tested showed one to five months of innate dormancy in light and partial shade (Zhuang, 1993). However, other experiments have shown that the seeds of many pioneer tropical tree species have the potential for long-term viability in the soil (Dalling *et al.*, 1998). Hong Kong's flora is largely tropical at the generic level but several strictly tropical families of Asian plants such as Burseraceae, Dipterocarpaceae and Myristicaceae are absent from Hong Kong (Dudgeon and Corlett, 1994). Instead, a number of largely extra-tropical families or genera such as Ericaceae, *Machilus* and *Ilex* are better represented. Corlett (1993) showed that there was a fruiting maximum in December-January in secondary shrubland in Hong Kong, which is a time not suitable for seed germination. Most of these species will germinate at the onset of the wet season in April. However experience in the nursery shows that many of these species could germinate in the dry season when irrigated (Unpublished data), suggesting that insufficient water rather than low temperature is limiting seed germination. On the other hand, some other species such as *Machilus breviflora*, *Cratoxylum cochichinensis*, *Melicope pteleifolia* and *Styrax suberifolius* displayed an innate dormancy from 4 months to a year. Summer fruiting-species tend to germinate immediately after dispersal, e.g. *Cordia dichotoma*, *Pygeum topengii*, *Sterculia lanceolata*, *Xylosma longifolium* (Unpublished data). A seed trapping study in degraded hillsides in Hong Kong showed that the most common tree seed species trapped were all pioneer species that produced fruits in the winter (Chapter 2). The most common species, *Schefflera octophylla*, bore ripe fruits near the end of the dry season (February-March), and readily germinated in the nursery when water was given (Unpublished data).

Similar to the abandoned pastures in tropical America, a typical feature of degraded hillside in Hong Kong is the dense coverage of grass species. Many studies have shown that grass cover can promote woody species germination (Aide & Cavelier, 1994; De Steven, 1991; Holl, 1999; Montagut, 1996) although, at the same time, it may inhibit seedling establishment and growth (see Chapter 6). Gerhardt, (1996) demonstrated that tree seed germination in unmown rainforest pasture in Costa Rica was higher than in mown pastures during the initial rains at the beginning of the wet season, but mowing did not affect germination rates of tree seeds sown in the peak of the rainy season. This suggests that soil moisture is an important factor affecting seed germination in degraded forest land. Montagut (1996) showed that high soil humidity, such as that found among the ground cover vegetation in the abandoned pasture, may help to break the nut covering of the *Helicocarpus appendiculatus* seeds and allow complete seed imbibition to occur. Holl (1999) showed that woody species placed in forest pasture in Costa Rica had higher germination rates than when placed in the forest, suggesting that the lack of seed germination is not a major factor limiting recovery.

Whilst there have been a lot of studies on the factors affecting early survival and establishment of tropical tree seedlings transplanted into degraded forest land in the tropics (see Chapter 6), there have been fewer studies that examine seed germination and the early phase of establishment, i.e. from seed germination to the emergence of the first pair of true leaves. The period from germination to establishment is the most critical phase for many plant populations, when they are susceptible to disturbance and environment effects (Itoh, 1995). As the first study on tree seed germination on a degraded hillside, this experiment aimed to compare the germination success of different tree species in degraded hillside sites and in the nursery, to determine whether tree seed germination is low in degraded hillside sites.

2. Materials and Method

Two separate germination tests were conducted. The first one involved 12 native tree species at one hillside site and the second one involved 6 native tree species at 3 hillside sites. The species were selected on the basis of seed availability in that year.

The first trial was conducted at the KFBG grassland in May 1996 (see Chapter 2 for the details of this site). Twelve native tree species were tested, which were all collected in the previous winter from November 95 to February 1996 (Table 4.1). The seeds of all species were stored in open plastic boxes at room temperature without any treatment before the experiment was conducted. However, all floating seeds were discarded as not viable. Sixty seeds of each species were buried 1 – 2 cm below the soil surface in 10 columns and 6 rows in each plot at 1 – 2 cm spacing. Each plot thus had a total of 12 subplots. Each sub-plot was enclosed by a 20 cm x 20 cm x 20 cm wire cage of mesh size 1 cm x 1 cm. All cages could be closed on top by folding up the wire mesh. The grasses in each subplot were clipped to just lower than 20 cm. The cages served to eliminate rodent seed predation, which was found to be a significant factor in reducing seed abundance at this site (see Chapter 3). Each subplot was haphazardly placed in each plot. The set up was replicated 5 times and each plot was haphazardly spread across the grassland. One hundred seeds of each species were sown in budding trays in sandy-loam soil in the nursery to determine seed germination rate under favourable conditions. As such, seeds were irrigated daily and put under 50 % shade. Seed germination was not counted as successful until the first pair of true leaves had emerged. Seed germination was checked and recorded once a month for four months until October 1996.

The second trial was conducted in March 1997 at the KFBG grassland, KARC shrubland and HSH grassland (see Chapter 2 & 3 for the details of the latter two sites). Only six of the 12 species were used in this test (Table 4.1). Ten seeds of each species were used instead of 60 and all 6 species were sown inside the same cage. The set-up was replicated 5 times as in the first trial. In the KARC shrubland, cages were set between woody stems of the shrubs and any grasses inside the cage were clipped as described above. Nursery plots were set up in the same manner but for *Choerospondias axillaris*, 80 seeds were used instead of 100. Seed germination was checked and recorded once a month for four months until July 1997.

3. Results

In the first trial, only 7 species germinated in the field plots and 6 species germinated in the nursery (Table 4.2). The mean germination rate in the field varied from 0.3 - 53.3 %. No *Mallotus paniculatus* seed germinated in the nursery but one *Mallotus* seed germinated in one of the field plots. The mean percentage germination of *Sapium discolor*, *Cyclobalanopsis myrsinifolia*, and *Microcos paniculata* in the field was comparable to the percentage germination in the nursery, but for *Choerospondias axillaris*, *Machilus breviflora* and *Reevesia thyrsoidea*, percentage germination in the nursery was over three times higher.

In the second trial, all six species germinated in the in the nursery and all except *Castanopsis fissa* germinated in the field plots. For all species, the germination rate in the nursery was 4 - 15 times higher than in the field plots (Table 4.3).

Comparing the germination rate of the same species in the two years in the nursery, *Choerospondias* had similar germination rates. *Sapium* and *Microcos* had almost 3 times and 5 times higher germination in 1996 than 1997. On the contrary, *Mallotus*, *Castanopsis* and *Cyclobalanopsis* had over 3 times higher germination in 1997 than 1996.

4. Discussion

The low or zero germination rates of *Choerospondias*, *Castanopsis*, *Cyclobalanopsis championii*, *Lithocarpus glaber*, and *Machilus* even in the nursery might be attributed to low seed viability. The germination rates of these five species, when sown immediately after collection, could reach 70 % in normal nursery conditions with a dormancy period varying from one to four months (unpublished data). Most of the Fagaceae seeds in Hong Kong are particularly susceptible to desiccation when kept in open air (less so for *Cyclobalanopsis myrsinifolia*). *Cyclobalanopsis championii* and *Lithocarpus* could lose their viability in dry weather within a week. This highlights the importance of proper seed storage conditions even for a short period, which was over-looked. The low germination of *Mallotus*, as well as *Gordonia*, in the nursery was a surprise as these species could normally maintain very high viability (up to 50 %) even after being stored at room conditions for months (unpublished data). *Elaeocarpus sylvestris* is a difficult species to germinate. No attempts have so far been successful in germinating this species in standard nursery conditions. It did not germinate in either of the field and control plots in this study.

Due to the problems of low seed viability in the first trial, no attempt was made to draw any conclusions from it. Because of this, the second trial was conducted in March, two months earlier than the first trial, to reduce the seed storage period. All six species showed much higher germination in the nursery (4 - 15 times) than in the field plots, suggesting that seed germination on degraded hillside was limited by some factors other than seed viability.

In summary, this experiment shows that tree seed germination on degraded hillside in Hong Kong, like in abandoned forest pastures in tropical America, is low (Aide & Caveliar, 1994; Holl, 1998c, 1999). This is particularly significant for *Sapium* and *Mallotus* because they were amongst the most common tree species collected in seed traps in the study sites at KARC and KFBG (see Chapter 2). However, this experiment was not designed to assess seedling recruitment beneath tree islands, which is needed to determine if elevated seed input in tree islands would increase seedling recruitment. In addition, this experiment was not designed to assess what attributes on degraded hillside cause low germination. Future seed germination studies on degraded hillside in Hong Kong should test the effect of factors such as soil surface temperature, soil moisture and photon flux density on seed germination. This should be the subject of future seed germination studies on degraded hillside in Hong Kong. Such studies would assist the development of appropriate methods of reforestation by direct seeding, which is considered cost-effective and more practical over large areas and in remote and steep sites where planting seedlings is not practical or too expensive (Cremer, 1990; Sun & Dickson, 1995; 1996). This study also highlights the importance of proper seed storage even for a short period, which should be accounted for in future studies.

Table 4.1

The collection dates of the seeds of the native tree species used in the seed germination tests. All seeds were collected from the wild in Hong Kong. "-" indicates that the species was not tested in that year. Mean seed mass was determined for 50 seeds.

Species	Family	Mean seed mass (g)	Seed collection dates: 1996 trial	1997 trial
Mallotus paniculatus	Euphorbiaceae	0.0148	15 Feb 1996	2 Mar 1997
Gordonia axillaris	Theaceae	0.0158	2 Nov 1996	-
Reevesia thyrsoidea	Sterculiaceae	0.0311	9 Jan 1996	-
Sapium discolor	Euphorbiaceae	0.0466	2 Dec 1995	15 Dec 1996
Microcos paniculata	Tiliaceae	0.1276	2 Dec 1995	15 Dec 1996
Elaeocarpus sylvestris	Tiliaceae	0.2900	2 Nov 1996	-
Machilus breviflora	Lauraceae	0.3390	2 Nov 1996	-
Cyclobalanopsis championii	Fagaceae	0.8068	9 Jan 1996	-
Cyclobalanopsis myrsinifolia	Fagaceae	0.8100	9 Jan 1996	15 Dec 1996
Castanopsis fissa	Fagaceae	1.6300	26 Feb 1996	2 Mar 1997
Lithocarpus glaber	Fagaceae	1.6624	9 Jan 1996	-
Choerospondias axillaris	Anacardiaceae	2.4648	2 Nov 1995	28 Oct 1996

Table 4.2

Percentage seed germination of each species in the seed germination test at KFBG grassland in 1996.

Family	Species	Plot 1	Plot 2	Plot 3	Plot 4	Plot 5	Mean	Nursery
Anacardiaceae	*Choerospondias axillaris*	6.7	5.0	6.7	5.0	1.7	5.0 ± 2.0	16
Euphorbiaceae	*Mallotus paniculatus*	0	0	0	0	1.7	0.3 ± 0.8	0
	Sapium discolor	50.0	46.7	56.7	68.3	45.0	53.3 ± 9.5	59
Fagaceae	*Castanopsis fissa*	0	0	0	0	0	0	0
	Cyclobalabopsis championii	0	0	0	0	0	0	0
	Cyclobalanopsis myrsinifolia	36.7	15.0	5.0	5.0	6.7	13.7 ± 13.5	21
	Lithocarpus glaber	0	0	0	0	0	0	0
Lauraceae	*Machilus breviflora*	1.3	0	1.7	1.7	10	2.9 ± 4.0	11
Theaceae	*Gordonia axillaris*	0	0	0	0	0	0	0
Tiliaceae	*Elaeocarpus sylvestris*	0	0	0	0	0	0	0
	Microcos paniculata	28.3	31.7	31.7	36.7	40.0	33.7 ± 4.6	52
Sterculiaceae	*Reevesia thyrsoidea*	16.7	15.0	11.7	13.3	8.3	13.0 ± 3.2	43

Table 4.3

Percentage seed germination of each species in the field (mean) and in the nursery (actual) in 1997.

Family	Species	KFBG grassland	HSH grassland	KARC shrubland	Nursery
Anacardiaceae	*Choerospondias axillaris*	1.0 ± 1.4	3.4 ± 1.5	0.4 ± 0.9	18.8
Euphorbiaceae	*Mallotus paniculatus*	4.8 ± 2.9	9.0 ± 0.7	1.4 ± 0.9	35.0
	Sapium discolor	1.4 ± 1.1	3.8 ± 1.6	0.4 ± 0.9	20.0
Fagaceae	*Castanopsis fissa*	0	0	0	10.0
	Cyclobalanopsis myrsinifolia	0.8 ± 1.8	5.2 ± 3.4	1.4 ± 0.9	75.0
Tiliaceae	*Microcos paniculata*	1.0 ± 1.4	1.4 ± 1.7	0.4 ± 0.9	10.0

Chapter 5

A Survey of Woody Species on Degraded Hillsides in Hong Kong

Abstract

A survey of woody species in six hillside shrublands in Hong Kong was conducted. The aim of this study was to identify shrub and tree species that are successful in invading degraded hillside shrubland sites in Hong Kong. Twenty-four tree and 29 shrub and vine species were recorded. The dominating shrubs were *Rhodomyrtus tomentosa* and *Baeckea frutescens*. Other common shrubs and vines included *Litsea rotundifolia*, *Rhaphiolepis indica*, *Melastoma sanguineum* and *Embelia laeta*. Most tree species recorded showed some 'pioneer' characters. The most widespread tree species was *Itea chinensis*. Other widespread tree species were *Acronychia pedunculata*, *Schefflera octophylla* (4 sites), *Rhus chinensis*, *Archidendron lucidum*, *Machilus chekiangensis* and *Cratoxylum cochinchinense* (3 sites). The potential of these tree species, especially *Cratoxylum*, *Machilus*, and *Schefflera*, in direct planting for reforestation is discussed.

The height distributions of the trees and shrubs in these shrublands were similar, with most individuals in the height ranges 50 – 150 cm and there were more trees over 2 m in height than shrubs. However, the overall percentage of plants recorded under the canopy of other individuals was small (only 7.6%).

1. Introduction

Less degraded hillside sites in Hong Kong could potentially be restored to closed secondary forest in 30 to 40 years in the absence of fire and other disturbances (Zhuang, 1997; Wong, 1999). Despite many years of reforestation efforts, a vast land area of Hong Kong is still covered by secondary grassland (16.5 %) and shrubland (25 %; Ashworth *et al.*, 1993). Corlett (1999) has pointed out that the repeated failures of planting trials using native tree species at degraded hillsides in Hong Kong in the past suggest that the optimism about our ability to restore species-rich native forests to degraded hillsides by direct planting may be misplaced. The occurrence of natural forest succession on similar sites forming secondary forests with many native species suggests that the problem is establishment rather than growth and survival. These trees invade grassland together with the light-demanding shrubs, which appear to act as invasion foci and nurse species (Zhuang & Corlett, 1997; Corlett, 1999). Shrubs act as nurse crops or foster ecosystems for species-rich native forests by suppressing competing grasses, ameliorating soil and microclimate conditions, and by attracting seed-dispersing vertebrates (Corlett, 1999; Vieira *et al.*, 1994). Yet, no planting trials in Hong Kong have used a mixture of the common shrub and short-lived pioneer tree species of spontaneous shrublands.

Zhuang (1993) suggests that a study on the earliest stage of forest succession, when the first woody species are invading grassland, is needed in Hong Kong. Understanding this stage appears to be essential for the efficient reforestation of the vast areas of secondary grassland in Hong Kong and South China. The aim of this study is to identify shrub and tree species that are successful in invading degraded hillside sites in Hong Kong.

2. Materials and Methods

Six hillside shrubland sites were surveyed between November 1996 and November 1997 (Table 5.1; Figure 5.1). The 6 sites were spread over Hong Kong; dominated by low shrubs (with the majority of woody plants lower than 2 m and the canopy not closed); had no obvious signs of human disturbance such as cutting and firewood collection and with natural forest in the vicinity. At each site, the heights of all woody species in two 4 m x 50 m quadrats (a total of 400 m^2) were measured and each plant was identified to species. Any individual that was obviously growing under the canopy of another individual was noted. The quadrats were haphazardly set at each site. Each site was checked carefully prior to setting the quadrats to make sure that there was no sign of recent burning such as burnt woody stem remains.

3. Results

Twenty-four tree species and 25 shrubs and four climbers (*Embelia laeta*, *E. ribes*, *Dalbergia benthamii* and *Gnetum luofuense*) were recorded in this study. The definition of a tree species follows Zhuang *et al.* (1997) which states that a tree is a self-supporting woody plant attaining a potential maximum height of greater than 5 m.

The percentage of trees was a lot lower than shrubs and climbers at all sites (Table 5.2). Eighteen of these tree species are known to be dispersed by vertebrates, mainly birds, another six species are wind-dispersed (Table 5.3). For shrubs and climbers, 25 species are known to be dispersed by vertebrates (once again, mainly birds), only three species are wind-dispersed and the dispersal agents of two species are not clear but is probably bird as well

All six shrublands were dominated by two shrub species: the animal- dispersed *Rhodomyrtus tomentosa* and the wind-dispersed *Baeckea frutescens*. They together accounted for 37 - 80% of all woody individuals recorded at the six sites (Table 5.4). *Rhodomyrtus* tended to have a more even distribution while *Baeckea* was clumped. Three other shrubs *Litsea rotundifolia, Rhaphiolepis indica and Melastoma sanguineum* and one climber, *Embelia laeta*, also occurred at all sites but in smaller numbers than *Rhodomyrtus* and *Baeckea*. The shrubs *Eurya nitida* and *Breynia fruticosa* were also common but only recorded at five sites.

The most widespread tree species was the wind-dispersed *Itea chinensis*, which occurred at five sites and was second in total abundance (Table 5.4). Other widespread tree species were *Acronychia pedunculata*, *Schefflera octophylla* (all four sites), *Rhus chinensis*, *Archidendron lucidum*, *Machilus chekiangensis* and *Cratoxylum cochinchinense* (three sites). *Zanthoxylum avicennae* and *Glochidion wrightii* also occurred at three sites but in smaller numbers. The wind-dispersed *Gordonia axillaris* was the most abundant tree species but was only recorded at two sites.

For both trees and shrubs, the majority were between 50 - 150 cm in height (Table 5.5). The High West shrubland was the lowest one with no shrubs or trees taller than 150 cm (Appendix 5.1). The Ngau Ngak Shan and Kwun Yam Shan shrubland were relatively taller than the other three sites. There were far more shrubs in the height classes 151 – 200 cm than trees. However, there were more trees over 2 m in height than shrubs. For the 24 tree species, *Litsea glutinosa* and *Artocarpus hypargyrea* were the only two species where the heights of all individuals recorded were less than 50 cm.

A total of 552 woody plants were recorded under the canopy of other individuals, only 7.6 % of the total number recorded (Appendix 5.2). The highest percentage was at Kwun Yam Shan (31%), mostly recorded under *Eurya nitida* and *Rhodomyrtus tomentosa*. In addition, *Acronychia pedunculata* at four sites, *Diospyros morrisiana* and *Itea chinensis* at one site, and *Baeckea frutescens* at two sites, were the main species found shading other plants.

4. Discussion

Although this survey was conducted at six sites in different parts of Hong Kong at different altitudes with different slope orientation, all sites are dominated by more or less the same shrub species. They are, in descending order of abundance, *Rhodomyrtus tomentosa, Baeckea frutescens, Litsea rotundifolia, Embelia laeta, Rhaphiolepis indica* and *Melastoma sanguineum*. In a more comprehensive survey on Lamma Island, *Rhodomyrtus* was also the most common species recorded in grassland and shrubland (Wong, 1999). This may be partly because it is the most fire tolerant species, which regenerates after fire from the woody stem base (Dudgeon & Corlett, 1994). In addition, its seeds are widely dispersed by many vertebrates in Hong Kong, including birds (at least 12 species), civets, macaques (Corlett, 1996; 1998) and rats (see Chapter 2). In fact, this is among the most.abundant species in the seed trapping study in Chapter 2. Zhuang and Corlett (1997) also state that *Rhodomyrtus* is the most abundant light-demanding shrub species that invades fire-protected grassland in Hong Kong. Young secondary forest can be distinguished by the presence of this species, which is eliminated when a closed tree canopy is formed. For the other five species, all except *Baeckea* were categorized as very common or common by Wong (1999). *Baeckea* is the only wind-dispersed species, which may explain its clumped distribution. It is also highly tolerant of fire. The other four species are dispersed by vertebrates, mainly birds. A study showed that *Litsea rotundifolia, Rhaphiolepis indica, Melastoma sanguineum* and *Embelia laeta* showed 60 – 100% regeneration after fire (Chau, 1994). All except *Rhaphiolepis* were collected in the seed trapping study in Chapter 2 and *Melastoma* was amongst the most common seed species collected.

All 24 tree species recorded are locally common in shrubland and secondary forest. Although Zhuang (1993) shows that Swaine and Whitmore's classification of tropical rainforest tree species into pioneers and climax species is not fully applicable in Hong Kong (Swaine & Whitmore, 1988; Swaine, 1996), all of these 24 tree species (with possible exception of *Artocarpus hypargyrea* and *Schima superba*) showed at least some "pioneer" characters, such as small seeds with dormancy, which are copiously produced from early in life and well-dispersed (usually by birds), and short life span.

An obvious weakest of this study is the small number of sites surveyed. As a result, some common tree species in Hong Kong shrubland and young secondary forest, such as *Aporosa dioica*, *Garcinia oblongifolia* and *Sterculia lanceolata* were not recorded in this survey. A comprehensive survey of thirty-one 400 m^2 forest plots in Hong Kong shows that lowland secondary forest is dominated by *Machilus chekiangensis* (and/ or *M. breviflora*), which usually contributed more than half of the total basal area while *Acronychia pedunculata* and *Schefflera octophylla* are common canopy species (Zhuang & Corlett, 1997). All of these three species were amongst the more widespread tree species recorded in this survey. Both *Machilus* and *Schefflera* were collected from the seed trapping study at a shrubland site and *Schefflera* was the second most abundant seed species trapped (Chapter 2). This indicates that at least *Schefflera* and *Machilus* start to invade degraded hillside in the earlier stages of forest succession.

The most common tree species in this study, *Itea chinensis*, is not usually a dominant species in lowland secondary forest. Zhuang (1993) found it in nine out of 25 plots (Zhuang, 1993). The other 3 more common tree species in this study (*Rhus chinensis*, *Archidendron lucidum* and *Cratoxylum cochinchinense*) were also not very common in lowland secondary forest (Zhuang 1993). However, *Cratoxylum* occurs in some tall secondary forests. *Rhus* is confined to shrubland and forest edges.

Finally, although *Gordonia axillaris* was the most abundant tree species in this survey, it was only recorded in two sites. Dudgeon and Corlett (1994) and Chau (1994) have already pointed out that *Gordonia*, being a wind-dispersed species with rather heavy seeds (mean seed mass = 0.0158g; Hau, 1997), has a highly-clumped distribution in Hong Kong. *Gordonia* is also not a very common species in lowland secondary forest (Zhuang, 1993).

This survey shows that the height distributions of the trees and shrubs in these shrublands are similar, with most individuals in the height ranges 50 – 150 cm, and there are more trees over 2 m in height than shrubs. However, the overall percentage of plants recorded under the canopy of other individuals was small (only 7.6%). While it has been shown in Chapter 2 that more tree seeds are dispersed under tree islands in secondary grassland in Hong Kong, this survey does not show more seedling recruitment under tree islands. However, this may be due to the fact that the taller trees or shrubs found in this study were in fact not much taller (less than 1 m on average) than other woody species. The tree islands in the seed trapping study were over 2 m in height; some were 3 m in height. Toh *et al.* (1999) show that scattered, low-growing trees smaller than 3 m in height act as the initial focus for the activities of seed-dispersing birds, but this process is accelerated by the development of taller trees greater than 6 m in height that act as bird perches. To test the theory that seedlings are better established under perches in secondary grasslands (Callaway, 1992; Verdú & García-Fayos, 1998; Vieira *et al.*, 1994) in Hong Kong, a survey of woody species seedling under scattered, tall (>3 m) tree islands in secondary grassland is needed.

Among the common shrubs and tree species found in this survey, *Rhodomyrtus, Litsea, Embelia, Rhaphiolepis, Melastoma*, *Eurya*, *Schefflera*, *Rhus*, and *Machilus* appear to have good potential in direct planting. They all produce fleshy fruits that are dispersed by vertebrates, especially birds. They are common and their seeds are mostly small, making them easy to collect in large quantities. This could significantly reduce the production cost of these species in the nursery and make large-scale production possible. On the other hand, the wind-dispersed species, *Itea*, *Gordonia* and *Cratoxylum* also have rather good potential in direct planting. They can establish well on degraded hillside sites but are apparently limited in distribution due to poor seed dispersal capability (None of their seeds were collected in the seed trapping study in Chapter 2). Their seeds are also very easy to collect in large quantities. *Gordonia*, though slow-growing, has been proven to be successful when planted in a degraded hillside site (see Chapter 7). *Cratoxylum* grows fast in the nursery in comparison with many native tree species in Hong Kong: it can reach 30 cm in less than 6 months (unpublished data). It was included in a planting trial in May 1999 by the Kadoorie Farm and Botanic Garden and had been shown to be rather successful in overcoming the planting shock (unpublished data).

In summary, this and another previous survey (Wong, 1999), have shown that shrubland in Hong Kong is dominated by *Rhodomyrtus tomentosa.* The minor disagreement between these surveys in the other common species recorded may be due to difference in sampling effort. The two common secondary forest tree species *Machilus chekiangensis* and *Schefflera octophylla* were also among the most common tree species in this shrubland survey. This suggests that they start to invade degraded hillside sites at the earlier stages of forest succession. The potential to plant these two species on a large scale is high, especially *Schefflera,* because it produces a large quantity of fruits rather steadily each year, which are easy to collect. Germination and growth rates in the nursery are also high (unpublished data). This survey failed to assess the role of tree islands in accelerating forest regeneration in Hong Kong. A specially designed survey is needed as the understanding of the role of tree islands could assist the formulation of a cost-effective reforestation strategy.

Figure 5.1

Locations of the six study sites

a. High West

b. Ngau Ngak Shan

c. Tai Ho Wan

d. Pak Ngau Shek

e. Ho Sheung Heung

f. Kwun Yam Shan

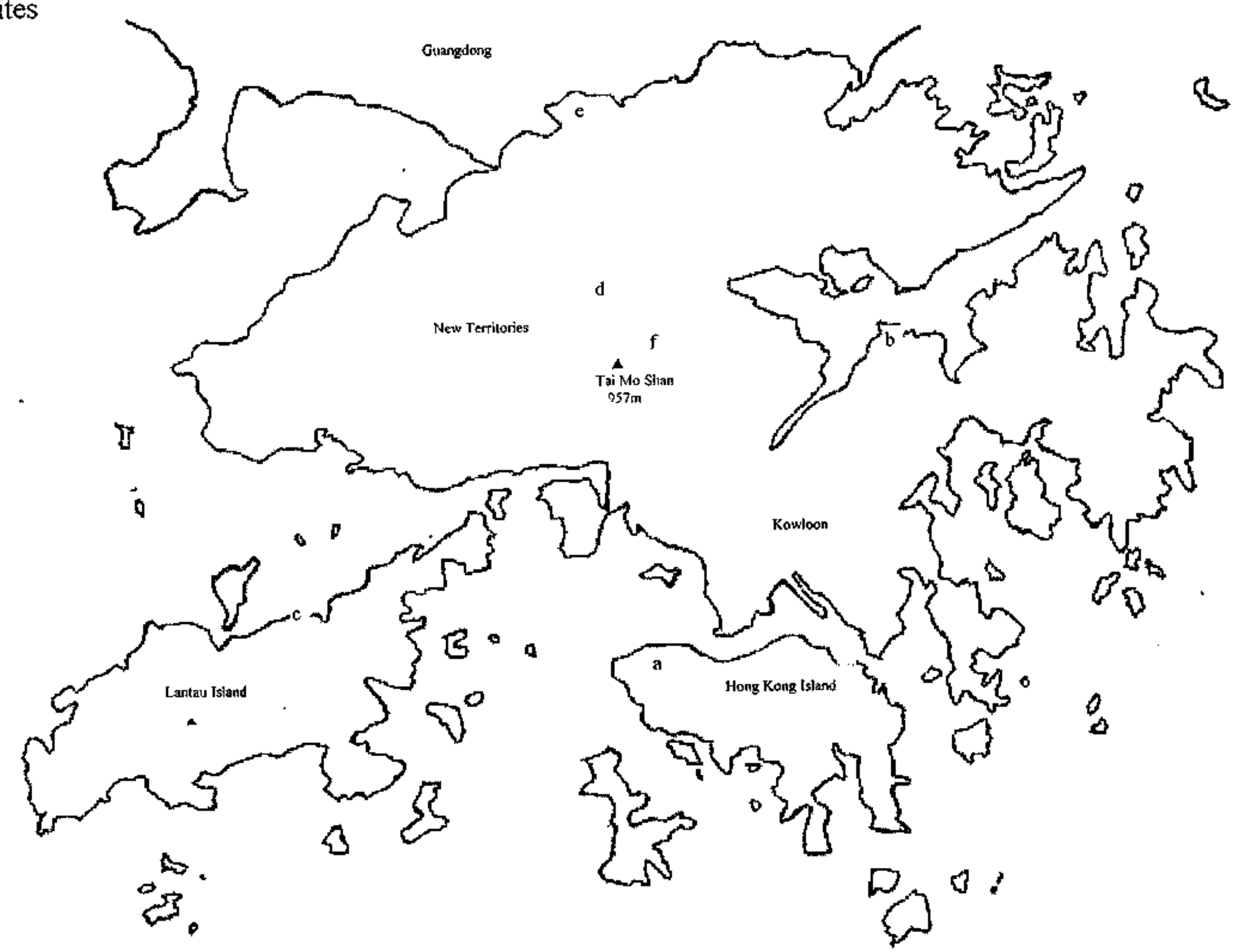


Table 5.1

The dates and locations of the shrubland sites surveyed in this study.

Site	District	Altitude (m)	Slope orientation	Survey dates
Kwun Yam Shan	Central New Territories	400	Southwest facing	26 & 27 NOV 96
Ho Sheung Heung	Northern New Territories	20	Southeast facing	27 & 28 JAN 97
Pak Ngau Shek	Central New Territories	400	East facing	30 & 31 JAN 97
Tai Ho Wan	North Lantau	100	North facing	7 & 8 NOV 97
High West	Western Hong Kong Island	250	Southwest facing	12 & 13 NOV 97
Ngau Ngak Shan	Eastern New Territories	450	Southwest facing	14 & 15 NOV 97

Table 5.2

The number and percentage of trees and shrubs or climbers recorded at different sites in this study.

Site	Trees		Shrubs or climbers		Total Number
	Number	%	Number	%	
Kwun Yam Shan	123	14.9	824	85.1	947
Ho Sheung Heung	27	3.8	707	96.2	734
Pak Ngau Shek	54	5.3	961	94.7	1015
Tai Ho Wan	371	29.7	878	70.3	1249
High West	573	31.6	1240	68.4	1813
Ngau Ngak Shan	404	27.4	1073	72.6	1477

Table 5.3

Known dispersal agents of the woody species recorded in this survey. Trees and shrubs/climbers are listed in descending order in abundance. "?" indicates that the dispersal agents of that species is unclear.

Trees	Dispersal agent	Shrubs	Dispersal agent
Gordonia axillaris	Wind	*Rhodomyrtus tomemtosa*	Bird, civet, macaque, rat
Itea chinensis	Wind	*Baeckea frutescens*	Wind
Rhus chinensis	Bird	*Litsea rotundifolia* [2]	Bird
Archidendron lucidum	Bird[1]	*Embelia laeta**	Civet
Phyllanthus emblica	Civet	*Rhaphiolepis indica* [2]	Bird
Machilus chekiangensis	Bird	*Melastoma sanguineum*	Bird
Cratoxylum cochinchinense	Wind	*Eurya nitida*	Bird
Acronychia pedunculata	Bird	*Phyllanthus cochinchinense*	?
Diospyros morrisiana	Civet[1]	*Breynia fruticosa*	Bird[1]
Schefflera octophylla	Bird	*Clerodendrum fortunatum*	Bird[1]
Zanthoxylum avicennae	Bird[1]	*Ficus variolosa* [2]	Bird
Litsea cubeba	Bird	*Ardisia crenata*	Bird
Litsea glutinosa	Bird	*Melicope pteleifolia*	Bird
Glochidion wrightii	Bird[1]	*Dalbergia benthamii**	Wind
Pentaphylax euryoides	Wind	*Eurya chinensis*	Bird
Schima superba	Wind	*Embelia ribes**	Bird
Sapium discolor	Bird	*Ilex asprella*	Bird
Adinandra millettii	Bird[1]	*Symplocos paniculata* [2]	Bird[1]
Daphniphyllum calycinum	Bird	*Myrsine seguinii*	Bird[1]
Pinus massoniana	Wind	*Mussaenda pubescens*	Bird[1]
Artocarpus hypargyrea	Macaque, civet	*Sarcandra glabra*	Bird
Mallotus paniculatus	Bird	*Croton lachnocarpus*	?
Glochidion lanceolarium	Bird[1]	*Glochidion eriocarpum*	Bird[1]
Bridelia tomentosa	Bird	*Rhamnus crenata* [2]	Bird[1]
		Ficus hirta	Bird
		Gardenia jasminoides	Bird
		Dendrotrophe frutescens	Bird[1], civet
		*Gnetum luofuense**	Macaque, civet
		Strophanthus divaricatus	Wind

* Climbers
[1] Inferred
[2] Tree species that usually grow as shrub in Hong Kong (Zhuang *et al.*, 1997).
Source: Corlett, 1992b, 1996, 1998a and my observations.

Table 5.4

The species and numbers of woody species recorded at each site.

Species	High West	Ngau Ngak Shan	Tai Ho Wan	Pak Ngan Shek	Ho Sheung Heung	Kwun Yam Shan	Total	No. of sites recorded
<u>Trees</u>								
Gordonia axillaris	509	251					760	2
Itea chinensis	9	102	68	18		26	223	5
Rhus chinensis	19		134		18		171	3
Archidendron lucidum			52	3		3	58	3
Phyllanthus emblica			54				54	1
Machilus chekiangensis	10			4		36	50	3
Cratoxylum cochinchinense			36		2	5	43	3
Acronychia pedunculata	7	25		4	5		41	4
Diospyros morrisiana			25			1	26	2
Schefflera octophylla	4	4		3		11	22	4
Zanthoxylum avicennae			2	7		4	13	3
Litsea cubeba				4		8	12	2
Litsea glutinosa						12	12	1
Glochidion wrightii		3		3		5	11	3
Pentaphylax euryoides		11					11	1
Schima superba	8	3					11	2
Sapium discolor				1		8	9	2
Adinandra millettii				6			6	1
Daphniphyllum calycinum	4			1			5	2
Pinus massoniana		5					5	1
Artocarpus hypargyrea	3						3	1
Mallotus paniculatus						3	3	1
Glochidion lanceolarium					2		2	1
Bridelia tomentosa						1	1	1
Total	573	404	371	54	27	123	1552	
<u>Shrubs and Climbers*</u>								
Rhodomyrtus tomentosa	934	360	316	489	270	414	2783	6
Baeckea frutescens	83	491	151	320	307	17	1369	6
Litsea rotundifolia[1]	16	12	17	32	3	179	259	6
*Embelia laeta**	68	37	72	23	40	9	249	6
Rhaphiolepis indica[1]	35	49	80	3	13	13	193	6
Melastoma sanguineum	5	29	12	47	3	53	149	6
Eurya nitida	47	35	2	4		49	137	5
Phyllanthus cochinchinense			112		2		114	2
Breynia fruticosa	4		16	30	45	27	105	5
Clerodendrum fortunatum	26	3	17				46	3
Ficus variolosa[1]	7	34	2			1	44	4
Ardisia crenata		8	23	9		1	41	4
Melicope pteleifolia				1		30	31	2
*Dalbergia benthamii**			9		1	18	28	3
Eurya chinensis	6	5	11				22	3
*Embelia ribes**			1	1		18	20	3
Ilex asprella			17			1	18	2

Cont….

Table 5.4 Cont.

Species	High West	Ngau Ngak Shan	Tai Ho Wan	Pak Ngau Shek	Ho Sheung Heung	Kwun Yam Shan	Total	No. of sites recorded
Symplocos paniculata[1]			17			1	18	2
Myrsine seguinii	9	7					16	2
Mussaenda pubescens					13		13	1
Sarcandra glabra						8	8	1
Croton lachnocarpus		3			4		7	2
Glochidion eriocarpum			3				3	1
Rhamnus crenata[1]				1		2	3	2
Ficus hirta					2		2	1
Gardenia jasminoides					2		2	1
Dendrotrophe frutescens					1		1	1
*Gnetum luofuense**				1			1	1
Strophanthus divaricatus					1		1	1
Total	1240	1073	878	961	707	824	5683	

[1] Tree species that grow as a shrub in Hong Kong; Zhuang *et al.*, 1997.

Table 5.5

Height distribution of the woody species recorded in this study.

Species	Number recorded					Total	No. of sites recorded
cm:	<50	50-100	101-150	151-200	>200		
Trees							
Gordonia axillaris	256	335	127	25	17	760	2
Itea chinensis	35	109	65	8	6	223	5
Rhus chinensis	60	107	3	1	0	171	3
Archidendron lucidum	4	50	4	0	0	58	3
Phyllanthus emblica	0	15	26	11	2	54	1
Machilus chekiangensis	23	22	4	1	0	50	3
Cratoxylum cochinchinense	3	26	14	0	0	43	3
Acronychia pedunculata	2	8	20	8	3	41	4
Diospyros morrisiana	2	5	9	8	2	26	2
Schefflera octophylla	9	10	1	0	2	22	4
Zanthoxylum avicennae	1	2	6	1	3	13	3
Litsea cubeba	3	4	1	4	0	12	2
Litsea glutinosa	12	0	0	0	0	12	1
Glochidion wrightii	0	1	3	6	1	11	3
Pentaphylax euryoides	0	5	3	1	2	11	1
Schima superba	0	10	1	0	0	11	2
Sapium discolor	4	4	1	0	0	9	2
Adinandra millettii	1	1	2	2	0	6	1
Daphniphyllum calycinum	0	4	0	1	0	5	2
Pinus massoniana	0	0	0	3	2	5	1
Artocarpus hypargyrea	3	0	0	0	0	3	1
Mallotus paniculatus	0	0	2	0	1	3	1
Glochidion lanceolarium	0	0	2	0	0	2	1
Bridelia tomentosa	0	0	0	1	0	1	1
Total	418	718	294	81	41	1552	
Shrubs and Climbers*							
Rhodomyrtus tomentosa	839	1486	366	92	0	2783	6
Baeckea frutescens	305	756	272	31	5	1369	6
Litsea rotundifolia[1]	139	65	40	13	2	259	6
*Embelia laeta**	56	157	22	11	3	249	6
Rhaphiolepis indica[1]	49	121	18	5	0	193	6
Melastoma sanguineum	25	84	33	7	0	149	6
Eurya nitida	39	54	30	12	2	137	5
Phyllanthus cochinchinense	9	68.	37	0	0	114	2
Breynia fruticosa	46	56	10	0	0	105	5
Clerodendrum fortunatum	22	19	5	0	0	46	3
Ficus variolosa[1]	4	14	24	2	0	44	5
Ardisia crenata	8	32	1	0	0	41	4
Melicope pteleifolia	29	2	0	0	0	31	2
*Dalbergia benthamii**	6	19	2	0	1	28	3
Eurya chinensis	3	13	6	0	0	22	3
*Embelia ribes**	4	5	8	3	0	20	3
Ilex asprella	2	6	10	0	0	18	2
Symplocos paniculata[1]	5	10	3	0	0	18	2

Cont….

Table 5.5 Cont.

Species	Number recorded					Total	No. of sites recorded
cm:	<50	50-100	101-150	151-200	>200		
Myrsine seguinii	0	14	2	0	0	16	2
Mussaenda pubescens	2	4	6	1	0	13	1
Sarcandra glabra	6	2	0	0	0	8	1
Croton lachnocarpus	4	1	2	0	0	7	2
Glochidion eriocarpum	0	3	0	0	0	3	1
Rhamnus crenata[1]	0	3	0	0	0	3	2
Ficus hirta	0	0	1	0	1	2	1
Gardenia jasminoides	0	2	0	0	0	2	1
Dendrotrophe frutescens	0	1	0	0	0	1	1
*Gnetum luofuense**	0	0	1	0	0	1	1
Strophanthus divaricatus	0	0	0	1	0	1	1
Total	1599	2996	896	178	14	5683	

[1] Tree species that grow as a shrub in Hong Kong; Zhuang *et al.*, 1997.

Chapter 6

A Field Experiment Examining the Factors Affecting the Early Survival and Growth of Four Native Tree Species Planted on a Degraded Hillside Grassland in Hong Kong

Abstract

The effects of seasonal drought, weed competition and soil fertility on the early survival and growth of four native tree species over two years on a degraded hillside grassland in Hong Kong were studied in a field transplant experiment using multiway factorial analysis of variance. The results suggested that the removal of grass competition by the use of herbicides was very effective in enhancing the growth of *Schima superba* and *Schefflera octophylla* seedlings. However, the survival of the other two species, *Castanopsis fissa* and *Sapium discolor*, was significantly impaired and the growth rate of *Castanopsis* was negatively affected by herbicide treatment. While the survival responses of the species to fertiliser treatment cannot be distinguished, it was found that adding fertiliser either had positive or no effect on seedling growth. Providing irrigation in the dry season had no effect on seedling survival but enhanced basal diameter increment in *Schima* and *Castanopsis*. The growth responses of the seedlings to various combinations of treatment suggested that herbicide had the overriding effect in enhancing growth, whereas dry season irrigation played a minor role. Fertiliser application had an intermediate effect. This suggests that the current practice of manual weeding in forest rehabilitation projects in Hong Kong may not be effective in eliminating belowground competition, which may have led to slow growth and high mortality of many native species in past planting trials. The light-demanding, early pioneer species, *Schefflera*, had the highest overall growth rate. This supports the use of more native species with similar characteristics in future planting trials on degraded hillside in Hong Kong. In contrast, the poor survival of another early pioneer species, *Sapium*, in this and in another study in Chapter 7, requires further investigation. Transplantation losses were less than 2 % in all

species and survival remained above 75 % before the onset of the dry season. This indicates that good post-nursery care can minimise transplantation stress and reduce losses.

1. Introduction

Hill soils in Hong Kong are generally described as thin, acidic, low in organic matter, and nutrient poor, with low to very low levels of nitrogen, phosphorus and calcium (Grant, 1983; Dudgeon & Corlett, 1994). Marafa and Chau (1999) have shown that hill fires can raise soil pH and significantly reduce the total soil nitrogen (75%), phosphorus (66%), and other nutrients, as well as the cation exchange capacity. Some studies in the tropics have found that low soil nutrients on degraded lands significantly limit tree seedling growth (Bowman & Panton, 1993; Ferraz, 1993; Fetcher *et al.*, 1996). Aide and Cavelier (1994) showed that the slow growth rate of tree seedlings in the degraded grassland soils compared to forest soils in a lowland tropical forest in Sierra Nevada de Santa Marta, Colombia, was associated with lower cation-exchange capacity, calcium, magnesium and potassium. On the other hand, others have found that soil nutrients were not a limiting factor (Holl, 1998a; Nepstad, *et al.* 1991; Zeng & Whelan, 1993). In some studies, adding fertilisers significantly enhanced tree seedling growth rates (Dalling & Tanner, 1995; Nussbaum *et al.*, 1995; Otsamo *et al.*, 1995). A study in Mexico found that the growth responses of 34 woody seedling species from a tropical deciduous forest to fertiliser addition displayed a continuum, suggesting that resource utilisation and tolerance are different between the species (Huante *et al.*, 1995). However, another study, in the Amazon rainforests, found that species did not differ significantly in their growth responsiveness to increased nutrient supply (Coomes & Grubb, 1998). None of the above studies reported any correlation between soil nutrients and seedling mortality.

Grasses grow from the protected base of the leaf, making them particularly tolerant of repeated burning, which accounts for the large extent of grassland on degraded hillsides in Hong Kong (Dudgeon & Corlett, 1994). The most common hillside grasses in Hong Kong are *Ischaemum* spp. and *Arundinella setosa* (Chau, 1994; Wong, 1999). *Imperata cylindrica* dominates the grasslands that cover large areas of former forest land in the moist tropical regions, especially in Southeast Asia. In Indonesia alone, the estimated *Imperata* grassland cover varies from 20 to 64.5 million hectares (Kosonen *et al.*, 1997). These grasslands are formed after repeated logging and fires and are difficult to reforest because of physical difficulty in planting, grass competition and allelopathy, fire susceptibility, as well as soil degradation and compaction (Otsamo *et al.*, 1995). In Hong Kong, *Imperata cylindrica* is common only on recently-burnt hillside sites but does not cover extensive areas (Griffiths, 1983; Ng Sai Chit, pers. comm.).

Apart from fire, invasion by grasses is the most serious contemporary ecological threat to the restoration and maintenance of degraded land in the tropics (Janzen, 1988; Otsamo *et al.*, 1995). Nepstad *et al.* (1990) point out that the dense root systems of grasses produce severe soil moisture deficits during the dry season and compete with tree seedlings for available soil nutrients in abandoned Amazonian forest pastures. Competition with grasses has also been suggested as an important factor limiting tree seedling survival and growth on degraded hillsides in Hong Kong (Lay *et al.*, 1999). Many studies in the tropics have already shown that competition with grasses is a significant factor limiting tree seedling establishment on degraded forest lands (Nepstad *et al.*, 1991; Chapman & Chapman, 1997; Coomes and Grubb, 1998; Davis *et al.*, 1998; Holl, 1998a; Lowery *et al.*, 1993; Steven, 1991). Relatively fewer studies have found that competition with grasses did not affect tree seedling establishment (Aide & Cavelier, 1994; Coomes & Grubb, 1998). Killing grasses with herbicide in degraded pasture in Brazil was also found to enhance tree seed germination (Holl, 1998b). Dominant grasses species on degraded hillside in Hong Kong seem less aggressive than those species (e.g. *Imperata cylindrica*) in other parts of the tropics. Tree seedlings planted on degraded hillside in Hong Kong were usually not overgrown by grass species (personal observation). In this respect, grass competition with tree seedlings likely occur underground.

Seasonal drought has been found to be another limiting factor to tree seedling growth and survival on degraded forest lands in the wet-dry tropics (Bowman & Panton, 1993; Gerhardt, 1993; Nepstad *et al.*, 1991). Nepstad *et al.* (1996) found that the pre-dawn xylem pressure potential (a measurement of drought stress) of all three tested tree seedling transplants was significantly lower (2 to 5 times) in the abandoned pastures than in experimental tree fall gaps in the dry season. Drought stress in the dry season was one of the factors correlated with higher levels of seedling mortality in abandoned pastures. Uhl *et al.* (1981) also showed that seasonal drought reduced seedling survival in cut and burn sites in the Amazon.

The aims of the experiment reported here were to test if the survival and initial establishment of native tree seedlings planted in a degraded upland grassland in Hong Kong are affected by low soil nutrients, competition with weeds and seasonal drought.

2. Materials and Methods

2.1 Study sites

The study was conducted at a grassland site on the northeast side of Tai Mo Shan in the northern New Territories. It is situated within the Kadoorie Farm and Botanic Garden and just outside the Tai Mo Shan Country Park (see Figure 2.1, Chapter 2). The grassland is on a northwest-facing steep slope (slope gradient is about 40°) at 550 m above sea level. It is dominated by *Arundinella* sp., *Ischaemum* sp., *Eulalia* sp., *Eragrostis* sp., *Cymbopogon* sp. and *Miscanthus sinensis*. There are some small trees and shrubs (up to 1 m. in height) in the slightly more sheltered part of the grassland. They include *Archidendron lucidum*, *Litsea cubeba*, *Rhodomyrtus tomentosa* and *Melastoma sanguineum*. In a ravine less than 1 km to the southwest of this grassland is a secondary woodland which is dominated by *Machilus* spp. The grassland is not managed, except that the grasses within a 2 m wide strip above and below the footpath are removed annually to promote the growth of wild orchids. Other than this, there is no human disturbance at this site. The planting plots were set below the footpath and outside the managed area.

The evergreen broadleaf forest remnants in steep ravines on Tai Mo Shan suggest that this grassland site would have been covered by dense forest in the past. Old tea terraces can still be seen today after hill fire when the grass cover is burnt away, and Tai Mo Shan was once famous for the production of green "Cloud and mist tea". The earliest record of tea cultivation on Tai Mo Shan is in the Xin An Gazetteer of the Qing Dynasty in 1688 (Dudgeon & Corlett, 1994). This suggests that the original forests at this site might have been destroyed for tea production. However, the period when tea production was active at this site is not known. Like many other hillside grasslands in Hong Kong, this site is also prone to wild fire. Although there are no nearby fire sources such as hiking trails and graves, it has been affected by the spread of fires started on the southern slopes of Tai Mo Shan. Prior to the start of this experiment, the last fire recorded at this site was 1989 and it was burnt again on 9 February 1999, after the field experiment was completed. This grassland is a typical degraded hillside site in Hong Kong where natural colonisation is extremely slow or absent. The grassland site is underlain by volcanic rocks and soils are volcanic-derived (Geotechnical Control Office, 1988a).

2.2 Soil analysis

The total nitrogen and available phosphorus of the soil in the study site were determined in December 1997. A 50 m transect was haphazardly placed across the study site and twenty surface soil samples (0-20cm), about one litre each, were collected at positions selected by random numbers.

The total nitrogen was determined using the Kjeldahl procedure (Allen, 1989; Bremner & Mulvaney, 1982). Each soil sample was ground and sieved through a 100 mesh (0.14 mm) and 0.5 to 1.0 g (±0.1 mg) was added to a 250 ml L digestion tube. Two Kjeltabs Cu 3.5 were added to the soil sample and 12 ml L concentrated sulphuric acid was slowly added to the digestion tube. The mixture was digested for 60 minutes at 400 °C in the Tecator 2000 Digestion System. Cooled digest was diluted with distilled water for filtration. Filtered digest was made up to 100 ml L with distilled water in a volumetric flask to form the aliquot. Three ml of the aliquot were mixed with 50 ml L 40% NaOH in a digestion tube for steam distillation using the Tecator Kjeltec System 1026 Distilling Unit. The distillate was titrated with M/70 HCl using 7 ml L boric acid indicator. For each sample, two steam distillations and titrations were conducted to obtain a mean % N figure.

The total nitrogen of the soil samples was calculated as % N according to the following formula (Tecator 1026 manual):

$$\% \text{N} = \frac{14.01 \times \left[\begin{matrix} \text{ml of titrant} \\ \text{of sample} \end{matrix} - \begin{matrix} \text{ml of titrant} \\ \text{of blanks} \end{matrix} \right] \times \text{Molarity of standard acid}}{\text{Sample weight (g)} \times 10}$$

Available phosphorus of the soil samples was determined by the Mehlich No.1 (Double Acid) extraction (Anonymous, 1992). Each soil sample was ground and sieved through a 100 mesh (0.14 mm) and 5 g was added into a 50 ml L extraction bottle. Twenty-five ml L of extracting reagent (0.05N HCl in 0.025N H_2SO_4) were then added and the mixture was shaken for 5 minutes on a reciprocating shaker at 200 oscillations per minute. Two ml of the filtered extract were pipetted into a spectrophotometer cuvet with 23 ml L working solution (10 ml L ascorbic acid + 20 ml L sulphuric-molybdate solution + 970 ml L distilled water). The mixture was allowed to stand for 20 minutes before the absorbance at 880 nm was measured with the spectrophotometer. The spectrophotometer was zeroed against a blank consisting of the extracting reagent. The soil phosphorus content (in mg/L) was obtained using a calibration curve. The absorbance at 880 nm of standard phosphorus solutions containing 1, 2, 5, 10, 15 and 20 mg/L phosphorus (prepared by dissolving ammonium dihydrogen phosphate in the extracting reagent) was measured with the spectrophotometer. On a semi-log graph paper, the absorbency was plotted on the logarithmic scale and the standard phosphorus content (in mg/L) was plotted on the linear scale to form the calibration curve.

2.3 Experimental design

This experiment was a four-factor analysis of variance design with fixed effects, comprising a total of seven treatments and one control (8 planting plots) with five replications (5 blocks, Figure 6.1) and four native tree species. The factors were irrigation in the dry seasons (factor A), herbicide application (factor B), fertilisation (factor C), and tree species. The first three factors had two levels, the presence and absence of the factor, while the last factor had four.

In each planting plot, ten 18-month old seedlings of each of the four native tree species were planted at 1.2 – 1.5 m spacing to form a rectangular grid with five rows and eight columns, in June/July 1995. Planting holes were created by a pickaxe and about two times the size of the seedling soil tube, which was about 10 cm x 10 cm x 15 cm. Starting at the top left point of each grid, from left to right and then top to bottom, the species to be planted at each point was pre-determined by drawing a named paper from a black box, which contained 10 papers of each of the four species. The eight treatments (planting plots) in each block were randomly mixed in the same way, except that the irrigated plots were grouped together for ease of management (Figure 6.1). Since each of the 8 treatments was replicated 5 times, a total of 1,600 seedlings were planted.

The four native tree species used in this study were *Schima superba* (Theaceae), *Castanopsis fissa* (Fagaceae), *Schefflera octophylla* (Araliaceae) and *Sapium discolor* (Euphorbiaceae). Apart from *Pinus massoniana*, *Schima superba* was the most widely planted native species and most successful native species in the afforestation history of Hong Kong (Corlett, 1999). *P. massoniana* has not been planted for many years due to serious pest problems but *Schima* is still commonly planted (Dudgeon & Corlett, 1994). *Castanopsis fissa* is one of the ten most commonly planted in Hong Kong in the last 5 years, with a total of 110,000 seedlings (Agriculture and Fisheries Department homepage, www.info.gov.hk/afd/trees/treedata.htm). A 35-year long monitoring study at the Dinghushan Biosphere Reserve in Guangdong, South China, revealed that *Castanopsis* was a dominant species between the succession from coniferous/broadleaf mixed forest to the subtropical broadleaf evergreen forests (Peng & Fang, 1994; Peng, 1996b). *Castanopsis fissa* was gradually replaced by the less light demanding species, like *Schima superba* and *Castanopsis chinensis*, before the true climax, shade-tolerant species *Cryptocarya chinensis* and *C. concinna* were established. Both *Sapium discolor* and *Schefflera octophylla* are common pioneer tree species in Hong Kong. The light-demanding, deciduous, *Sapium* is a dominant species in tall shrubland and very young secondary forest and, when planted in open habitats, it has high survival and growth rates but is strictly shade intolerant (Zhuang, 1993). *Schefflera* is a dominant species in young secondary forests in Hong Kong (Dudgeon & Corlett, 1994) but was found to be among the more shade tolerant species of the 42 native tree species in a field trial (Zhuang, 1993). *Sapium discolor* has been used in afforestation in Hong Kong since the early 1990s but with limited success (Corlett, 1999). *Schefflera* has not been commonly used in afforestation but has been tried in some planting trials (Corlett, 1999; Zhuang, 1993; Lay *et al.*, 1999).

All seedlings were purchased from a local commercial nursery but were produced in Guangdong Province. The 18-month old container-grown seedlings (container size: 6 cm x 6 cm x 10 cm) were delivered to the planting site in April 1995. The heights of *Schima* and *Sapium* ranged from 30-50 cm, and *Schefflera* and *Castanopsis* ranged from 30-40 cm. Common precautions during transportation and transplantation as described in Forest Restoration Research Unit (1998) were taken. In addition, all seedlings that were damaged during transportation were discarded. They were hardened on site without shade and with a minimum amount of water given only after two consecutive dry days. All dead and sick-looking seedlings were discarded. The seedlings were planted between 6 June and 6 July 1995. To standardise the workmanship of planting, I planted over 95 % of the seedlings. The other 5 % were planted by my friends and KFBG workers under my instructions. Seedlings planted on dry days were watered to reduce transplantation loss.

2.3.1 Irrigation treatment

A three metre vegetated gap was left between the irrigated and non-irrigated plots in each block to avoid water percolation through the soil to the non-irrigated plots. Water was given once per week in the dry season (from November to March inclusive). During this period, the soil suction, which is a direct measure of the availability of soil moisture for plant growth (Rowell, 1994), was determined by a tensiometer (Model 2900F Quickdraw Soil Moisture Probe form Soilmoisture Equipment Corporation, P.O. Box 30025, Santa Barbara, CA 93105, U.S.A.). Measurements at 20 cm below the soil surface were taken three times per month after at least two consecutive rainless days. Five measurements were randomly (by random numbers from a calculator) taken along a 100 m transect which was set haphazardly across the site but not crossing the planting plots. A preliminary field test at the study site on 29 October 1995 indicated that the mean soil suction reading[1] 20 cm below the soil surface decreased from 54 kPa to 25.8 kPa five minutes after one litre of water was given. The mean soil suction then gradually increased to 37.2 kPa in 60 minutes. In medium-textured soils, most plants grow best where the soil suction readings are kept between 20 and 60 kPa (Anonymous 1987). A zero soil suction reading indicates that the soil is saturated. If soil suction readings reach 80 kPa, it can be detrimental to the plant. There was no water supply system on site and water had to be brought in by trolleys and buckets. Irrigation was done by a hose driven by gravity from a large bucket. Water was hosed smoothly around the stem of each seedling in a circle of about 0.5m in diameter and surface runoff was minimized by regulating the tap of the hose. Approximately 0.75 L of water was given to each plant once a week in the 95/96 dry season (equivalent to 15 mm rainfall/month if the water does not spread horizontally outside a vertical cylinder). A preliminary analysis of the data from the first year indicated that irrigation was not significantly affecting the survival and growth of the seedlings so that the water supply was doubled to 1.5 L per plant per week in the 96/97 dry season (equivalent to 30 mm rainfall/month). No water was given when it rained on the scheduled irrigation day. In the 95/96 dry season, each plant was irrigated 19 times (with one rainy day) with a total of 14.25 L of water. In the 96/97 dry season, each plant was irrigated 14 times (with four rainy days) with a total of 21 L of water. Although the volume of water given in the second year does not sound too bad, the real rainfall-equivalent will be considerably less than 30 mm rainfall/month.

2.3.2 Herbicide treatment

The grass cover of the plots with herbicide treatment was clipped and removed manually by a sickle prior to application of the post-emergence, non-selective herbicide Roundup (Monsanto Company, U.S.A.). The active ingredient is glyphosate, which has proven to be an effective herbicide in forestry in South America and Papua New Guinea (Lowery *et al.*, 1993). Herbicide was applied once with a back-pack hand sprayer in April 95, 96 and 97. The first application was two months before the seedlings were planted. For the second and third applications, a plastic shield was used to protect the seedlings

[1] Note that the soil suction readings given throughout this thesis refer to the values given by the Quickdraw Soil Moisture Probe. These are not the same as the soil water potential, which is always negative.

from direct contact with the spray and the area within 20 cm radius from each seedling was not sprayed but pulled up by hand. Fifty ml of Roundup was added to an eight-litre sprayer, which was then filled up with water. Each herbicide treated planting plot (about 90 m^2) was covered by one sprayer load in each application. This rate, which is approximately half that recommended, appeared to be adequate as grass did not reappear until the following March.

2.3.3 Fertilisation treatment

A slow release NPK complex fertiliser named Nitrophoska Permanent 15-9-15-2 (BASF, Germany) was used (Table 6.1). Twelve g Nitrophoska were buried about 5 cm below the soil surface 10 cm up slope of each seedling in July 1995, soon after all seedlings were planted. A preliminary analysis of the data from the first year indicated that fertilisation was not significantly affecting the survival and growth of the seedlings, so an additional 25 g Nitrophoska was applied in a similar manner in March 1996 and 1997.

2.4 Data collection and statistical analysis

Transplantation losses were assessed in July/August 1995, one month after planting. Survival was recorded again in October 1995, March and October 1996, and June/July 1997. The basal diameter and stem height of each seedling was recorded one month after planting as the baseline for determining the growth rate. The same measurements were repeated in October 1995, March and October 1996 and June/July 1997. The relative height increment per year (RHI) was calculated using RHI = $[\ln(H_2) - \ln(H_1)]$ / Time in years, where H_1 and H_2 were seedling heights in July 95 and 97 respectively (Coomes & Grubb, 1998). The relative basal diameter increment per year (RBDI) was calculated in the same way.

In early June 1995, just before the seedlings were planted in the field, thirty of each species were haphazardly selected and set aside for biomass determination. These seedlings were dried in an oven at 70 °C for 72 hours in July 1995 and the dry weights of leaves, stems and roots were measured. The mean weight of each species was taken as the initial dry weight for the growth trials and was used to calculate the relative biomass increment per year (RBI) by the same formula as RHI and RBDI. In September 1997, two *Schima* and *Schefflera* seedlings were randomly harvested (by a draw of numbered papers) from each treatment plot. Each seedling was carefully dug up by a hoe and a pickaxe so as to minimise the loss of roots in the ground, and the dry weights determined in the same way. The other two species were not harvested because too few seedlings had survived in certain treatment plots.

The mean percentage seedling survival and growth rate of each species in each treatment were compared by 4-way ANOVA using Minitab Release 12.1. Percentage seedling survival data were arcsine transformed following standard practice (Fry, 1993). Seedling growth data were transformed according to Taylor's power law. The power (p) for the transformation (x^p) was calculated by $p = 1 - b/2$, where b was the slope of the regression line of the $\log_{10}$ of the group variances upon $\log_{10}$ group means (Fry, 1993). In all ANOVA, the homogeneity of variance was checked by manually calculating Hartley's F_{max} and Cochran's C (Fry, 1993). Normal probability plots produced by Minitab Release 12.1 were used to check the residuals for normality (Fry, 1993). When the assumptions of an ANOVA did not hold, outliers were removed. A new power of transformation was determined and a new ANOVA was conducted. These steps were repeated until the assumptions of the ANOVA were maintained. The Newman-Keuls multiple range (SNK) test was used for multiple comparisons.

3. Results

The total nitrogen content of the soil at the study site ranged from 0.007 to 0.043% (0.0204 ± 0.0084%, n = 20) and the available soil phosphorus content ranged from 0.0010 to 0.0040% (0.00225 ± 0.00079%, n = 20). The soil suction readings of the study site never reached the lethal range of about 80 kPa during either winter dry season. The highest single reading was 72 kPa on 19 November 1996 (Figure 6.2). The soil suction readings in the dry season of 1996/97 (58.427 ± 10.692) were significantly higher than in 1995/96 (44.95 ± 8.031; $F = 90.71$, $p < 0.001$; Figure 6.2).

Transplantation losses were less than 2 % in all species and survival remained over 75 % before the onset of the first dry season in November 1995 (Table 6.2). Overall two-year seedling survival rates ranged from 97.8% in *Schima* to 15.8% in *Sapium* (Table 6.3). All seedling mortality appeared to be due to desiccation, with no visible pest or herbicide damage. Fertiliser had a small but significant negative effect on seedling survival. The significant interaction between herbicide and species results from a large and significant negative herbicide effect on the survival of *Castanopsis* and *Sapium* (both with over 20% lower mean percentage survival when herbicide was applied) but no effect on the other two species. *Schefflera* had similar survival to *Castanopsis* when no herbicide was given but significantly higher survival (>20%) than *Castanopsis* when herbicide was applied. The significant interaction between irrigation and species was weak ($p = 0.051$) and irrigation had no significant effect on any one species although the effects are generally negative. However, irrigation had a similar effect as herbicide on between species difference. *Schefflera* had significantly higher survival (>15%) than *Castanopsis* when irrigation was provided, while there was no significant difference between the two when irrigation was absent.

Sapium was excluded from the statistical analysis of the growth parameters as its survival rate was zero in 40% of the plots. Most of the surviving seedlings decreased in height due to broken stems (wind damage).

The increase in mean stem height and basal diameter of *Schima* in 2 years was 43.43 cm ± 26.90 (N=371) and 9.75 mm ± 6.71 (N=384) respectively. The increase in mean stem height and basal diameter of *Schefflera* in 2 years was 12.25 cm ± 14.30 (N=262) and 8.16 mm ± 6.26 (N=263) respectively. Finally, the increase in mean stem height and basal diameter of *Castanopsis* in 2 years was 29.04 cm ± 22.46 (N=137) and 3.43 mm ± 2.57 (N=228) respectively.

Overall mean RHI over two years ranged from 0.29 in *Schima* to 0.63 in *Schefflera* (Table 6.4). The significant interaction between herbicide and fertiliser was because when herbicide and/or fertiliser treatments were provided, seedlings had much higher mean RHI (> 0.1) than when neither treatments was given. The significant interaction between fertiliser and species results from a significant and strong positive fertiliser effect on *Schima* and *Castanopsis* (≥0.1 difference in mean RHI) but no effect on *Schefflera*. The effect of fertiliser on the between species difference was not significant. The interpretation of the significant interaction term between irrigation, herbicide and species is complex. While different treatment combinations of irrigation and herbicide had no effect on *Schefflera*, the treatment combinations with herbicide generally had positive effects on *Schima* but negative effects on *Castanopsis*. The effects of different treatment combinations of irrigation and herbicide on between species difference was varied but *Schefflera* always had significant and much higher mean RHI (>0.2) than the other two species.

Overall mean RBDI over two years ranged from 0.16 in *Castanopsis* to 0.46 in *Schima* (Table 6.5). The interpretation of the significant interaction term between irrigation, herbicide and fertiliser on mean RBDI is very complex. In general, treatment combinations with both herbicide and fertiliser had significantly higher mean RBDI than all other treatment combinations. In contrast, treatment combinations without either herbicide or fertiliser had significantly lower mean RBDI than all other treatment combinations. Irrigation showed either no or weak positive effects on mean RBDI in all treatment combinations. The significant interaction between irrigation and species results from a significant but weak positive irrigation effect on the mean RBDI of *Schima* and *Castanopsis* (a difference of only 0.04 in RBDI for both species), but no effect on *Schefflera*. The effect of irrigation on between species difference in RBDI was not significant. The significant interaction between herbicide and species results from a significant and strong positive effect on the mean RBDI of *Schima* and *Schefflera* (both with a >0.3 difference in mean RBDI), but no effect on *Castanopsis*. The interpretation of the effect of herbicide on between species difference in RBDI is very complex and no generalisation can be made. The significant interaction term between fertiliser and species shows that fertiliser had a significant and moderate effect on *Schima* and *Castanopsis* but no effect on *Schefflera*. *Schima* had significantly higher RBDI than

Schefflera (a difference of 0.12 in mean RBDI) when fertiliser was applied, but these species were not significantly different when no fertiliser was given.

Overall mean RBI of *Schefflera* was much higher than *Schima* (Table 6.6). Overall, fertiliser had a significant but weak positive effect on the mean RBI. The significant interaction term between irrigation, herbicide and species shows that for both species, treatment combinations with herbicide had significant and strong positive effects on RBI (a difference of about 1.0 in RBI in both cases). On the other hand, irrigation showed no effect on either species in all treatment combinations. The differences in mean RBI between the two species were significant only when both irrigation and herbicide were present, and *Schefflera* had slightly higher mean RBI than *Schima* (a difference of only 0.35 in mean RBI).

4. Discussion

The soil analysis results indicated that the total nitrogen (mean 0.02%) and available phosphorus (mean 0.002%) of the study site were lower than reported for many other sites in Hong Kong and Guangdong. The total soil nitrogen of several degraded hillside sites studied previously in Hong Kong ranged from 0.027 to 0.28% (Marafa & Chau, 1999; Lay, *et al.*, 1999; Chen *et al.*, 1996; Wong, 1999). Wong (1999) showed that hill soil nutrient contents are negatively correlated with human disturbance and suggests that deforestation, trampling and hill fire remove or decrease vegetative ground cover and accelerate soil erosion. The available soil phosphorus of several other sites studied previously in Hong Kong ranged from 0.005 to 0.0975% (Lay, *et al.*, 1999; Chen *et al.*, 1996). A comprehensive study of soils in the nearby Guangdong Province showed that the mean total nitrogen of the top soil of eleven degraded hillside sites at 300 – 700 m above sea level with granite-derived soils ranged from 0.043 to 0.195% (the A horizon) and 0.032 to 0.063% (the B horizon) respectively (Liu, 1996). For mean available phosphorus, it ranged from 0.0010 to 0.0083% (the A horizon) and 0.0013 to 0.0045% (the B horizon) respectively (Liu, 1996). The lower levels were recorded on badly eroded sites with sparse grass cover and the higher levels were recorded on sites with coniferous/broadleaf mixed forest. Sites covered with grass, grass/shrub mix and shrubland had relatively low total nitrogen and available phosphorus contents. The study site has long been deforested and subject to fire disturbance and the soil nutrients were expected to be low.

The almost 100% seedling survival for all species after 30 days and the still very high survival rates before the first dry season (Table 6.2) shows that the post-nursery care and the precautions during seedling transportation and transplantation are effective in minimising transplantation loss.

The results on seedling survival are both interesting and surprising. Firstly, while *Schima* and *Schefflera* showed no significant response to herbicide treatment, *Castanopsis* and *Sapium*, surprisingly, had significantly lower survival in herbicide treated plots. This is in contrast to the results of similar studies elsewhere in the tropics (e.g. Chapman & Chapman, 1997; De Steven, 1991). It is possible that due to the lack of a ground cover in herbicide treated plots, evaporation from the soil surface is higher and this, in turn leads to higher mortality of the less drought tolerant species, especially in the winter dry season. Unfortunately, the soil suction measurements were taken at the site where the grass cover was intact and therefore cannot tell the difference in soil moisture between the herbicide treated plots and the plots without herbicide treatment. Nevertheless, this suggests that below ground competition does not limit early seedling survival at this site. Secondly, the reason for the significant but weak negative effects of fertiliser on seedling survival are not known. Nevertheless, this shows that low soil nutrients also do not limit early seedling survival. Finally, no one species showed significant difference in survival with and without irrigation. The weak significant interaction term between irrigation and species (P=0.051) was accounted for by the difference between *Schefflera* and *Castanopsis*, where *Schefflera* showed significantly higher survival than *Castanopsis* (>15%) when irrigation was provided. In general, the weak or zero effect of irrigation on seedling survival suggests that seasonal drought also does not limit early seedling survival at this site (provided that seedlings were not growing on exposed soils). In fact, the mean monthly soil suction reading in both dry seasons was less than 60 kPa (Figure 6.2), which was below the lethal range for plants

(~80 kPa). This indicates that the seedlings at this site were not under severe water stress in the winter dry season.

Sapium discolor had very low survival in this study, which was much lower than in two other planting trials (Lay *et al.*, 1999; Zhuang, 1993). Since most of the surviving seedlings have broken stems, the low survival is probably attributed to wind damage as the study site is very exposed. It is also possible that this site (550 m a.s.l.) is above its natural altitudinal range. However, its survival was also very low at two other sites with much lower altitudes (see Chapter 7).

The significant interactions between herbicide and fertiliser on seedling height increment and between irrigation, herbicide and fertiliser on basal diameter increment show that herbicide and fertiliser together effectively promote seedling growth. This suggests that both belowground competition and low soil nutrients together significantly impair seedling growth at this site. Irrigation in the winter dry season had shown either weak or no effect on seedling growth, which may be because the growth rate in the cool winter is low anyway. However, it is also possible that the amount of water added was not sufficient to bring about notable benefits to tree growth.

The results on seedling growth have also shown that the three species react differently to the experimental treatments. While herbicide could enhance the growth of *Schima* (all 3 parameters) and *Schefflera* (all except height), *Castanopsis* showed either no (basal diameter) or negative (height) response. These results show that removing below ground competition can significantly improve early seedling growth at this site (*Schima* and *Schefflera* in this study). This suggests that below ground competition is an important barrier to forest succession. By slowing down seedling growth, the chance of fire arresting succession is higher. Irrigation had either no or weak positive effects on the three species in the three growth parameters, which also suggests that seasonal drought is not a major filter-barrier to forest succession. Finally, fertiliser had positive effects on biomass and basal diameter increment on *Schima* and *Castanopsis* but no effect on

Schefflera. This suggests that low soil nutrient is an important selective filter on tree species that could establish on this hillside site.

The negative response of *Castanopsis* to herbicide is interesting. Zhuang (1993) suggested that the higher mortality and poorer growth of *Castanopsis* in the open site than in the forest in her planting trials might be due to desiccation and high temperature in the open. Without grass cover in the herbicide treated plots, the soil surface tends to be hotter and drier, which may explain why *Castanopsis* had slower growth. It is also interesting that while both *Schima* and *Castanopsis* showed positive growth responses in height and basal diameter to fertiliser treatment, *Schefflera* showed no response. In fact, *Schefflera* had the highest overall height and biomass increment. *Schefflera* was also the most abundant tree species in the seed trapping study (see Chapter 2) and was among the most common tree species in hillside shrublands (see Chapter 5). All these results highlight the very high potential of *Schefflera* in reforestation in Hong Kong. This also suggests that species with similar growth habits such as light-demanding, fast-growing, common on shrublands and young secondary forests, should be explored for wider-use in reforestation in Hong Kong (e.g. *Evodia lepta*, *Mallotus paniculatus, Diospyros morrisiana*). Alias *et al.* (1998) also found that light-demanding native tree species grow faster and supported their selection for trial planting.

In conclusion, a major weakness of this study is that only one site and very few species were involved. Nevertheless, the results of this experiment suggest that underground competition and low soil nutrients in hillside grassland limit early seedling growth but not survival in Hong Kong. Seasonal drought apparently did not limit seedling survival and only weakly affected seedling growth. This study also highlights the difference in response of different species to the treatments. This suggests that no single planting method is suitable for all species and that concentration on a group of more successful species appears necessary. To this end, the framework species method developed in the wet tropics of north Queensland, is worthy of consideration (Forest Restoration Research Unit, 1998; Goosem & Tucker, 1995).

Figure 6.1

Design and layout of the seedling transplant experiment. "1" indicates that the factor is present and "0" indicates that the factor is absent. Each treatment plot has four species.

Design:

Treatment U_{ABC}		Factor A: Irrigation in the dry season			
		1		0	
		Factor B: Herbicide		Factor B: Herbicide	
		1	0	1	0
Factor C: Fertilisation	1	U_{111}	U_{101}	U_{011}	U_{001}
	0	U_{110}	U_{100}	U_{010}	U_{000} Control

Treatment mix in each block:

Block 1

Plot 1 U_{101}	Plot 2 U_{111}
Plot 3 U_{110}	Plot 4 U_{100}
Plot 5 U_{000}	Plot 6 U_{010}
Plot 7 U_{001}	Plot 8 U_{011}

Block 2

Plot 1 U_{010}	Plot 2 U_{000}
Plot 3 U_{011}	Plot 4 U_{001}
Plot 5 U_{101}	Plot 6 U_{110}
Plot 7 U_{100}	Plot 8 U_{111}

Block 3

Plot 1 U_{100}	Plot 2 U_{110}
Plot 3 U_{111}	Plot 4 U_{101}
Plot 5 U_{001}	Plot 6 U_{000}
Plot 7 U_{010}	Plot 8 U_{011}

Block 4

Plot 1 U_{000}	Plot 2 U_{011}
Plot 3 U_{010}	Plot 4 U_{001}
Plot 5 U_{100}	Plot 6 U_{101}
Plot 7 U_{111}	Plot 8 U_{110}

Block 5

Plot 1 U_{110}	Plot 2 U_{101}
Plot 3 U_{111}	Plot 4 U_{100}
Plot 5 U_{001}	Plot 6 U_{011}
Plot 7 U_{010}	Plot 8 U_{000}

Block layout in the field (not in scale):

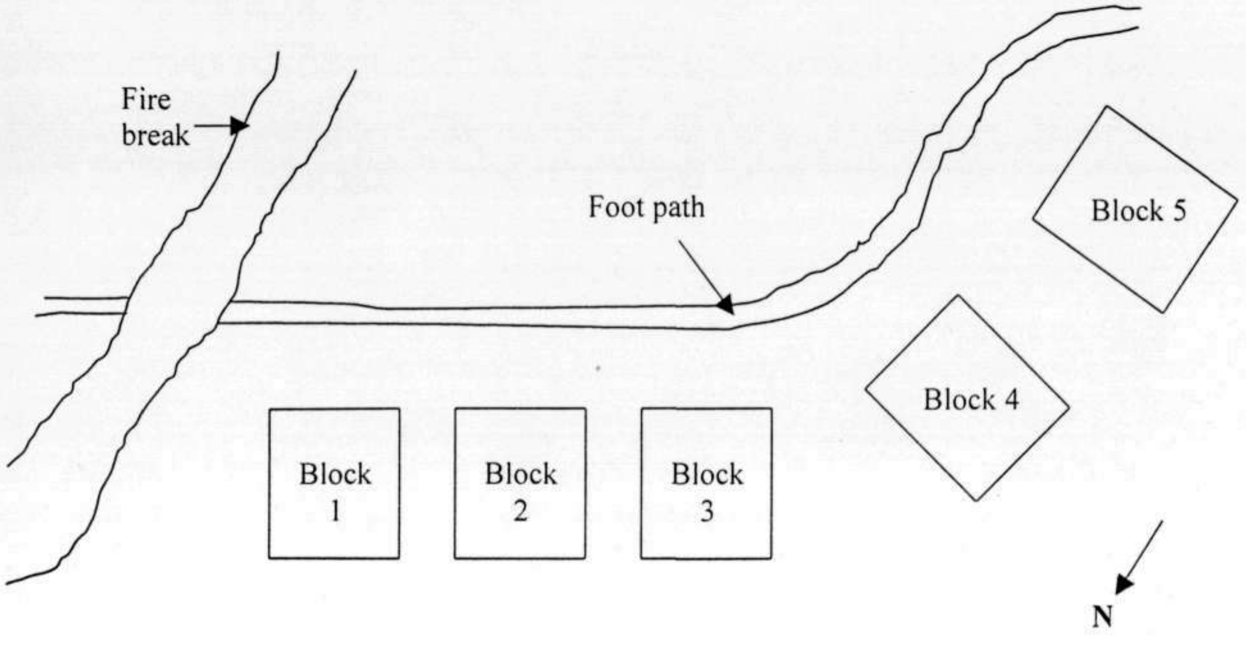


Figure 6.2

Mean monthly soil suction readings at the study site in the 1995-96 and 1996-97 dry seasons. A zero soil suction reading indicates that the soil is saturated. If the reading reaches 80 kPa, it can be detrimental to the plant. The total rainfall in these periods were recorded at the Hong Kong Observatory in Kowloon.

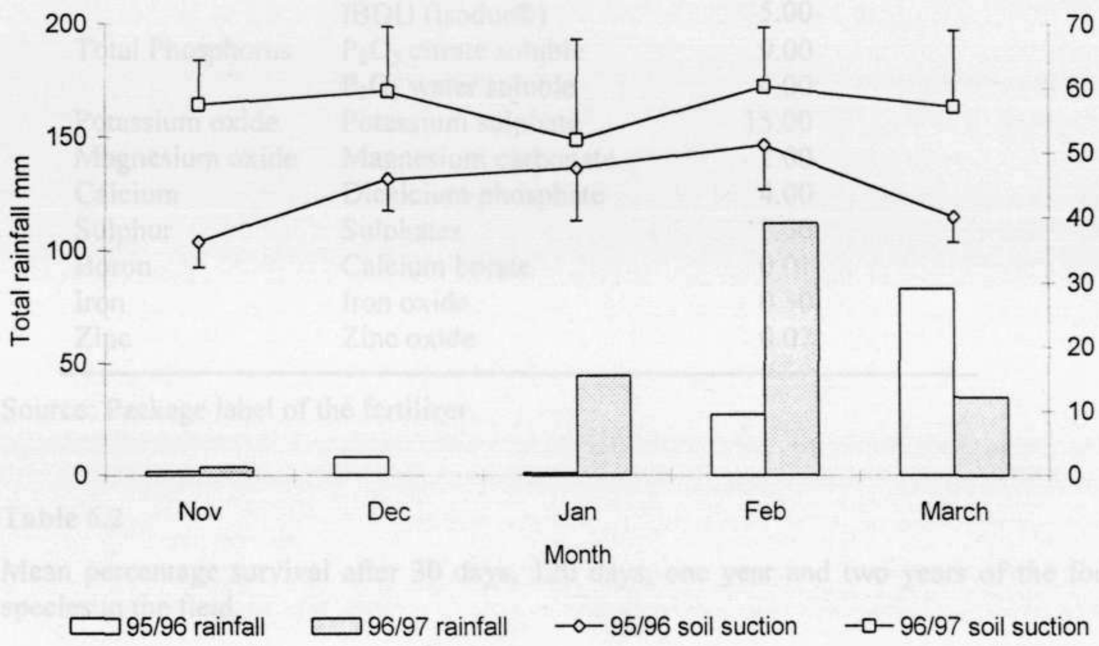


Table 6.1

Nutrient content of Nitrophoska Permanent 15-9-15-2 slow-release fertiliser.

Nutrient	Form	Percentage composition
Total Nitrogen	Ammonium (NH_4)	5.80
	Nitrate (NO_3)	4.20
	IBDU (Isodur®)	5.00
Total Phosphorus	P_2O_5 citrate soluble	9.00
	P_2O_5 water soluble	4.00
Potassium oxide	Potassium sulphate	15.00
Magnesium oxide	Magnesium carbonate	2.00
Calcium	Dicalcium phosphate	4.00
Sulphur	Sulphates	7.00
Boron	Calcium borate	0.01
Iron	Iron oxide	0.30
Zinc	Zinc oxide	0.07

Source: Package label of the fertilizer.

Table 6.2

Mean percentage survival after 30 days, 120 days, one year and two years of the four species in the field.

Species	Mean % survival in			
	30 days Jul 1995	120 days Oct 1995	1 year Jul 1996	2 years Jul 1997
Schima superba	100.00 ± 0.00	100.00 ± 0.00	97.75 ± 4.80	97.75 ± 5.30
Schefflera octophylla	98.25 ± 4.46	93.50 ± 8.93	87.25 ± 14.14	71.00 ± 18.37
Castanopsis fissa	98.75 ± 4.04	90.00 ± 11.77	75.75 ± 17.96	63.75 ± 21.57
Sapium discolor	98.75 ± 4.04	75.75 ± 15.51	47.00 ± 23.77	15.75 ± 17.67

Table 6.3

Four-way ANOVA of the survival rates over two years of the four native tree species under different treatments. The percentage survival data were arcsine transformed. Significant differences were further analyzed using SNK tests. Mean percentage survival is included in the SNK results.
*** $p < 0.001$, ** $p < 0.01$, * $p < 0.05$; NS, not significant.

ANOVA

Factor	df	MS	F	P	Significant
Irrig	1	118.7	0.80	0.373	NS
Herb	1	1288.3	8.68	0.004	**
Fert	1	697.1	4.70	0.032	*
Species	3	31575.3	212.68	0.000	***
Irrig x Herb	1	110.0	0.74	0.391	NS
Irrig x Fert	1	67.3	0.45	0.502	NS
Irrig x Species	3	395.3	2.66	0.051	*
Herb x Fert	1	233.0	1.57	0.213	NS
Herb x Species	3	1550.7	10.45	0.000	***
Fert x Species	3	137.5	0.93	0.430	NS
Irrig x Herb x Fert	1	249.8	1.68	0.197	NS
Irrig x Herb x Species	3	107.3	0.72	0.540	NS
Irrig x Fert x Species	3	202.5	1.36	0.257	NS
Herb x Fert x Species	3	11.6	0.08	0.972	NS
Irrig x Herb x Fert x Species	3	118.8	0.80	0.496	NS
Error	128	148.5			
Total	159				

SNK tests

1. Herbicide: With herbicide < No herbicide
   58.5 65.6

2. Fertiliser: With fertiliser < No fertiliser
   59.4 64.8

3. Species: *Schima* > *Schefflera* = *Castanopsis* > *Sapium*
   97.8 71.0 63.8 15.8

4. Herbicide x Species

Schima + herbicide 98.5	=	*Schima* - herbicide 97.0
Schefflera + herbicide 76.0	=	*Schefflera* - herbicide 66.0
Castanopsis + herbicide 53.0	<	*Castanopsis* - herbicide 74.5
Sapium + herbicide 6.5	<	*Sapium* - herbicide 25.0

Cont...

Table 6.3 Cont.

Schima + Herbicide 98.5	>	*Schefflera* + Herbicide 76.0	>	*Castanopsis* + Herbicide 53.0	>	*Sapium* + Herbicide 6.5
Schima – Herbicide 97.0	>	*Schefflera* – Herbicide 66.0	=	*Castanopsis* – Herbicide 74.5	>	*Sapium* – Herbicide 25.0

5. Irrigation x Species

Schima + irrigation 97.5	=	*Schima* - irrigation 98.0
Schefflera + irrigation 73.5	=	*Schefflera* - irrigation 68.5
Castanopsis + irrigation 56.5	=	*Castanopsis* - irrigation 71.0
Sapium + irrigation 15.0	=	*Sapium* - irrigation 16.5

Schima + Irrigation 97.5	>	*Schefflera*+ Irrigation 73.5	>	*Castanopsis* + Irrigation 56.5	>	*Sapium* + Irrigation 15.0
Schima – Irrigation 98.0	>	*Schefflera* – Irrigation 68.5	=	*Castanopsis* – Irrigation 71.0	>	*Sapium* – Irrigation 16.5

Table 6.4

Four-way ANOVA of the relative height increment per year (RHI) of the three native tree species under different treatments over two years. The power of transformation of the RHI data was 0.6381 according to Taylor's power law. Forty-three outliers were removed. Significant differences were further analyzed using SNK tests. Mean RHI is included in the SNK results.
*** $p < 0.001$, ** $p < 0.01$, * $p < 0.05$; NS, not significant.

ANOVA

Factor	df	MS	F	P	Significant
Irrig	1	0.07915	4.63	0.032	*
Herb	1	0.07669	4.48	0.035	*
Fert	1	1.33416	78.01	0.000	***
Species	2	7.93333	463.86	0.000	***
Irrig x Herb	1	0.00322	0.19	0.664	NS
Irrig x Fert	1	0.00045	0.03	0.871	NS
Irrig x Species	2	0.00119	0.07	0.933	NS
Herb x Fert	1	0.13032	7.62	0.006	**
Herb x Species	2	0.86794	50.75	0.000	***
Fert x Species	2	0.15614	9.13	0.000	***
Irrig x Herb x Fert	1	0.01039	0.61	0.436	NS
Irrig x Herb x Species	2	0.05509	3.22	0.040	*
Irrig x Fert x Species	2	0.00179	0.10	0.901	NS
Herb x Fert x Species	2	0.00122	0.07	0.931	NS
Irrig x Herb x Fert x Species	2	0.03474	2.03	0.132	NS
Error	746	0.01710			
Total	769				

SNK tests

1. Irrigation: With irrigation > No irrigation
   0.42 0.39

2. Herbicide: With herbicide > No herbicide
   0.45 0.36

3. Fertiliser: With fertiliser > No fertiliser
   0.45 0.37

4. Species: *Schefflera* > *Schima* = *Castanopsis*
   0.63 0.29 0.30

5. Herbicide x Fertiliser

+ herbicide + fertiliser		+ herbicide – fertiliser		– herbicide + fertiliser		– herbicide – fertiliser
	>		=		>	
0.48		0.43		0.42		0.31

Cont....

Table 6.4 Cont.

6. Herbicide x species

Schima + herbicide 0.36	>	*Schima* – herbicide 0.22		
Schefflera + herbicide 0.65	=	*Schefflera* – herbicide 0.61		
Castanopsis + herbicide 0.21	<	*Castanopsis* – herbicide 0.33		
Schefflera + Herbicide 0.65	>	*Schima* + Herbicide 0.36	>	*Castanopsis* + Herbicide 0.21
Schefflera – Herbicide 0.61	>	*Castanopsis* – Herbicide 0.33	>	*Schima* – Herbicide 0.22

7. Fertiliser x species

Schima + fertiliser 0.34	>	*Schima* – fertiliser 0.24		
Schefflera + fertiliser 0.66	=	*Schefflera* – fertiliser 0.61		
Castanopsis + fertiliser 0.37	>	*Castanopsis* – fertiliser 0.22		
Schefflera + Fertiliser 0.66	>	*Castanopsis* + Fertiliser 0.37	=	*Schima* + Fertiliser 0.34
Schefflera – Fertiliser 0.61	>	*Castanopsis* – Fertiliser 0.22	=	*Schima* – Fertiliser 0.24

8. Irrigation x Herbicide x species

Schima + Irrigation + Herbicide 0.37	=	*Schima* – Irrigation + Herbicide 0.35	>	*Schima* + Irrigation – Herbicide 0.24	=	*Schima* – Irrigation – Herbicide 0.20
Schefflera + Irrigation + Herbicide 0.64	=	*Schefflera* + Irrigation – Herbicide 0.65	=	*Schefflera* – Irrigation + Herbicide 0.66	=	*Schefflera* – Irrigation – Herbicide 0.58
Castanopsis – Irrigation – Herbicide 0.36	=	*Castanopsis* + Irrigation – Herbicide 0.30	>	*Castanopsis* + Irrigation + Herbicide 0.23	=	*Castanopsis* – Irrigation + Herbicide 0.19

Table 6.5

Four-way ANOVA of the relative basal diameter increment per year (RBDI) of the three native tree species under different treatments over two years. The power of transformation of the RBDI data was 1.0528 according to Taylor's power law. Thirty-five outliers were removed. Significant differences were further analysed using SNK tests. Mean RBDI is included in the SNK results.
*** $p < 0.001$, ** $p < 0.01$, * $p < 0.05$; NS, not significant.

ANOVA

Factor	df	MS	F	P	Significant
Irrig	1	0.1488	6.85	0.009	**
Herb	1	9.5165	438.25	0.000	***
Fert	1	1.3736	63.26	0.000	***
Species	2	1.9001	87.50	0.000	***
Irrig x Herb	1	0.1422	6.55	0.011	*
Irrig x Fert	1	0.0003	0.02	0.900	NS
Irrig x Species	2	0.0802	3.69	0.025	*
Herb x Fert	1	0.0106	0.49	0.485	NS
Herb x Species	2	1.7707	81.54	0.000	***
Fert x Species	2	0.1265	5.82	0.003	**
Irrig x Herb x Fert	1	0.0922	4.25	0.040	*
Irrig x Herb x Species	2	0.0218	1.00	0.366	NS
Irrig x Fert x Species	2	0.0137	0.63	0.533	NS
Herb x Fert x Species	2	0.0320	1.47	0.230	NS
Irrig x Herb x Fert x Species	2	0.0100	0.46	0.630	NS
Error	851	0.0217			
Total	874				

SNK tests

1. Irrigation: With irrigation > No irrigation
   0.41 0.37

2. Herbicide: With herbicide > No herbicide
   0.52 0.27

3. Fertiliser: With fertiliser > No fertiliser
   0.44 0.35

4. Species: *Schima* > *Schefflera* > *Castanopsis*
   0.46 0.39 0.16

5. Irrigation x Herbicide

+ irrigation + herbicide		− irrigation + herbicide		+ irrigation − herbicide		− irrigation − herbicide
0.53	=	0.51	>	0.30	>	0.24

Cont….

Table 6.5 Cont.

6. Irrigation x species

Schima + irrigation 0.48	>	*Schima* – irrigation 0.44		
Schefflera + irrigation 0.40	=	*Schefflera* – irrigation 0.38		
Castanopsis + irrigation 0.31	>	*Castanopsis* – irrigation 0.27		
Schima + Irrigation 0.48	>	*Schefflera* + Irrigation 0.40	>	*Castanopsis* + Irrigation 0.31
Schima – Irrigation 0.44	>	*Schefflera* – Irrigation 0.38	>	*Castanopsis* – Irrigation 0.27

7. Herbicide x species

Schima + herbicide 0.62	>	*Schima* – herbicide 0.29		
Schefflera + herbicide 0.53	>	*Schefflera* – herbicide 0.23		
Castanopsis + herbicide 0.29	=	*Castanopsis* – herbicide 0.27		
Schima + Herbicide 0.62	>	*Schefflera* + Herbicide 0.53	>	*Castanopsis* + Herbicide 0.29
Schima – Herbicide 0.29	=	*Castanopsis* – Herbicide 0.27	>	*Schefflera* – Herbicide 0.23

8. Fertiliser x species

Schima + fertiliser 0.52	>	*Schima* – fertiliser 0.40		
Schefflera + fertiliser 0.40	=	*Schefflera* – fertiliser 0.38		
Castanopsis + fertiliser 0.33	>	*Castanopsis* – fertiliser 0.24		
Schima + Fertiliser 0.52	>	*Schefflera* + Fertiliser 0.40	>	*Castanopsis* + Fertiliser 0.33
Schima – Fertiliser 0.40	=	*Schefflera* – Fertiliser 0.38	>	*Castanopsis* – Fertiliser 0.24

Cont….

Table 6.5 Cont.

9. Irrigation x Herbicide x Fertiliser

+ irrigation + herbicide + fertiliser 0.57	=	− irrigation + herbicide + fertiliser 0.56
	∨	
+ irrigation + herbicide − fertiliser 0.49	=	− irrigation + herbicide − fertiliser 0.47
	∨	
	+ irrigation − herbicide + fertiliser 0.36	
	∨	
	− irrigation − herbicide + fertiliser 0.28	
	∨	
+ irrigation − herbicide − fertiliser 0.22	=	− irrigation − herbicide − fertiliser 0.20

Table 6.6

Four-way ANOVA of the relative biomass increment per year (RBI) of *Schima superba* and *Schefflera otophylla* seedlings under different treatments over two years. The power of transformation of the RBI data was 1.02535 according to Taylor's power law. Eighteen outliers were removed. Significant differences were further analysed using SNK tests. Mean RBI is included in the SNK results.
*** $p < 0.001$, ** $p < 0.01$, * $p < 0.05$; NS, not significant.

ANOVA

Factor	df	MS	F	P	Significant
Irrig	1	0.0034	0.08	0.777	NS
Herb	1	37.2775	867.04	0.000	***
Fert	1	3.2926	76.58	0.000	***
Species	1	0.4245	9.87	0.002	**
Irrig x Herb	1	0.0869	2.02	0.158	NS
Irrig x Fert	1	0.0551	1.28	0.260	NS
Irrig x Species	1	0.0838	1.95	0.165	NS
Herb x Fert	1	0.0036	0.08	0.772	NS
Herb x Species	1	0.7517	17.48	0.000	***
Fert x Species	1	0.0386	0.90	0.345	NS
Irrig x Herb x Fert	1	0.0347	0.81	0.371	NS
Irrig x Herb x Species	1	0.2887	6.72	0.011	*
Irrig x Fert x Species	1	0.1297	3.02	0.085	NS
Herb x Fert x Species	1	0.0822	1.91	0.169	NS
Irrig x Herb x Fert x Species	1	0.1215	2.83	0.095	NS
Error	123	0.0430			
Total	138				

SNK tests

1. Herbicide: With herbicide > No herbicide
   1.67 0.64

2. Fertiliser: With fertiliser > No fertiliser
   1.33 1.03

3. Species: *Schefflera* > *Schima*
   1.24 0.11

Cont....

Table 6.6 Cont.

4. Herbicide x species

Schima + Herbicide 1.56	>	*Schima* – Herbicide 0.67
Schefflera + Herbicide 1.53	>	*Schefflera* – Herbicide 0.60
Schima + Herbicide 1.56	=	*Schefflera* + Herbicide 1.53
Schima – Herbicide 0.67	=	*Schefflera* – Herbicide 0.60

5. Irrigation x Herbicide x species

Schima + Irrigation + Herbicide 1.51	=	*Schima* – Irrigation + Herbicide 1.62	>	*Schima* + Irrigation – Herbicide 0.65	=	*Schima* – Irrigation – Herbicide 0.68
Schefflera + Irrigation + Herbicide 1.86	=	*Schefflera* – Irrigation + Herbicide 1.69	>	*Schefflera* + Irrigation – Herbicide 0.57	=	*Schefflera* – Irrigation – Herbicide 0.65

Schefflera + Irrigation + Herbicide 1.86	>	*Schima* + Irrigation + Herbicide 1.51
Schefflera + Irrigation – Herbicide 0.57	=	*Schima* + Irrigation – Herbicide 0.65
Schefflera – Irrigation + Herbicide 1.69	=	*Schima* – Irrigation + Herbicide 1.62
Schefflera – Irrigation – Herbicide 0.65	=	*Schima* – Irrigation – Herbicide 0.68

Chapter 7

The Early Survival and Establishment of Ten Tree Species Planted in Three Degraded Hillside Sites in Hong Kong

Abstract

The early survival and growth of nine native and one naturalised exotic tree species planted in an upland grassland, a lowland grassland and a lowland shrubland were studied for two years. As in the previous study in Chapter 6, transplantation loss was successfully minimised by proper post-nursery care. All species except *Sapium discolor* had high survival rates over two years in both grassland sites (70 – 100 %) and most species showed no difference between the two grassland sites. In the shrubland, *Sapium*, *Castanopsis fissa* and *Sterculia lanceolata* had very low survival rate (0 - 26%). Other species had moderate to high survival rate (43 - 98%). The naturalised exotic *Cinnamomum camphora* had high survival at all sites (95-100%) but very low growth rates. The relative growth rates of *Cyclobalanopsis neglecta*, *Machilus breviflora*, *Choerospondias axillaris* and to a lesser extent, *Schefflera octophylla*, were higher than other species at all three sites. *Cyclobalanopsis* and *Choerospondias* also had the highest final mean stem height (96 – 140 cm and 100 – 156 cm). While these two species would never have dispersed to the study sites, others would have suffered from serious seed predation and low field germination rates. When these barriers were overcome, they grow well without any need to ameliorate other barriers such as low soil nutrients and below ground competition. This suggests that low seed input, seed predation and low field germination rates together form a rather tough barriers to forest succession on degraded hillside sites than poor soil nutrient and below ground grass competition. All except *Sapium* have the potential for wider use in reforestation in Hong Kong, especially the four faster growing species.

1. Introduction

It is estimated that there are 70 million hectares of forest requiring rehabilitation in the Asia-Pacific region (Ådjers *et al.*, 1996; Elliott *et al.*, 1995). Tropical and sub-tropical

China has over 460,000 km^2 of degraded lands that formerly supported forests and most of them require rehabilitation (Sun *et al.*, 1995). Grasses and shrubs cover about 51% of the land area of Hong Kong, mostly on degraded hillside sites that formerly supported forest (Ashworth *et al.*, 1993). There has been an increase in efforts to restore habitats degraded by human activities throughout the world (Lesica & Allendorf, 1999). In addition to the purpose of biodiversity conservation (Miyawaki, 1993), rehabilitation of degraded tropical forest lands is important in increasing the biological productivity of such lands so as to support people and reduce pressures for degradation of additional tropical forest lands (Lovejoy, 1985). In recent years, restoring degraded tropical forest lands has been an active area of scientific research (Brown & and Lugo, 1994; Holl & Kappelle, 1999; Lamb, 1994; Lamb *et al.*, 1997; Lovejoy, 1985; Sarre, 1995).

Exotic species have played a major role in industrial plantation in the tropics. Three genera, *Pinus*, *Eucalyptus* and *Tectona*, made up 85% of the industrial plantations in the tropics in 1980 (Evans, 1992). Until 1987, for example, the commercial reforestation programmes in the tropical lowlands of Costa Rica had principally used four exotic species (González & Fisher, 1994). Besides the use of exotic tree species in commercial forestry, there has been a rapid growth, mainly in developing countries, in the use of exotic species in non-conventional forestry, such as for fuel-wood production and for restoration of badly eroded or exhausted lands (Zobel *et al.*, 1987).

The heavy reliance on a limited number of mostly exotic species in industrial plantations has been attributed to the requirements of the manufacturing process (Fisher 1999). Most wood produced in industrial plantations is converted to pulp and pulping can be carried out most efficiently if all the wood is of one kind. This means not only the same species but also a few select varieties that have very similar specific gravity and secondary plant chemical content. In addition, these exotic species have been thoroughly domesticated so that foresters know how to reproduce and grow them with ease (Fisher, 1999). Domestication is a lengthy process that requires the understanding of the basic physical, chemical and biological properties of the species. This kind of knowledge exists for nearly all temperate zone tree species but only a few tropical tree species (Richardson, 1998). After many years of research, exotic species have been tested, and seed sources selected and genetically improved to produce stocks of very high productive potential (Haggar *et al.*, 1998). In many cases, exotic species are also preferred because they grow faster than native species (Richardson, 1998). They are also often better suited to planting in grassland and shrubland, where most afforestation is required. For example, to reforest grasslands dominated by *Imperata cylindrica*, the selected tree species must be able to occupy the site with vigorous early growth. Tolerant fast-growing exotic species are more effective in this, because the original rainforest species are not adapted to harsh climatic conditions and degraded soils generally prevailing on grasslands (Otsamo *et al.*, 1997). However, Butterfield (1993) comments that foresters tend to depend upon exotic species with proven management techniques regardless of their appropriateness for the intended end products and site conditions. For example, the approved list of 25 tree species identified for the eight forestry regions of Costa Rica are mostly well-known exotic genera, such as *Pinus*, *Eucalyptus*, *Gmelina*, and *Tectona*. All of these species have been tested to some extent in Costa Rica, but none of them have been tested extensively in all of the ecological zones for which they are recommended. Exotic species are rarely tested against local species that may be better adapted. The tendency to use exotic species is strengthened by the channelling of research funds almost exclusively towards the testing of exotic species in new environments. Until recently, little or no investment has been made available for the development of native species.

The use of genetic modification (GM) in forestry are taking place almost unnoticed in recent years (Owusu, 1999). Since 1988, there were 116 confirmed genetically modified (GM) tree trials around the world and the number of both trials and species used has risen sharply since 1995. Despite the ethical and ecological concerns on the problems of GM organisms, GM trees may dominate the forestry plantations in the near future.

More recently, the interest in the use of native tree species for commercial forestry and forest rehabilitation and restoration has been growing. With the decline of natural timber stocks in many tropical countries, plantation production of native hardwoods will be needed to sustain the supply of quality wood (Butterfield, 1993). The identification and domestication of native timber species may also provide alternative species for industrial wood production that are better adapted to the local environment and provide a wider range of quality hardwoods to broaden the forestry production base (Appanah & Weinland, 1996; Butterfield, 1993; Haggar *et al.*, 1998). With the rising concern for biodiversity conservation, habitat restoration is regarded a complementary measure to preserving existing habitats for native species (Jordan, 1997). To conserve natural forests, and their biological diversity and genetic constituents, protecting the remaining forest as well as restoring indigenous forests based on the knowledge of vegetation ecology is equally important (Goosem & Tucker, 1995; Miyawaki, 1992; 1993). Within national parks and protected areas, where the primary objectives are biodiversity conservation, afforestation should aim to permanently restore the original forest ecosystems, as closely as possible, by accelerating the natural processes of forest regeneration (Elliott *et al.*, 1995). Community forestry, in buffer zones around protected areas and elsewhere, also requires restoration of near-natural forest ecosystems to provide a diverse range of forest products and ecological services to local people. The importance of native tree species, both in commercial forestry and in forest restoration for biodiversity conservation, will likely increase in the new millennium.

Currently, the problems with the wider use of native tree species in commercial and environmental forestry are yet to be overcome. There is a general lack of reliable knowledge of basic biology and nursery operations for native tree species in the tropics because of the large number of species and contradictory information from different areas (Richardson, 1998). Information on seed collection, storage, germination and seedling growth conditions for many native species is not yet available. Native species are thus more difficult to manage silviculturally than the exotics. This makes the production of native seedlings more expensive (Elliott *et al.*, 1995; Richardson, 1998). However, investment in nursery practices is necessary. For large-scale plantation programmes, seedling production by combining all of the available methods (seeds, wildings, cuttings) seems to be the most appropriate strategy (Ådjers *et al.*, 1996). A strong emphasis on continued research and development activities is recommended. More knowledge is needed, but this fact should not prevent anyone from starting to plant native species. In addition, studies of adaptability and growth of native tree species established on degraded land are still scarce (González & Fisher, 1994). There is an increasing need in the lowland humid tropics for promoting land-use systems, which can contribute to supplying the demands for timber, fuel wood, and other tree products without continuing the well-documented patterns of deforestation and land resource degradation. To achieve this, more information is needed on the performance of native tree species grown in plantations or in agroforestry systems. Information on the impacts of trees on soil fertility should be a determining factor on species choice for these systems (Montagnini & Sancho, 1990).

Research on the use of native species, both in nursery production and planting method has already started throughout the tropics (Appanah & Weinland, 1996; Ådjers & Otsamo, 1996; Butterfield & Fisher, 1994; Forest Restoration Research Unit, 1998; Goosem & Tucker, 1995; Miyawaki, 1992). Some studies have already shown that certain native species are promising (Holl, 1998b). For example, studies in Costa Rica have demonstrated that native species which had never been used before in reforestation, such as *Hyeronima alchorneoides*, *Virola koschnyi*, *Vochysia ferruginea* and *V. guatemalensis*, grow successfully on degraded soils (González & Fisher, 1994). A study of forest restoration on bauxite-mined lands in Brazilian Amazon showed that mixed native species plantings of more than 70 tree species appeared to be at low risk of arrested succession in comparison with traditional single species planting (Parrotta & Knowles, 1999).

Forestry in Hong Kong has always been for environmental reasons, such as soil erosion control, watershed protection, landscape repair and more recently biodiversity conservation, rather than timber production (Corlett, 1999). Despite the fact that the most commonly planted tree species in the early afforestation history of Hong Kong (1871-1965) was a native pine, *Pinus massoniana* (it lost its importance due to pest problems and its susceptibility to fire), Hong Kong has relied heavily on a limited number of exotic species in the 1970s and 1980s. Between 1871 and 1990, a total of 150 tree species were named in Hong Kong forestry reports, only 33 were native species (Corlett, 1999). An increasing number of native species has been tried in recent years (Lay *et al.*, 1999; http:\\www.info.gov.hk\afd\trees\treedata.htm), but for reclamation projects on barren lands, a much higher percentage of exotic species are used (Chong, 1996; Webb, 1993). It has been suggested that the dominating use of exotic species in afforestation programmes in Hong Kong is attributed to the ease of nursery production, tolerance of bad planting techniques and the ability of exotic tree species to survive better and grow faster than natives on degraded lands (Corlett, 1999; Chong, 1996; Webb, 1993; Corlett, pers. comm.).

The aim of this study was to test the early performance of ten tree species in three different degraded hillside sites: an upland grassland, a lowland shrubland and a lowland grassland. All except *Cinnamomum camphora* are native to Hong Kong. *Cinnamomum* has been generally considered native to Hong Kong, and is certainly native in the region, it is probably present only as an escape from cultivation. This study was designed as an extension of the study reported in Chapter 6 using more species, more sites but no treatments.

2. Materials and Methods

The majority of the seedlings were planted in the summer of 1996 and were observed for two years until July 1998. A smaller number were planted in August 1995, soon after the seedling transplant experiment in Chapter 6, and were observed for two years until August 1997.

2.1 Study sites

The study was conducted at an upland grassland site (KFBG grassland), a lowland grassland site (HSH grassland) and a lowland shrubland site (KARC shrubland). All three sites were described in the seed predation experiment in Chapter 3 (see also Table 7.1)

2.2 Soil

The total nitrogen and phosphorus contents of the soil at all three sites were determined in December 1997. Sampling and analysis methods were described in Chapter 6.

2.3 Experimental design

Due to the limitations in the availability of tree seedlings and manpower, only four species were planted in July and August 1995 at the KFBG grassland and the KARC shrubland, after the trial planting described in Chapter 6. They were *Castanopsis fissa*, *Sapium discolor*, *Schefflera octophylla* and *Schima superba*. These four species were planted again in September 1996 at the KFBG grassland and the HSH grassland. In late May, July and August 1996, another six species were planted, in separate plots from the previous four species, at the KFBG grassland, HSH grassland and KARC shrubland respectively. They were *Choerospondias axillaris*, *Cyclobalanopsis neglecta*, *Machilus breviflora*, *Mallotus paniculatus*, *Sterculia lanceolata* and *Cinnamomum camphora*. These are best viewed as two separate planting trials using four and six species respectively. The species used have a range of ecological characteristics, from classic, well-dispersed, fast-growing, shade-intolerant pioneers to species which persist in the oldest and least disturbed forests in the region.

The experiments were two-factor analysis of variance designs with fixed effects (site and species). No treatment was given to any planting plot throughout the study period, not even the removal of grass cover prior to seedling transplantation. The experimental designs of the four- and the six-species trial plantings were different only in terms of the number of replicates. Ten individuals of each species were randomly planted at 1.2 - 1.5 m spacing in a rectangular grid to form a planting plot (5 rows and 8 columns in the four-species trial; 6 rows and 10 columns in the six-species trial). For each planting plot, ten folded papers with the same number to represent a species were put in a box and a paper was drawn from the box to determine which species was planted at each square of the grid. In 1995, each plot was replicated five times at KFBG grassland (these were actually the control plots in the seedling transplant experiment in Chapter 6). At KARC shrubland, each plot was replicated ten times (i.e. 4 x 10 x 10 = 400 seedlings). In 1996, the four-species trial was replicated five times (i.e. 2 x 4 x 10 x 5 = 400 seedlings) and the six-species trial was replicated four times (i.e. 3 x 6 x 10 x 4 = 720 seedlings). The differences in the number of replications were due to seedling availability. As a result, a total of 1,520 seedlings were planted in this experiment. All replicates were haphazardly spread across each site.

In the four-species trial, eighteen-month old container-grown seedlings were used. They were obtained from a commercial nursery and delivered to a small nursery at KARC, which was set up specially for this project, in April 1995 and 1996, respectively. In the six-species trial, seedlings were propagated from seeds collected from the wild in Hong Kong. Most seeds were collected in the winter of 1994 and sown immediately after collection at the KARC nursery (Table 7.2). At the time of planting, the seedlings were twelve to twenty months old, with mean stem heights of 13.1 - 64.2 cm (Table 7.3).

Seedlings were hardened-off in the nursery one month prior to being taken to the planting site. Shade was gradually removed and irrigation was gradually reduced. All seedlings were delivered to the planting site at least two weeks prior to transplantation and were monitored every day. Water was given to seedlings that started to wilt. All dead and sick-looking seedlings were discarded. This screened out the weak ones and allowed the seedlings to acclimatise to the physical environment of the study site prior to transplantation. Seedlings planted on dry days were irrigated to reduce transplantation loss. Common precautions during transportation and transplantation as described in Forest Restoration Research Unit (1998) were taken. To standardise the workmanship of planting, I planted all of the seedlings in this experiment.

2.4 Data collection and statistical analysis

The survival of each seedling was recorded one month after they were transplanted. Any mortality during this period was assumed to be due to the stress in transportation and transplantation, and was thus counted as the transplantation loss. Seedling survival was recorded again in approximately four months, one year, and two years after they were planted. The mean percentage survival of each species at each site at the end of two years was compared by two-way ANOVA. Since there were three sets of planting plots, i.e. the six-species trial in 1996, the four-species trials in 1995 and 1996, three ANOVAs were conducted separately.

The basal diameter and stem height of each seedling was measured one month after transplantation and taken as a baseline for determining the growth rate. The same measurements were repeated in approximately four months, one year and two years. As in Chapter 6, the relative height increment per year (RHI) and relative basal diameter increment per year (RBDI) were calculated and used in the statistical analysis.

The total nitrogen and phosphorus of the soil at the three sites were compared by one-way ANOVA. Minitab Release 12.1 was used in all analyses of variances. Data transformation and assumption-checking was the same as described in Chapter 6.

3. Results

The KFBG grassland had significantly higher mean soil nitrogen than the KARC shrubland, which, in turn, had significantly more nitrogen than the HSH grassland (Tables 7.4 & 7.5). The KFBG grassland also had significantly higher mean soil phosphorus but there was no significant difference between the other two sites (Tables 7.4 & 7.6). Overall, the soil nitrogen and phosphorus at all three sites were considered low (see Chapter 6).

In all planting trials, over 80 % of all species survived for 30 days at all sites, and many species had 100 % survival (Table 7.7). The survival rates remained at a high level at 120 days before entering the first winter dry season. After two years, no *Sapium* seedlings survived in either grassland site in the 1996 trial or in the KARC shrubland in the 1995 trial. Its survival in the KFBG grassland in 1995 was only 24 %. For all other species, the mean percentage survival in both grassland sites was 70 - 100 %, but the survival in the shrubland site was more variable, ranging from 20 % to 100 %. The low survival rate of *Sterculia* in the KARC shrubland was because 21 seedlings (51.5 %)

were dug up in the first winter dry season and had their roots chewed off. Other species were not affected. A previous planting trial at this site also had seedling damage in a similar manner, which was attributed to wild boar *Sus scrofa* (Zhuang Xueying, pers. comm.)

Both the four- and six-species planting trials showed significant differences in survival after two years between species and between sites. However, only the six-species trial showed a significant interaction between sites and species. In no planting trial was there a significant difference in seedling survival between the two grassland sites but the survival rates in the KARC shrubland were significantly lower in all cases (Tables 7.8 & 7.9). In the four-species trials, *Sapium* had significantly lower survival than the other three species in both years. The difference between *Schefflera* and *Castanopsis* was not significant in either year. *Schima* had the highest survival in the 1995 trial but was the same as *Schefflera* and *Castanopsis* in 1996 (Table 7.8). Except for *Sterculia*, the survival rates of all species in the six-species trial did not differ significantly between the three study sites (Table 7.9). *Sterculia* had a significantly lower survival rate in the KARC shrubland while there was no significant difference between the two grassland sites. In the two grassland sites, the survival rates did not differ significantly between species, while in the KARC shrubland, the six species were divided into two groups with *Choerospondias*, *Cinnamomum*, *Cyclobalanopsis* and *Mallotus* having significantly higher survival than *Machilus* and *Sterculia*.

In the four-species planting trials, *Sapium* was excluded from the ANOVA on relative growth because few or no seedlings survived after two years (Table 7.7). Similarly, the survival rate of *Sterculia* was very low in the KARC shrubland in the six-species planting trials so that its growth data had a very large influence on the ANOVA. This made the variances non-homogenous and the residuals badly non-normal, even after the removal of many outliers. *Sterculia* was thus also excluded from the ANOVA.

There were significant differences in RHI in the interaction between sites and species in all planting trials (Table 7.10 & 7.11). *Schima* and *Castanopsis* had significantly lower

RHI in the KFBG grassland in both years (Table 7.10). *Schefflera* had significantly higher RHI in the HSH grassland than the KFBG grassland in 1996 but the difference between KARC shrubland and KFBG grassland in 1995 was not significant. In the six species planting trials, all except *Choerospondias* showed no significant difference in RHI between grassland sites (Table 7.11). The RHI of *Cheorospondias* in the KFBG grassland and KARC shrubland was not significantly different but it was significantly higher than in the HSH grassland. The differences in RHI of the other species between grassland and shrubland sites varied. At all three sites in the four-species planting trials, *Schefflera* had significantly higher RHI than *Castanopsis*, which in turn, had higher RHI than *Schima* (Table 7.10). In the six-species planting trials, the differences in RHI between the five species in all three sites were similar, with *Machilus* and *Cyclobalanopsis* had significantly higher RHI than all other species and *Cinnamomum* had significantly lower (Table 7.11).

Since the nine species were not planted together in a mix, it is inappropriate to compare statistically the RHIs of all nine species but *Schefflera*, *Cyclobalanopsis* and *Machilus* had relatively higher RHI than other species, while the RHIs of *Schima* and *Cinnamomum* were relatively lower. *Castanopsis*, *Choerospondias*, *Mallotus* and *Sterculia* were in the middle range (Table 7.12). The actual increase in mean stem height in two years ranged from 14 cm in *Cinnamomum* to 108 cm in *Cyclobalanopsis* (Table 7.13 & 7.14). The actual increase in mean basal diameter in two years was small, ranging from 2.2 mm in *Castanopsis* to 16.2 mm in *Choerospondias* (Tables 15 & 16).

All planting trials showed a significant interaction between site and species for RBDI. While *Schima* showed no difference between sites (Table 7.17), the difference in RBDI of *Schefflera* and *Castanopsis* between sites varied. While *Choerospondias* and *Cinnamomum* showed no significant differences in RBDI between sites, the other three species all had significantly higher RBDI at KFBG grassland (Table 7.18). The difference in RBDI between species was complex. In general, *Castanopsis* had higher RBDI than *Schefflera* and *Schima* at most sites. *Choerospondias* and *Cyclobalanopsis* had significantly higher RBDIs in most sites than other species and *Cinnamomum* was significantly lower (Table 7.18).

Like in RHI, *Cyclobalanopsis* and *Machilus* also had relatively higher RBDI than other species (Table 7.19).

4. Discussion

The total nitrogen and phosphorus of the all three sites appeared very low in comparison with several other studies on degraded hillside in Hong Kong (see Chapter 6). From a nutrient perspective, this study can therefore be seen as a "worst-case" scenario.

The almost 100% seedling survival for all species in 30 days and the still very high survival rates before the first dry season (Table 7.7) again indicates that the post nursery care and all the precautions during seedling transportation and transplantation are effective in minimising transplantation loss (see Chapter 6). In a previous planting trial in a degraded hillside in Tung Chung, the same species (except *Sapium discolor*) had much lower survival after two and three years (Lay *et al.*, 1999). At Tung Chung, *Schima* had the best survival among the 54 species planted (about 75% after one year; 65 % after two years and 55 % after three years), however, it was much lower than the survival rate in this study. In another planting trial in a grassland at KARC, the survival rates of the same species were comparable to this study (Zhuang, 1993). The seedlings in the Tung Chung planting trial were planted by contractors while the seedlings in this and Zhuang's study were planted by the researcher. Lay *et al.* (1999) also suggest that poor post-nursery care may be a significant factor affecting seedling survival and growth (see also Chapter 1).

All species except *Sapium discolor* had very high survival rates in this study. *Sapium* also had exceptionally low survival in the planting trial in Chapter 6. However, its survival rate over two years was about 40 % in the Tung Chung trial (Lay *et al.*, 1999) and about 60 % in another planting trial in KARC (Zhuang, 1993). The *Sapium* seedlings in these two studies were grown from seeds collected locally in the Government nursery while *Sapium* seedlings in the current study were obtained from China. This suggests that seedlings from different sources may have different performances. In view of this potential problem, it has been suggested that seedlings grown from seeds collected from the nearest forests should be used in reforestation projects in the tropics (Goosem & Tucker, 1995; Forest Restoration Research Unit, 1998). However, three other species used in this study that were imported from China had very good performance. This suggests that further study is needed to determine if there is significant different in the performance of native tree seedlings produced in Hong Kong and in mainland China, and which species have more serious problems in this regard.

Wild boar was previously not documented as seedling predators. Its role in seedling predation requires further investigation. The use of *Sterculia* at shrubland sites where wild boar occurs should be avoided.

Cyclobalanopsis, *Machilus*, *Choerospondias* and to a lesser extent, *Schefflera*, had higher relative growth rates than other species in this trial at all three sites. The growth in height of most species was comparable to Zhuang (1993) but slightly better than Lay *et al.* (1999). The growth rate is also comparable to 25 native species in a planting trial in Taiwan (Lee *et al.*, 1993) but between 1 - 2 m less than exotic species in degraded sites in the nearby Guangdong Province (Yang *et al.*, 1995; Wang & Wang, 1989). Native tree seedling growth in degraded sites in Hong Kong is much slower than reported for truly tropical climates (Ådjers *et al.*, 1996; Butterfield & Espinoza C., 1995; Haggar *et al.*, 1998; Otsamo *et al.*, 1997). For example, a planting trial of 11 native tree species in degraded pasture in Costa Rica showed that the mean increase in stem height over two years was 200 - 1000 cm (González & Fisher, 1994).

There was no significant difference between the growth rates in upland and lowland grassland for most species, despite an expected difference in mean and minimum temperature of 3 °C (calculated from a lapse rate of 0.6 °C per 100 m altitude).

An obvious limitation of the current study is the very short study period (a constraint of postgraduate research in Hong Kong). In a study of the performance of 12 native, shade tolerant tree species planted on a grassland that was dominated by *Imperata* *cylindrica* in Indonesia, all species had high survival after two years but the mortality of several species increased considerably in the third year due to a severe drought (Otsamo *et al.*, 1996). This can also happen in Hong Kong, and at the upland site, occasional frost may also cause high seedling mortality.

Apart from *Sapium* and *Sterculia* (in KARC shrubland only), there was no significant different in survival between sites and all species had high survival rates without any treatment. In addition, several species including *Cyclobalanopsis*, *Machilus*, *Choerospondias* and *Schefflera* grew much faster than the other species. These reinforce the results of the earlier chapters that the major filter barriers to tree invasion to grassland occur before seedling establishment. *Cyclobalanopsis* and *Choerospondias* could never have been dispersed to the study sites and other species such as *Schefflera* and *Machilus*, though dispersed, may suffer from serious seed predation. These, plus low germination rate in the field, will significantly reduce the chance for these species to establish on the study sites. However, once the seedlings are established, other barriers including seasonal drought, low soil nutrient and below-ground competition are unlikely to limit seedling survival.

This study also indicates that species common in closed secondary forests such as *Cyclobalanopsis* and *Choerospondias* could also have good performance on open, degraded sites. More of the 390 native tree species in Hong Kong should be tested for forest restoration. In conclusion, the results suggest that all except *Sapium* have the potential for wider use in forest restoration Hong Kong, especially the four faster growing species.

Table 7.1

The characteristics of the three study sites.

Site	District	Altitude (m)	Steepness	Slope orientation
Kadoorie Farm and Botanic Garden (KFBG) grassland	Central New territories	550	40°	Northwest facing
Ho Sheung Heung (HSH) grassland	Northern New territories	20	15°	Southeast facing
Kadoorie Agricultural Research Centre (KARC) shrubland	Central New territories	200	25°	Northeast facing

Table 7.2

The dates of seed collection, germination and transplantation of seedlings into growth containers of the six tree species in the nursery at KARC.

Species	Collection	Germination	Transplanting
Choerospondias axillaris	29/10/94; 02/11/94	16/03/95	05-08/05/95
Cinnamomum camphora	04/12/94	19/03/95	01-03/05/95
Cyclobalanopsis neglecta	26/02/95	07/03/95	09-10/05/95
Machilus breviflora	02/11/94	02/04/95	01-03/06/95
Mallotus paniculatus	02/02/95	04/03/95	15-16/05/95
Sterculia lanceolata	02/08/95	05/08/95	04/09/95

Table 7.3

Age and mean stem heights of the six tree species when they were transplanted into KFBG grassland, HSH grassland and KARC shrubland.

Site	Species	Planting Date	Age (month)	Mean stem height (cm) (n = 40)
KFBG	*Choerospondias axillaris*	25-29/5/96	14	47.83 ± 7.60
	Cinnamomum camphora		14	40.93 ± 6.53
	Cyclobalanopsis neglecta		14	31.29 ± 6.76
	Machilus breviflora		13	13.09 ± 3.23
	Mallotus paniculatus		14	50.12 ± 12.49
	Sterculia lanceolata		10	21.01 ± 3.28
HSH	*Choerospondias axillaris*	27-28/7/96	16	53.51 ± 11.76
	Cinnamomum camphora		16	38.74 ± 5.79
	Cyclobalanopsis neglecta		16	22.69 ± 6.25
	Machilus breviflora		15	18.40 ± 4.50
	Mallotus paniculatus		16	64.18 ± 16.07
	Sterculia lanceolata		12	22.44 ± 5.83
KARC	*Choerospondias axillaris*	2-3/8/96	17	61.63 ± 6.78
	Cinnamomum camphora		17	40.41 ± 8.12
	Cyclobalanopsis neglecta		17	22.69 ± 6.25
	Machilus breviflora		16	15.58 ± 2.63
	Mallotus paniculatus		17	53.75 ± 9.66
	Sterculia lanceolata		14	24.29 ± 4.97

Table 7.4

The soil nitrogen and phosphorus contents of the three study sites.

Site	Total Nitrogen (%)			Total Phosphorus (%)		
	Minimum	Maximum	Mean (n=40)	Minimum	Maximum	Mean (n=20)
KFBG grassland	0.0070	0.0430	0.0204 ! 0.0084	0.0025	0.0049	0.00328 ! 0.00065
HSH grassland	0.0061	0.0229	0.0090 ! 0.0029	0.0015	0.0040	0.00264 ! 0.00048
KARC shrubland	0.0081	0.0168	0.0108 ! 0.0022	0.0010	0.0040	0.00225 ! 0.00079

Table 7.5

One-way ANOVA to compare the total soil nitrogen (%) of the KFBG grassland, HSH grassland and KARC shrubland. The power of transformation of total soil nitrogen data was minus 0.511 according to Taylor's power law. Significant differences were further analyzed using SNK test.

Factor	df	MS	F	p
Site	2	126.22	52.71	0.000
Error	117	2.39		
Total	119			

SNK tests

KFBG > KARC > HSH

Table 7.6

One-way ANOVA to compare the total soil phosphorus (%) of the KFBG grassland, HSH grassland and KARC shrubland. The power of transformation of total soil phosphorus data was 1.3805 according to Taylor's power law. Significant differences were further analyzed using SNK test.

Factor	df	MS	F	p
Site	2	0.0000001	12.31	0.000
Error	57	0.0000000		
Total	59			

SNK tests

KFBG > HSH = KARC

Table 7.7

The mean percentage survival of ten tree species after 30, 120 days and two years in the KFBG grassland, HSH grassland and KARC shrubland.

Species	Mean percentage survival (30 days, 120 days, 2 years)																	
	1995 - 1997						1996 - 1998											
	KFBG grassland			KARC shrubland			KFBG grassland			HSH grassland			KARC shrubland					
Schima superba	100	100	96	100	99	78	100	-[b]	80	98	-	86	_	_	_
Schefflera octophylla	98	92	72	98	94	59	98	-	82	88	-	70	_	_	_
Castanopsis fissa	100	94	82	100	68	26	100	-	70	94	-	74	_	_	_
Sapium discolor	98	78	24	98	26	0	96	-	0	86	-	0	_	_	_
Choerospondias axillaris	_[a]	_	_	_	_	_	100	100	97.5	100	100	100	100	100	97.5
Cinnamomum camphora	_	_	_	_	_	_	100	95	95	100	100	97.5	100	100	100
Cyclobalanopsis neglecta	_	_	_	_	_	_	97.5	97.5	97.5	95	90	90	100	97.5	80
Machilus breviflora	_	_	_	_	_	_	100	100	87.5	100	97.5	95	100	100	70
Mallotus paniculatus	_	_	_	_	_	_	100	100	100	95	95	87.5	100	100	42.5
Sterculia lanceolata	_	_	_	_	_	_	97.5	95	92.5	97.5	97.5	87.5	100	45	20

[a] _, indicates that the species was not tested at that site in that period.
[b] -, indicates that no data was taken at that time.

Table 7.8

Two-way ANOVA to compare the survival rates of the four native tree species in the KFBG grassland and KARC shrubland in the 1995 planting trials and in the KFBG grassland and HSH grassland in the 1996 planting trials. The percentage survival data were arcsine transformed. Significant differences were further analyzed using SNK tests. The mean percentage survival (± s.d.) is shown under each treatment in the SNK tests. * $p < 0.001$; NS, not significant.

1995

Factor	df	MS	F	p	Significant
Site	1	6531.2	25.01	0.000	*
Species	3	9418.6	36.06	0.000	*
Site x Species	3	557.3	2.13	0.107	NS
Error	52	261.2			
Total	59				

SNK tests

Site:

KFBG grassland	>	KARC shrubland
68.00 ± 32.22		40.75 ± 36.19

Species:

Schima	>	*Schefflera*	>	*Castanopsis*	>	*Sapium*
83.33 ± 23.50		64.00 ± 22.93		44.67 ± 31.82		7.33 ± 17.10

1996

Factor	df	MS	F	p	Significant
Site	1	10.8	0.13	0.718	NS
Species	3	9832.3	120.10	0.000	*
Site x Species	3	75.9	0.93	0.439	NS
Error	32	81.9			
Total	39				

SNK tests

Species:

Schima	=	*Schefflera*	=	*Castanopsis*	>	*Sapium*
83.00 ± 8.23		76.00 ± 9.66		72.00 ± 19.89		0.00 ± 0.00

Table 7.9

Two-way ANOVA to compare the survival rates of *Choerospondias axillaris* (CA), *Cinnamomum camphora* (CC), *Cyclobalanopsis neglecta* (CN), *Machilus breviflora* (MB), *Mallotus paniculatus* (MP) and *Sterculia lanceolata* (SL) in the KFBG grassland, HSH grassland and KFBG shrubland in the 1996 planting trials. The percentage survival data were arcsine transformed. Significant differences were further analyzed using SNK tests. The mean percentage survival (± s.d.) is shown under each treatment in the SNK results. * $p < 0.01$, ** $p < 0.001$.

Factor	df	MS	F	p	Significant
Site	2	1013.8	7.44	0.001	*
Species	5	938.9	6.89	0.000	**
Site x Species	10	400.4	2.94	0.005	*
Error	54	136.3			
Total	71				

SNK tests

Sites:

KFBG	=	HSH	>	KARC
93.33 ± 11.29		92.08 ± 8.33		79.17 ± 24.83

Species:

CA	=	CC	>	MP	=	CN	=	MB	=	SL
98.33 ± 3.89		96.67 ± 4.92		90.00 ± 9.53		85.83 ± 16.76		83.33 ± 18.26		75.00 ± 28.44

Site x Species:

KFBG CA 97.50 ± 5.00	=	HSH CA 100.00 ± 0.00	=	KARC CA 97.50 ± 5.00
KFBG CC 92.50 ± 5.00	=	HSH CC 97.50 ± 5.00	=	KARC CC 100.00 ±0.00
KFBG CN 90.00 ± 20.00	=	HSH CN 87.50 ± 9.57	=	KARC CN 80.00 ± 21.60
KFBG MB 87.50 ± 18.93	=	HSH MB 95.00 ± 5.77	=	KARC MB 67.50 ± 17.08
KFBG MP 100.00 ± 0.00	=	HSH MP 85.00 ± 10.00	=	KARC MP 85.00 ± 5.77
KFBG SL 92.50 ± 5.00	=	HSH SL 87.50 ± 5.00	>	KARC SL 45.00 ± 33.17

KFBG CA	=	KFBG CC	=	KFBG CN	=	KFBG MP	=	KFBG MB	=	KFBG SL
97.50 ± 5.00		92.50 ± 5.00		90.00 ± 20.00		100.00 ± 0.00		87.50 ± 18.93		92.50 ± 5.00
HSH CA	=	HSH CC	=	HSH CN	=	HSH MP	=	HSH MB	=	HSH SL
100.00 ± 0.00		97.50 ± 5.00		87.50 ± 9.57		85.00 ± 10.00		95.00 ± 5.77		87.50 ± 5.00
KARC CA	=	KARC CC	=	KARC CN	=	KARC MP	>	KARC MB	=	KARC SL
97.50 ± 5.00		100.00 ±0.00		80.00 ± 21.60		85.00 ± 5.77		67.50 ± 17.08		45.00 ± 33.17

Table 7.10

Two-way ANOVA to compare the relative height increment per year (RHI) of *Schima superba*, *Schefflera octophylla* and *Castanopsis fissa* in the KFBG grassland and KARC shrubland in the 1995 planting trials; and in the KFBG grassland and HSH grassland in the 1996 planting trials. The power of transformation was 0.551 in 1995 and 0.3356 in 1996 according to Taylor's power law. Significant differences were further analysed using SNK tests. The mean RHI (± s.d.) is shown under each treatment in the SNK tests.
* $p < 0.01$, ** $p < 0.001$.

1995

Factor	df	MS	F	P	Significant
Site	1	0.1766	9.07	0.003	*
Species	2	2.9650	152.21	0.000	**
Site x Species	2	0.0954	4.90	0.008	*
Error	240	0.0195			
Total	245				

SNK tests

Site:	KARC shrubland 0.3399 ± 0.2179	>	KFBG grassland 0.3067 ± 0.2343		
Species:	*Schefflera* 0.5345 ± 0.1836	>	*Castanopsis* 0.3172 ± 0.1836	>	*Schima* 0.1684 ± 0.1063
Site x Species:	KARC *Schima* 0.1856 ± 0.1051	>	KFBG *Schima* 0.1402 ± 0.1033		
	KARC *Schefflera* 0.5242 ± 0.1886	=	KFBG *Schefflera* 0.5506 ± 0.1771		
	KARC *Castanopsis* 0.3880 ± 0.1778	>	KFBG *Castanopsis* 0.2464 ± 0.1587		
	KFBG *Schefflera* 0.5506 ± 0.1771	>	KFBG *Castanopsis* 0.2464 ± 0.1587	>	KFBG *Schima* 0.1402 ± 0.1033
	KARC *Schefflera* 0.5242 ± 0.1886	>	KARC *Castanopsis* 0.3880 ± 0.1778	>	KARC *Schima* 0.1856 ± 0.1051

Cont....

Table 7.10 Cont.

1996

Factor	df	MS	F	P	Significant
Site	1	0.82169	143.72	0.000	**
Species	2	1.44985	253.59	0.000	**
Site x Species	2	0.04619	8.08	0.000	**
Error	207	0.00572			
Total	212				

SNK tests

Site:	HSH grassland 0.4886 ± 0.2849	>	KFBG shrubland 0.2848 ± 0.1532		
Species:	*Schefflera* 0.6247 ± 0.2517	>	*Castanopsis* 0.3416 ± 0.1305	>	*Schima* 0.1887 ± 0.0847
Site x Species:	HSH *Schima* 0.2314 ± 0.0875	>	KFBG *Schima* 0.1504 ± 0.0614		
	HSH *Schefflera* 0.8346 ± 0.1909	>	KFBG *Schefflera* 0.4424 ± 0.1220		
	HSH *Castanopsis* 0.4162 ± 0.1231	>	KFBG *Castanopsis* 0.2580 ± 0.0775		
	KFBG *Schefflera* 0.4424 ± 0.1220	>	KFBG *Castanopsis* 0.2580 ± 0.0775	>	KFBG *Schima* 0.1504 ± 0.0614
	HSH *Schefflera* 0.8346 ± 0.1909	>	HSH *Castanopsis* 0.4162 ± 0.1231	>	HSH *Schima* 0.2314 ± 0.0875

Table 7.11

Two-way ANOVA to compare the relative height increment per year (RHI) of *Choerospondias axillaris*, *Cinnamomum camphora*, *Cyclobalanopsis neglecta*, *Machilus breviflora* and *Mallotus paniculatus* in the KFBG grassland, HSH grassland and KARC shrubland in the 1996 planting trials. The power of transformation was 0.27695 according to Taylor's power law. Significant differences were further analyzed using SNK tests. The mean RHI (± s.d.) is shown under each treatment in the SNK tests.
* p < 0.01, ** p < 0.001.

Factor	df	MS	F	p	Significant
Site	2	0.02510	6.22	0.002	*
Species	4	1.55016	384.29	0.000	**
Site x Species	8	0.03168	7.89	0.000	**
Error	485	0.00404			
Total	499				

SNK tests

Site:	KFBG grassland 0.4296 ± 0.2473	>	HSH grassland 0.3883 ± 0.2380	=	KARC shrubland 0.3832 ± 0.2013

Species:

Cyclobalanopsis 0.6511 ± 0.1492	=	*Machilus* 0.6215 ± 0.1657	>	*Choerospondias* 0.3566 ± 0.1317	>	*Mallotus* 0.2826 ±0.1041	>	*Cinnamomum* 0.1471 ± 0.0596

Site x Species:

KFBG *Choerospondias* 0.3771 ± 0.1408	=	KARC *Choerospondias* 0.4066 ± 0.1118	>	HSH *Choerospondias* 0.2868 ± 0.1132
KFBG *Cinnamomum* 0.1270 ± 0.0405	=	HSH *Cinnamomum* 0.1187 ± 0.0428	<	KARC *Cinnamomum* 0.1900 ± 0.0622
KFBG *Cyclobalanopsis* 0.6355 ± 0.1707	=	HSH *Cyclobalanopsis* 0.6522 ± 0.1221	=	KARC *Cyclobalanopsis* 0.6717 ± 0.1519
KFBG *Machilus* 0.6807 ± 0.1764	=	HSH *Machilus* 0.6333 ± 0.1294	>	KARC *Machilus* 0.5015 ± 0.1489
KFBG *Mallotus* 0.2972 ± 0.1103	=	HSH *Mallotus* 0.2634 ± 0.0852	=	KARC *Mallotus* 0.2861 ± 0.1145

KFBG *Machilus* 0.6807 ± 0.1764	=	KFBG *Cyclobalanopsis* 0.6355 ± 0.1707	>	KFBG *Choerospondias* 0.3771 ± 0.1408	>	KFBG *Mallotus* 0.2972 ± 0.1103	>	KFBG *Cinnamomum* 0.1270 ± 0.0405
HSH *Machilus* 0.6333 ± 0.1294	=	HSH *Cyclobalanopsis* 0.6522 ± 0.1221	>	HSH *Choerospondias* 0.2868 ± 0.1132	=	HSH *Mallotus* 0.2634 ± 0.0852	>	HSH *Cinnamomum* 0.1187 ± 0.0428
KARC *Cyclobalanopsis* 0.6717 ± 0.1519	>	KARC *Machilus* 0.5015 ± 0.1489	>	KARC *Choerospondias* 0.4066 ± 0.1118	>	KARC *Mallotus* 0.2861 ± 0.1145	>	KARC *Cinnamomum* 0.1900 ± 0.0622

Table 7.12

The relative height increment per year (± s.d.) of different species at different sites. KFBG grassland had two sets of data for the first three species because there were two separate planting trials.

Species\ site	Mean RHI			
	KFBG grassland		HSH grassland	KARC shrubland
Schima superba	0.14 ± 0.10	0.15 ± 0.06	0.23 ± 0.09	0.19 ± 0.11
Schefflera octophylla	0.55 ± 0.18	0.44 ± 0.12	0.83 ± 0.19	0.52 ± 0.19
Castanopsis fissa	0.25 ± 0.16	0.26 ± 0.08	0.42 ± 0.12	0.39 ± 0.18
Choerospondias axillaris	0.38 ± 0.14		0.29 ± 0.11	0.41 ± 0.11
Cinnamomum camphora	0.13 ± 0.04		0.12 ± 0.04	0.19 ± 0.06
Cyclobalanopsis neglecta	0.64 ± 0.17		0.65 ± 0.12	0.67 ± 0.15
Machilus breviflora	0.68 ± 0.18		0.63 ± 0.13	0.50 ± 0.15
Mallotus paniculatus	0.30 ± 0.11		0.26 ± 0.09	0.29 ± 0.11
Sterculia lanceolata	0.38 ± 0.14		0.44 ± 0.14	0.34 ± 0.14

Table 7.13

The initial mean stem height; final minimum, maximum and mean stem height; and the mean relative height increment/year (RHI) of the three native tree species in the KFBG grassland and KARC shrubland in the 1995 planting trials, and in the KFBG grassland and HSH grassland in the 1996 planting trials.

Year	Site	Species	Stem height (cm)						Increase in mean stem height (cm) in 2 years	Mean RHI
			Initial mean	N	Final					
					Minimum	Maximum	Mean	N		
1995	KFBG	*Schima superba*	51.73 ± 8.01	50	47.0	115.0	69.60 ± 15.69	44	17.87	0.1402 ± 0.1033
		Schefflera octophylla	12.02 ± 4.30	49	15.0	59.0	38.27 ± 14.56	35	26.25	0.5506 ± 0.1771
		Castanopsis fissa	36.20 ± 15.59	50	24.5	81.0	56.68 ± 15.80	20	20.48	0.2464 ± 0.1587
	KARC	*Schima superba*	51.90 ± 6.73	100	49.0	136.0	75.97 ± 17.34	72	24.07	0.1856 ± 0.1051
		Schefflera octophylla	14.42 ± 4.13	98	20.0	98.0	45.46 ± 17.80	55	31.04	0.5242 ± 0.1886
		Castanopsis fissa	40.74 ± 13.76	100	22.0	122.0	80.65 ± 25.03	20	39.91	0.3880 ± 0.1778
1996	KFBG	*Schima superba*	97.08 ± 22.20	50	89.0	182.0	129.42 ± 21.94	38	32.34	0.1504 ± 0.0614
		Schefflera octophylla	21.10 ± 7.90	49	27.0	62.0	47.47 ± 7.87	38	26.37	0.4424 ± 0.1220
		Castanopsis fissa	34.73 ± 6.65	50	38.0	79.5	58.26 ± 9.87	33	23.53	0.2580 ± 0.0775
	HSH	*Schima superba*	68.94 ± 11.34	49	77.0	145.0	106.72 ± 17.52	34	37.78	0.2314 ± 0.0875
		Schefflera octophylla	9.51 ± 4.66	44	23.0	70.0	46.86 ± 9.92	33	37.35	0.8346 ± 0.1909
		Castanopsis fissa	29.56 ± 6.17	46	40.0	91.0	67.87 ± 14.68	37	38.31	0.4162 ± 0.1231

Table 7.14

The initial mean stem height; final minimum, maximum and mean stem height; and the mean relative height increment per year (RHI) of the six tree species in the KFBG grassland, HSH grassland and KARC shrubland in the 1996 planting trials.

Site	Species	Stem height (cm)						Increase in mean stem height (cm) in two years	Mean RHI
		Initial mean	N	Final					
				Minimum	Maximum	Mean	N		
KFBG	*Choerospondias axillaris*	47.83 ± 7.60	40	77.0	232.0	117.41 ± 40.86	35	69.58	0.3771 ± 0.1408
	Cinnamomum camphora	40.93 ± 6.53	40	30.0	71.0	54.90 ± 8.98	31	13.97	0.1270 ± 0.0405
	Cyclobalanopsis neglecta	31.29 ± 6.76	39	64.0	230.0	134.41 ± 39.67	35	103.12	0.6355 ± 0.1707
	Machilus breviflora	13.09 ± 3.23	40	32.0	110.0	63.34 ± 19.76	35	50.25	0.6807 ± 0.1764
	Mallotus paniculatus	50.12 ± 12.49	40	60.0	276.0	102.38 ± 41.16	36	52.26	0.2972 ± 0.1103
	Sterculia lanceolata	21.01 ± 3.28	39	22.5	85.0	51.42 ± 16.34	37	30.41	0.3768 ± 0.1397
HSH	*Choerospondias axillaris*	53.51 ± 11.76	40	57.5	171.0	100.36 ± 25.98	36	46.85	0.2868 ± 0.1132
	Cinnamomum camphora	38.74 ± 5.79	40	35.0	66.0	51.14 ± 7.22	37	12.40	0.1187 ± 0.0428
	Cyclobalanopsis neglecta	31.92 ± 7.45	38	70.0	244.0	139.98 ± 38.12	32	108.06	0.6522 ± 0.1221
	Machilus breviflora	18.40 ± 4.50	40	41.5	169.0	78.76 ± 22.76	38	60.36	0.6333 ± 0.1294
	Mallotus paniculatus	64.18 ± 16.07	38	87.0	155.0	118.53 ± 18.51	33	54.35	0.2634 ± 0.0852
	Sterculia lanceolata	22.44 ± 5.83	39	31.0	92.0	64.17 ± 18.79	33	41.73	0.4418 ± 0.1371
KARC	*Choerospondias axillaris*	61.63 ± 6.78	40	104.0	252.0	156.46 ± 37.63	36	94.83	0.4066 ± 0.1118
	Cinnamomum camphora	40.41 ± 8.12	40	45.5	88.0	62.50 ± 10.14	39	22.09	0.1900 ± 0.0622
	Cyclobalanopsis neglecta	22.69 ± 6.25	40	28.5	151.0	96.20 ± 32.75	25	73.51	0.6717 ± 0.1519
	Machilus breviflora	15.58 ± 2.63	40	21.0	92.0	49.57 ± 17.83	21	33.99	0.5015 ± 0.1489
	Mallotus paniculatus	53.75 ± 9.66	40	58.0	182.0	103.13 ± 30.65	31	49.38	0.2861 ± 0.1145
	Sterculia lanceolata	24.29 ± 4.97	40	26.0	73.0	57.36 ± 17.36	7	33.07	0.3403 ± 0.1435

Table 7.15

The initial mean basal diameter (± s.d.); final minimum, maximum and mean basal diameter; and the mean relative basal diameter increment per year (RBDI) of the three native tree species in the KFBG grassland and KARC shrubland in the 1995 planting trials, and in the KFBG grassland and HSH grassland in the 1996 planting trials.

Year	Site	Species	Basal diameter (mm)						Increase in mean basal diameter (mm) in 2 years	Mean RBDI
					Final					
			Initial mean	N	Minimum	Maximum	Mean	N		
1995	KFBG	*Schima superba*	6.56 ± 1.17	50	5.4	19.3	9.70 ± 3.21	48	3.14	0.2175 ± 0.1384
		Schefflera octophylla	6.29 ± 1.21	46	4.5	18.2	9.94 ± 4.09	36	3.65	0.2373 ± 0.1657
		Castanopsis fissa	5.30 ± 1.17	47	3.5	17.7	7.78 ± 2.84	41	2.48	0.1892 ± 0.1499
	KARC	*Schima superba*	5.49 ± 1.07	99	4.9	14.5	7.96 ± 1.82	78	2.47	0.2253 ± 0.1186
		Schefflera octophylla	5.74 ± 0.87	94	4.8	17.0	8.49 ± 2.23	59	2.75	0.2118 ± 0.1304
		Castanopsis fissa	4.66 ± 1.17	68	3.5	10.5	6.84 ± 1.86	26	2.18	0.3092 ± 0.1403
1996	KFBG	*Schima superba*	9.64 ± 2.62	50	7.7	20.6	12.40 ± 2.39	40	2.76	0.1412 ± 0.0662
		Schefflera octophylla	6.63 ± 1.29	49	7.1	12.3	9.35 ± 1.48	41	2.72	0.1726 ± 0.0693
		Castanopsis fissa	4.52 ± 0.93	50	5.1	9.5	7.06 ± 1.11	35	2.54	0.2334 ± 0.0835
	HSH	*Schima superba*	8.82 ± 1.78	49	7.0	16.3	12.03 ± 1.81	43	3.21	0.1698 ± 0.0744
		Schefflera octophylla	4.80 ± 1.10	44	6.2	12.0	8.26 ± 1.24	35	3.46	0.2758 ± 0.0661
		Castanopsis fissa	4.32 ± 1.59	47	5.3	15.0	7.98 ± 1.82	37	3.66	0.3301 ± 0.0843

Table 7.16

The initial mean basal diameter (± s.d.); final minimum, maximum and mean basal diameter; and the mean relative basal diameter increment per year (RBDI) of the six tree species in the KFBG grassland, HSH grassland and KARC shrubland in the 1996 planting trials.

Site	Species	Basal diameter (mm)						Increase in mean basal diameter (mm) in two years	Mean RBDI
		Initial mean	N	Final					
				Minimum	Maximum	Mean	N		
KFBG	*Choerospondias axillaris*	6.23 ± 1.09	40	12.0	43.8	22.47 ± 8.22	36	16.24	0.5490 ± 0.1506
	Cinnamomum camphora	4.35 ± 0.56	40	5.5	13.0	8.11 ± 1.50	35	3.76	0.2683 ± 0.0740
	Cyclobalanopsis neglecta	4.62 ± 1.01	39	9.5	32.1	19.38 ± 5.60	35	14.76	0.6113 ± 0.1188
	Machilus breviflora	2.71 ± 0.67	40	4.6	19.7	10.10 ± 3.32	31	7.39	0.5664 ± 0.1474
	Mallotus paniculatus	6.60 ± 1.69	40	8.8	41.0	14.86 ± 5.84	33	8.26	0.3499 ± 0.0950
	Sterculia lanceolata	5.36 ± 1.45	39	5.6	17.7	9.03 ± 2.68	37	3.67	0.2378 ± 0.1134
HSH	*Choerospondias axillaris*	5.30 ± 1.03	40	9.1	25.9	15.50 ± 3.76	37	10.20	0.4741 ± 0.1062
	Cinnamomum camphora	4.22 ± 0.50	40	5.1	10.4	7.10 ± 1.13	36	2.88	0.2330 ± 0.0505
	Cyclobalanopsis neglecta	3.14 ± 0.70	40	9.5	20.5	14.57 ± 3.15	32	11.43	0.5110 ± 0.1023
	Machilus breviflora	3.32 ± 0.53	40	5.4	14.5	8.74 ± 2.01	35	5.42	0.4150 ± 0.0904
	Mallotus paniculatus	7.65 ± 9.71	38	7.5	15.2	10.59 ± 1.76	31	2.94	0.2549 ± 0.0639
	Sterculia lanceolata	4.44 ± 0.93	39	3.8	10.0	6.84 ± 1.57	34	2.40	0.1836 ± 0.0870
KARC	*Choerospondias axillaris*	6.02 ± 1.32	40	8.8	37.0	18.40 ± 6.69	37	12.38	0.4855 ± 0.1123
	Cinnamomum camphora	4.08 ± 0.66	40	5.7	9.2	7.44 ± 0.90	40	3.36	0.2689 ± 0.0541
	Cyclobalanopsis neglecta	3.14 ± 0.70	40	4.5	10.7	7.24 ± 1.68	28	4.10	0.3865 ± 0.1019
	Machilus breviflora	2.95 ± 0.45	40	3.9	9.3	5.98 ± 1.47	19	3.03	0.3015 ± 0.0794
	Mallotus paniculatus	5.34 ± 0.75	40	5.6	12.0	9.02 ± 1.70	28	3.68	0.2419 ± 0.0656
	Sterculia lanceolata	5.10 ± 1.87	40	5.2	7.9	6.20 ± 0.97	7	1.10	0.1445 ± 0.0505

Table 7.17

Two-way ANOVA to compare the relative basal diameter increment per year (RBDI) of *Schima superba, Schefflera octophylla* and *Castanopsis fissa* in the KFBG grassland and KARC shrubland in the 1995 planting trials; and in the KFBG grassland and HSH grassland in the 1996 planting trials. The power of transformation was 0.9816 in 1995 and 0.8169 in 1996 according to Taylor's power law. Significant differences were further analyzed using SNK tests. The mean RBDI (± s.d.) is shown under each treatment in the SNK results.
* $p < 0.01$, ** $p < 0.001$; NS, not significant.

1995

Factor	df	MS	F	p	Significant
Site	1	0.07098	3.72	0.055	NS
Species	2	0.01575	0.82	0.439	NS
Site x Species	2	0.09690	5.08	0.007	*
Error	268	0.01909			
Total	273				

SNK tests

Site x Species:

KFBG *Schima* 0.2175 ± 0.1384	=	KARC *Schima* 0.2253 ± 0.1186		
KFBG *Schefflera* 0.2373 ± 0.1657	=	KARC *Schefflera* 0.2118 ± 0.1304		
KFBG *Castanopsis* 0.1829 ± 0.1499	<	KARC *Castanopsis* 0.3092 ± 0.1403		
KFBG *Schima* 0.2175 ± 0.1384	=	KFBG *Schefflera* 0.2373 ± 0.1657	=	KFBG *Castanopsis* 0.1829 ± 0.1499
KARC *Schima* 0.2253 ± 0.1186	=	KARC *Schefflera* 0.2118 ± 0.1304	<	KARC *Castanopsis* 0.3092 ± 0.1403

Cont....

Table 7.17 Cont.

1996

Factor	df	MS	F	p	Significant
Site	1	0.38357	59.55	0.000	**
Species	2	0.35784	55.55	0.000	**
Site x Species	2	0.03645	5.66	0.004	*
Error	223	0.00644			
Total	228				

SNK tests

Site:

KFBG grassland	<	HSH grassland		
0.1801 ± 0.0814		0.2537 ± 0.1012		

Species:

Schima	<	*Schefflera*	<	*Castanopsis*
0.1559 ± 0.0715		0.2201 ± 0.0850		0.2824 ± 0.0965

Site x Species:

KFBG *Schima*	=	HSH *Schima*		
0.1412 ± 0.0662		0.1698 ± 0.0744		
KFBG *Schefflera*	<	HSH *Schefflera*		
0.1726 ± 0.0693		0.2758 ± 0.0661		
KFBG *Castanopsis*	<	HSH *Castanopsis*		
0.2334 ± 0.0835		0.3301 ± 0.0843		
KFBG *Schima*	=	KFBG *Schefflera*	<	KFBG *Castanopsis*
0.1412 ± 0.0662		0.1726 ± 0.0693		0.2334 ± 0.0835
HSH *Schima*	<	HSH *Schefflera*	<	HSH *Castanopsis*
0.1698 ± 0.0744		0.2758 ± 0.0661		0.3301 ± 0.0843

Table 7.18

Two-way ANOVA to compare the relative basal diameter increment per year (RBDI) of *Choerospondias axillaris*, *Cinnamomum camphora*, *Cyclobalanopsis neglecta*, *Machilus breviflora* and *Mallotus paniculatus* in the KFBG grassland, HSH grassland and KARC shrubland in the 1996 planting trials. The power of transformation was 0.08975 according to Taylor's power law. Significant differences were further analyzed using SNK tests. The mean RBDI (± s.d.) is shown under each treatment in the SNK results.

Factor	df	MS	F	p
Site	2	0.027792	68.16	0.000
Species	4	0.069623	170.74	0.000
Site*Species	8	0.004127	10.12	0.000
Error	478	0.000408		
Total	492			

SNK tests

<u>Site</u>:

KFBG grassland	>	HSH grassland	>	KARC shrubland
0.4685 ± 0.1806		0.3784 ± 0.1415		0.3424 ± 0.1266

<u>Species</u>:

Choerospondias	=	*Cyclobalanopsis*	>	*Machilus*	>	*Mallotus*	>	*Cinnamomum*
0.4952 ± 0.1405		0.4431 ± 0.1751		0.3677 ± 0.1684		0.3303 ±0.1177		0.3324 ± 0.1253

<u>Site x Species</u>:

KFBG *Choerospondias*	=	KARC *Choerospondias*	=	HSH *Choerospondias*
0.5490 ± 0.1506		0.4855 ± 0.1123		0.4741 ± 0.1062
KFBG *Cinnamomum*	=	KARC *Cinnamomum*	=	HSH *Cinnamomum*
0.2683 ± 0.0740		0.2689 ± 0.0541		0.2330 ± 0.0505
KFBG *Cyclobalanopsis*	>	HSH *Cyclobalanopsis*	>	KARC *Cyclobalanopsis*
0.6113 ± 0.1188		0.5110 ± 0.1023		0.3865 ± 0.1019
KFBG *Machilus*	>	HSH *Machilus*	>	KARC *Machilus*
0.5664 ± 0.1474		0.4150 ± 0.0904		0.3015 ± 0.0794
KFBG *Mallotus*	>	KARC *Mallotus*	=	HSH *Mallotus*
0.3499 ± 0.0950		0.2419 ± 0.0656		0.2549 ± 0.0639

KFBG *Choerospondias*	=	KFBG *Cyclobalanopsis*	=	KFBG *Machilus*	>	KFBG *Mallotus*	>	KFBG *Cinnamomum*
0.5490 ± 0.1506		0.6113 ± 0.1188		0.5664 ± 0.1474		0.3499 ± 0.0950		0.2683 ± 0.0740
HSH *Choerospondias*	=	HSH *Cyclobalanopsis*	>	HSH *Machilus*	>	HSH *Mallotus*	=	HSH Cinnamomum
0.4741 ± 0.1062		0.5110 ± 0.1023		0.4150 ± 0.0904		0.2549 ± 0.0639		0.2330 ± 0.0505
KARC *Choerospondias*	>	KARC *Cyclobalanopsis*	>	KARC *Machilus*	=	KARC *Mallotus*	>	KARC *Cinnamomum*
0.4855 ± 0.1123		0.3865 ± 0.1019		0.3015 ± 0.0794		0.2419 ± 0.0656		0.2689 ± 0.0541

Table 7.19

The relative basal diameter increment per year (RBDI ± s.d.) of different species at different sites. KFBG grassland had two sets of data for the first three species because there were two separate planting trials.

Species\ site	Mean RBDI			
	KFBG grassland		HSH grassland	KARC shrubland
Schima superba	0.22 ± 0.14	0.14 ± 0.07	0.17 ± 0.07	0.23 ± 0.12
Schefflera octophylla	0.24 ± 0.17	0.17 ± 0.07	0.28 ± 0.07	0.21 ± 0.13
Castanopsis fissa	0.19 ± 0.15	0.23 ± 0.08	0.33 ± 0.08	0.31 ± 0.14
Choerospondias axillaris	0.55 ± 0.15		0.47 ± 0.11	0.49 ± 0.11
Cinnamomum camphora	0.27 ± 0.07		0.23 ± 0.05	0.27 ± 0.05
Cyclobalanopsis neglecta	0.61 ± 0.12		0.51 ± 0.10	0.39 ± 0.10
Machilus breviflora	0.57 ± 0.15		0.42 ± 0.09	0.30 ± 0.08
Mallotus paniculatus	0.35 ± 0.10		0.25 ± 0.06	0.24 ± 0.07
Sterculia lanceolata	0.24 ± 0.11		0.18 ± 0.09	0.14 ± 0.05

Chapter 8

Overall Discussion

1. Filter-barriers to Forest Succession in Hong Kong

The results of this study suggest that low seed dispersal, poor seed germination and seed predation together significantly reduce woody species seed availability on degraded grasslands in Hong Kong, especially on open grassland without woody species cover, and therefore delay the rate of forest succession. Once a seed finds its way to germinate, survival is not affected by seasonal drought, grass competition and low soil nutrients (Chapter 6). *Sapium discolor* appeared to be seriously damaged by wind at the exposed study site while nine other species appeared not to be (Chapter 7). Taking the open grassland at Tai Mo Shan (i.e. the KFBG grassland in this study) as an example, seed dispersal into this grassland was approximately one woody species seed per 5 m^2 per year (Chapter 2), that is 2000 seeds per ha. The mean percentage seed removal by seed predators over 60 days at this site was 74% (s.d. = 23%, 12 seed species) in 1996 (Chapter 3). Seed germination rate at this site varied from 0 to 53% depending on species. Thus, for the 520 seeds per ha that survived seed predation, between zero and 244 seeds may germinate. Seedling survival in the first two years was generally high at this site varing from 70 - 100% (except *Sapium discolor*, Chapter 7). Therefore, between zero and 244 woody plants per hectare per year will be able to establish at the KFBG grassland in this model. While early seedling survival does not seem to be limited by seasonal drought, weed competition and low soil nutrients, the latter two factors tend to slow down seedling growth (Chapter 6). All these together explain the low rate of early succession at this site.

Seed trapping study shows that after some woody species are established and grow taller than the grass cover, they will act as bird perches and significantly increase the seed rain on the degraded hillside sites and thus lead to a higher chance of seedling establishment. Less degraded grassland and shrubland in Hong Kong could develop into secondary forest in 30 - 50 years in the absence of disturbance (Zhuang, 1997). The current fire frequency at this site is one every ten years, which is considered low by Hong Kong standards (Chau, 1994), however, forest succession will rarely be fast enough to escape from the impact of fire.

In addition to delaying forest succession, seed dispersal, predation and germination are also important selective filters on species that could establish on degraded hillside in Hong Kong. Seed dispersal seems to be the most important filter. Birds and rodents were found the major seed dispersers in the seed trapping study and most seeds dispersed are limited to small-seeded and fleshy-fruited species. Although the role of bats and civets in seed dispersal on degraded hillside is not certain, both disperse only fleshy-fruited species (see Chapter 2). While fruit bats are known to disperse mainly smaller seeds, such as, *Ficus* spp., civets are known to disperse also large-seeded species such as *Choerospondias axillaris* and *Gnetum luofuense*. Although no wind-dispersed seeds of woody species were collected in the seed trapping study, the seedling survey suggests that they (e.g. *Cratoxylum cochinchinense*, *Gordonia axillaris* and *Ilea chinensis*) are important on certain sites and tend to have highly clumped distribution. Non-wind dispersed dry seeds, such as those of the Fagaceae and Camellia, are apparently not dispersed in Hong Kong. Dudgeon and Corlett (1994) suggest that the original dispersal agents for these seeds, mainly forest specialists, are locally extinct. The seed predation study shows that seed predation is not selective on seed size but selective on seed texture (Chapter 3). Seed species with tough and thick seed coats, such as *Choerospondias axillaris* and *Elaeocarpus sylvestris*, tend to be less affected by seed predation. This is possibly because only rats, not ants, were found to be important seed predators on degraded hillsides in Hong Kong. The seed germination study shows that tree seed germination rates on degraded hillsides were low and varied between species, e.g. the germination rate of *C. axillaris* was between 1 - 3.4 % at different hillside sites (Chapter 4). Seasonal drought, grass competition and low soil nutrients of degraded hillside sites are not significantly affecting seedling survival (Chapters 6 & 7). However, the latter two factors are significantly affecting growth. Therefore, they appear to be moderate filters in affecting species dominance on degraded hillside sites in Hong Kong.

2. Comparison with the Amazonian Forest Pastures

Forest succession on degraded Amazonian forest pastures with histories of heavy use is, in many ways, similar to the secondary grasslands on degraded hillside in Hong Kong. Both are facing similar filter-barriers, however, their relative importance was found to be different. Tree seedling establishment in highly degraded Amazonian pastures is limited by a lack of seed dispersal, seed predation, seedling predation, and seasonal drought (Nepstad *et al.*, 1990). In terms of seed dispersal, the situation in Hong Kong and the Amazonian pastures are similar, birds are the major seed dispersers, and bird-dispersed seed species are concentrated beneath perches (see Section 8.1 above and Chapter 2). For seed predation, apart from rats, ants, especially the leaf cutter ant *Atta sexdens*, were found to be very important seed predators in the Amazon pastures (Nepstad *et al.*, 1990, 1991). The presence of ants as seed predators leads to a much higher seed predation intensity on small seeds (<0.01g). Apart from this, other aspects of seed predation between the two places are similar.

While seedling predation does not appear to be an important filter-barrier in Hong Kong (at least in the three study sites), the occurrence of leaf cutter ants in the Amazonian pastures leads to very serious seedling predation (Nepstad *et al.*, 1990). As in Hong Kong, rats sometimes eat tree seedlings in Amazonian pastures but the seedling predation intensity does not appear to be very high. In fact, no seedling damage by rats was observed in the planting trials in this study.

Nepstad *et al.* (1990; 1991) have not looked into seed germination as a filter-barrier in Amazonian pastures. However, a study in abandoned pastures in Costa Rica found that the percentage seed germination varied from 0 - 53% (Holl, 1999), which was the same range as in my study. Holl (1999) further shows that whilst seed germination in abandoned pasture is significantly lower than in the forest floor in the dry season, there is no difference in the wet season. The tree seed germination experiment in this study did not go into such a detailed level and therefore cannot be compared.

While seasonal drought is found to be a very important filter-barrier to tree establishment in the Amazonian pastures (Nepstad *et al.*, 1990, 1991), it is not an important factor in abandoned pastures in Costa Rica (Holl, 1999). This suggests that the importance of seasonal drought could be very different in different sites. While the current study suggests that seasonal drought is not a filter-barrier to seedling establishment in the three study sites, it may be an important factor in other, drier sites in Hong Kong.

3. Implications for Efficient and Economical Reforestation in Hong Kong and the Surrounding Region

Despite the fact that various reforestation methods have been tried in Hong Kong, including direct seeding, planting bare-rooted seedlings and container-grown seedlings. No one, single method seemed to be highly successful in reforesting Hong Kong with species-rich secondary forests (see Chapter 1). Measures to promote natural forest regeneration has been considered a cost-effective reforestation strategy in South China (Peng, 1996; Yang *et al.*, 1995; 1999; Zhou, 1995) as well as in other parts of the tropics (e.g. Alias *et al.*, 1998; Forest Restoration Research Unit, 1998; Goosem & Tucker, 1995; Moline, 1999), but this has not been seriously considered in Hong Kong. Considering the rugged terrain and very high labour costs in Hong Kong, it is reasonable to assume that reforestation by promoting natural forest regeneration will also be cost-effective. It is possible that the various filter-barriers to forest regeneration on degraded hillsides in Hong Kong identified in this study can be overcome, making forest regeneration possible. Here, I proposed a low input and efficient planting strategy for Hong Kong, which may also be used in South China. In addition, recommendations are made concerning the existing planting practices in Hong Kong.

3.1 A cost-effective reforestation strategy for Hong Kong

Fire is the major threat in the terrestrial ecology of Hong Kong (Chau, 1994). No reforestation strategy can be successful without effective fire-control measures. Therefore, for any given reforestation site, the first step is to conduct a fire hazard assessment. It should include a fire history review and the identification of fire sources at or near the planting site. A properly-designed firebreak network should then be set up. The firebreak should consist of a planted tree belt no less than 15 m wide and a no-vegetation belt 10 - 20 m wide outside the tree belt. Tree species in the firebreak should be fast growing. Apart from exotic species such as *Acacia confusa*, native species may also be considered. Having said that, no native species have been tested as firebreak trees in Hong Kong. However, densely planted *Schima superba* (<1 m spacing) has been widely used in South China forests as a firebreak. A complementary publicity campaign on fire prevention should be launched at the rural villages near the planting site. Fire hazard warning signs should be put up along the paths in the planting site to alert countryside visitors.

The second step is to accelerate forest succession by increasing the input of tree propagules to the site. The lack of seed dispersal can be overcome by promoting natural seed dispersal and by direct seeding. The natural seed rain could be increased by the provision of bird perches to reinforce any island vegetation on site. This could be accomplished by planting tree seedlings in patches across the site and using artificial perches (Holl, 1998c). Since only a small area (c. 5%) relative to the total area of the site needs to be planted for this purpose, a higher planting standard can be achieved. Native species should be used and higher planting requirements should be set to ensure high survival and growth. All seedling transportation and planting precautions should be strictly enforced (see 8.3.2 below) and post-planting irrigation in extremely dry weather should be provided to minimise seedling mortality. To promote faster seedling growth, bigger planting holes should be prepared and fertilizers should be provided. Below-ground competition can be removed by herbicide or by manual weeding if herbicide use is considered too controversial.

The overall diversity of the site can be increased by direct seeding, especially by species that are not well-dispersed naturally, e.g. the wind-dispersed species, the Fagaceae, the Camellia, *Styrax suberifolius* and *Tutcheria championii*. To do this, a seed predation experiment and rodent trapping study should be conducted in advance to assess the extent of seed predation. Should seed predation be important, seeds for direct seedling should be chemically treated with rodent repellent (see Chapter 3). Since seed germination in the field could be very low, most seeds used for direct seeding should consists of seed species that at least are known to have high germination rates in the nursery. For example, *Schefflera octophylla*, *Castanopsis fissa*, *Cyclobalanopsis myrsinifolia*, *C. neglecta*, *C. edithiae*, *Machilus berviflora*, *Reevesia thyrsoidea*, *Sterculia lánceolata*, *Pygeum topengii* and *Cordia dichotoma* (unpublished data). The last three species are worthy of special consideration. They fruit either at the beginning or in the middle of the summer wet season, and in addition to high germination rates, they germinate immediately after dispersal and have high initial growth rate (they could reach 30 cm in two months). In theory, these attributes would allow them to establish better on degraded hillside sites. However, field trials are needed to verify this.

Ideally, the fire prevention plan should be in force until a closed shrubland or secondary forest is formed, which will then be relatively immune to fire (Chau, 1994). This would probably take 15 to 20 years. However, the fire prevention plans may be suspended after the firebreak is fully functional (5 - 10 years). Management of the vegetation island will only be needed for three to five years. Apart from this, the site should require no further management. Additional direct seeding could be done as boosters if resources are available.

3.2 Recommendations concerning existing tree planting practices

The above proposed reforestation strategy will be cost-effective, especially on large sites. However, this strategy may be criticised by the public for a longer lag phase before the forest is formed. The existing way of covering the whole site with tree seedlings seems inevitable, at least in the near future. Therefore, recommendations for improving the existing reforestation practices, basing on the results of this study, are made. However, the above proposed fire management plan should apply to all reforestation projects in Hong Kong.

In recent years, with an increasing concern for ecology, there is a bigger social pressure on using native tree species in reforestation. The Government is now formulating a sustainable development strategy in Hong Kong for the 21st century (Hong Kong Government, 1999). One of the guiding principles of the strategy, focusing on biodiversity, is "to maintain the biodiversity of Hong Kong and to minimize any threat which consumption in Hong Kong may have on biodiversity elsewhere". The emphasis on using native tree species in reforestation is likely to increase in the future. In fact, the Agriculture and Fisheries Department has been actively promoting native species since the beginning of the 1990s. Apart from trying more native species in Country' Parks, a new programme, known as the Native Species Planting Scheme – Schools, was launched in June 1999 to enhance the young people's understanding of the importance of reforestation and biodiversity conservation (http://www.afdparks.gov.hk/nativespp/nativesch.htm). The Kadoorie Farm and Botanic Garden, a public conservation and education centre in Hong Kong, has also set up a native tree nursery [managed by myself] in late 1997 to facilitate scientific research on native tree species and promote the use of native species in forest rehabilitation.

To date, with a few exceptions, the survival of all native species has been reported to be very low (e.g. Lay *et al.*, 1999). However, this study shows that good post-nursery care and precautions can greatly enhance the survival of native tree species in the first two years (Chapter 6 & 7). In addition to being a confounding factor in scientific experiments, poor post-nursery care and precautions could lead to very high seedling mortality. This is a possible explanation for the failure of many planting trials using native tree species in Hong Kong. On the other hand, the high tolerance of exotic tree species to the planting shock may be one of the reasons why they can do better on degraded hillsides in Hong Kong. Due to the high cost of maintaining forester teams in Hong Kong, it seems that the involvement of contractors is necessary. The existing forestry contract already has very detailed conditions governing post-nursery care and planting precautions (Appendix 1.2), however, they are not normally observed (see Chapter 1). It is therefore recommended that the Government should increase the level of monitoring of the planting project to make sure that all of these precaution measures are implemented in all planting projects.

The results of the study also suggest that removing below ground competition and adding fertilizers could significantly enhance seedling growth (see Chapter 6). Whilst the current practices have already included fertilizer applications, weeding is mainly accomplished manually by sickles or mechanically by back-pack grass cutters. However, both methods fail to remove below ground competition, which is also considered far more important than above ground competition in the reforestation of grassland (Nepstad *et al.*, 1990, 1991; Davis *et al.*, 1998). In addition, Zhuang (1997) shows that mechanical weeding can actually hamper natural tree regeneration as naturally occurring young seedlings are removed together with the weeds. Tjitrosemito (1986) suggested that using herbicides in grassland reforestation had an additional advantage of reducing soil erosion over mechanical weeding methods. The use of herbicides for forest site preparation has been increasing in plantation forestry (Lowery & Gjerstad, 1991). Peng *et al.* (1993) also found that chemical weed control could be used to promote forest regeneration in degraded forest land and was far more cost effective, with up to 70% reduction in cost and 80% reduction in labour. However, herbicide usage in reforestation in the tropics is still problematic due to the lack of effective regulatory programmes, limited herbicide research, few people knowledgeable and trained in herbicide use, and misinformation regarding herbicides (Lowery *et al.*, 1993). But, In Hong Kong, apart from the legal prohibition on the use of herbicides in water catchments, it is likely that the environmental problems and social concerns relating to the use of herbicide have restricted its uses in reforestation projects. However, this should not forbid scientific studies on chemical weed control methods in reforestation. It is recommended that low-toxicity herbicide use is considered on less sensitive sites, such as those outside water catchment areas. In sites covered with grass species that are short or not too dense, such as *Ischaemun* spp. and *Arundinella* spp., weeding may not be necessary.

For very degraded sites, if planting exotic species or mixtures of native and exotic species is necessary, longer-term measures to enrich the diversity of the planting site should be included. Apart from thinning the exotic species, direct seeding and planting of islands of native trees as described above could be applied.

4 Limitations to This Study and Gaps in Knowledge

The major limitations of this study are the small number of sites and species tested, especially for the study on factors affecting seedling growth and survival, where only one site and four species was used (Chapter 6). It is uncertain if the results are true at other sites with different settings e.g. altitude, slope orientation or vegetation cover. In addition, seedling growth and survival were only monitored for two years. Mortality and/or growth rates may change in subsequent years. Casual observations at both planting sites that were not affected by fire recently found many individuals reaching over 2 m in height, three years after planting.

Whilst this study has shown that bird perches could significantly enhance the seed rain, it has not assessed whether fruit-bearing perches could further increase the seed rain or increase the diversity of the seed rain. A further field study on this should be conducted, as this would affect the species choice for the island vegetation to be established on the reforestation site to accelerate natural succession. A large scale field experiment to demonstrate the effectiveness of vegetation island in accelerating natural forest succession is also needed to convince the administrators and the public that this is a suitable and cost-effective reforestation strategy in Hong Kong. Further seed predation experiments using chemical rodent repellants should be conducted so as to find out if it is possible to minimize seed predation. Currently, direct seeding has only been successful with one species (*Pinus massoniana*) in the reforestation history of Hong Kong. Similarly in South China, only this species was found successful in aerial seeding although another native species, *Schima superba*, was also found successful when mixed with the pine species (Huang & Shen, 1993). However, in view of the far lower cost of direct seeding and the advantage of this method at remote and steep sites, direct seeding experiments using different native species at different times of the year are needed to find out the best species and timing for successful seed germination and seedling establishment.

Because biodiversity conservation is one of the main aims of reforestation in Hong Kong, the planting of native tree species seedlings, either at the beginning or in the middle stages of reforestation projects is inevitable. However, basic information on the more than 390 native tree species in Hong Kong is still insufficient. The nursery techniques required to produce most native species are unknown. Many planting trials were conducted in the past but were not adequately documented (Corlett, 1999). More properly-designed planting trials on native species with adequate precautions and documentation are needed. One of the aims of these planting trials should be to identify framework species that could facilitate forest formation (Forest Restoration Research Unit, 1998; Goosem & Tucker, 1995).

5. A Reforestation Plan for Hong Kong

The current forest cover, including both spontaneous and planted forests, is about 14% of the land area of Hong Kong (Ashworth *et al.*, 1993). The secondary shrubland cover is 37 % (19% are tall shrub and 18% are low shrub) and grassland cover is 16.5%. The reforestation target should be to replace most of the grasslands and some of the low shrubland with forests within 20 to 30 years so that the forest cover could reach 30%, i.e. reforesting an additional 170 km^2 of land. The priority areas should be lands within the Country Parks (c. 40% of the land area) and the primary aim should be for biodiversity conservation. If the recommended reforestation strategy is adopted (see 8.3.1 above), the area of firebreaks and vegetation island needed will be around 10 %, i.e. about 17 km^2 of land needs to be planted with trees. Spreading it over ten years and for a 1 m x 1m spacing, 1.7 million tree seedlings will have to be planted per year. Just for planting trees, the annual cost will be 34 million Hong Kong dollars

(4.4 million US dollars) for HK$ 20 per tree, which is a small amount of money by Hong Kong standards. For instance, it is less than 10% of the annual budget for Country Parks. In addition, the current interests in reforestation have made fund-raising from the society possible. The good responses to the Agriculture and Fisheries Department's Corporate Afforestation Scheme suggest that part of this money can be accomplished by private sponsorship.

Appendices

Appendix 1.1

Number of patches planted by direct seeding and number of bare-rooted seedlings planted between 1873-1938 in Hong Kong hillsides for afforestation purpose. Data extracted from the forestry reports 1873 – 1938. The number in bracket is the percentage of *Pinus massoniana*. "?" indicates that number is not specified in the forestry report. Those years where no data are available are not listed in the table.

Year	Patches planted by spot sowing	Bare-rooted seedlings	Notes
1873	0	10,337	Numbers included the roadside trees planted
1874	0	4,970	Numbers included the roadside trees planted
1875	0	59132	Numbers included the roadside trees planted
1876	0	10,770	Numbers included the roadside trees planted
1882	808490 (98%)	288740 (70%)	
1883	845646 (98%)	311963 (98%)	
1884	384140 (100%)	330019 (97%)	
1885	209838 (98%)	363338 (94%)	
1886	?	299911 (98%)	
1887	217738 (100%)	157144 (96%)	
1888	289997 (100%)	392328 (97%)	50000 seeds by broadcast sowing
1889	0	601211 (91%)	
1890	0	556982 (93%)	
1891	0	115081 (87%)	
1892	197889 (100%)	158774 (97%)	
1893	86596 (100%)	193052 (94%)	
1894	0	63607 (56%)	32% was *Cunnghamia lanceolata*
1895	0	55664 (63%)	19% was *Cunnghamia lanceolata*
1896	0	29949 (29%)	More species planted this year
1897	0	26066 (41%)	24% was *Lophostemon confertus*
1898	0	33923 (71%)	
1899	0	54582 (81%)	The New Territories (N.T.) added to the colony
1900	48520 (95%)	38214 (85%)	94% were planted in N.T.
1901	0	139048 (90%)	99% were planted in N.T.
1902	137000 (100%)	39438 (81%)	58% were planted in N.T.
1903	0	16099 (80%)	Planting this year was limited by the partial failure of the pine seedling crop.
1904	0	163265 (97%)	All tree planted in Kowloon and Hong Kong
1905	0	344200 (97%)	Only 18 % were planted in N.T.
1906	88137 (100%)	129368(98%)	Only 0.7 % were planted in N.T.
1907	178361 (100%)	57905 (98%)	
1909	?	?	All trees planted and sown were the pine *P. massoniana*
1910	528200 (100%)	0	
1911	> 418915 (?)	?	Exact numbers not available in the report
1912	175000 (0%)	10200	7 broad-leaved species sown as an experiment
1913	?	80000 (89%)	420 pounds pine seeds sown by broadcast
1914	31500 (100%)	32000 (84%),	Another 185 pounds pine seeds sown by broadcast
1915	36000 (100%)	57494 (82%)	Another 351 pounds pine seeds sown by broadcast
1916	?	103721 (87%)	62 pounds pine seeds sown by broadcast
1917	?	63050 (78%)	52 pounds pine seeds sown by broadcast
1918	114000 (100%)	55590 (79%)	Another 85 pounds pine seeds sown by broadcast
1919	55960 (100%)	30024 (75%)	Another 196 pounds pine seeds sown by broadcast
1920	?	10568 (80%)	Another 90 pounds pine seeds sown by broadcast
1921	50000 (100%)	3400 (85%)	Another 384 pounds pine seeds sown by spot and broadcast
1922	129455 (100%)	5167 (100%)	Another 86 pounds pine seeds sown by spot and broadcast
1923	103077 (100%)	15218 (0%)	Another 718 pounds pine seeds sown by broadcast
1924	120000 (100%)	8696 (23%)	Another 917 pounds pine seeds sown by broadcast
1925	216500 (100%)	7708 (0%)	

Appendix 1.1 Cont.

Year	Patches planted by spot sowing	Bare-rooted seedlings	Notes
1926	70500 (100%)	9264 (0%)	Another 1100 pounds pine and 110 pounds *Leucaena leucocephala* seeds sown by broadcast. Some *Pinus radiata* were sown at Tai Po Kau Forest Reserve but numbers not specified.
1927	197476 (100%)	?	Another 995 pounds pine and 110 pounds *L. leucocephala* seeds sown by broadcast
1928	196000 (100%)	?	Another 1666 pounds pine and 55 pounds *L. leucocephala* seeds sown in mix by broadcast
1929	250480 (100%)	26000 (0%)	Another 1725 pounds pine and 250 pounds *L. leucocephala* seeds sown in mix by broadcast
1930	277193 (100%)	?	Another 2018 pounds pine and 200 pounds *L. leucocephala* seeds sown in mix by broadcast
1931	94753 (100%)	?	Another 2678 pounds pine seeds sown by broadcast
1932	259085 (100%)	12740 (4%)	Another 2031 pounds pine seeds sown by broadcast
1933	331658 (100%)	11588 (0%)	Another 2089 pounds pine seeds sown by broadcast
1934	232488 (100%)	> 823 (0%)	Another 2931 pounds pine seeds sown by broadcast
1935	273852 (100%)	21500 (0%)	Another 3591 pounds pine seeds sown by broadcast
1936	306102 (100%)	12900 (0%)	An unknown amount of pine seeds were sown by broadcast
1937	352758 (100%)	16894 (?)	Another 5309 pounds pine and 94 pounds unspecified seeds sown by broadcast
1938	261834 (100%)	771 (0%)	Over 3500 pounds pine seeds sown by broadcast
1939	79345 (100%)	>5600 (?)	Around 3000 pounds pine seeds sown by broadcast

Appendix 1.2

The terms and conditions of a typical afforestation contract in Hong Kong.

I. Seedling Tree Planting (1st year)

1. Seedlings tree	A seedling tree shall be a tree with all the following characteristics: - a single slender stem - a well developed vigorous root system - aged between 1 – 2 years old - a height of above soil level of 150-600 mm - grown in a container not less than 75 mm in diameter and 100 mm deep or a tube not less than 60 mm in diameter and 150 mm long.
2. Species required	List of species with scientific names, common names, and numbers required.
3. Materials to be as specified	Once the plant species was selected, they shall be true to species and healthy and shall not be less than the minimum size specified.
4. Storage of plant materials	Seedling trees, which are not immediately planted in their permanent positions, shall be stood upright on level ground, protected and maintained in good condition by the Contractor.
5. Damaged plant material	The Contractor shall immediately replace all plant material which has become damaged or has fallen below the specified standard due to poor transportation, handling, maintenance and storage, and shall immediately clear all substandard plant material off site at his own expenses.
6. Planting work	All plants shall be planted to accommodate the spreading root system of the plant to the same soil depth as in the nursery, and shall be watered before removing them from containers. Plants are to be positioned upright and soil firmed around the roots.
7. Pit planting	a. Seedling tree species shall be pit planted at 1000 mm distance apart in the fire-break zone and at 1200 mm distance apart in non-fire-break zone. The minimum dimensions of planting pit shall be as follows: Width and length of pit: 275 mm Depth of pit: 300 mm b. On slopes the depth will be measured from the lowest adjacent ground level on the download side of the pit. The pit shall be backfilled to a suitable level to maintain the same relationship between the root collar of the plant and soil surface as exist in the nursery. The plant shall be set upright in the pit, the roots spread carefully and backfilled with soil shall be completed, firming first with the knuckles and then finally with the feet.

I. Seedling Tree Planting (1st year) – Cont.

8. Backfilling operation	In all backfilling operations adequate care shall be taken to avoid any damage to the root system, branches and leaves. The soil shall be consolidated in layers to ensure that the plant is firmly held in the ground and that air pockets are not left around the roots. The backfilled soil shall be free of weeds and rubbish.
9. Pre-planting fertilizing	Apply slow release granular fertilizer in the ratio 15:9:15:2 (N-P-K/Mg) at 25 g per seedling and spread around each pit.
10. Replacement of plants due to poor material or workmanship	The Contractor shall on his own expenses replace all plants which are dead, dying or otherwise unsatisfactory, if the cause is a consequence of the use of poor materials or workmanship.
11. Daily planting progress report	The Contractor shall submit the daily planting progress report to the Client showing the planting dates and respective quantities of seedling tree planted on site.
12. Time of completion	The Contractor shall execute and complete the planting works in the manner specified and within the period as instructed by the Client.
13. Weeding	The planting area shall be maintained by removal of all competing and overhanging weed growth and by keeping all areas within a radius of 600 mm of each seedling a weed free and tidy condition to prevent the seedlings from hill fire.
14. Maintenance achievement at the end of 1st year	Fair wear and tear (e.g. damage due to hill fire, cow bite) excepted, a survival rate of 75% of the seedlings planted shall be achieved at the end of the 12-month period after the planting. If the survival rate is below 75%, the Contractor shall at his own expenses replace the dead seedlings planted to achieve the 75% survival rate.

II. Establishment Works (2nd year)

1. Weeding	The planting area shall be maintained by removal of all competing and overhanging wed growth and by keeping all areas within a radius of 600 mm of each seedling a weed free and tidy condition. The Contractor shall weed planting areas twice in the year i.e., one between January and March and one Between October and November. The implementation dates of the weeding operation shall be approved by the Client in advance.
2. Adding fertilizer	The Contractor shall add 50 g of slow releasing granular compound fertilizer in a ratio 20:11:11 (N-P-K) spread around the ground level of each seedling.

III. Establishment Works (3rd year)

1. Weeding	The Contractor shall clear the area within a radius of 600 mm of each seedling once between the period October and November of the year. The implementation dates of the weeding operation shall be approved by the Client in advance.
2. Adding fertilizer	The Contractor shall add 50 g of slow releasing granular compound fertilizer in a ratio 20:11:11 (N-P-K) to the area around the seedling right after the weeding operation.
3. Pruning	The Contractor shall prune all seedling trees, when agreed with by the Client during the same period of weeding and adding fertilizer operation. Pruning and removal of unwanted lower branches/shoots to maintain a good shape shall be carried out by cutting flush with the adjoining stem and in such a way that no part of the stem is damaged or torn. Ragged edges of bark shall be trimmed with a sharp knife.

Appendix 2.1

Number of seeds of each species collected by month at each site. "S" indicates traps placed under tree canopy. "O" indicates traps not shaded by any vegetation. Underlined numbers indicate that the seeds were recovered from rats' faecal pellets in the seed traps. The rest were recovered from bird droppings.

		Number of seeds collected by month																								
		KARC Shrubland													KFBG Grassland											
Species		**10**	**11**	**12**	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**		**10**	**11**	**12**	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**
Tree species:																										
Aporosa dioica	S											13	17													
	O												2													
Aralia armata	S				6																					
	O																									
Bridelia tomentosa	S			8	21	20																				
	O																									
Ficus fistulosa	S														<u>216</u>	<u>43</u>										
	O																									
Fiscus variolosa	S									7																
	O																									
Litsea cubeba	S											7													12	5
	O											1														
Litsea rotundifolia	S		1	1	3								45													
	O												5													
Mallotus paniculatus	S		2	28	10																					
	O																									
Memecylon sp.	S	1									2	2														
	O																									
Microcos paniculata	S		1	5	1	7	1																			
	O																									

Appendix 2.1 Cont.

		Number of seeds collected by month																								
		KARC Shrubland													KFBG Grassland											
Species		**10**	**11**	**12**	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**		**10**	**11**	**12**	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**
Machilus chekiangensis	S								1						1								1			
	O																									
Rhus sp.	S			6	12	67	5	7	1			1	4											1		
	O						2		1																	
Rhus succedanea	S				3							5	23													
	O												8													
Sapium discolor	S	4	18	12	6	5																				
	O					1																				
Schefflera octophylla	S			77	294	693	592	34	9											19						
	O					6	2																			
Symplocos sp.	S											1												1		
	O																									
Syzygium levinei	S																					1				
	O																									
Shrub species:																										
Eurya chinensis	S														21+5	*16*	1	8								23
	O																									
Glochidion eriocarpum	S							2	23																	
	O																									
Glochidion puberum	S		1																							
	O																									

Appendix 2.1 Cont.

		Number of seeds collected by month																								
		KARC Shrubland													KFBG Grassland											
Species		**10**	**11**	**12**	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**		**10**	**11**	**12**	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**
Ilex asprella	S								17	153	24															
	O									12		1														
Ilex pubescens	S			10	32	1									1		2	1								
	O																									
Lantana camera	S								4				7									2		1		
	O												<u>1</u>													
Ligustrum sinense	S				4	1																				
	O																									
Melastoma candidum	S	50		357	274	180	160									535	1800+<u>1328</u>	429+<u>1240</u>	60+<u>44</u>							
	O																									
Melastoma sanguineum	S					349												100+<u>796</u>								
	O																									
Psychotria rubra	S	5		1	6	27	5						13													
	O																									
Rhodomyrtus tomentosa	S	31	14	1		1			1		1	68	997+<u>18</u>		11+<u>377</u>	5+<u>99</u>	23+<u>1</u>	9	1						1	48
	O	2						2	4	1	3	19	228+<u>1</u>		1											
Sarcandra glabra	S					1																				
	O																									
Viburnum sempervirens	S			5																						
	O																									

Appendix 2.1 Cont.

		Number of seeds collected by month																								
		KARC Shrubland													KFBG Grassland											
Species		**10**	**11**	**12**	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**		**10**	**11**	**12**	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**
Climbers:																										
Embelia laeta	S								2	15																
	O									1																
Embelia ribes	S								88	8	6												1	3		
	O										1															
Psychotria serpens	S							2																		
	O																									
Smilax china	S				2																					
	O																									
Tetracera asiatica	S		3																							
	O																									
Herbaceous species:																										
Carex baccans	S															6										
	O																									
Carex cruciata	S												1													1
	O										3		4													
Cassytha filiformis	S					2																				
	O																									
Dianella ensifolia	S			1	1				2						1											
	O														14											
Rubus reflexes	S											4											33	5		
	O																									

Appendix 3.1

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Tree seed predation on degraded hillsides in Hong Kong

Chi Hang Hau*

Department of Ecology and Biodiversity, The University of Hong Kong, Pokfulam Road, Hong Kong, P.R. China

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Tree seed predation on degraded hillsides in Hong Kong

Chi Hang Hau*

Department of Ecology and Biodiversity, The University of Hong Kong, Pokfulam Road, Hong Kong, P.R. China

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Abstract

Hong Kong was once entirely covered in subtropical rain forest but almost all of this had been cleared by the 17th century. Today around 9% of the land area is covered by secondary forest, mostly developed since 1945, with an additional 5% under plantations, largely of exotic species. The use of mixed, native species for reforestation has only started recently. The aim of this study was to determine whether seed predation was a barrier to natural forest regeneration on degraded hillsides. Removal of seeds of eight tree species in the winter of 1995 and 12 in 1996 at four Hong Kong hillside sites was monitored. Seed removal at the two shrubland sites was higher than at the two grassland sites in both 1995 and 1996. Most seeds placed in the shrubland sites in 1996 were removed; 11 of 12 species were totally removed from one shrubland site within 60 days, while only one of 12 species was totally removed in one grassland site. Rats were found to be the major seed predator. They included *Niviventer fulvescens* and *Rattus rattus flavipectus*. The tough/thick coated seeds of *Choerospondias axillaris* and *Elaeocarpus sylvestris* had the lowest mean percentage removal. Currently, all reforestation efforts in Hong Kong use container-grown seedlings, which is expensive even on accessible sites and impractical in remote areas. The results of this study suggest that direct seeding may be possible if species with tough/thick coated seeds are used. © 1997 Elsevier Science B.V.

Keywords: Seed predation; Forest regeneration; Reforestation; Degraded hillsides

1. Introduction

Archaeological evidence indicates that about 6000 years ago Hong Kong was covered in dense forest where the tree families Lauraceae and Fagaceae were probably dominant (Dudgeon and Corlett, 1994). The original forest has now disappeared, except in some steep ravines, and has been replaced by a variety of secondary vegetation types. Clearance for cultivation, cutting for firewood and hill fires were probably the main causes of the disappearance of forest. Though firewood collection no longer threatens the survival of the forest, hill fires remain important (Dudgeon and Corlett, 1994) and urban development has become an additional threat. For example, some 20 ha of forest was cleared in 1993 for building a new airport. Today, around 9% of the land area is covered by secondary forest, mostly developed since 1945, with an additional 5% under plantations, largely of exotic species (Dudgeon and Corlett, 1994). Recently, the Hong Kong government has made the first attempt to plant mixed, native species for reforestation, albeit on a trial and error basis (very little research work on planting native tree species has been done in Hong Kong to date). A total of 0.26 million container-grown seedlings of a variety of native species will be used to recreate a 60 ha woodland at a cost of over US$1.2 million (Agriculture and Fisheries Department, 1995).

This is the first study in Hong Kong examining seed predation in degraded hillsides. It aims to determine if post-dispersal seed predation is a barrier to natural

* Fax: +852 26727715; e-mail: billyhau@ibm.net

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forest regeneration in hillside grassland and shrubland. This is part of a wider project examining the barriers to natural forest regeneration in Hong Kong. Other experiments have been and will be conducted at hillside grassland and shrubland to examine tree seed dispersal, seed fate and seedling growth rate and survival.

Seed predation is a significant factor in causing demographic and evolutionary changes in plant populations (Janzen, 1971; Louda, 1989; Hulme, 1993). Nepstad et al. (1991) have identified seed predation as one of the biotic barriers to natural forest regeneration in abandoned Amazon pasture derived from rainforest. Many studies have shown that seed size is related to the vulnerability of a seed species to seed predators (Louda, 1989; Nepstad et al., 1991; Reader, 1993; Osunkoya, 1994). Small mammals and ants are typical seed predators (Janzen, 1971; Louda, 1989; Hulme, 1993). Nepstad et al. (1991) showed that small seeds (<0.02 g) are more vulnerable to fire ants (*Solenopsis* sp.), cutter ants (*Atta sexdens*) and harvester ants (*Pheidole* sp.) than larger seeds (>5 g).

2. Materials and methods

2.1. Study sites

The study was conducted at two hillside grassland and two hillside shrubland sites in the northern New Territories of Hong Kong. Hong Kong was previously a Dependent Territory of the United Kingdom in southern China but it became a Special Administrative Region of the People's Republic of China in July 1997. It lies between latitudes 22°09′N and 22°37′N and is more than 100 km south of the Tropic of Cancer. The total land surface area of Hong Kong is 1091 km^2, about three quarters of which is countryside.

The mean annual temperature in Hong Kong is 23°C (1961–1990). January has the lowest mean temperature (15.8°C) and July has the highest (28.8°C). The mean annual precipitation is 2214 mm (1961–1990) and it is highly seasonal; monthly precipitation from November through February averages <50 mm (accounting for only 6% of the annual total). Over 77% of the annual total rainfall falls between May and September with the highest rainfall in August (18%). Most hill soils in Hong Kong are formed from weathering of granite or volcanic rocks. Many granitic areas are badly eroded and support only sparse vegetation while most volcanic areas have a continuous vegetation cover (Dudgeon and Corlett, 1994).

The soils at all four sites are volcanic-derived. The Kadoorie Farm and Botanic Garden (KFBG) grassland is on a 40° slope at 550 m and is dominated by *Arundinella* sp., *Ischaemum* sp., *Eulalia* sp., *Eragrostis* sp., *Cymbopogon* sp. and *Miscanthus sinensis*. In a ravine about 300 m away from this grassland is a secondary woodland which is dominated by *Machilus* sp. The Ho Sheung Heung (HSH) grassland and shrubland are next to each other on a 15° slope at 20 m. The grassland is dominated by *Arundinella* sp., *Eulalia sp.* and *Miscanthus sp.* The dominant shrubs are *Baeckea frutescens* and *Rhodomyrtus tomentosa*. These sites are immediately next to a secondary woodland which is dominated by *Microcos paniculata*, *Acronychia pedunculata*, *Schefflera octophylla* and *Machilus chinensis*. A few young trees of *Schefflera octophylla* and *Cratoxylum cochinchinense* occur in the shrubland. The Kadoorie Agricultural Research Center (KARC) shrubland is on a 25° slope at 200 m and is dominated by *Rhodomyrtus tomentosa*, *Litsea rotundifolia* and *Rhaphiolepis indica*. It is adjacent to a young secondary woodland which is dominated by *Schefflera octophylla*, *Mallotus paniculatus*, *Evodia lepta* and *Sapium discolor*. Young trees of these species also occur in the shrubland.

2.2. Experimental design

Since the majority of tree species in Hong Kong produce fruits/seeds in winter, this study was conducted in winter and only seeds from the same winter were used to make sure that more viable seeds were used. In addition, all floating seeds were discarded. A total of 16 seed species with different seed size were tested in this study. Ten species were used in the winter of 1995 and 12 in 1996 (see Table 1). The differences in the number of species used at each site in each year was due to seed availability. For example, *Acronychia pedunculata* and *Choerospondias axillaris* were used in only one site in 1995 because there were not enough seeds for all sites in both years.

Table 1

Mean percentage seed removal at two shrubland and two grassland sites in 1995 and 1996

No.	Species	Family	Mean seed mass (g)[a]	Mean percentage seed removal (60 days) at shrubland sites in:			Mean percentage seed removal (60 days) at grassland sites in:		
				1995	1996	1995 + 1996	1995	1996	1995 + 1996
1	*Evodia lepta*	Rutaceae	0.0066	–[b]	99.0	–	70.5	96.1	83.3
2	*Bischofia javanica*	Euphorbiaceae	0.0101	99.0	–	–	26.0	–	–
3	*Mallotus paniculatus*	Euphorbiaceae	0.0148	–	100	–	–	96.3	–
4	*Gordonia axillaris*	Theaceae	0.0158	70.5	98.0	84.25	19.5	85.1	52.3
5	*Reevesia thyrsoidea*	Sterculiaceae	0.0311	–	95.5	–	–	90.0	–
6	*Sapium discolor*	Euphorbiaceae	0.0466	–	91.0	–	81.5	87.95	84.73
7	*Microcos paniculata*	Tiliaceae	0.1276	63.0	99.5	81.25	26.0	55.7	40.85
8	*Diospyros morrisiana*	Ebenaceae	0.1383	45.0	–	–	2.0	–	–
9	*Acronychia pedunculata*	Rutaceae	0.1611	31.0	–	–	–	–	–
10	*Elaeocarpus sylvestris*	Tiliaceae	0.2900	–	64.0	–	–	28.4	–
11	*Machilus breviflora*	Lauraceae	0.3390	72.0	100	86.0	6.5	58.55	32.53
12	*Quercus championii*	Fagaceae	0.8068	–	100	–	–	81.45	–
13	*Quercus myrsinaefolia*	Fagaceae	0.8100	–	100	–	–	80.95	–
14	*Castanopsis fissa*	Fagaceae	1.6300	–	100	–	–	83.0	–
15	*Lithocarpus glaber*	Fagaceae	1.6624	46.5	100	73.25	12.0	81.7	46.85
16	*Choerospondias axillaris*	Anacardiaceae	2.4648	14.5	–	–	–	–	–

Non-parametric sign tests show that the mean percentage seed removal at shrubland sites was higher than at grassland sites for 1995 ($P < 0.05$) and 1996 ($P < 0.001$).
[a]Mean seed mass was the average weights of 50 seeds.
[b]–, indicates that the species was not tested at that site in that year.

Ten 25 × 30 × 3 cm plastic seed trays were evenly spaced along two 50 m transects, 10 m apart at each site. All seed trays were shaded by vegetation at all sites. The seed trays were lined with a piece of hemp cloth. Ten seeds of each species were placed in each seed tray at the same time. In 1995, seeds were simply placed on seed trays. In 1996, the seed tray was covered with a layer of nursery soil and the seeds were half-buried. Trays were checked daily for the first week, twice a week for the next 3 weeks and once a week afterwards for a total of 60 days. The numbers of missing seeds were counted and any evidence of seed predator occurrence (e.g. the presence of broken seed coats and faecal pellets) was also noted.

Five 10 × 10 × 30 cm cage-traps were set along a 50 m transect at each site for 7 consecutive nights after there had been evidence of rodent seed predators visiting the seed trays. Rodent trapping was conducted only at HSH shrubland in 1995 using bread as bait. *Quercus myrsinaefolia* seeds, which had a high rate of removal, were used as bait at all sites in 1996. Traps were checked every morning and all rodents trapped were identified, measured and kept in the laboratory. They were fed with the same seed species to see how readily they would consume the seeds and what evidence they left behind. They were released back to where there were trapped at the end of the trapping study.

3. Results

Non-parametric sign tests (Ryan et al., 1976) show that the mean percentage seed removal at shrubland sites was higher than at grassland sites in both 1995 ($P < 0.05$) and 1996 ($P < 0.001$) (Table 1). The pattern of seed removal at all sites was very similar in both years. The shrubland sites almost always had higher initial and total percentage seed removal than grassland sites (Fig. 1). In 1996, most seeds at shrubland sites were removed in the first week (Fig. 1). Eleven of 12 species were totally removed in 1996 from KARC shrubland but there was only one species (*Evodia lepta*) totally removed in one grassland site (Table 2). In 1996, *Mallotus paniculatus*, *Machilus breviflora* and all four species of Fagaceae (*Quercus*

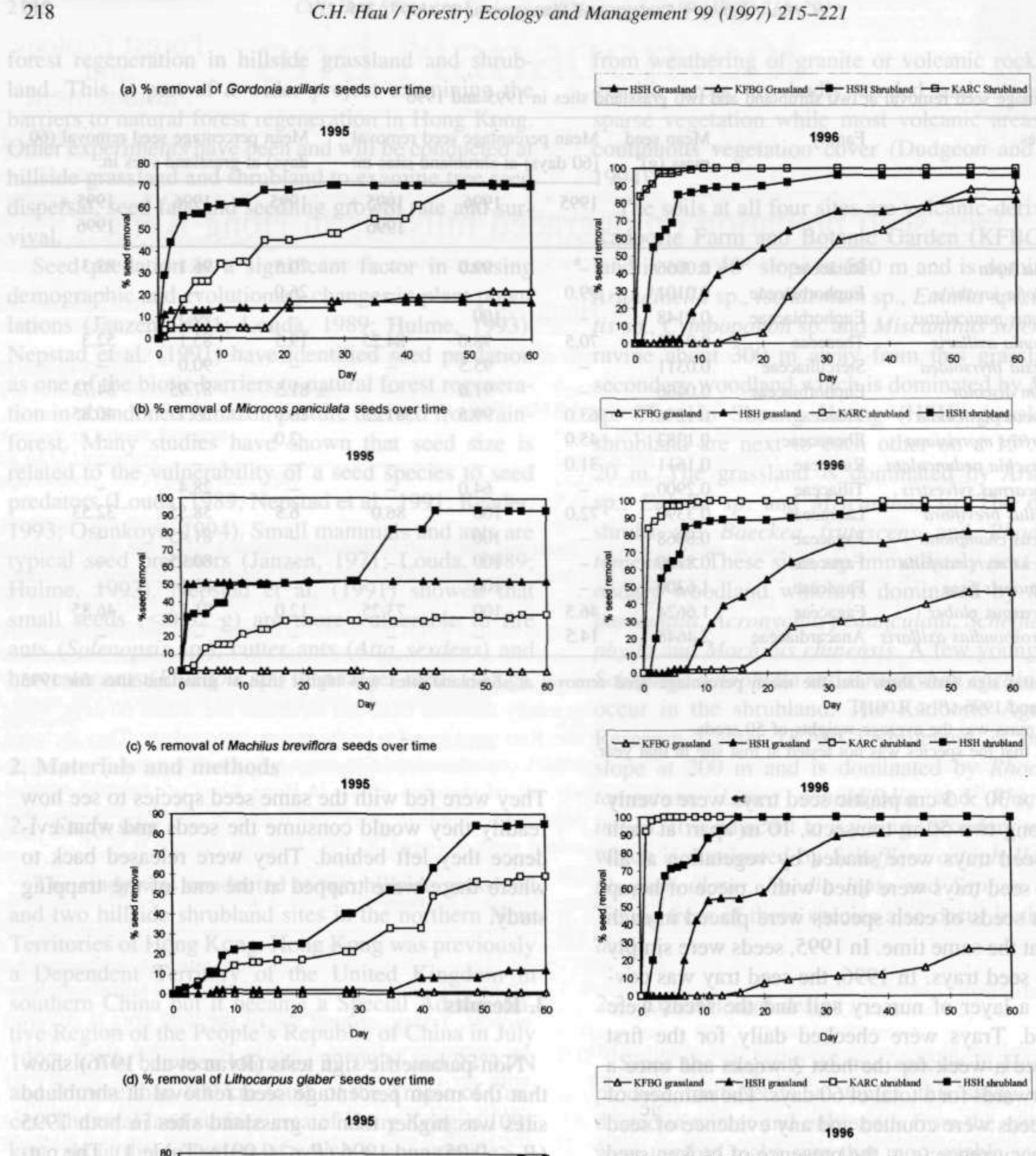

Fig. 1. Percentage seed removal over time for (a) *Gordonia axillaris*, (b) *Microsos paniculata*, (c) *Machilus breviflora* and (d) *Choerospondias axillaris* at all sites in both 1995 and 1996.

Appendix 3.1

Table 2
Percentage seed removal at two shrubland and two grassland sites in 1995 and 1996

		Percentage seed removal (60 days)							
		KARC shrubland		HSH shrubland		KFBG grassland		HSH grassland	
No.	Species	1995	1996	1995	1996	1995	1996	1995	1996
1	*Evodia lepta*	–[a]	100	–	98	71	100	70	92.2
2	*Bischofia javanica*	98	–	100	–	25	–	27	–
3	*Mallotus paniculatus*	–	100	–	100	–	97	–	95.6
4	*Gordonia axillaris*	71	100	70	96	22	88	17	82.2
5	*Reevesia thyrsoidea*	–	100	–	91	–	90	–	90
6	*Sapium discolor*	–	100	–	82	64	97	99	78.9
7	*Microcos paniculata*	33	100	93	99	0	47	52	64.4
8	*Diospyros morrisiana*	38	–	52	–	1	–	3	–
9	*Acronychia pedunculata*	10	–	52	–	–	–	–	–
10	*Elaeocarpus sylvestris*	–	94	–	34	–	49	–	7.8
11	*Machilus breviflora*	59	100	85	100	1	26	12	9.1
12	*Quercus championii*	–	100	–	100	–	74	–	88.9
13	*Quercus myrsinaefolia*	–	100	–	100	–	73	–	88.9
14	*Castanopsis fissa*	–	100	–	100	–	76	–	90
15	*Lithocarpus glaber*	15	100	78	100	1	79	23	84.4
16	*Choerospondias axillaris*	13	–	16	–	–	–	–	–

[a]–, indicates that the species was not tested at that site in that year.

championii, *Quercus myrsinaefolia*, *Lithocarpus glaber* and *Castanopsis fissa*) were totally removed at both shrubland sites (Table 2). It was not clear why the percentage seed removal of *Microcos paniculata*, *Machilus breviflora* and *Lithocarpus glaber* at KFBG grassland was exceptionally low in 1995.

Choerospondias axillaris, which has tough/thick coated seeds, had the lowest mean percentage removal at shrubland sites in 1995 (14.5%). *Elaeocarpus sylvestris*, which also has tough/thick coated seeds, had the lowest mean percentage removal at both grassland (28.4%) and shrubland (64%) sites in 1996 (Table 1). For both species, a lot of the intact seeds left on the seed trays had clear rodent teeth marks.

The mean percentage seed removal was found not to be related to seed size ($r^2 = 0.1165$, $P > 0.05$ for shrubland and $r^2 = 0.0188$, $P > 0.05$ for grassland).

Two rat species, *Rattus rattus flavipectus* (body length 16 cm, tail length 20 cm) and *Niviventer fulvescens* (body length 10.5–13 cm, tail length 11–16 cm) were the only seed predators caught at all sites. Both species readily ate the same seed species in the laboratory and left the same kinds of remains as observed in the field. The overall trapping success in shrubland was higher than in grassland (less small rodents were trapped in HSH shrubland than HSH grassland in 1996 but there was 10 records of baits in the traps of the former site being taken without triggering the traps). No rats were trapped at the KFBG grassland even after two further 7-night trappings with double the number of cages (Table 3). In 1996, trapping success was higher at KARC shrubland where mean percentage seed removal was also the highest (all except *E. sylvestris* were totally removed).

4. Discussion

The results of this study indicate that seed predation by small rodents at hillside grassland and shrubland could significantly reduce the availability of tree seeds for regeneration. Both pioneer species (*Evodia lepta*, *Mallotus paniculatus* and *Sapium discolor*) and potential climax species (the Fagaceae and *Machilus breviflora*) were heavily consumed by seed predators. A study of small mammal distribution in Hong Kong has shown that rodent densities in degraded habitats (i.e. hillside shrubland and grassland) were relatively

Table 3
Rodent trapping results[a]

Site	Number of rodents trapped							
	KARC shrubland		HSH shrubland		KFBG grassland		HSH grassland	
Species	1995	1996	1995	1996	1995	1996	1995	1996
Rattus rattus flavipectus	–	0	2	1	–	0	–	0
Niviventer fulvescens	–	9	4	2	–	0	–	5
Total	–	9	6	3	–	0	–	5

[a]Trapping was carried out at Ho Sheung Heung (HSH) shrubland only in 1995.

high in comparison with other countries. It was estimated that small rodent density could reach as high as 60/ha in lowland grassland and 30/ha in shrubland (Rao, 1994). High rodent densities in Hong Kong may be due to low predator populations and high ground cover of shrubland and grassland. Though rodents form a substantial portion of the diets of civets (*Viverricula indica* and *Paguma larvata*) and leopard cats (*Felis bengalensis*), these mammals have been significantly reduced by trapping, habitat destruction and fragmentation (Rao, 1994). This may have significant implications for similar degraded habitats elsewhere since seed predation by small mammals would likely become a more important factor in affecting the species composition and limiting natural forest regeneration.

Unlike similar seed predation studies elsewhere (Louda, 1989; Nepstad et al., 1991; Reader, 1993; Osunkoya, 1994), the probability of a seed being eaten by seed predators was found not to be related to seed mass. Ants were found to be a major seed predator for very small seeds (<0.01 g) in other studies (Andersen, 1987; Nepstad et al., 1991; Reader, 1993) but no ants were observed carrying seeds away in this study.

All species, except *C. axillaris* and *E. sylvestris*, had broken seed coats found at the seed trays at all sites. The missing *C. axillaris* and *E. sylvestris* seeds were probably carried away whole by rodents as their tough seed coats would take them a significantly longer time to handle. The low seed removal rates of these two species have significant implications on reforestation in Hong Kong. At present, the Hong Kong government uses only container-grown seedlings from its tree nurseries for reforestation. Since 1980, about 330 ha of borrow areas and erosion control sites have been restored using nearly 1.5 million container-grown seedlings produced in government tree nurseries (Webb, 1993). Since it is expensive to produce container-grown seedlings, the government has been striving to reduce production costs as well as improving productivity. In recent years, field trials in sowing seeds directly into polythene seedling containers had been proven successful in producing more cost-effective seedlings (Agriculture and Fisheries Department, 1996). Direct seeding using tough/thick coated species at target reforestation sites, if proved possible, would further reduce the cost of reforestation and allow reforestation to be conducted at remote areas and steep slopes. A direct seeding experiment has just been started using *C. axillaris* and *E. sylvestris* (with tough seed coats) and *C. fissa* and *Machilus breviflora* (with soft seed coats) at the KARC shrubland and KFBG grassland, respectively.

Higher percentage seed removal at all sites for all species in 1996 implies that half-burying the seeds did not reduce the chance of seeds being eaten. The study will be repeated next spring by totally burying the seeds. If the percentage of seed removal is significantly lower, it would imply that direct seeding using species with a soft seed coat may also be possible.

In summary, these results indicate that seed predation is a significant barrier to natural forest regeneration in degraded hillsides in Hong Kong. Seed predation is probably also affecting the species composition of natural forest regeneration. The vulnerability of a seed species to seed predators was not related to its size. Seeds with tough seed coats were less attractive to seed predators.

Appendix 3.1

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Appendix 5.1 Numbers and height distribution of the woody species recorded at each site.

a. High West, West Hong Kong Island.

Tree species	Height in cm					Total	Shrub/ Climber species	Height in cm					Total
	<50	50-100	101-150	151-200	>200			<50	50-100	101-150	151-200	>200	
Gordonia axillaris	197	281	31			509	*Rhodomyrtus tomemtosa*	489	438	7			934
Itea chinensis	1	8				9	*Baeckea frutescens*	18	65				83
Rhus chinensis		18	1			19	*Litsea rotundifolia*	3	13				16
Archidendron lucidum							*Embelia laeta*	21	46	1			68
Phyllanthus emblica							*Rhaphiolepis indica*	16	19				35
Machilus chekiangensis	4	5	1			10	*Melastoma sanguineum*		5				5
Cratoxylum cochinchinense							*Eurya japonica*	13	28	6			47
Acronychia pedunculata	2	3	2			7	*Phyllanthus cochinchinense*						
Diospyros morrisiana							*Breynia fruticosa*	1	3				4
Schefflera octophylla	2	2				4	*Clerodendrum fortunatum*	18	8				26
Zanthoxylum avicennae							*Ficus variolosa*	1	5	1			7
Litsea cubeba							*Ardisia crenata*						
Litsea glutinosa							*Melicope pteleifolia*						
Glochidion wrightii							*Dalbergia benthami*						
Pentaphylax euryoides							*Eurya chinensis*	1	5				6
Schima superba		8				8	*Embelia ribes*						
Sapium discolor							*Ilex asprella*						
Adinandra millettii							*Symplocos paniculata*						
Daphinphyllum calycinum		4				4	*Myrsine seguinii*		7	2			9
Pinus massoniana							*Mussaenda pubescens*						
Artoacrpus hypargyrea	3					3	*Sarcandra glabra*						
Mallotus paniculatus							*Croton lachnocarpus*						
Glochidion lanceolarium							*Glochidion eriocarpum*						
Bridelia tomentosa							*Rhamnus crenata*						
Total	209	329	35			573	*Ficus hirta*						
							Gardenia jasminoides						
							Dentrotrophe frutescens						
							Gnetum luofuense						
							Strophanthus divaricatus						
							Tot	[illegible]	642	17			1240

Appendix 5.1 Cont.

b. Ngau Ngak Shan, East New Territories.

Tree species	Height in cm					Total	Shrub/ Climber species	Height in cm					Total
	<50	50-100	101-150	151-200	>200			<50	50-100	101-150	151-200	>200	
Gordonia axillaris	59	54	96	25	17	251	*Rhodomyrtus tomemtosa*	79	135	117	29		360
Itea chinensis	13	45	31	7	6	102	*Baeckea frutescens*	74	186	215	15	1	491
Rhus chinensis							*Litsea rotundifolia*	3	2	7			12
Archidendron lucidum							*Embelia laeta*	13	18	2	2	2	37
Phyllanthus emblica							*Rhaphiolepis indica*	11	30	5	3		49
Machilus chekiangensis							*Melastoma sanguineum*		14	15			29
Cratoxylum cochinchinense							*Eurya japonica*	13	12	9	1		35
Acronychia pedunculata		5	16	4		25	*Phyllanthus cochinchinense*						
Diospyros morrisiana							*Breynia fruticosa*						
Schefflera octophylla		3	1			4	*Clerodendrum fortunatum*	2	1				3
Zanthoxylum avicennae							*Ficus variolosa*	3	6	23	2		34
Litsea cubeba							*Ardisia crenata*		8				8
Litsea glutinosa							*Melicope pteleifolia*						
Glochidion wrightii				3		3	*Dalbergia benthami*						
Pentaphylax euryoides		5	3	1	2	11	*Eurya chinensis*	2	2	1			5
Schima superba		2	1			3	*Embelia ribes*						
Sapium discolor							*Ilex asprella*						
Adinandra millettii							*Symplocos paniculata*						
Daphinphyllum calycinum							*Myrsine seguinii*		7				7
Pinus massoniana				3	2	5	*Mussaenda pubescens*						
Artoacrpus hypargyrea							*Sarcandra glabra*						
Mallotus paniculatus							*Croton lachnocarpus*		1	2			3
Glochidion lanceolarium							*Glochidion eriocarpum*						
Bridelia tomentosa							*Rhamnus crenata*						
Total	72	114	148	43	27	404	*Ficus hirta*						
							Gardenia jasminoides						
							Dentrotrophe frutescens						
							Gnetum luofuense						
							Strophanthus divaricatus						
							Tot	[illegible]	422	396	52	3	1073

Appendix 5.1 Cont.

c. Tai Ho Wan, North Lantau.

Tree species	Height in cm					Total	Shrub/ Climber species	Height in cm					Total
	<50	50-100	101-150	151-200	>200			<50	50-100	101-150	151-200	>200	
Gordonia axillaris							*Rhodomyrtus tomemtosa*	47	245	23	1		316
Itea chinensis	5	43	19	1		68	*Baeckea frutescens*	18	75	38	16	4	151
Rhus chinensis	57	77				134	*Litsea rotundifolia*	2	13	2			17
Archidendron lucidum	2	49	1			52	*Embelia laeta*	10	53	7	2		72
Phyllanthus emblica		15	26	11	2	54	*Rhaphiolepis indica*	9	62	8	1		80
Machilus chekiangensis							*Melastoma sanguineum*		10	2			12
Cratoxylum cochinchinense	2	22	12			36	*Eurya japonica*			2			2
Acronychia pedunculata							*Phyllanthus cochinchinense*	8	67	37			112
Diospyros morrisiana	2	5	9	7	2	25	*Breynia fruticosa*	3	7	6			16
Schefflera octophylla							*Clerodendrum fortunatum*	2	10	5			17
Zanthoxylum avicennae		2				2	*Ficus variolosa*		2				2
Litsea cubeba							*Ardisia crenata*	7	16				23
Litsea glutinosa							*Melicope pteleifolia*						
Glochidion wrightii							*Dalbergia benthami*		9				9
Pentaphylax euryoides							*Eurya chinensis*		6	5			11
Schima superba							*Embelia ribes*			1			1
Sapium discolor							*Ilex asprella*	2	5	10			17
Adinandra millettii							*Symplocos paniculata*	4	10	3			17
Daphinphyllum calycinum							*Myrsine seguinii*						
Pinus massoniana							*Mussaenda pubescens*						
Artoacrpus hypargyrea							*Sarcandra glabra*						
Mallotus paniculatus							*Croton lachnocarpus*						
Glochidion lanceolarium							*Glochidion eriocarpum*		3				3
Bridelia tomentosa							*Rhamnus crenata*						
Total	68	213	67	19	4	371	*Ficus hirta*						
							Gardenia jasminoides						
							Dentrotrophe frutescens						
							Gnetum luofuense						
							Strophanthus divaricatus						
							Total	112	593	149	20	4	878

Appendix 5.1 Cont.

d. Pak Ngau Shek, Central New Territories.

Tree species	Height in cm					Total
	<50	50-100	101-150	151-200	>200	
Gordonia axillaris						
Itea chinensis	1	6	11			18
Rhus chinensis						
Archidendron lucidum		1	2			3
Phyllanthus emblica						
Machilus chekiangensis		4				4
Cratoxylum cochinchinense						
Acronychia pedunculata				3	1	4
Diospyros morrisiana						
Schefflera octophylla		1			2	3
Zanthoxylum avicennae			3	1	3	7
Litsea cubeba				4		4
Litsea glutinosa						
Glochidion wrightii			2	1		3
Pentaphylax euryoides						
Schima superba						
Sapium discolor			1			1
Adinandra millettii	1	1	2	2		6
Daphinphyllum calycinum				1		1
Pinus massoniana						
Artoacrpus hypargyrea						
Mallotus paniculatus						
Glochidion lanceolarium						
Bridelia tomentosa						
Total	2	13	21	12	6	54

Shrub/ Climber species	Height in cm					Total
	<50	50-100	101-150	151-200	>200	
Rhodomyrtus tomemtosa	83	363	41	2		489
Baeckea frutescens	96	217	7			320
Litsea rotundifolia	3	11	14	4		32
Embelia laeta	2	16	4	1		23
Rhaphiolepis indica		2	1			3
Melastoma sanguineum	7	27	7	6		47
Eurya japonica		2		2		4
Phyllanthus cochinchinense						
Breynia fruticosa	7	23				30
Clerodendrum fortunatum						
Ficus variolosa						
Ardisia crenata	1	7	1			9
Melicope pteleifolia		1				1
Dalbergia benthami						
Eurya chinensis						
Embelia ribes			1			1
Ilex asprella						
Symplocos paniculata						
Myrsine seguinii						
Mussaenda pubescens						
Sarcandra glabra						
Croton lachnocarpus						
Glochidion eriocarpum						
Rhamnus crenata		1				1
Ficus hirta						
Gardenia jasminoides						
Dentrotrophe frutescens						
Gnetum luofuense			1			1
Strophanthus divaricatus						
Tot	[illegible]	670	77	15		961

Appendix 5.1 Cont.

e. Ho Sheung Heung, North New Territories.

Tree species	Height in cm					Total	Shrub/ Climber species	Height in cm					Total
	<50	50-100	101-150	151-200	>200			<50	50-100	101-150	151-200	>200	
Gordonia axillaris							*Rhodomyrtus tomemtosa*	81	181	7	1		270
Itea chinensis							*Baeckea frutescens*	91	207	9			307
Rhus chinensis	3	12	2	1		18	*Litsea rotundifolia*	1	1	1			3
Archidendron lucidum							*Embelia laeta*	8	19	7	5	1	40
Phyllanthus emblica							*Rhaphiolepis indica*	2	7	3	1		13
Machilus chekiangensis							*Melastoma sanguineum*	2	1				3
Cratoxylum cochinchinense		2				2	*Eurya japonica*						
Acronychia pedunculata			2	1	2	5	*Phyllanthus cochinchinense*	1	1				2
Diospyros morrisiana							*Breynia fruticosa*	27	17	1			45
Schefflera octophylla							*Clerodendrum fortunatum*						
Zanthoxylum avicennae							*Ficus variolosa*						
Litsea cubeba							*Ardisia crenata*						
Litsea glutinosa							*Melicope pteleifolia*						
Glochidion wrightii							*Dalbergia benthami*					1	1
Pentaphylax euryoides							*Eurya chinensis*						
Schima superba							*Embelia ribes*						
Sapium discolor							*Ilex asprella*						
Adinandra millettii							*Symplocos paniculata*						
Daphinphyllum calycinum							*Myrsine seguinii*						
Pinus massoniana							*Mussaenda pubescens*	2	4	6	1		13
Artoacrpus hypargyrea							*Sarcandra glabra*						
Mallotus paniculatus							*Croton lachnocarpus*	4					4
Glochidion lanceolarium			2			2	*Glochidion eriocarpum*						
Bridelia tomentosa							*Rhamnus crenata*						
Total	3	14	6	2	2	27	*Ficus hirta*			1		1	2
							Gardenia jasminoides		2				2
							Dentrotrophe frutescens		1				1
							Gnetum luofuense						
							Strophanthus divaricatus				1		1
							Tot	[illegible]	441	35	9	3	707

Appendix 5.1 Cont.

f. Kwun Yam Shan, Central New Territories.

Tree species	Height in cm					Total	Shrub/ Climber species	Height in cm					Total
	<50	50-100	101-150	151-200	>200			<50	50-100	101-150	151-200	>200	
Gordonia axillaris							*Rhodomyrtus tomemtosa*	60	124	171	59		414
Itea chinensis	15	7	4			26	*Baeckea frutescens*	8	6	3			17
Rhus chinensis							*Litsea rotundifolia*	127	25	16	9	2	179
Archidendron lucidum	2	0	1			3	*Embelia laeta*	2	5	1	1		9
Phyllanthus emblica							*Rhaphiolepis indica*	11	1	1			13
Machilus chekiangensis	19	13	3	1		36	*Melastoma sanguineum*	16	27	9	1		53
Cratoxylum cochinchinense	1	2	2			5	*Eurya japonica*	13	12	13	9	2	49
Acronychia pedunculata							*Phyllanthus cochinchinense*						
Diospyros morrisiana				1		1	*Breynia fruticosa*	5	5				10
Schefflera octophylla	7	4				11	*Clerodendrum fortunatum*						
Zanthoxylum avicennae	1		3			4	*Ficus variolosa*		1				1
Litsea cubeba	3	4	1			8	*Ardisia crenata*		1				1
Litsea glutinosa	12					12	*Melicope pteleifolia*	29	1				30
Glochidion wrightii		1	1	2	1	5	*Dalbergia benthami*	6	10	2			18
Pentaphylax euryoides							*Eurya chinensis*						
Schima superba							*Embelia ribes*	4	5	6	3		18
Sapium discolor	4	4				8	*Ilex asprella*		1				1
Adinandra millettii							*Symplocos paniculata*	1					1
Daphinphyllum calycinum							*Myrsine seguinii*						
Pinus massoniana							*Mussaenda pubescens*						
Artoacrpus hypargyrea							*Sarcandra glabra*	6	2				8
Mallotus paniculatus			2		1	3	*Croton lachnocarpus*						
Glochidion lanceolarium							*Glochidion eriocarpum*						
Bridelia tomentosa				1		1	*Rhamnus crenata*		2				2
Total	64	35	17	5	2	123	*Ficus hirta*						
							Gardenia jasminoides						
							Dentrotrophe frutescens						
							Gnetum luofuense						
							Strophanthus divaricatus						
							Tota	[illegible]	228	222	82	4	824

Appendix 5.2

Numbers of individuals of different species that were recorded under the canopy of other individuals at each site

Shading species	Shaded plant	Form	High West	Ngau Ngak Shan	Tai Ho Wan	Pak Ngau Shek	Ho Sheung Heung	Kwun Yam Shan	Total
Tree									
Acronychia pedunculata	*Acronychia pedunculata*	Tree					3		3
	Gordonia axillaris		3	3					6
	Itea chinensis					1			1
	Machilus chekiangensis					3			3
	Rhus chinensis						1		1
	Baeckea frutescens	Shrub		1					1
	Breynia fruticosa					2			2
	Gardenia jasminoides						2		2
	Litsea rotundifolia		2						2
	Rhaphiolepis indica			2					2
	Rhodomyrtus tomentosa		1				3		4
Cratoxylum cochichinense	*Archidendron licidum*	Tree			1				1
	Itea chinensis				1				1
	Rhus chinensis				5				5
	Rhaphiolepis indica	Shrub			2				2
Diospyros morrisiana	*Archidendron licidum*	Tree			1				1
	Diospyros morrisiana				5				5
	Itea chinensis				4				4
	Rhus chinensis				3				3
	Baeckea frutescens	Shrub			3				3
	Eurya chinensis				2				2
	Melastoma sanguineum				1				1
	Phyllanthus cochinense				1				1
	Rhaphiolepis indica				1				1

Appendix 5.2 Cont.

Shading species	Shaded plant	Form	High West	Ngau Ngak Shan	Tai Ho Wan	Pak Ngau Shek	Ho Sheung Heung	Kwun Yam Shan	Total
	Embelia laeta	Climber			1				1
Glochidion wrightii	*Glochidion wrightii*	Tree						1	1
	Schefflera octophylla							2	2
	Litsea rotundifolia	Shrub						1	1
Gordonia axillaris	*Gordonia axillaris*	Tree	1	4					5
	Itea chinensis			2					2
	Rhaphiolepis indica	Shrub		3					3
	Rhodomyrtus tomentosa		1	6					7
	Embelia laeta	Climber		1					1
Itea chinensis	*Archidendron licidum*	Tree			2				2
	Diospyros morrisiana				1				1
	Gordonia axillaris			2					2
	Itea chinensis			1	1				2
	Rhus chinensis				2				2
	Baeckea frutescens	Shrub			4				4
	Embelia laeta				3				3
	Phyllanthus cochinense				1				1
	Rhaphiolepis indica			1					1
	Rhodomyrtus tomentosa			2	3				5
	Symplocos paniculata				1				1
Litsea cubeba	*Baeckea frutescens*	Shrub				6			6
	Rhodomyrtus tomentosa					3			3
Mallotus paniculatus	*Litsea cubeba*	Tree						1	1
	Baeckea frutescens	Shrub						1	1
	Melicope pteleifolia							3	3
	Embelia ribes	Climber						1	1
Phyllanthus emblica	*Archidendron licidum*	Tree			1				1

Appendix 5.2 Cont.

Shading species	Shaded plant	Form	High West	Ngau Ngak Shan	Tai Ho Wan	Pak Ngau Shek	Ho Sheung Heung	Kwun Yam Shan	Total
	Rhus chinensis				3				3
	Baeckea frutescens	Shrub			2				2
	Clerodendrum fortunatum				1				1
	Phyllanthus cochinense				1				1
	Rhaphiolepis indica				1				1
	Rhodomyrtus tomentosa				5				5
Pinus massoniana	*Gordonia axillaris*	Tree		3					3
	Rhodomyrtus tomentosa	Shrub		2					2
Schefflera octophylla	*Schefflera octophylla*	Tree				1			1
Shrub									
Baeckea frutescens	*Cratoxylum cochichinense*	Tree			1				1
	Gordonia axillaris			13					13
	Itea chinensis			8	2				10
	Rhus chinensis				3				3
	Baeckea frutescens	Shrub			2				2
	Litsea rotundifolia			1					1
	Phyllanthus cochinense				2				2
	Rhaphiolepis indica			2					2
	Rhodomyrtus tomentosa			29	3				32
	Embelia laeta	Climber		3	2			1	6
Eurya nitida	*Cratoxylum cochichinense*	Tree			1				1
	Itea chinensis							2	2
	Machilus chekiangensis							4	4
	Schefflera octophylla							1	1
	Litsea cubeba							2	2
	Litsea glutinosa							3	3

Appendix 5.2 Cont.

Shading species	Shaded plant	Form	High West	Ngau Ngak Shan	Tai Ho Wan	Pak Ngau Shek	Ho Sheung Heung	Kwun Yam Shan	Total
	Mallotus paniculatus							1	1
	Baeckea frutescens	Shrub						3	3
	Eurya nitida							4	4
	Litsea rotundifolia							11	11
	Melastoma sanguineum							1	1
	Melicope pteleifolia							2	2
	Rhodomyrtus tomentosa				1			7	8
	Sarcandra glabra							1	1
	Embelia laeta	Climber						1	1
Ficus variolosa	*Rhodomyrtus tomentosa*	Shrub		1					1
Litsea rotundifolia	*Ardisia crenata*	Shrub				5		1	6
	Litsea rotundifolia							3	3
	Melicope pteleifolia							7	7
	Rhodomyrtus tomentosa				1				1
	Sarcandra glabra							2	2
	Embelia laeta	Climber		1					1
Melastoma sanguineum	*Machilus chekiangensis*	Tree				1			1
	Melastoma sanguineum	Shrub						4	4
Phyllanthus cochinense	*Archidendron licidum*	Tree			2				2
	Cratoxylum cochichinense				2				2
	Rhus chinensis				3				3
	Baeckea frutescens	Shrub			1				1
	Rhodomyrtus tomentosa				5				5
Rhaphiolepis indica	*Baeckea frutescens*	Shrub			1				1
	Rhodomyrtus tomentosa				1				1
Rhodomyrtus tomentosa	*Archidendron licidum*	Tree			1			3	4
	Gordonia axillaris			4					4

Appendix 5.2 Cont.

Shading species	Shaded plant	Form	High West	Ngau Ngak Shan	Tai Ho Wan	Pak Ngau Shek	Ho Sheung Heung	Kwun Yam Shan	Total
	Itea chinensis			1				13	14
	Litsea cubeba							4	4
	Litsea glutinosa							8	8
	Machilus chekiangensis							10	10
	Rhus chinensis				3				3
	Sapium discolor							4	4
	Schefflera octophylla							3	3
	Baeckea frutescens	Shrub		1				4	5
	Breynia fruticosa							3	3
	Eurya nitida							12	12
	Ficus variolosa			1					1
	Ilex asprella							1	1
	Litsea rotundifolia							83	83
	Melastoma sanguineum			1		3		8	12
	Melicope pteleifolia							11	11
	Rhamnus crenata							1	1
	Rhaphiolepis indica			3	2			1	6
	Rhodomyrtus tomentosa			1	5		1	28	35
	Sarcandra glabra							5	5
	Symplocos paniculata							1	1
	Dalbergia benthamii	Climber						12	12
	Embelia laeta			1				3	4
	Embelia ribes							7	7
Symplocos paniculata	*Symplocos paniculata*	Shrub			1				1
	Baeckea frutescens				1				1
		Total	8	104	109	25	10	296	552
		% of the total number of woody plants recorded at that site	0.4	7.0	8.7	2.5	1.4	31.3	7.6

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