Ecological Assessment of Restored Subtropical Forests in Hong Kong

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Abstract of thesis entitled:

"Ecological Assessment of Restored Subtropical Forests in Hong Kong"

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The present study investigates the successional development of three restored subtropical forests in Hong Kong, with special emphasis on the development of an assessment system suitable for the evaluation of restoration progress. The restored forests, aged 8 (R00), 13 (R95) and 19 (R89) years, were established on severely degraded lands in Tai Tong East Borrow Area (TTEBA) depleted of vegetation and top soil cover. The vegetation, soil and nutrient cycling processes of these restored forests were benchmarked against a mature forest (TO), aged 300+ years, characterized by high biodiversity and undisturbed soil. The restored forests are dominated by exotic species, mostly nitrogen-fixing legumes, and the mature forest by broad-leaved native species. The objectives of this thesis are threefold: (a) to assess the ecological development of the restored forests; (b) to develop an assessment system that can be employed to assess restoration progress; and (c) to revisit the goals and strategy of ecological restoration in Hong Kong.

Ecological succession is happening in all the restored sites and the evidence of proof include: stem exclusion (declining density); steady increase of tall trees (>12m); invasion of grass, fern, vine and climber though few; increase in species diversity;

canopy closure; decreasing standing litter but a simultaneous increase in fruits and seeds; increase of DBH, transaction areas and crown area, etc. Overall, the restored forests are inferior to the mature forest in terms of biodiversity and stand complexity.

Ecological rehabilitation with exotic species, particularly nitrogen-fixing legumes, is capable of ameliorating the degraded soils. While there is no change in the soil texture, ecological rehabilitation lowered pH and bulk density, increased organic carbon (OC) and total Kjeldahl nitrogen (TKN) contents, and elevated field capacity and available water contents of the soil. Intra-layer differences in OC and TKN were widened in R89, a trend comparable to TO. Ammonification and nitrification were detected in the restored soils, where NH_4-N predominated over NO_3-N and net N mineralization was higher in R89 than R95 and R00. In contrast, TO is dominated by NO_3-N and net N mineralization is higher than the restored soils.

Litterfall production was characterized by bimodal peaks in TTEBA and by a single peak in TO. Peak production in August through September in Tai Tong was caused by mechanical breakage during the passage of tropical cyclones. In contrast, litterfall in TO was less easily affected by cyclones. Total litterfall production was in the order of TO $\geq$ R00 > R89 $\geq$ R95. Litterfall production was only positively correlated with canopy closure of the forests. Nutrient return was higher in the dry season than the wet season, which is governed by the quantity of litterfall.

The decomposition constants (k) for *Acacia mangium* and *Schima superba* foliage litter decreased in the order of TO > R89 > R00 > R95. It is positively correlated with SOC, TKN, TP, clay percentage and FC water of the soils but negatively with soil bulk density ($p<0.05$). Among the restored sites, the half life (T_{50}) of leaf litter was shortest for *Acacia mangium* (92.4 days) and *Schima superba* (105.0 days) in R89. Overall, litter decomposition is faster in TO than any of the restored sites.

After 8-19 years of restoration, therefore, there were improvements in the quality of the restored sites in terms of vegetation structure, soil productivity and nutrient cycling. While the degree of improvement increases with age of the plantations, the qualities of the restored sites still lag behind that of the mature forest.

Based on the above findings, an assessment system is developed for the evaluation of restoration progress in the subtropical region. It includes a set of ecological indicators including: species diversity, timber transaction areas, standard deviation of tree height, total crown area, sapling density, species invasion, soil bulk density, soil water-holding capacity, organic carbon content, TKN, pH, seed bank, soil mineralization rate and litter decomposition rate. These findings have, therefore, filled the knowledge gap that there is virtually no study on the ecological assessment of restored subtropical forests in the literature. In addition, the goals and strategy of ecological restoration in Hong Kong are also discussed in the thesis.

Keywords: ecological assessment, forest rehabilitation, soil destruction sites, subtropical area

論文摘要

香港亞熱帶森林復修的生態評估

本研究調查了香港三個亞熱帶復修林的演替發展，著重于發展生態評估系統以適合評價生態恢復過程。三個研究樣區建立在植被和土壤嚴重退化的采泥區，分別復修了8年（R00），13年（R95）和19年（R89）。對比本地高生物多樣性無人爲擾動的成熟森林（超過300年的大庵風水林TO），研究了修復林區的植被，土壤和養分循環過程。研究目的包括三方面：(a)評價復修區的生態發展過程；(b)建立可以用於評價森林生態恢復過程的評價系統；(c)回顧香港生態恢復的目的和策略。

結果顯示三個修復樹林都出現了不同程度的植被生長生態演替，證據包括：莖桿排斥（下降的林間密度）；大于12米樹木的穩定增加；草本，蕨類植物，蔓生植物和攀緣植物的入侵；物種多樣性的增加；林冠覆蓋率；落葉層生物量減少同時種子和果實所占比例增加；胸徑，橫截面積和樹冠面積增加，等等。総的來説，復修林區在生物多樣性和結構複雜性上遠劣于成熟森林。

利用外來樹種，尤其是固氮植物的生態修復能夠改善退化的土壤。盡管土壤質地沒有變化，生態復修降低了土壤酸堿度，土壤容重，增加了土壤有機碳和総氮含量，提高了田間持水量和可利用水含量。修復地均檢測出氨化和硝化過程，然而氨態氮在修復林區占主導地位。R89樣區氮淨礦化率高于R95和R00。作爲比較，TO土壤中硝態氮佔主導並且淨礦化率高於修復樣區。

大棠采泥區的落葉量呈雙峰特征，而風水林是單峰。八、九月份大棠地區的高落葉量是熱帶風暴的機械破壞引起。作為對比，大庵風水林的落葉更不易受臺風的影響。研究區的年總落葉量順序是：TO ≥ R00 > R89 ≥ R95。落葉量只和樹林的林冠覆蓋度正相關。干季營養元素返還高于濕季，主要由落葉量決定。

馬佔相思和木荷兩种落葉降解常數(k)在四個研究區有相同的大小排序：TO > R89 > R00 > R95。這與土壤有機碳，全氮，全磷，粘土含量以及田間持水量成正相關，和土壤容重成負相關。復修林區中，R89有最短半分解期(T_{50})，馬佔相思92.4天，木荷105.0天。而R95樹葉的半分解期比R00樣區長，分別為馬佔相思210.0天，256.7天。大庵風水林的樹葉分解速率高過任何一個復修樣區。

總而言之，經過8-19年的修復過程，土壤破壞嚴重的修復陽區的質量在一定程度上得到了改善，包括土壤特性，植被覆蓋和養分循環。隨著修復期的增

長，樹林恢復程度也有所改善。即使如此，復修區的質量還是遠遠落後本地頂級森林生態群落。

基於以上結果，建立了亞熱帶地區的生態恢復評估系統。包括一系列的生態因子的評價因子包括，物種多樣性，樹木橫截面積，樹高的標準偏差，總林冠面積，樹苗的密度，物種入侵，土壤容重，土壤持水量，有機碳含量，総氮，酸堿度，土壤氮礦化以及落葉分解速率。本研究填補了亞洲亞熱帶地區森林生態恢復系統評估這一領域研究的空白。另外，本文最後也回顧並討論了香港生態復修的目標和策略。

關鍵詞：生態修復，土壤破壞地，亞熱帶森林，生態評估

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CHAPTER 1

INTRODUCTION

1.1 Research background

The rapid industrialization and urbanization, together with economic development in both developed and developing countries, have resulted in great improvement of living standards. One of the costs of rapid growth is environmental and ecological degradation, resulting in the depletion of natural resources. As an important natural resource, forest lands suffer from diminishment and deterioration. The world is losing over 9 million hectares of forest each year and at least twice that amount is being fragmented and degraded. With nearly half the world's original forest cover gone and a third of remaining forests under threat from climate change, decision-makers must consider not only how to halt forest loss and degradation but also how to reverse the trend (Gautam, 2003). Although it is widely recognized that degradation is occurring and that biodiversity is being lost worldwide, it is very difficult to obtain the quantitative data that can effectively reflect the scale of the problem. One possible reason is that different countries report the problem in different ways. Another possible reason is that the rate of change is extremely rapid, so that available statistics usually lag behind the field situation (Lamb, 1994). Large areas of the world's forest have been degraded; some degraded ecosystems are able to recover naturally but many do not. In some cases, there are simply too few original plant and animal biota remaining at the site or the site has been dominated by pests and weeds. In other cases, either some component of the biophysical environment such as soil fertility has changed or repeated disturbances prevent successional development of the community. In sites where natural recovery is able to take place, the process may be slow. This can increase the chance of further disturbances, thus perpetuating the degradation process. Because of these limitations, human intervention is needed to either initiate the recovery process or to accelerate the rate at which it proceeds.

1.1.1 Classification of landscape degradation

Almost all the tropical and subtropical forests are naturally subjected to disturbance and destruction from a small scale, which includes single tree dying and falling down, to a large scale, which includes landslides and volcanic eruptions (John, 1986). Human activities add a new range of disturbances to these naturally occurring events. On the basis of the severity of the disturbance levels, land degradation can be classified into three levels: vegetation disturbance, soil disturbance and soil destruction (Figure 1.1) (Aber, 1987).

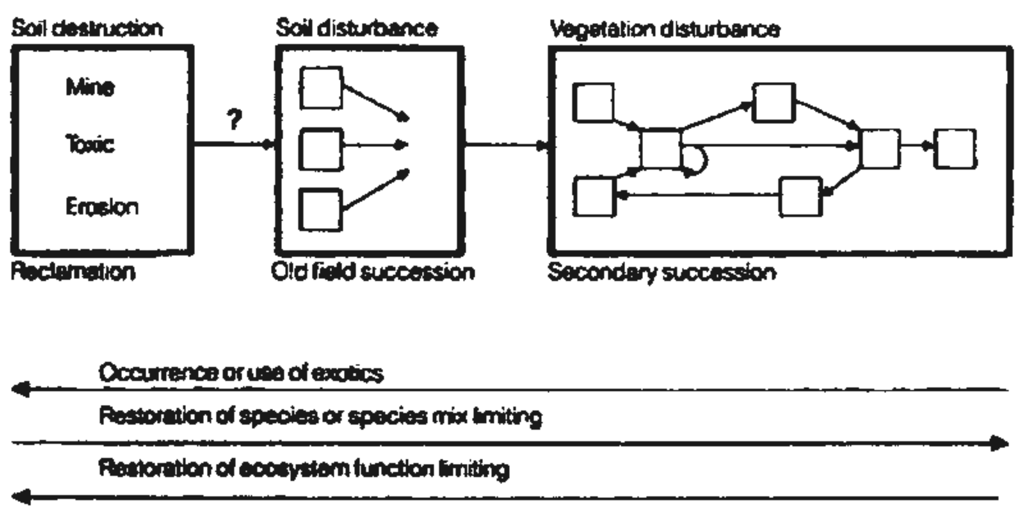


Figure 1.1 Types of land degradation according to the severity of the disturbance (Aber, 1987)

Vegetation disturbance, the mildest level of disturbance, involves disruption or removal of the native plant community. Recently cutover area or fire-affected areas are typical examples of vegetation disturbance, in which the soil profile is largely unaltered. The major problems associated with vegetation disturbance include disturbance of nutrient cycling and shrinkage of seed bank (Table 1.1). It is possible to introduce native species, which existed before disturbance, directly to the site. The introduction of exotic species, however, can pose serious problems because they can change the structure and function of the original ecosystem. There exists the possibility that the exotics can outcompete the natives, too.

Table 1.1 Comparison of different types of degraded lands

	Vegetation disturbance	Soil disturbance	Soil destruction
Soil	Intact profile	Intact profile	Top soil removed
Likely problems	Disturbance of nutrient cycling, seed bank shrinkage	Nutrient imbalance, nutrient toxicity	Decline of soil physical, biological and chemical properties
Severity	Mildest	Medium	Severest
Examples	Fire-affected slopes	Abandoned agriculture	Mine, quarry, borrow area, badland

Soil disturbance involves both soil and vegetation perturbation, and the best example is the abandoned farmland. The likely problems are nutrient imbalance, nutrient toxicity and water availability. In this case, the feasibility of introducing exotic species to the site will depend on the specific conditions of soil nutrients. To speed up the restoration of soil disturbance site, a mixture of native and exotic species is considered appropriate. Available nitrogen is usually lacking in a soil disturbance site as a result of prolonged cultivation; it is critical to successful restoration (Aber, 1987). Species with a capacity to fix atmospheric nitrogen, exotics or natives, are a definite advantage in the restoration programme.

Soil destruction involves complete removal of the vegetation and the topsoil, which is undoubtedly the most valuable cover of the earth. It represents the severest type of land degradation, resulting in rapid decline of soil physical, biological and chemical properties. The soil is lost both in pedological and biological senses. Mine sites, quarries, borrow areas and badlands are typical examples of soil destruction. The restoration of a soil destruction site requires an in-depth understanding of the environmental conditions, and species tolerance and requirements. Besides nutrient scarcity, trace amount of toxic substances may be harmful to ecological rehabilitation. For instance, heavy metals of iron, copper and manganese are commonly found in mine tailings. In the preliminary stage of restoration, the selection of pioneer species tolerant to heavy metals and with a capacity to fix atmospheric nitrogen (e.g. legumes) should be accorded higher priority. Because of these specific requirements, the probability of introducing exotic species to the soil destruction site is higher than in the vegetation and soil disturbance sites. Notwithstanding this, there are also native legumes that possess these properties, and the only drawback limiting their wider use is limited stock supply.

1.1.2 Trajectories of degraded ecosystem development

A common response to forest degradation is to simply abandon the land. The land may then continue to degrade if there are further disturbances; otherwise, it may begin to slowly recover its productivity and biodiversity if some of the original biota remains in or near the site. The recovery time will depend on the extent of degradation, but it is likely to exceed 100 years (Riswan *et al.*, 1985). In order to accelerate the recovery of forest ecosystem including biodiversity and productivity, there are three broad approaches to deal with the degraded lands.

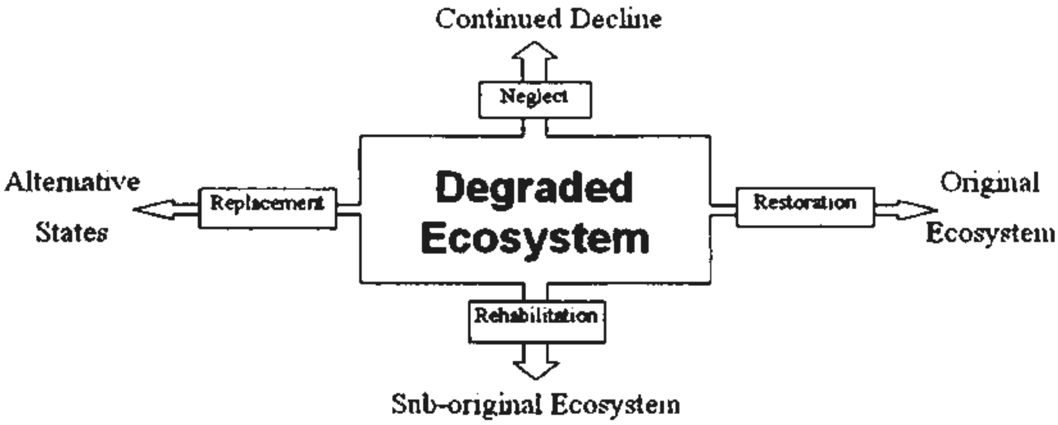


Figure 1.2 Approaches for the restoration of degraded land

The first and most ambitious approach is restoration (Figure 1.2). Restoration means attempting to recreate the original forest ecosystem by reassembling the native complement of plants and animals that once occupied the site. Ecological restoration is the progress of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed (SER, 2004). The ultimate goal of restoration is to create a self-supporting ecosystem that is resilient to perturbation without further assistance (Urbanska *et al.*, 1977; SER, 2004). Biological diversity is one of the most concerned aspects in the process of recovery. Therefore, native species are extremely desirable in species selection of revegetation.

The second approach is rehabilitation which means using some of the original species plus, where necessary, exotic species to reforest the site. In this case, there is no attempt to recreate the original ecosystem. The objective is to return the forest to a stable and productive condition. Rehabilitation returns major ecosystem functions and recreates social and economic benefits. Some but not all original biodiversity can be achieved. So, rehabilitation may also be the part of an overall restoration strategy.

The final choice is replacement, also known as reclamation (Lamb, 2003). This means utilizing one or more exotic species to achieve productivity and stability. There is no attempt to restore any of the original biodiversity at the site; instead, an alternative to the original ecosystem is produced. This could be simpler in structure but more productive (for instance, woodland replaced by agricultural grassland), or could be simpler in structure but less productive (for instance, woodland replaced by an amenity grassland).

Ecosystem development can be quantified in two dimensions, structure and function. Ecosystem structure refers to the physiognomy or architecture of the ecosystem with respect to the density, horizontal stratification, and frequency distribution of species-populations, and the sizes and life forms of the organisms that comprise the ecosystem. Ecosystem functions, also known as ecological processes, are the dynamic attributes of ecosystems, including interactions among organisms and interactions between organisms and their environment. Ecological processes are the basis for self-maintenance in an ecosystem. In natural succession, the two dimensions are increasing with time. Meanwhile, when the system is degraded by mining or other destructive forces, the two dimensions will decrease even to zero. The trajectories of the different approaches (restoration, rehabilitation, replacement and do-nothing) in relationship to ecosystem structure and function are illustrated in Figure 1.3.

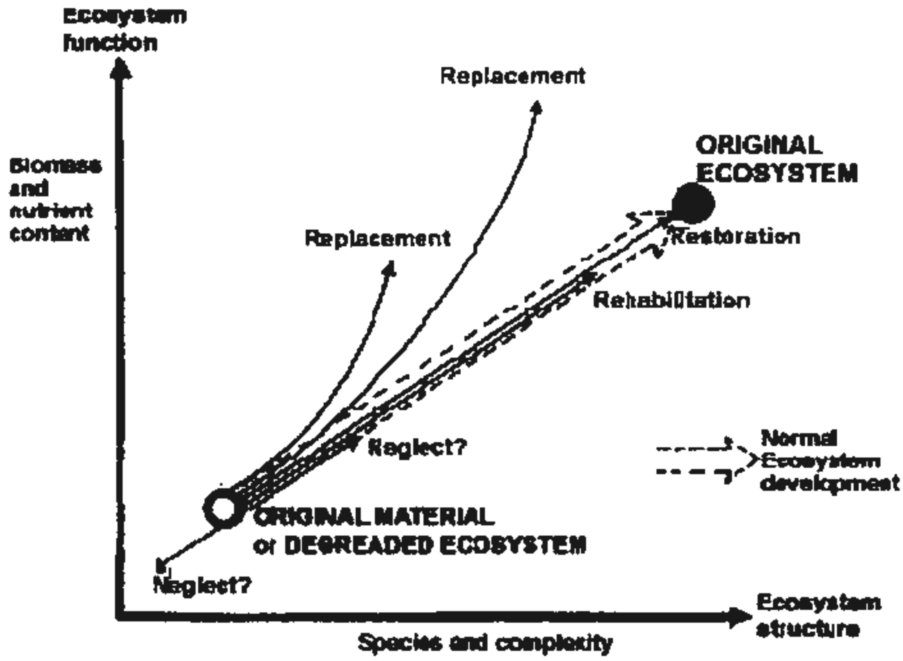


Figure 1.3 Trajectories of restoration (Bradshaw, 1987)

While ecosystem structure and function will increase with time under natural succession, the choice of restoration approach would depend *inter alia* on objectives, cost, and the availability of nursery stock. The bottom line is to strike an equilibrium among the environmental value, economic value, and social requirement of ecological restoration, while the ultimate goal is to create a self-sustainable ecosystem.

1.1.3 Reference ecosystem site

A reference usually serves as a model for planning a restoration project, and later for its evaluation (SER, 2004). The present study focuses on evaluating the success of ecological restoration. The designated ecosystem, landscape or unit can be called the reference site. It represents a point of advanced development that lies somewhere along the intended trajectory of the restoration. In other words, the restored ecosystem is eventually expected to emulate the attributes of the reference, and project goals and strategies are developed in light of that expectation. The reference, in its simplest form, is an actual site, or a written description, or both. A reference usually represents a single state or illustration of ecosystem attributes. Information collected on the reference includes both biotic and abiotic components.

To ensure the quality and usefulness of selecting a reference site, three factors are considered. First, the reference site must be well-developed with goals, strategies or services beneficial to human beings. Second, if the goal of restoration is a natural ecosystem, human-mediated impacts should not be considered. Third, the reference site must be in the same life zone as the restored ecosystems. For these reasons, the selection of the reference requires experience and sophisticated ecological judgment.

1.2 Conceptual framework

Plenty of the restoration projects on forest recovery are undertaken by different stakeholders with different goals. The strategies for forest recovery are site-specific and vary with environmental, social and economical profits. The ultimate goal of forest restoration is to create a self-supporting ecosystem, just like the climax forest ecosystems. And, the first step in any restoration strategy is to protect the disturbed habitats and communities from further degradation and from losing the extant genes. But how do we know when we have reached the goal? And how can we measure the processes of restoration success? Some authors have suggested that restoration success could be based on vegetation characteristics (Walters, 2000; Wilkins, 2003), species diversity (van Aarde *et al.*, 1996; Reay and Norton, 1999; Passell, 2000; McCoy and Mushinsky, 2002), or ecosystem progresses (Rhoades *et* *al*., 1998).

Restoration of degraded lands is a challenging task, and requires a thorough understanding of the ecological principles. Bradshaw (1987) called restoration an "acid test of ecology", and Ewel (1987) the "ultimate test for ecological theory". As we all know, restoration ecology is multidisciplinary, involving the interplay of soil science, silviculture, botany, zoology, etc. Moreover, different restoration strategies are adopted according to the severity of land degradation and the socio-economic benefits. In this research, I shall focus on ecological restoration in the sub-tropical region of south China.

Hong Kong is located in the subtropical region of south China. With a well-developed economy, land degradation is a serious problem in this densely populated city. The original forest cover was lost in large scale deforestation before the 1950s, and more recently in quarrying, mining and infrastructure development. The degraded lands were subsequently reforested to improve the visual impact, protect the soil against erosion and conserve water resource until the 1960s. Thereafter, the purpose of reforestation is changed to conservation planting to preserve local biodiversity. Despite changes in the objectives of planting in the past, stereotype procedures are followed in terms of species choice, soil treatment, fertilization programme, and post-planting care. For instance, there is no differentiation in the pre-planting treatment of the soils between a fire-affected slope and a quarry site although the former is relatively rich in soil organic matter and nitrogen contents. Worse still, no attempts are made to monitor the progress and success of restoration against the pre-determined goals. We believe systematic evaluation of ecological restoration is crucial in bettering our understanding of ecosystem re-creation, not to mention the potential socio-economic benefits. Indeed, most restoration projects in Hong Kong are undertaken by Agriculture, Fisheries and Conservation Department (AFCD) of the government. They are disinterested in research work, except a few small scale planting trials. Several research questions were raised as follows:

- Hong Kong is a densely populated city where 40% of the land is placed under the protected area system locally known as country parks. What are the goals of ecological rehabilitation?
- How to measure the success or failure of ecological rehabilitation?
- What indicators can be used for the assessment of restored ecosystems?

The countryside is a valuable asset to the people of Hong Kong by providing a wide range of ecosystem services to the society. Specifically, people seek recreational pursuits in country parks, where naturalness and biodiversity are key attractions. It is therefore a common practice for the government to repair any degraded parts of the landscape, especially after enactment of the County Parks Ordinance in 1976. Nowadays, about 40 % of the vegetated area in Hong Kong is dominated by exotic plantations and yet their success or failure has never been systematically evaluated.

It is, therefore, necessary to reverse the situation and build up an effective indicator system to monitor the processes of forest recovery, and set the criteria for assessment of forest restoration or rehabilitation. To fulfill this objective, a conceptual framework of the study is proposed below (Figure 1.4).

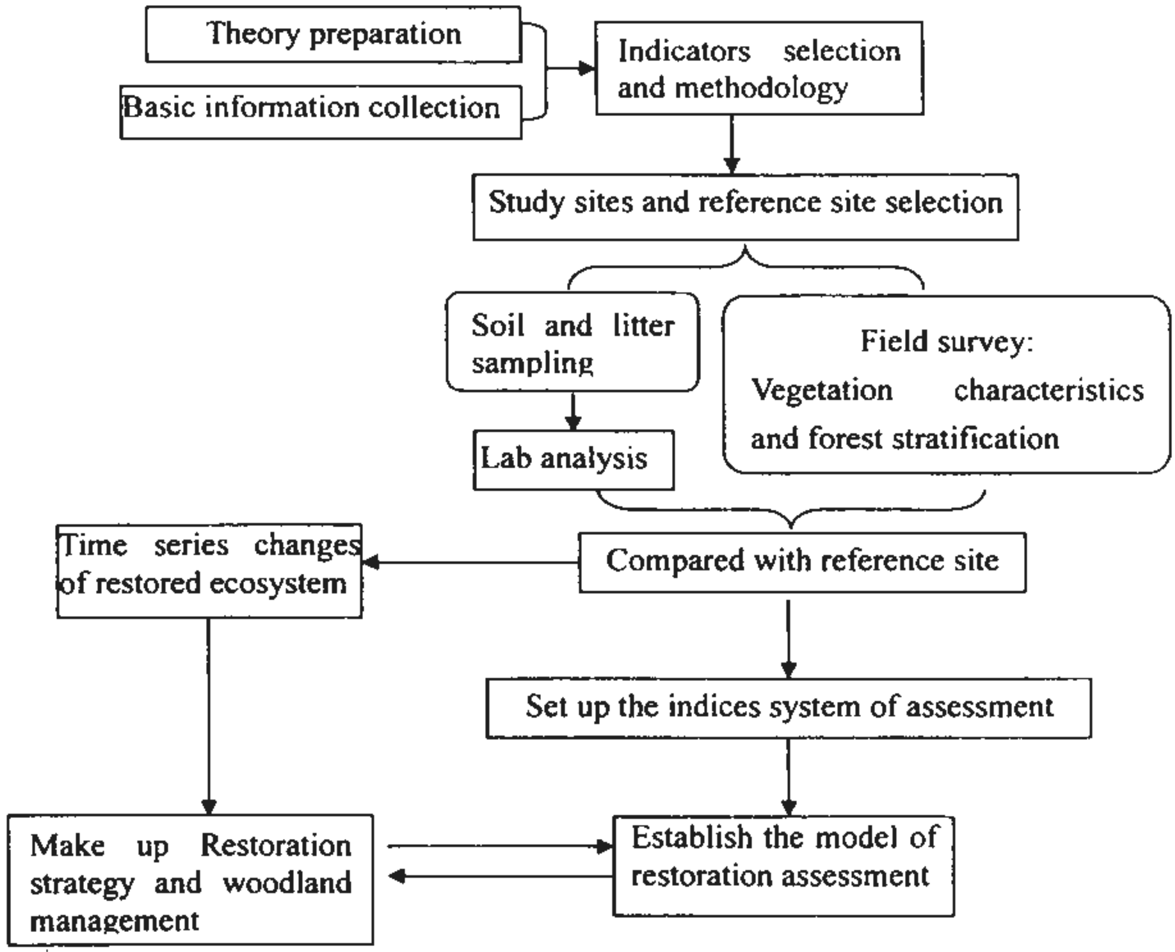


Figure 1.4 Conceptual framework of the study

The study begins with a desk-top review of the theories and principles of ecological restoration, as well as known assessment indicators in the literature. In analyzing the known indicators, special attention would be paid to the objectives of restoration and where it is implemented. The similarities and differences between the different indicators and places of application will be analyzed against the objectives of restoration. The analysis shall provide some rules or guidelines to the present study in the selection of appropriate indicators, study sites and reference site. In the selection of study sites, a worst scenario approach is adopted so that only soil destruction sites are selected for this study. This is because restoration of the soil destruction sites would naturally start from scratch, including recovery of soil productivity and vegetation structure. Furthermore, restored sites of known history are readily available in Hong Kong. These uneven-aged plantation sites would be similar to a vegetation gradient where climate, soil types and the restored species are similar.

The selection of assessment indicators would inevitably involve the soil and vegetation at the sub-system level, and structure and function at the ecosystem level. The results obtained from field survey and laboratory analysis will then be compared to those of the reference site, a feng shui woodland, which represents a climax or near-climax state of the local forest. The purpose of this comparison is to find out if there is evidence that trajectory development of the uneven-aged plantations is moving towards an appropriate path. In this regard, the evidence of occurrence of a structural development or a process is more important than their magnitude. The arguments can be further strengthened by analyzing the changes of indicators with time among the uneven-aged plantations. Through these analyses, it is possible to identify indicators that are suitable for inclusion in the indices system of assessment, which is, in turn, the cornerstone towards the building of a model for restoration assessment (Figure 1.4). As the findings of this thesis are related to ecological restoration, they can help us re-define the local restoration strategy and formulate a viable woodland management plan.

1.3 Objectives of the study

The main theme of this thesis is to set up an indices system and model for the assessment of ecological restoration in subtropical China. The study sites are actually borrow areas (from which land fill materials are excavated for infrastructure development) that have been restored for 8-19 years in Hong Kong. The specific objectives are as follows:

- To review the current indicators used for the assessment of ecological restoration;
- To investigate the soil, vegetation and ecological processes of three uneven-aged woodland plantations developed on soil destruction sites;
- To identify viable attributes for the assessment of ecological restoration in subtropical China; and
- To revisit the strategy of ecological rehabilitation, and woodland management measures on soil destruction sites in Hong Kong.

1.4 Significance of the study

With a growing concern of accelerated ecosystem destruction and the lack of sustainability of modern societies, there has been a tremendous surge in interest in restoration as a technique for reversing habitat degradation world-wide (Jordan *et al.* 1988). Hence, research on assessment of rehabilitation or restoration is getting popular both in theory and in practice.

In theory, restoration is "an acid-test of ecology" (Bradshaw, 1987). The borrow areas are typical soil destruction sites in Hong Kong, in which the vegetation and topsoil up to an average depth of 8 m are removed. It is an arduous task to rehabilitate this severely degraded landscape, and proactive inputs are reminiscent of ecosystem re-creation. Unfortunately, the local rehabilitation strategy and objectives are ill-defined so that greening of the derelict landscape to remove the eyesore is the government's prime concern. In the present study, the borrow areas have been restored for 8-19 years. An in-depth study into their structure and functions, in the form of identifiable indicators, shall reveal their trajectory of development along the vegetation chronosequence. As unveiled in Chapter 2, very few of the assessment studies on ecological restoration are conducted in Asia and still less in the subtropical region. The present study shall, therefore, fill in an important knowledge gap on the assessment of restoration in the subtropics. In addition to this, the findings can help us understand better the recovery of subtropical forests on severely degraded sites. This will, in turn, advance our knowledge on the ecology of subtropical forest system.

In practice, the study of restoration assessment is lacking in Hong Kong. The restored plantations are rarely monitored for their growth performance except occasional pruning and thinning after typhoons. There is a need to build a systematic and effective indices system to monitor the success of restoration. It will yield quantitative and objective information about the rehabilitation projects, including their trajectory in the nearest future. Bearing in mind that the ultimate goal of ecological rehabilitation is to recreate self-sustainable ecosystems, the model of assessment can be a useful tool for woodland management. It can identify restored sites that are not heading towards a self-sustainable ecosystem and that some intervention is needed to correct the trajectory of development. This is reminiscent of woodland transformation, involving clearfelling of the overstorey exotic species and enrichment planting of native species in the understorey stratum.

1.5 Organization of the thesis

The thesis consists of eight chapters. Chapter 1 is the introductory chapter, which describes the concepts of ecological restoration and assessment, as well as the objectives and conceptual framework of the study. The significance and organization of this study are also explained. Chapter 2 is the literature review, which summarizes the assessment of ecological restoration of former researches. Chapter 3 is the description of the study areas and reference site. Chapter 4 examines the vegetation dynamics in the uneven-aged restored sites reminiscent of a vegetation chronosequence. Chapter 5 investigates the recovery of soil productivity and soil nitrogen mineralization. Monthly leaf litterfall and the nutrient dynamics are investigated in Chapter 6. Leaf litter decomposition rate of local and adventive species are compared in Chapter 7. Finally, Chapter 8 summarizes the research findings and implications, redefines the goals of ecological restoration in Hong Kong and reviews the indicators system for ecological assessment, together with a summary of the limitations of the study as well as suggestions for future research.

CHAPTER 2

LITERATURE REVIEW

2.1 Introduction

Restoration of degraded lands is a challenging task, which requires a thorough understanding of the ecological principles (Singh, 1993). Bradshaw (1987) regards restoration as an "acid-rest for ecology", and Ewel (1987) the "ultimate test for ecological theory." Ecosystem restoration, on the one hand, is merely a technical business with an ecologically crude objective. On the other hand, it provides us a good opportunity to test our understanding of ecosystems. We can benefit from both failure and success of restoration. If we succeed, we are rewarded with a repaired ecosystem similar to its former state, implying that we can fully master the theory and manipulate it. If we fail, we still learn a lot how ecosystems work provided that we know the causes of failure. In either case, the experience can help us develop and advance the theory of ecology. But the important question is: how to judge the success or failure of ecological restoration? Should there be a set of indicators that are universally applicable, or are the indicators site-specific? There are no easy answers to these questions. Scientists differ in opinions in terms of their research interests and knowledge background.

2.2 Qualitative criteria

These criteria are usually suggested by stakeholders with different research interests or purposes.

(a) The Commonwealth EPA recognizes three generic success criteria (Bellairs, 1999):

- The site can be managed for its designated land use without any greater management inputs than other land in the area being used for a similar purpose;
- Restored native ecosystems may be different in structure to the surrounding native ecosystems, but there should be confidence that they will change with time along with or toward the makeup of the surrounding area; and
- The rehabilitated land should be capable of withstanding normal disturbances such as fire or flood.

Recommended by the regulatory agency, they are generic criteria targeted at ecological rehabilitation instead of ecological restoration, and ecosystem structure and management instead of ecosystem functions. They also emphasize the importance of low maintenance of the restored ecosystem although the impact of fire and flood are difficult to test proactively. Thus, there are no specific indicators and attributes to follow.

(b) Five criteria proposed by Ewel (1987) in assessing the success of ecosystem restoration

- Sustainability: the reconstructed community is capable of perpetuating itself, without further management.
- Invasibility: intact, nature communities are less easily invaded than ones that have been damaged. Invasion can be symptoms of incomplete use of

light, water and nutrients.

◆ Productivity: A restored ecosystem should be as productive as the original. It integrates many progresses, including photosynthesis, respiration, herbivory and death.

◆ Nutrient retention: a reconstructed community that loses greater amount of nutrients than the original is a defective imitation.

◆ Biotic interactions: plants, animals and microbes assemble in functional integrity.

The five criteria give equal emphasis on ecosystem structure and functions. They target not only on forests assessment, but also the abiotic environment and biotic interaction. It is more comprehensive than the one proposed by EPA, which is incidentally also its weakness. There are too many indicators or attributes to follow, which makes evaluation time consuming and expensive. We believe there are key indicators and they vary with different types of forest ecosystem and different levels of degradation.

There are other sets of criteria, for example, Lamd (1994) proposed that restored ecosystems should be evaluated according to productivity, ecological indices and socioeconomic indices. All these criteria were established from different points of view, and they are usually abstract and not systematic. To resolve this problem, it is necessary to gather more information from site-specific studies. These shall include inter alia different types of degraded landscapes, location and even the purposes of restoration.

2.3 Quantitative criteria

Progress and success of restoration can be measured in many ways. Compositional and structural indicators such as the quantity, size and mix of native species of plants compared to a reference system are the most frequently used approach (e.g. Kentula, 2000; McCoy and Mushinski, 2002). However, many authors feel that success in ecological restoration can only be realized when ecological functions of the system have been restored (Bradshaw, 1993; Toth *et al.*, 1995; Hobbs and Norton, 1996; King and Keeland, 1999; Stanturf *et al.*, 2001). Among the different ecological functions, erosion control, control of hydrology and nutrient cycling, capture of energy, and return of key fauna to the ecosystem are considered to be essential aspects of a fully operational ecosystem. Many researchers also believe that successful restoration projects must meet all their pre-set goals and objectives (Hobbs and Harris, 2001; McCoy and Mushinski, 2002). The goals and objectives will, of course, vary with the restoration projects. This being the case, should there be only one set of evaluation criteria?

The success of restoration can be defined differently at different times throughout its development. Successful progress at later stages in the life of a project depends very much on that of the earlier phases. At the very early stage of development, survival rates of plantings might be sufficient for an assessment of "success" (King and Keeland, 1999). As a stand starts to mature, however, more information is required to make judgments about whether the development of ecosystem components are on a trajectory that will lead to the restoration of a fully functioning, self-sustaining natural system adapted to the study site (Reay and Norton, 1999).

2.3.1 Methods for measurement

Current researches focus on the measurement of success or progress in restoration and describe the success of the restoration in conceptual terms, such as the meeting of structural, functional and social goals and objectives. There are a number of published papers, on the other hand, that provide concrete examples of indicators of success for individual projects. The methods used range from very simple to quite complex, and from a broad coverage of many restorations to detailed investigations of a few. Examples include the visual assessment of growth and density of native vegetation, sometimes assisted with photographic records; direct counting of percent survival trees (Sweeney *et al.*, 2002; Matthes *et al.*, 2003); detailed measurement of species and abundance of vertebrate wildlife communities (McCoy and Mushinski, 2002); and remote sensing methods to measure the overall return of forest vegetation on the landscape (Betts *et al.*, 2003).

Reay and Norton (1999) pointed out that by the end of the 20th century there are few established general criteria for assessing forest restoration progress, especially those dealing with ecosystem functions. They use species richness, the Shannon-Weiner index of diversity, and the Jaccard coefficient of similarity to track the progress of uneven-aged restoration sites, and compare them with the floristic compositional similarity of a reference mature forest site. They also assess the similarity of invertebrate populations between the restoration and reference sites as a measure of habitat function. The situation of not having any new general indicators in the literature continued until to date.

2.3.2 Indicators for ecological structures and functions

There are a small number of papers in the restoration literature that directly address the problem of devising more general measures of success for particular ecosystems. For example, Spencer *et al.* (2001) attempted to find a group of reference parameters that could be used for monitoring restoration projects in bottomland hardwood forests in southeastern Virginia, based on existing naturally regenerating sites. They studied the development of vegetation, soils and hydrology in a chronosequence of 17 clear-cut sites which had been undergoing natural regeneration for between 2 and 20 years. Unfortunately, among the parameters that they chose, including population densities and dominance values for vegetation, chemical analysis of soils, and biweekly water levels, they found no general indicators that could be "scientifically justified" as reliable measures of success. Two factors contributed to this conclusion. One was the natural variability in rates of succession due to a complex mix of influencing factors for aggradations in these wetland forests. The other was the ongoing demand for monitoring results early in the life of the project when variability is at its greatest."

Stanturf *et al.* (2001) have proposed several possible criteria for the measurement of restoration progress in bottomland hardwood forests in the Lower Mississippi Valley. Simple indicators such as canopy closure, the measure of soil organic carbon and soil dry bulk density appear to be possible candidates. Reference data for soil organic carbon and soil dry bulk density are obtained from a chronosequence of similar sites in the region.

Inouye (1988) discussed the question of how to evaluate whether a restoration project can be deemed to fall within an appropriate range of recovery trajectories given that variability is a constant in nature. Sources of variation in populations of plants and animals are discussed. The idea of evaluating the status of the earliest stage of succession as a way of determining whether a project is on its way to the desired end point is introduced. Data were presented to illustrate the difficulty of obtaining representative population measurements over a short time period. Inouye (1988) concluded that the evaluation of success using only short-term population measurements of key indicator species is difficult if not impossible.

In the reforestation literature, Hession *et al.* (2000) investigated the ecological benefits of riparian reforestation in urban watersheds. They collect a variety of baseline structure and function data for paired forested and unforested stream reaches in southeastern Pennsylvania and then use these data to determine the effectiveness of riparian reforestation on stream function for a range of urbanization levels in the Philadelphia area. None of these papers have addressed the problem of general indicators for evaluation of upland forest restoration independent of watershed data. Nor are there any articles that discuss indicators for upland forest communities.

Some recent work has been done in developing a set of general scientific criteria to measure success in the restoration of wetland ecosystems. Muotka and Laasonen (2002) used leaf retention as an indicator to compare the progress of restored headwater streams with natural streams and unrestored streams. Rheinhardt *et al.* (1999) found six indicators, including presence or absence of channelization, total basal area of trees, and percent litter cover, would jointly give robust assessments of ecological function in headwater ecosystems. Zedler and Callaway (1999) found that soil organic matter content, total Kjeldahl nitrogen, and plant growth could be used to compare restored and natural coastal marshes. Interestingly, Zedler and Callaway (1999) mentioned that success in the restoration of wetland ecosystems is much easier to follow and monitor than forests, due to the much longer recovery times for the latter.

A great deal of information about appropriate criteria to measure the progress of forest recovery through natural regeneration can be found in the literature, especially on the monitoring of ecosystem structure and functions. Bormann and Likens (1979) have covered comprehensively the subject of monitoring natural regeneration in Eastern Deciduous Forest ecosystems. They reference hundreds of articles giving details for methods of monitoring the recovery of vegetation and the control of functions such as nutrient cycling, hydrology and energy capture. A selection of more recent articles includes deGruchy *et al*. (2001) who studied the recovery of woody species on disturbed talus at Niagara Falls using various soil characteristics, species richness and percent aliens as indicators; Qi and Scarrett (1998) who tracked species richness and density of seed banks in an Ontario boreal forest; and Harvey and Brais (2002) who used species densities and height growth to measure the relative recovery of vegetation in disturbed and relatively undisturbed sample plots after careful logging in boreal forest in Quebec.

Larson *et al*. (1999) surveyed over 30 mature forests in Ontario. The methods they used, such as the Pointcentred Quarter Method (PQM), were robust and straightforward, and several of the criteria they measured, such as the population densities of trees and seedlings, the species and diversity of understory vegetation, and the presence of coarse woody debris, are readily transferrable to measuring progress in forest restoration. Keddy and Drummond (1996) described 10 possible indicators that might be used to characterize the condition of eastern deciduous forests in North America. These include inter alia tree size, canopy composition, amount of coarse woody debris, the number of spring ephemerals, and the density of wildlife trees (snags). They reviewed the literature to identify a typical range of values for each of these indicators in eastern deciduous forests.

McCarthy *et al.* (1987) studied the patterns and structure of woody species in an old-growth forest in Ohio. They used stratified sampling to locate transects within different communities in the forest, and set up circular quadrats along these transects to survey tree populations. The forest vegetation was characterized using stem density, basal area, species richness and importance percentages for the tree and sapling components of the vegetation. They also correlated soil and vegetation variables using multivariance analysis. Boerner and Cho (1987) used randomly placed quadrats along randomly placed transects to determine stem densities and size-frequency distributions of trees species in an old-growth forest in northwestern Ohio. Quigley and Pratt (2003) surveyed a sample of forest types at latitudes ranging from 0° to 40° north. They used replicate quadrats to count woody stems and characterized stem densities, basal area, and species richness at the different latitudes.

2.4 Summary of indicators and criteria in previous studies

As we have discussed before, the selection of assessment criteria and indicators must be linked to the goals set for particular restoration projects (Hobbs and Harris 2001). If this is done effectively, then the problems identified above largely disappear. This being the case, it may not be necessary to develop general ecosystem health indicators; instead, what we need is a toolbox of reliable indicators that can be selected for use in specific instances. The indicators, therefore, need to be tailor-made to suit the specific goals of restoration. They must be capable of indicating the improvement or otherwise of restoration progress.

Aronson *et al.* (1993) have identified a set of vital ecosystem attributes (VEA) which are correlated with and which can serve as indicators of ecosystem structure and functions. The attributes related to ecosystem structure include: (1) perennial species richness, (2) annual species richness, (3) total plant cover, (4) aboveground phytomass, (5) beta diversity, (6) life form spectrum, (7) keystone species, (8) microbial biomass, and (9) soil biota diversity. Vital Ecosystem Attributes related to ecosystem functions are: (1) biomass productivity, (2) soil organic matter, (3) maximum available soil water reserves, (4) coefficient of rainfall efficiency, (5) rain use efficiency, (6) length of water availability period, (7) nitrogen use efficiency, (8) microsymbiont effectiveness, and (9) cycling indices.

The Society of Ecological Restoration international (SER) (2004) produced a primer that provides a list of nine attributes as a guideline for measuring restoration success. They suggested that a restored ecosystem should have the following attributes: (1) similar diversity and community structure in comparison with reference sites; (2) presence of indigenous species; (3) presence of functional groups necessary for long-term stability; (4) capacity of the physical environment to sustain reproduction populations; (5) normal functioning; (6) integration with the landscape; (7) elimination of potential threats; (8) resilience to natural disturbances; and (9) self-sustainability.

Ruiz-Jaen *et al.* (2005) reviewed articles published in "*Restoration Ecology*" from volume 1(1) to volume 11 (4) dealing with the assessment of ecological restoration. They found that most of the studies assessed measures that can be categorized into three major ecosystem attributes: diversity, vegetation structure, and ecological processes. Based on similar approach, I have reviewed 86 papers published from 1993 to 2007 on the same subject matter but not limited to "*Restoration Ecology*", and the results are summarized in the ensuing paragraphs.

Attributes **Indicators**

Assessment

- Diversity
  - Plants
  - Arthropods
  - Birds
  - Guilds or trophic level
  - Others: microbes, fungi, earthworms, reptiles and small mammals
- Vegetation structure
  - Plant cover
  - Plant density
  - Plant biomass
  - Plant height
  - Litter: litter layers, cover, biomass and production
- Ecological progress
  - Biological interactions: herbivory, mycorrhizae, pollination, predation, seed dispersal, parasitism, or competition.
  - Nutrient pool
  - Soil organic matter

Figure 2.1 Summary of indicators contained in 86 papers, 1993-2007

All the indicators adopted by former researchers can be categorized into three ecosystem attributes; namely, diversity, vegetation structure and ecological progress (Figure 2.1). Among them, the diversities of plants and arthropods (diversity), plant cover and plant density (vegetation structure), and biological interactions (ecological progress) are most popularly used.

The majority of these studies were carried out in North America, Europe and Australia, mostly temperate regions while tropical to subtropical studies were few. Wetlands (16%), grassland (13%) as well as montane forest (13%) were the most frequently studied habitats. Antecedent land use or degradation activities were mainly associated with mining (28%) and agriculture (20%); and more importantly, most studies measured two ecosystem attributes in ecosystem assessment.

Restoration ecologists need a robust conceptual framework, realistic timetable, and accurate, reliable and transferable criteria for evaluation of their efforts (Clewell and Rieger, 1997). Higgs (1997) claimed that restoration success has not been thoroughly measured unless a full range of structural or compositional, functional, social, cultural, historical, political, aesthetic and moral indicators have been applied. But in practice, no study can measure all the indicators mentioned above in one restoration project because of financial and manpower constraints. Furthermore, many attributes related to ecosystem functions require long period of study.

2.5 Knowledge gaps

After reviewing the literature on restoration assessment, several knowledge gaps are identified. First, most of the assessment studies were conducted in North America, Europe and Australia, while only two were carried out in Asia (Indonesia and Philippines). There were virtually no studies dealing with subtropical forest ecosystems. Land degradation is a serious problem in subtropical China, where Hong Kong is located, because of rapid economic growth and urbanization. Second, ecological processes are rarely measured in the assessment due to time and manpower constraints. Third, most studies focus on one-off measurement of a single site that cannot depict changes with time reminiscent of the successional process. On top of these knowledge gaps, still less studies are concerned with the strategy of ecological restoration assessment (SERA).

2.6 Summary

Dozens of indicators or criteria have been proposed for the evaluation of restoration projects, yet we still do not have a set of standard criteria that is universally applicable to date. More specifically, assessment criteria or indicators for subtropical forest ecosystems are lacking in the literature. Different ecosystems have their own special properties, so are the different stages during ecosystem development. In Hong Kong as in other places, there are different types of land degradation and some are more severe and difficult to restore than others. It may be unrealistic to apply the same set of indicators to vastly different ecosystems. For instance, the recovery of soil productivity as represented by nitrogen accumulation must be faster on vegetation disturbance site (e.g., fire-disturbed slope) than soil destruction site (e.g., abandoned quarry). Whatever we measure will only reflect a point in time the particular developmental stage of a restored ecosystem. A viable solution to this problem is conducting the assessment study on a vegetation gradient recovering from the same perturbation. Furthermore, the selected indicators should be able to reflect the objectives of restoration. For instance, providing a green cover to a severely disturbed site is much easier than the re-creation of a self-sustainable ecosystem with maximum biodiversity. With different objectives, the indicators are expected to be different. Realizing these knowledge gaps in the literature, the present study would focus on subtropical forest ecosystem that has recovered from soil destruction, with equal weighting given to ecosystem structure and functions. A vegetation gradient consisting of uneven-aged plantations is chosen for the present study to depict their trajectory development compared to a carefully chosen reference site. This shall remove any uncertainties arising from one-off measurement.

CHAPTER 3

STUDY AREA

3.1 Introduction

This chapter describes the geographical setting of Hong Kong in general and the study area in particular. The information shall include the climate, soil and vegetation characteristics of Hong Kong and the study area. They are relevant to our understanding of the local ecological environment, including natural succession of the local forest ecosystem and developmental constraints of restored communities. Besides, this baseline information is also crucial to our selection of appropriate indicators for the assessment of ecological restoration. Lastly, the selection of a reference site for this study is also justified.

3.2 The geographical setting of Hong Kong

This section summarizes some key geographical elements of Hong Kong, with special emphasis on physical attributes that are related to the local ecology.

3.2.1 Climate of Hong Kong

The Hong Kong Special Administrative Region (HKSAR) has a total land area of 1096 km^2. It lies on the south-eastern coast of China, between latitudes 22°09' and 22°37'N, and longitudes 113°52' and 114°30'E. Hong Kong experiences a subtropical monsoon climate, with cool dry winter from November to February and a hot wet summer from May to September. The spring and autumn seasons are relatively short.

The mean annual temperature of Hong Kong ranges from 16°C in winter to 29 °C in summer. It is frost free all year round. The mean monthly sunshine is highest in July but lowest in March, reaching 1,950 hours per annum. Hong Kong is relatively humid, and mean monthly humidity ranges from 68 – 83% throughout the year but can drop to below 50% in winter (HKO, 2008).

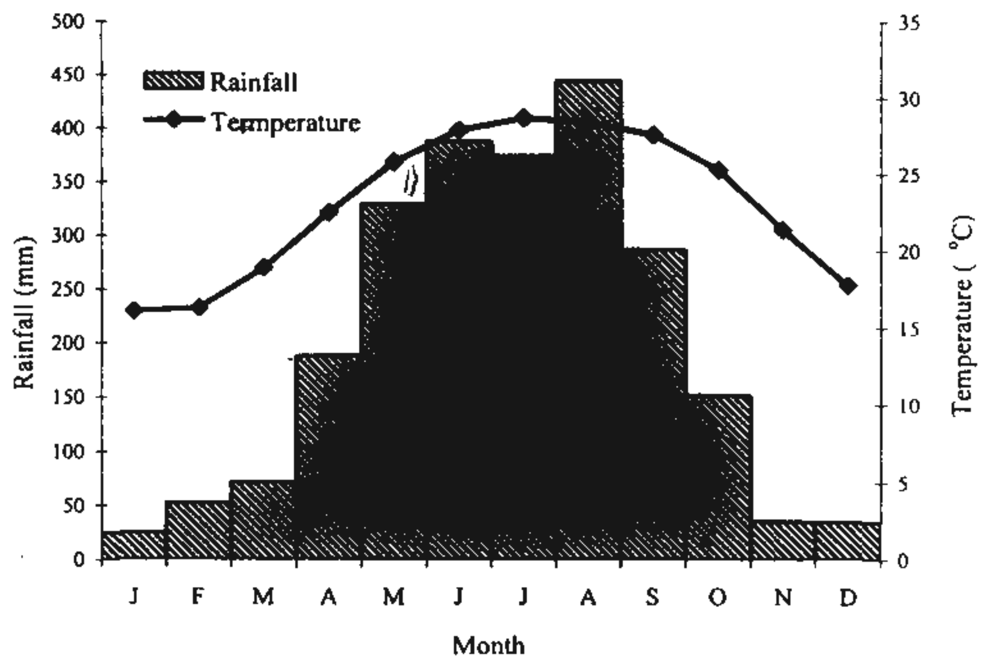


Figure 3.1 Mean monthly rainfall and temperature in Hong Kong, 1971-2000 (Source: HKO, 2008)

Hong Kong has a mean annual rainfall of 2,300 mm, which is unevenly distributed throughout the year as more than 80% of the total is recorded in summer (Figure 3.1). The mean monthly rainfall fluctuated from a maximum of 445 mm in August to a minimum of 25 mm in January. Other forms of precipitation include fog and mist that are prevalent from February to April. Occasional tropical cyclones are encountered in summer, which bring about very strong winds and abundant rainfall.

Other severe weather phenomena include monsoon troughs with strong winds in winter and thunderstorms with squalls in summer. Snow and tornados are, however, rare (HKO, 2008).

3.2.2 Geology of Hong Kong

Eighty-five percent of the land area is underlain by igneous rocks (Dudgeon and Corlett, 2004). They were formed from the vigorous volcanic activities associated with the Yanshanian tectonic movement in the Paleozoic Era. Peak volcanic activities occurred at Late Jurassic and Early Cretaceous during which large amount of lava and ash were erupted, and substantial granite batholiths intruded underground (Peng, 1986). The Repulse Bay Formation is the most common rock types found in Hong Kong, which spread over the eastern peninsulas of the New Territories, Tai Mo Shan, western Lantau Island and about half of Hong Kong Island. The rocks, siliceous and rhyolitic in chemical composition, are dominated by volcanic tuffs with some lava in the marginal areas.

Two types of igneous rocks, extrusive volcanic rocks and intrusive granite, are found in Hong Kong, accounting for around 50% and 35% of the total land area, respectively (Sewell, 1999). The former was formed from rapid cooling of lava and ash emitted through volcanic eruption while the later was formed from intrusion underneath. Both rocks were similar in chemical composition yet granite is more coarsely grained because of the slower cooling and crystallization of the minerals underground (Dudgeon and Corlett, 2004).

Non-igneous rocks scatter as tiny patches in the territory; they include sedimentaries, metamorphosed sedimentaries and Quaternary deposits. These sedimentary rocks account for approximately 10% of Hong Kong's total surface area, while the remaining is reclaimed land (CEDD, 2008a).

3.2.3 Soils of Hong Kong

Soil is the product of weathering and pedogenesis processes. In Hong Kong, more than 80% of the land area is covered by hill soils of intrusive granite or extrusive volcanic. Granite and volcanic rocks have similar chemical composition but different granular sizes. With well-defined joints, granite is intensely weathered to form a coarse-textured, loose and structureless substrate. Soils derived from granite are classified as ferrisol, and those derived from volcanic rocks are classified as ferrisol (<500 m) or ferrallisols (>500 m) according to the China Soil Classification System since the early 21st century (Luo *et al.*, 2007).

Ferrisols derived from granite are leached and depleted of organic matter. The soil is dominated by kaolinite, with a low cation exchange capacity (CEC) of 37.24 mmol/kg, a base saturation of 16.63%, and an average SiO_2:Al_2O_3 ratio of 2.0. The soil is acidic in reaction and contains low levels of organic matter, nitrogen (N), phosphorus (P) and cation nutrients (Shaw, 1993; Wong, 2004). Gullying and badlands formation is thus common on barren slopes underlain by granite. Where the lands are opened up for development, including Borrow Area and quarry, the decomposed granite (DG) is sometimes retained for sale as a growth substrate in horticulture planting.

Volcanic rocks are less easily weathered compared to granite (Irfan, 1999). Soils derived from volcanic rocks are granular in structure and has higher clay content. Volcanic soils have poorly developed profile but are generally more fertile than the granitic soils, primarily due to their finer texture and higher water- and nutrient-holding capacities (Luo *et al.*, 2007; Anthelme *et al.*, 2008).

3.2.4 Vegetation of Hong Kong

With a subtropical monsoon climate, Hong Kong used to support a climax vegetation of monsoonal forest or evergreen broad-leaved forest (Chang *et al.*, 1989). As a result of human disturbances in the 19^{th} century and extensive clearfelling during Japan's occupation of Hong Kong from 1941-1945, most of the original forests had disappeared. These forest lands were later replaced by grassland, scrubland and secondary woodland, depending on the pressure of human interferences. To date, remnants of the climax forest exist as *feng shui* woodlands behind Hakka villages in the New Territories. They are characterized by a high biodiversity of local fauna and flora species.

According to the Hong Kong Herbarium (2004), Hong Kong has a diverse flora of 261 families, 1,374 genera, and 3,164 species and varieties. There are 390 native tree species belonging to 192 genera and 67 families (Zhuang *et al.*, 1997).

Currently, about one-third of the existing forests are plantation forests dominated by monoculture of exotic species, such as *Lophostemon confertus*, *Acacia confusa*, *A. auriculiformis*, *A. mangium*, *Eucalyptus* spp., *Melaleuca quinquenervia* and *Pinus elliottii*. In recent decades, there is an increase in the planting of native species to attract wildlife and improve biodiversity.

3.3 Reforestation in Hong Kong

The vegetation of Hong Kong has been subject to long period of disturbance by people. Before founding of the colony in 1842, *Aquilaria sinensis* was cut and exported to south China and Southeast Asia for the manufacturing of joss sticks. The wood of this species is fragrant, from which Hong Kong attained its present name. As early as the late 19th century, trees were planted for amenity purpose around the urban development on Hong Kong Island. The main species planted by then was *Pinus massoniana*, which is a pine species native to south China. It adapts particularly well on strongly acid hill soils deficient in nitrogen and phosphorus (Chinese Tree Species Editorial Committee, 1976). The pine was also planted in "forest plots" in the rural area to provide fuel and timber.

During the Japanese occupation of Hong Kong from 1941-1945, the countryside suffered from uncontrolled cutting and forest lands were reduced to barren slopes. Trees were logged to provide fuel wood and energy. Immediately after World War II and defeat of the Kuomintang in China, there was an influx of over 1 million refugees from across the border. Because of this, there was a great demand for fresh water supply by the inflated population. New reservoirs were built in Shek Pik on Lantau Island and Tai Lam Chung in the New Territories in the early 1950s. This was accompanied by large scale reforestation of the waste hill side and catchment areas to protect the soil and conserve water resource, while demand for timber and fuel was reduced (Daley, 1970). The dominant species used to revegetate the barren hills were *Melaleuca quinquenervia*, *Eucalyptus robusta*, *E. citriodora*, *E. tereticornis*, *Lophostemon confertus* and *Acacia confusa*. They are fast growing pioneer species with an ability to grow on severely impoverished soils. Native species were few with the exception of *Pinus massoniana*.

Restoration of the eroded slopes continued into the mid-1960s, using more or less the same species mix. *P. massoniana* was found to be not suitable for slope revegetation because it was susceptible to damage by fire (Thrower, 1970), attack by insect (Daley, 1970) and pine wilt nematode (Winney, 1980). The pine litters are rich in polyphenols, which can acidify and further impoverish the soil (Ross, 1989). It was gradually replaced by the aforesaid exotic species, which accounted for 85% of the annual planting in Hong Kong.

With enactment of the Country Parks Ordinance in 1976, 40% of the territory was designated as country parks. This protected area system fulfills three objectives: conservation, education, and recreation. Annually, about 12 million visitors are recorded in Hong Kong's 23 country parks. Besides the long-term objective of soil and water conservation, restoration planting has been extended to cover fire-effected areas, landfill sites, borrow areas, and grassy slopes in an attempt to improve visual impact, accelerate ecological succession, and preserve the local biodiversity (AFD, 1988). Because of this changing need, more native species (e.g. *Castanopsis fissa, Schima superba, Cinnamomum camphora, Tutcheria spectabilis, Szygium jambos*) and exotic woody legumes (e.g., *Acacia confusa, A. mangium, and A. auriculiformis*) are used in restoration planting. More recently, there are field trials on the transformation of senile exotic woodland plantations by adding native species to the understorey stratum.

3.4 Selection of study site

With a population of 7 million and a total land area of 1,060 square kilometers, Hong Kong is one of the most densely populated areas in the world. The population pressure poses a demand for infrastructure development, such as new towns, highways, public utilities and transport terminals. However, due to the hilly nature of the territory, land scarcity becomes a major obstacle of infrastructure and economic development. In order to provide more utilizable land in Hong Kong, reclamation is practiced using either land-derived fill or marine fill. The use of land-derived fill leads to the occurrence of borrows area from where decomposed granite is excavated. A borrow area refers to a site, mostly eroded granitic hill, from where the topsoil is excavated to provide fill materials for infrastructure development. It is necessary to obtain approval from the authority, such as Agriculture, Fisheries and Conservation Department (AFCD) of Hong Kong, if the site is located within country parks, before excavation of the soil. This involves the preparation of a master plan by the project proponent to ensure proper planning, rehabilitation and management of the borrow area. Specifically, the plan gives details of land reinstatement by engineering and biological means during and after excavation. In this research, I shall focus on the study of restored borrow areas in the New Territories of Hong Kong for four reasons. First, the borrow areas are typical soil destruction sites, where the soil suffers from decline in physical, biological and chemical properties. Among the different types of degraded land in subtropical China, borrow area is one of the most difficult derelict landscapes to restore. This is reminiscent of a worst scenario approach in ecosystem re-creation studies. Any uncertainties arising from the short history of restoration could perhaps be resolved by this approach. Second, the demand of fill materials is ever-increasing in south China as a result of rapid economic development. There is an urgent need to also repair the degraded landscapes as early as practicable. The quality of restoration is of paramount importance to local ecology; hence assessment criteria are needed to fulfill this task. There is no reason why results obtained from this study cannot be applied to other areas in south China. Third, restoration planting in the borrow area is undertaken in different phases. Uneven-aged woodland plantations are, therefore, available for this study, providing an excellent opportunity to monitor their trajectory development in respect of ecosystem structure and functions. Fourth, in the restoration of borrow areas in Hong Kong as other degraded landscapes, the objectives are loosely defined as to provide a rapid vegetative cover (greening) to the re-contoured landscape. Land reinstatement, planting strategy, species selection and post-planting care are geared towards this loosely defined objective. The results obtained from this study can perhaps help us re-define more clearly the planting objectives and shed light on the management of restored woodlands.

3.4.1 Tai Tong East Borrow Area

The study area is located in Tai Tong East Borrow Area (TTEBA), south of Yuen Long in northwestern New Territories (Figure 3.2). The site comprises a peak rising to 267 m, and dissected ridges radiating from it to the Yuen Long Plain. Besides the deeply incised valleys, which run parallel in a northwesterly direction, gullies with precipitous sidewalls are abundant in the borrow area. The site is underlain by fine- to medium-grained granite while the peak is capped by volcanic tuff. Some of the valleys contain colluviums, which are slope wash deposits derived from the eroded ridges.

Before excavation, the land was severely eroded and dissected by gullies, being comparable to the badlands now seen east of the site. The vegetation was dominated by the secondary growth of *Pinus massoniana*, majority of which had been killed by the nematode disease (*Busaphelenchus xylophilus*) since the mid-1960s. Acid soil indicator plants are abundant, including *Baeckea frutescens, Dicranopteris linearis, Lycopodium cernuum, Melastoma candidum, M.*

dodecandrum and *Rhodomyrtus tomentosa*. These species can tolerate strong acidity and low fertility soils.

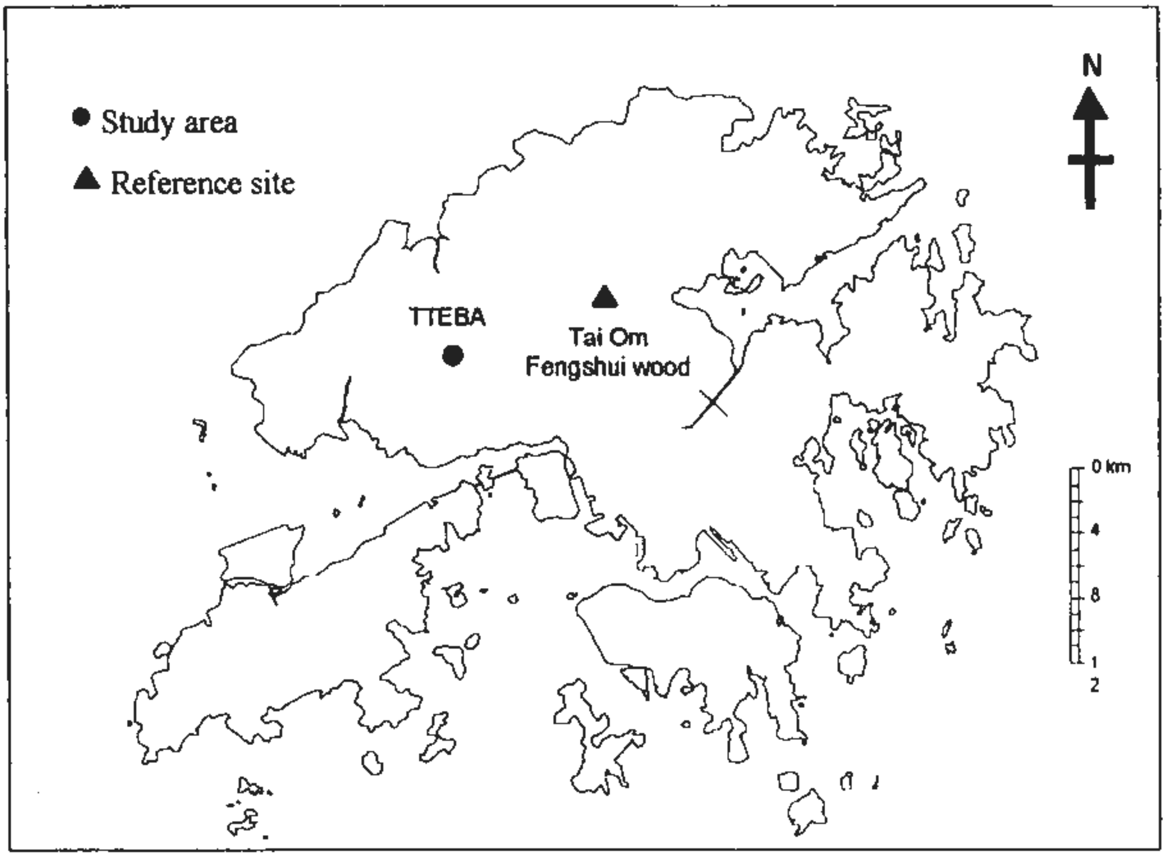


Figure 3.2 Location of the study areas and reference site

To kill two birds with one stone, excavation began in the late 1980s and has already been completed now. The site was excavated to an average depth of 8 meters, yielding 3.5 M m3 fill materials. These materials were used for the construction of the Yuen Long Bypass Highway and reclamation of land for the Tin Shui Wai new town. This resulted in complete removal of the vegetation and topsoil, which is a typical example of soil destruction in land degradation.

Figure 3.3 Location of the three study sites in TTEBA

The TTEBA was rehabilitated in different phases from 1989 to 2000, as required by the contract. This practice is necessitated by the need to protect to site from soil erosion and to remove the visual impact. Three sites restored in different years were selected for the present study so that the current strategy of forest rehabilitation can be evaluated within a vegetation chronosequence. The three sites were separately restored in 1989 (R89 hereafter), 1995 (R95 hereafter) and 2000 (R00 hereafter) (Table 3.1). The three sites are located within TTEBA, not far away from each other. Sites R00 and R95 are flat, while site R89 is located on a 35° slope facing west.

We believe limitations arising from the relatively short history of restoration can be overcome by this vegetation chronosequence approach. With successional development of the restored woodlands, changes can be detected in the soil and vegetation structure. The evidence of the changes is undoubtedly more important than the magnitude of changes within a short period of time. Indeed, there are no better solutions than the chronosequence approach in this kind of ecological studies.

Table 3.1 Description of the three restored sites

Site	Grid reference	Altitude	Aspect	Slope angle	Geology	Planted species	Year of planting
R00	22°24'46.55"N 114° 2'41.71"E	143m	Flat	0	Granite		2000
R95	22°24'39.60"N 114° 2'30.81"E	153m	Flat	0	Granite	Predominant exotic species	1995
R89	22°24'44.54"N 114° 2'19.69"E	84m	West	35^0	Granite		1989

(a) R89, (19-year-old woodland)

(b) R95, (13-year-old woodland)

(c) R00, (8-year-old woodland)

Plate 3.1 The three woodland plantations in TTEBA

The three sites are dominated by granite, which is pinkish and grey in color and its main components are quartz, potassium feldspar and acid plagioclase feldspar with some biotite and occasional hornblende (Peng, 1986). This rock is susceptible to rapid weathering in the subtropical region, resulting in the formation of thick regolith (>60m) that can be differentiated into four distinct zones according to the degree of weathering (Ruxton and Betty, 1957; So, 1986). For this reason, it is a good source of fill materials. Gullies are formed as a result of deforestation and partial dissection by stream action, which removes the surface materials by dehydration, deflation and surface wash (Grant, 1962).

The dominant soil in the borrow area is ferrisol, which is acidic in reaction due to siliceous nature of the parent materials and intense leaching of the bases. During the excavation of fill materials, the vegetation was cleared and the topsoil beheaded, leaving behind the subsoil layer dominated by semi-weathered parent materials. It is a typical example of soil destruction in land degradation, in which soil organic matter and nutrients are lost, and ecological functions are disrupted. After excavation, the borrow area is re-contoured to blend in with the surrounding environment. Interception channels and herringbone drains were provided to divert any excessive runoff from the slope. An overburden layer of 500 mm is laid on top and compacted to form the planting surface. This overburden layer consists of *in situ* decomposed granite, which is coarse-textured but contains a high proportion of weatherable minerals. Owing to poor workmanship and supervision, the thickness of this overburden layer is not always uniform. In places, the thickness is less than 200 mm and root growth is constrained. The re-contoured surface was then hydroseeded with a mixture of grass seeds while tree planting was carried out on the grassed slope about 3-6 months afterwards. Ecological restoration of the borrow area, therefore, include land re-contouring, provision of drainage, hydroseeding and tree planting. The above processes were carried out throughout the year and planting was not confined to on-set of the rainy season even though this is a poor practice.

Before excavation of the site in the late 1980s, the vegetation in Tai Tong had suffered serious cutting during World War II. A secondary woodland dominated by *Pinus massoniana* had evolved thereafter through natural regeneration and enrichment planting of the badland slopes. Fire occurs frequently in Tai Tong due to the high volatile oil content of the pine and accumulation of needle litters on the forest floor. The secondary woodland was subsequently reduced to a species-poor community as in other parts of Hong Kong (Fung, 1995). It is undoubtedly an eyesore to the public, which has prompted the government to convert it to a borrow area. In so doing, the government is killing two birds with one stone. Through excavation, re-contouring of the land and revegetation, the environment is expected to improve. Ecosystem repair in TTEBA began with hydroseeding, a grass establishment technique introduced from overseas. There are two recipes of hydroseeding throughout the year (Table 3.2). Between April and August, a mixture of *Cynodon dactylon* (common Bermuda), *Paspalum notatum* (bahia grass) and *Eragrostis curvula* (weeping love grass) grasses seeds, together with inorganic fertilizers, soil solidifying agent and the dye malachite green, were applied by a high pressure jet on to the re-contoured slopes (Tsang, 1997). The grasses would germinate within 5-7 days, and under ideal conditions a complete grass cover can be established in 8 weeks time in summer. Between September and March, a different recipe is employed, with the cool season grass *Lolium perenne* replacing *Eragrostis curvula* in the previous recipe. This enables the cool season grass to germinate in winter.

Table 3.2 Seed mixtures and ingredients in hydroseeding

	Seed mixtures	Common name	g/m^2
1a.	Between September to March		
	• *Cynodon dactylon*	Common Bermuda grass	15
	• *Paspalum notadum*	Bahia grass	12
	• *Lolium perenne*	Perennial ryegrass	3
1b.	Between April to August		
	• *Cynodon dactylon*	Common Bermuda grass	15
	• *Paspalum notadum*	Bahia grass	10
	• *Eragrostis curvula*	Weeping love grass	0.5
2.	Other ingredients		
	• Mulch fiber		200
	• Fertilizer (NPK ratio 15:15:15)		100
	• Soil solidifying agent		50
	• Dyestuff (malachite green)		0.2

Regardless of the recipe adopted, tree seedlings up to 600 mm high were planted on the grassed slope 3-6 months after hydroseeding. Trees are introduced for three reasons: (a) they are more resistant to fire than grasses, (b) the exotic grasses are not sustainable under natural conditions, and (c) trees are believed to be the dominant vegetation type in Hong Kong.

The government department overseeing the excavation project was the then Territory Development Department (TDD), which was re-organized in 2004 to form a new department known as Civil Engineering and Development Department (CEDD). Because of this re-organization, most of the planting records were lost, except those in 1994 and 1995 (Table 3.3). As seen from the planting list, there were more exotic species than native species being planted in these two years. In 1994, for instance, exotic species accounted for 65.5% of the total compared to 34.5% for native species. The corresponding values for 1995 were 53.5% and 46.5%. Among the exotic species, there were also higher percentages of nitrogen-fixing legumes (e.g. *Acacia confusa, A. mangium* and *A. auriculiformis*) and non-legumes (e.g. *Casuarina equisetifolia*) than other broad-leaved species. The eucalypts are exotic pioneer species that can thrive on inhospitable sites. Nevertheless, species with a capacity to fix atmospheric nitrogen are preferred to late-successional native species in the borrow area.

Table 3.3 Tree species planted in Tai Tong East Borrow Area

Year	Species	Origin	Nitrogen fixing	% total
1994	*Acacia auriculiformis*	Exotic	Y	14.4
	A. confusa	Exotic	Y	13.4
	A. mangium	Exotic	Y	12.9
	Eucalyptus calophylla	*Exotic*	*N*	*11.1*
	E. torelliana	*Exotic*	*N*	*8.5*
	Cinnamomum burmannii	Native	N	8.2
	Ficus retusa	Native	N	8.2
	Casuarina equisetifolia	Exotic	Y	5.2
	Other broadleaves	Native	/	18.1
1995	*A. confusa*	Exotic	Y	17.3
	A. mangium	Exotic	Y	15.8
	Sapium discolor	Native	N	15.8
	E. camaldulensis	Exotic	N	7.1
	E. calophylla	Exotic	N	4.7
	A. auriculiformis	Exotic	Y	4.3
	Casuarina equisetifolia	Exotic	Y	3.4
	Pinus elliottii	Exotic	N	0.9
	Other broadleaves	Native	/	30.7

Source: Unpublished planting record of Territory Development Department, 1995 (Cited from Tsang, 1997)

Whip-size seedlings about 600 mm in height were planted on the grassed slope, at 0.75-1.0 m center to center. Dense planting of tree seedlings is a common practice in Hong Kong to ensure adequate survival of the species. Whip-size trees are also preferred to standard size trees (25-mm diameter) in hill slope planting because of the poor growth environment.

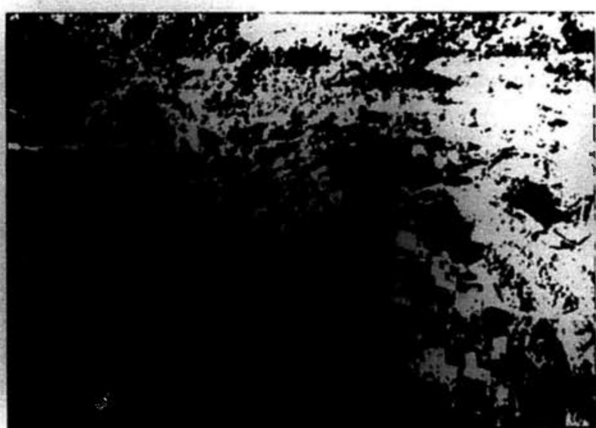
We believe the same practices were applied to sites R89, R95 and R00 selected in the present study despite the incomplete records. Once an agreement is signed between the landscape contractor and a government department, variations are extremely difficult to make.

3.4.2 Selection of a reference site

In addition to evaluating the attributes in the restored sites, it is necessary to compare them with that of a reference site to estimate the level of restoration success (Passell, 2000; Purcell *et al.*, 2002; SER, 2004). A reference site serves as a model for planning a restoration project, and later for its evaluation (SER, 2004). Two issues need to be considered in the selection of reference site. First, reference sites should occur in the same life zone, close to the restoration project, and should be exposed to similar natural disturbances (Hobbs and Harris, 2001; SER, 2004). This is a useful guideline that can avoid ambiguities arising from different life zones and different natural disturbances. Second, more than one reference site should be selected to avoid variations between different sites (Hobbs and Norton, 1996; SER, 2004).

In the present study, Tai Om feng shui woodland (TO) is selected as the reference site. Feng shui woodland refers to remnants of climax vegetation behind traditional Hakka villages in Hong Kong (Daly, 1970; Zhuang, 1993; Yip *et al.*, 2004). Chinese people, including the Hakka ethnic group, believe in feng shui, a supernatural power influencing the vitality of human being. Old trees are treated as a symbol of fortune and longevity, not to mention their ecological values. Therefore, trees behind Hakka villages have been protected from cutting, fire and other human disturbances for over several hundreds of years in Hong Kong. They become today's feng shui woods, which are highly endemic and characterized by high species diversity. In his study of 112 feng shui woods, Chu (1998) found a total of 567 vascular plants, accounting for over 25% of the total in Hong Kong. Feng shui woods are unique habitats with great ecological values in Hong Kong. It is like a living plant herbarium, offering valuable resources for both academic research and nature conservation (Joseph *et al.*, 2004). Some common species found in the feng shui woods include *Ficus microcarpa, Cinnamomum camphora, Dimocarpus longan, Gironniera nitida and Litchi sinensis*. Because of its endemic property, high biodiversity and long history of development, the feng shui woods represent a point of advanced development that lies somewhere along the intended trajectory of the restoration. It is, therefore, chosen as a reference site for this study.

The Tai Om feng shui woods (TO), is located in the Lam Tsuen Valley of Tai Po, in the New Territories of Hong Kong. It occupies a low hill about 80 m above sea level, and is underlain by sedimentary rocks. Tai Om village in front of this feng shui woods belongs to the Cheung clan, whose ancestors had moved to the place from Wuhua County in the early Qing dynasty. According to a village elder, the feng shui wood is more than 300 years old. Intense weathering of the parent material results in the formation of ferrisol as described in section 3.2.3. The top 10 cm layer of the ferrisol is strongly acidic in pH reaction (4.05), contains appreciable amount of organic carbon (43,503 kg/ha) and total Kjeldahl nitrogen (3,150 kg/ha), but low levels of available potassium (53 kg/ha), calcium (54 kg/ha) and magnesium (31 kg/ha) (Chau *et al.*, 2007). This feng shui wood, 20 km east of the three restored sites, is located in the same life zone as the TTEBA. Because of this, they are likely exposed to the same natural disturbances in the past although human disturbance between them in the last two decades was different. It is true that there are inherent variations between different reference sites. In this connection, there are nearly 600 feng shui woods in Hong Kong, ranging from 1 ha to 10 ha in area. Owing to the constraint of manpower and time, however, only Tai Om feng shui woodland is selected for this study. Reference will also be made to other feng shui woods in Hong Kong, where necessary.



(a) Layout

(b) Hakka village

(c) Structure

Plate 3.2 Tai Om feng shui woodland

CHAPTER 4

VEGETATION DYNAMICS IN RESTORED SITES

4.1 Introduction

Vegetation cover is the most important and indispensable part of forest ecosystem; it plays a vital role in ecosystem development. If we want to assess the success or failure of forest restoration, biodiversity and vegetation growth are perhaps the most appropriate indicators. Vegetation measurement involving the static and dynamic attributes is undoubtedly a powerful tool in ascertaining ecosystem development. This can be accomplished through continuous monitoring of a particular site or a vegetation chronosequence consisting of uneven-aged plantations.

Vegetation is a vital component of any terrestrial ecosystems. There are many attributes of vegetation characteristics; some are static (e.g., stand structure) while others are dynamic (e.g., species diversity). Vegetation structure and species diversity are the most popular ecosystem attributes mentioned in the literature (e.g. Ruiz-Jaen and Aide, 2005). Diversity is usually measured by determining richness and abundance of vegetation within different layers (Nichols and Nichols, 2003). In addition, it is useful to determine the diversity of species within different functional groups because this information provides an indirect measure of ecosystem resilience (Peterson *et al.*, 1998). Vegetation structure is, on the other hand, determined by measuring vegetation cover (e.g., herbs, shrubs and trees), woody plant density, biomass, or vegetation profiles (Salinas and Guirado, 2002; Kruse and Groninger, 2003; Wilkins *et al.*, 2003), and these attributes are useful for predicting the direction of plant succession. In previous studies, researchers measured at least the recovery of vegetation structure or species diversity after restoration. This is inevitable because laws governing rehabilitation always require the project proponent to monitor growth of the vegetation (Allen, 1992). Furthermore, it is assumed that the recovery of fauna and ecological processes will follow vegetation establishment automatically (Toth *et al.*, 1995; Young, 2000). For example, there is a strong correlation between vegetation structure (e.g., height, foliage layers, and basal area) and the recovery of bird population (Tilghman, 1987; George and Zack, 2001). Similarly, the development of ecosystem structure can enhance seed dispersal and nutrient availability (Robinson and Handel, 1993; Bradshaw, 1997; Rhoades *et al.*, 1998). Furthermore, vegetation structure is easy to measure and there are few seasonal variations in the measurements (Martin and Karr, 1986; Lefebvre and Poulin, 1996; Holmes and Sherry, 2001). In contrast, the measurements of dynamic attributes, such as productivity or decomposition, require at least a year to yield any meaningful results because of their seasonal variations.

In this study, I shall measure both the static and dynamic attributes of vegetation on the restored and reference sites. These include: plant composition, vegetation biomass, stratification and standing litter. In Hong Kong destructive sampling is prohibited on any ecosystems; hence vegetation biomass is estimated indirectly by measuring the tree height, diameter at breast height (DBH), transection area at breast height (TABH), crown area, tree density, and canopy closure of the uneven-aged plantations. The results obtained from this experiment shall provide answers to the following specific questions:

(1) What are the characteristics of the vegetation in the three restored sites?

(2) Is ecological succession happening in the restored sites?

(3) What kinds of vegetation indicators can be used for the assessment of restoration progress?

(4) Is Tai Om feng shui woods a suitable reference site?

4.2 Materials and Methods

This section describes the experimental design of the study, procedures for vegetation survey, collection of field data, and statistical analysis of the data.

4.2.1 Field measurement of vegetation

The quadrat method was adopted in vegetation measurement. It was carried out on the three restored sites in TTEBA (R89, R95 and R00) and the feng shui woodland in Tai Om (TO) from October to November, 2006. A 10m × 10m plot was demarcated randomly in each of the study sites. Precautionary measures were taken to avoid any unnecessary edge effects of the site. Every woody plant species in the plot was identified and recorded. The diameter at breast height (DBH), height, and crown area of these species were also measured. Plant height was measured either directly (<4.0 m) by measuring tape or indirectly (>4.0 m) by the trigonometric principles (Bonham, 1989). The height of each woody individual was measured to the nearest cm. DBH was measured by measuring tape, at 1.3 m above ground for plants taller than 100 cm and to the nearest 0.1 cm. The height of the woody species was divided into three classes (<3 m, 3-12 m, and >12 m) and the DBH into two classes (>2 cm and ≤2cm). Crown area was determined by measuring the longest and shortest diameter of the tree crown projected on to the ground to the nearest cm. The two values were then averaged for the estimation of crown area.

Canopy closure was measured at four spots in each study site, with a camera and a convex mirror engraved with 24 squares in March, 2008. The convex mirror was hand held at breast height so that the observer's head was reflected from the edge of the mirror just outside the graticule. The curved mirror reflected the canopy above, and canopy closure was estimated by calculating the number of squares (or quarters of squares) that the image of the canopy covers (Korhonen *et al.*, 2006).

Ten standing litter samples were collected from each of the four sites. A transect was laid on each study site, on which ten 0.5m × 0.5m quadrats were demarcated systematically. The litter inside each of the quadrats was collected by carefully eliminating the soil particles. Grass coverage was estimated by counting the actual number of individuals within the 10m x 10m plot. It was further verified with reference to photos taken for the individual site. There were very few grasses in the restored sites.

4.2.2 Species diversity

The species diversity of trees at each site was estimated by the Shannon–Wiener index below:

$$H' = -\sum_{i=1}^{S} p_i \ln p_i$$

Where: p_i is the relative abundance of species i calculated as the proportion of that individuals in the whole community; and S is the number of species. The Shannon diversity index, H', is higher for species-rich ecosystem than species-poor ecosystem.

4.2.3 Litter treatment

The standing litters were returned to the laboratory, rinsed with water to remove all the soil particles, and oven dried to constant weight at 85°C. They were then sorted into three components: foliage, branches (and stems) and fruits. The other components, such as flowers and fruss, were not included in the present study in view of their small quantities, and one-off collection of the litter in November-December, 2006. The biomass is expressed as tonnes per hectare on oven dry weight basis (ODW).

4.2.4 Statistical analysis

The data obtained from the fieldwork were calculated using the spreadsheet software Microsoft Excel and Statistical Package for Social Sciences (SPSS) 13.0 for Windows, for the numerical processing of mean values and standard deviations. The differences in vegetation and litter measurements among the four sites were tested by Duncan's Multiple Range Test at the confidence level of $p<0.05$, unless otherwise specified.

4.3 Results and discussions

4.3.1 Woody species composition

The number of woody species increased with age of the woodland plantations in TTEBA. *Acacia mangium* and *A. confusa* are the only species found in R00, reaching a density of 4,800 nos./ha (Table 4.1). They are exotic species planted in the site together with other species, at about 0.8-1 m centre to centre. Tree density in the site has, therefore, decreased drastically since planting, as a result of typhoon damage and stem exclusion among the species. Four tree species are found in R95, including

Acacia mangium, *Acacia confusa*, *Liquidambar formosana* and *Cinnamomu camphora*. The latter two are native, mid- to late-successional species associated with secondary and feng shui woods in Hong Kong. Hence, the two species are expected to grow better on fertile soils than infertile soils. For plants with a DBH ≤2cm, only two *Liquidambar formosana* species were recorded in the plot. Tree density of the site is equal to 2,900 nos./ha. In R89, there were four tree species: *Acacia confusa*, *Acacia mangium*, *Eucalyptus torelliana* and *Casuarina equisetifolia*. Unlike R95, all of them are exotic species and the DBH for three of them is ≤2cm. Tree density is highest among the three restored sites, reaching 5,400 nos./ha.

Overall, the three restored sites are dominated by exotic tree species and there appears no relationship between tree density and age of the plantations. Thus, tree density decreases in the order of R89 > R00 > R95. Given the same planting density to begin with, tree density is expected to decrease with age of the plantations as a result of stem exclusion. This is, however, not the case in the restored sites, implying that stem exclusion is not the only factor affecting tree density in TTEBA. Typhoon, uneven substrate depth, and even fire are also possible contributory factors. Indeed, typhoon damage is serious in TTEBA and the two legumes, *A. auriculiformis* and *A. mangium* are easily uprooted by typhoon because they are shallow-rooted species. This problem is further aggravated by poor workmanship in site preparation works by the contractor. The overburden layer is not always uniform and in places its thickness is less than 200 mm instead of 500 mm as is required by the contract. Although the restored sites are dominated by a few tree species, the percentages of trees with DBH>2 cm are higher than those with DBH≤2 cm. They increase in the order of R00 (87.5%), R89 (90.7%) and R95 (93.1%). Another interesting phenomenon is that no shrubs are found in any of the restored sites in TTEBA, suggesting that invasion of native species is difficult and that only tree species were planted during restoration.

The reference site at Tai Om (TO) differs greatly from the restored sites in species composition. It has a higher biodiversity, with a total of 17 native species compared to two only recorded in R95 (Table 4.1). This is expected because feng shui woods are highly endemic. Unlike the restored sites, the percentages of species with a DBH≤2cm (87.6%) are much higher than those with a DBH>2cm (12.4%). This clearly shows that the feng shui wood is undergoing rapid regeneration. Four out of the ten species with a DBH≤2cm are also represented in the DBH>2cm class. These include *Aporusa dioica*, *Psychotria asiatica*, *Sarcosperma laurinum* and *Sterculia lanceolata*. On the other hand, *A. confusa* and *A. mangium* planted in the restored sites are prolific litter producers. The thick litter layer can trap the seeds, preventing them to germinate on the forest floor. Allelopathic substances contained in the *A. confusa* litter can also inhibit seed germination (Chou *et al.*, 1998).

Out of the 17 species recorded in Tai Om, 8 are shrub species (species having a dual life form of tree and shrub are counted as shrub here), which is unrivalled by any of the restored sites. The restored sites in TTEBA, therefore, lag behind the feng shui woods in terms of biodiversity (species number and life forms) and regeneration potential.

Table 4.1 Woody species of the sites

Site	DBH>2cm					DBH≤2cm				
	Species	Family	Life form	Number	Total	Species	Family	Life form	Number	Total
R00	*Acacia mangium*	Mimosaceae	T	35	42	*Acacia mangium*	Mimosaceae	T	3	6
	Acacia confusa	Mimosaceae	T	7		*Acacia confusa*	Mimosaceae	T	3	
R95	*Acacia mangium*	Mimosaceae	T	14	27	*Liquidambar formosana*	Hamamelidaceae	T	2	2
	Acacia confusa	Mimosaceae	T	10						
	Liquidambar formosana	Hamamelidaceae	T	1						
	Cinnamomum camphora	Lauraceae	T	2						
R89	*Acacia mangium*	Mimosaceae	T	16	49	*Acacia Mangium*	Mimosaceae	T	2	5
	Eucalyptus torelliana	Myrtaceae	T	22		*Eucalyptus torelliana*	Myrtaceae	T	1	
	Casuarina equisetifolia	Casuarinaceae	T	10		*Acacia confusa*	Mimosaceae	T	2	
	Acacia confusa	Mimosaceae	T	1						
TO	*Psychotria asiatica*	Rubiaceae	S/T	5	31	*Ardisia quinquegona*	Myrsinaceae	S	112	220
	Aporusa dioica	Euphorbiaceae	S	8		*Sarcosperma laurinum*	Sapotaceae	T	4	
	Schima superba	Theaceae	T	3		*Psychotria asiatica*	Rubiaceae	S/T	16	
	Sarcosperma laurinum	Sapotaceae	T	5		*Aporusa dioica*	Euphorbiaceae	S	20	
	Machilus sp.	Lauraceae	T	1		*Sterculia lanceolata*	Sterculiaceae	T	12	
	Canthium dicoccum	Rubiaceae	S/T	3		*Syzygium hancei*	Myrtaceae	T	4	
	Elaeocarpus chinensis	Elaeocarpaceae	T	1		*Helicia reticulata*	Proteaceae	T	4	
	Canarium album	Burseraceae	T	1		*Syzygium jambos*	Myrtaceae	T	12	
	Machilus chinensis	Lauraceae	T	2		*Uvaria grandiflora*	Liliaceae	S	20	
	Aquilaria sinensis	Thymelaeaceae	T	1		*Desmos chinensis*	Annonaceae	S	16	
	Sterculia lanceolata	Sterculiaceae	T	1						

T: Tree; S: Shrub

4.3.2 Stand structure

Forest stratification or layering refers to the different layers of plants in a forest. In a mature forest, several layers of vegetation are identified from the forest floor to the canopy. Each layer can offer a unique set of habitat features. In this section, the vertical strata of the four sites will be analyzed. Five layers are divided according to the vertical structure of the study sites: canopy layer (≥12m), low tree layer (3m-12m), shrub layer (≤3m), herb/fern layer, and litter layer.

Although stand structure varies among the sites, some generalizations can still be made. The restored sites are dominated by low trees within the height class of 3-12 m, decreasing in the order of R00 (40), R89 (31), R95 (21) (Table 4.2). In terms of the percentage of total, however, the importance of low trees tended to decrease with age of the plantations. While canopy trees are not found in the youngest plantation, they increase steadily from R95 (5) to R89 (12). A similar pattern is found for the percentages of herbs and ferns, which increase in the order of R00 (3.1%), R95 (6.5%) and R89 (10%). Many of these herbs and ferns, such as *Dicranopteris linearis*, *Miscanthus floridulus* and *Palhinhaea cernua*, are shade-intolerant species. Their increase along the vegetation gradient clearly suggests that the canopy of the restored sites has not closed up after 8-19 years of establishment (see Table 4.1). In contrast to this, the feng shui woods recorded only 6 canopy trees compared to 12 for R89. Yet the site records the highest number of shrubs (229) but no herbs and ferns, which is indicative of a shade environment. This re-affirms the previous argument that regeneration of species is most active in the feng shui woods, where seed germination is easier than in the restored sites. As a corollary, species originally planted in TTEBA are still developing, resulting in an increase of canopy trees with time. The low trees with a height of 3-12 m were also planted, given the short history of restoration and limited seed germination in the restored sites.

Table 4.2 Stand structure of the sites

Composition		R00	R95	R89	TO
Number of canopy trees (≥12m)		0	5	12	6
Number of low trees (3m-12m)		40	21	31	16
Number of shrubs (≤3m)		8	3	11	229
Herb/fern coverage (%)		3.1	6.5	10.0	0
Standing litter (tonne/ha)* (n=10)	Foliage	16.26±4.14^{a} (67.8)	13.36±3.87^{b} (84.2)	6.11±1.54^{c} (77.0)	3.14±0.89^{d} (51.1)
	Branches	7.39 ± 3.95^{a} (30.8)	2.23±1.16^{b} (14.0)	1.59 ± 0.56^{c} (20.1)	1.83±0.34^{bc} (29.9)
	Fruits	0.34 ±0.18^{b} (1.4)	0.28±0.13^{a} (1.8)	0.23±0.07^{a} (2.9)	1.17±0.23^{c} (19.0)
	Total	23.99^{a}	15.87^{b}	7.93^{c}	6.14^{c}

* Standing litter was estimated in November-December, 2006
Values in brackets represent % total of the component in a site.
Row means sharing the same superscript were not significantly different at $p < 0.05$ by Duncan's Multiple Range Test.

The litter layer in the study sites mainly compose of foliage, branches and fruits (and seeds). Standing litter biomass decreased in the order of R00 (23.99 t/ha) > R95 (15.87 t/ha) > R89 (7.93 t/ha) > TO (6.14 t/ha) (Table 4.2). Standing litter is governed by the amount of litter fall and its decomposition rate within the same period of time. It is thus clear that standing litter decreases with age of the forests. While it is a norm for litter production to increase with complexity development of a forest, the low standing litter biomass in TO probably reflects a faster decomposition rate in the site than in TTEBA as a result of more favorable microclimate. This issue will be further examined in Chapter 7.

Two other phenomena are noted in the standing litter biomass. There is a tendency for the percentage of foliage to decrease with age of the forests while the opposite is true for the importance of the fruit component. For instance, the percentages of foliage litter decrease in the order of R95 (84.2%), R89 (77.0%), R00 (67.8%) and TO (51.1%). This is expected because, other factors being equal, the foliage would break down faster than other woody components in the forest due to its higher nutrient contents, especially nitrogen and cation nutrients. Mature ecosystems are stable and self-sustainable so that regeneration is uninterrupted. For this to happen, large amounts of fruits are produced resulting in their growing importance in the standing litter with age of the forest. Thus, the percentages of fruits decrease significantly in the order of TO (19.0%) > R89 (2.9%) ≥ R95 (1.8%) > R00 (1.4%). There are two implications pertaining to this finding. First, the seed bank in the restored sites is considerably small. Second, the large amount of fruits in the feng shui forest supports a rich flora, especially the shrub layer.

Overall, standing litter is likely a reliable indicator of restoration progress in terms of its total biomass and changing importance of components. This indicator reflects not only species difference of the communities but also different microclimatic conditions of the sites.

4.3.3 Vegetation performance

Height, DBH, density and canopy closure are parameters commonly employed to describe vegetation performance (Ruiz-Jaen and Aide, 2005). In this section, I shall analyze the suitability of the above criteria, together with transection area and total crown area, for restoration assessment.

In section 4.3.2, it was found that the number of canopy and low tree species did not increase with age of the forests. Forest growth is indeed a dynamic process involving competition, regeneration and nutrient cycling, etc. Under the present experimental design, therefore, developmental status of the canopy and low tree layers is not a reliable indicator for the measurement of restoration progress. Are the static attributes of tree height, DBH, canopy closure, transaction area and crown area better indicators for this purpose?

Among the static attributes, DBH, transaction area, crown area and, to some extent, canopy closure increased progressively with age of the forests (Table 4.3). For example, average DBH increased in the ascending order of 3.82 cm (R00), 8.13 cm (R95), 8.26 cm (R89) and 11.52 cm (TO). There appeared to be two breaks in the increase, between R00 and R95 and between R89 and TO. In this connection, DBH in the feng shui forest fluctuates greatly between 2.4 cm and 99.68 cm (Appendix 4.1).

The average transection area of an individual tree at breast height increases in the order of R00 (14.7 cm^2), R95 (71.04 cm^2), R89 (77.80 cm^2), and TO (434.65 cm^2). It follows closely the pattern of DBH in all the sites. These values, however, do not necessarily represent the total biomass of the forests due to the different tree densities. While R95 yields a transection area similar to R89, for instance, its biomass should be smaller due to a lower tree density.

Table 4.3 Vegetation dynamics of the sites

Parameters		R00	R95	R89	TO
Average DBH (DBH>2cm) (cm)		3.82(2.05)	8.13(5.02)	8.26(5.60)	11.52(20.85)
Average transection area at breast height (cm^2)		14.7	71.04	77.80	434.65
Transection area at breast height per square meter (cm^2/m^2)		7.05	20.60	42.01	134.74
Average height (DBH>2cm) (m)		5.37(1.82)	6.55(2.62)	9.21(3.85)	6.31(5.87)
Density (Nos./ha)*	DBH>2cm	4,200	2,700	4,900	3,100
	DBH≤2cm	600	200	500	22,000
Total crown area (DBH>2cm) (m^2/ha)		10,869	15,411	19,731	25,374
Canopy closure (%)		62.8	49.3	53.8	76.4

* Values extrapolated from Table 4.1
Values in bracket represent standard deviations.

The index of transection area per square meter (TASM), on the other hand, reflects the influence of both tree density and growth performance of every individual tree. During successional development of a forest, tree density is negatively correlated with the transection area of a single tree. An increase in tree density will result in poor growth of the species due to fierce competition for nutrients, water and space. It is not possible to compare the biomass of plantations from the measurement of tree density alone. Instead, TASM is a better index for the assessment of forest productivity if the goal of rehabilitation is targeted at wood production. TASM of the sites increase progressively in the order of: R00 (7.05 cm^2/m^2), R95 (20.60 cm^2/m^2), R89 (42.01 cm^2/m^2), and TO (134.74 cm^2/m^2). There was a 6-fold increase in TASM between R00 and R89, which are 11 years apart. Indeed, the exotic species planted in TTEBA are fast growers.

The average height of the restored tree species (DBH>2cm) increases in the order of R00 (5.37 m), R95 (6.55 m), R89 (9.21 m). This is expected because the plantations had only been established for 8-19 years, and the restored species are still developing. Unlike the aforesaid attributes, average height of the established species in Tai Om is astonishingly lower than R95 and R89. While tree height is a good measurement of growth increment, it is not suitable for monitoring restoration progress. Instead, the standard deviations of tree height can be a better alternative, which decrease in the order of TO (5.87) > R89 (3.85) > R95 (2.62) > R00 (1.82) (Table 4.3). The feng shui forest at Tai Om has the best developed structure or stratifications among the sites, resulting in the highest standard deviations in plant heights. It is truly a good indicator of the different structure and life forms of the forest.

Total crown area is defined as the summation of all the tree (DBH>2cm) crown areas in the forest, which reflects the growth characteristics and biomass of the ecosystems. Total crown area increases progressively from R00 (10,869 m^2/ha) to R95 (15,411 m^2 /ha), R89 (19,731 m^2/ha), and TO (25,374 m^2/ha) (Table 4.3). Like DBH, transaction area, TASM and standard deviation of heights, total crown area is probably a good indicator of restoration progress.

Canopy closure, the proportion of sky hemisphere obscured by vegetation when viewed from a single point (Jennings *et al.*, 1999), has been employed as an ecological indicator for the assessment of forest floor microclimate and light conditions (Korhonen *et al.*, 2006). Tree density, vegetation stratification, and plant growth performance are deterministic factors of canopy closure. Due to the relatively homogeneous structure of

R00, its percentage canopy closure is higher than R95 and R89 although the highest value (76.4%) coincides with the well established feng shui woods in Tai Om. No wonder the understory layer in Tai Om is dominated by shade-tolerant species.

The trend of tree density differs greatly between the DBH>2cm class and the DBH≤2 cm class in all the study sites. The relationship between tree density and age of the forests is inconspicuous in the DBH>2cm class. For trees with DBH≤2 cm, however, tree density rises exponentially from 200-600 nos./ha in the restored sites to 22,000 nos./ha in the TO site (Table 4.3). As an important index of regeneration potential, sapling density is also a suitable indicator for the evaluation of restoration project. The higher the sapling density, the more complex will be the structure of the restored ecosystem.

4.3.4 Species diversity

The Shannon diversity index, H', is a good indicator of the structural complexity of an ecosystem. In the present study, grasses and herbs were excluded in the estimate because they are perennials and their presence in a particular site could be highly seasonal. An attempt is made here to separate the woody species of each site into two classes, DBH>2 cm and DBH≤2 cm, and estimate their Shannon indices separately. In so doing, more information can be gathered about the stand characteristics of the sites in relation to ecosystem development.

Regardless of the difference in DBH class, species diversity tends to increase with age of the forests in the order of R00 (0.51), R95 (1.14), R89 (1.20) and TO (1.97) (Table 4.4). This pattern is comparable to composition of the woody species in the sites (see Table 4.1). This is expected because only limited numbers of species were planted in TTEBA and invasion of native species was difficult. In contrast, the feng shui woods have established on the site for more than 350 years resulting in the highest biodiversity among the sites. Lastly, the diversity indices are wider apart between sites for the DBH>2cm class than the DBH≤2cm class. Thus, the Shannon diversity index is a good indicator for the evaluation of ecosystem development.

Table 4.4 Shannon indices (*H'*) of woody species

Sites	R00	R95	R89	TO
H' (DBH>2cm)	0.45	1.02	1.13	2.12
H' (DBH≤2cm)	0.69	0	1.05	1.7
Total	0.51	1.14	1.2	1.97

4.3.5 Species invasion

There are diverse species composition and special phenomena in different stages of forest succession. Species invasion and the implications of dead trees are discussed below to further elucidate the process of ecosystem development in TTEBA.

Invasive indigenous or exotic plant species can alter the properties of plant communities, including species diversity, primary productivity, interactions between species, stability, and rates or pathways of successional recovery of a community following disturbance (Walker and Stanley, 1997).

Different species have invaded the uneven-aged plantations in Tai Tong East Borrow Area (Table 4.5). *Miscanthus floridulus* and *Dicranopteris linearis* are species that have invaded all the three restored sites aged 8-19 years. In addition to these species, vine and climber are also found in R95 (*Embelia laeta*) and R89 (*Psychotria serpens).* The cellulose-decomposing fungus, *Ganoderma applanatum*, is commonly found on dead tree trunks in R89 but rarely in R00 and R95. These invaders, including fern, grass, vine/climber and fungus, are native species and their numbers tend to increase with age of the plantations. Unlike the notorious exotic weed *Mikania micrantha*, these invaders can play an important role in recovering ecosystem structure and functions. They can fix nutrients, deepen soil layer, improve soil texture, balance soil water, increase biodiversity, attract soil fauna and birds, and accelerate native species regeneration (Williams, 1997). Therefore, native species invasion may be the impassable and functional stage during the trajectory of ecological rehabilitation.

Table 4.5 Species invasion in TTEBA

Invader	R00	R95	R89
Herb/fern	*Miscanthus floridulus*	*Miscanthus floridulus*	*Miscanthus floridulus*
	Dicranopteris linearis	*Dicranopteris linearis*	*Dicranopteris linearis*
		Dianella ensifolia	*Gahnia tristis Nees*
Vine/climber	*Nil*	*Embelia laeta*	*Psychotria serpens*
Fungi	*Nil*	*Nil*	*Ganoderma applanatum*

Dead wood might seem expendable in a forest or may even be regarded as unsightly; however, it plays an important role in supporting wildlife and assisting ecological processes. Dead trees provide habitat to numerous animal species, such as birds, mammals, amphibians, reptiles, and invertebrates; and play an important role in nutrient cycling. Additionally, fungi and mushrooms flourish on and around logs, breaking down the organic matter to release important nutrients back into the forest ecosystem (Santiago and Rodewald, 2004).

Another useful biological hint from dead trees is the cause of death in the forest.

Except for the reason of insect or disease attack, dead tree implies the intensive competition among the species for space, light, water and nutrients. Weaker trees are outcompeted during the process of stem exclusion, resulting in die-back and declining tree density.

The number of dead trees varies from 5 in R00 to 1 in R95 and 8 in R89. No attempt is made here to account for their death although high tree density is a possible cause. Indeed, tree density is higher for R00 (4,200 ha^{-1}) and R89 (4,900 ha^{-1}) than R95 (2,700 ha^{-1}). Dead trees and fallen logs in a forest are proof of ecological succession, yet their use as indicators in monitoring restoration progress is difficult to quantify.

4.4 Conclusions

The three woodland plantations have been restored on soil destruction sites dominated by decomposed granite for 8-19 years, while the reference site is an established feng shui woods that has established in Tai Om village for at least 300 years. From findings obtained in this study, the following conclusions can be summarized.

1. The vegetation on the three restored sites possesses the following characteristics:

   a) The overstorey species are dominated by the planted exotics of *Acacia confusa, A. mangium, Casuarina equisetifolia* and *Eucalyptus torelliana*; while planted native species are few and confined to *Cinnamomum camphora* and *Liquidambar formosana*. The number of woody species increased with age of the plantations, as was species diversity.

   b) Tree density decreased in the order of R89 > R00 >R95. The percentage of trees with DBH>2 cm was higher than those with DBH≤2cm in all the restored sites.

c) The restored sites, regardless of age, lag behind the feng shui woods in terms of biodiversity (species number and life forms) and regeneration potential.

d) The restored sites are dominated by low trees of 3-12 m in height, yet their importance decreased with age of the plantations. Hence, there is a steady increase of canopy trees >12m with time.

e) Unlike the feng shui woods, shrubs are absent in the restored sites, and the major invading species, though few, include grass, fern, vine, climber and cellulose-decomposing fungus.

f) The canopy of the exotic woodlands has not closed yet after 8-19 years of growth.

g) Standing litter on the forest floor decreased in the order of R00 (23.99 t/ha) > R95 (15.87 t/ha) > R89 (7.93 t/ha) > TO (6.14 t/ha). The importance of foliage declines with age of the forests while the reverse is true for fruits and seeds component, suggesting that the seed bank is limited in the restored sites.

h) The DBH, transaction areas, crown area and percent canopy closure seemed to increase with age of the woodland plantations.

i) The restored sites are invaded by a limited number of native species, which are, however, confined to grass (*Miscanthus floridulus*), fern (*Dicranopteris linearis*), vine (*Embelia laeta*), climber (*Psychotria serpens*) and the cellulose-decomposing fungus (*Ganoderma applanatum*). The latter three are more commonly found in R95 and R89 than R00.

2. Ecological succession is happening in all the restored sites and the evidence of proof include: stem exclusion of the planted exotic species (declining density); steady increase of tall trees (>12m); invasion of grass, fern, vine and climber though few; increase in species diversity; increase of percent canopy closure; decreasing total standing litter biomass but a simultaneous increase in fruit/seed component; increase of DBH, transaction areas and crown area, etc.

3. In terms of vegetation dynamics, useful indicators for the evaluation of restoration progress include: species diversity, standing litter biomass and its changing compositions, transaction areas, standard deviations of tree height, total crown area, saplings density and, to some extent, species invasion.

4. The established feng shui wood is a suitable reference site in view of its high biodiversity and endemic property. Since the goal of ecological restoration has never been clearly defined in Hong Kong, we believe assessment should then be bench marked against the best developed forest ecosystem in this part of the world.

CHAPTER 5

RECOVERY OF SOIL PRODUCTIVITY AND NITROGEN MINERALIZATION

5.1 Introduction

Forest soils serve multiple production and environmental functions, and they are highly manipulated by forest practices. Soil rehabilitation is an important component of forest management strategy that aims at reducing the effect of soil degradation on forest productivity (Plotnikoff *et al.*, 2002). Soil qualities, especially the chemical and biological properties, are not just monitored and studied routinely under the priority of soil fertility improvement (Johnston *et al.*, 2002). In addition to the improvement of plant productivity, there is recently a growing need to determine if management practices can improve soil quality and the sustainability of ecosystem functions. The effects of soil texture, water-holding capacity and nutrient bioavailability on plant growth have been well documented in the literature. On the other hand, there are relatively few studies on the assessment of soil productivity recovery in Hong Kong and South China, not to mention the identification of soil indicators suitable for the assessment of restoration progress. It is indeed inappropriate to ignore soil biology when dealing with ecological rehabilitation.

The concept of soil quality (SQ) includes both the soil properties and processes, which affect directly the normal functions of a healthy ecosystem. Soil quality, like site quality or forest productivity, is a value-based concept related to the objectives of ecosystem management. Hence, it is management- and ecosystem-dependent. It is defined as "the capacity of a soil to function within ecosystem boundaries to sustain biological productivity, maintain environmental quality and promote plant and animal health" (Soil Science Society of America, 1997). Soil quality is a combination of the physical, chemical and biological properties that contribute to soil functions. Indicators of soil quality should be responsive to management practices, integrated ecosystem processes, and be components of existing, accessible data bases. These indicators must be quantified to document the improvement, maintenance or degradation of soil quality (Larson and Pierce, 1994).

One main objective of the present research is to assess the resilience of a soil destruction site, where the decomposed granite suffers from a decline in physical, chemical and biological properties. Soil is an abiotic component of an ecosystem; it affects the survival, growth and productivity of vegetation. In return, vegetation cover can influence soil properties due to differences in litter quality, root activity, canopy interception of atmospheric deposition, nutrient uptake and growth (Alban, 1982; Miles, 1985; Binkley, 1995; Hagen-Thorn *et al.*, 2004). Successful forest restoration should inter alia lead to a recovery of soil productivity or improvements in physical, chemical and biological properties to its pre-disturbed level.

Investigating the relationship between soil properties and nutrient cycling requires multiple measurements and extended laboratory work, which is expensive and time consuming (Herrick, 2000). Degraded soils are slower to recover in comparison to vegetation structure and diversity (Chambers *et al.*, 1994; Kindscher and Tieszen, 1998; Morgan and Short, 2002). For example, the recovery of soils after mining needs to be monitored continuously for more than 15 years in order to obtain any reliable results. Soil characteristics and properties related to ecological processes are not measured as frequently as the measurement of vegetation structure and diversity (Ruiz-Jaen and Aide, 2005). Soil quality indicators are in fact the physical, chemical and biological properties, processes, and characteristics that can be measured to monitor changes in soil (Muckel and Mausbach, 1996). Although there is no single soil attribute or property that can be used to estimate soil quality, a group of soil attributes or properties that are sensitive and reliable for obtaining changes in soil physical, chemical and biological properties can be used to estimate soil quality and assess restoration progress. Quantifying these variables through long-term monitoring can improve our understanding of the effects land management practices and different disturbances have on the soil.

In Chapter 4, we have compared the vegetation structure and functions of the restored sites with a well developed feng shui woods. To understand better the soil-plant relationship on the restored sites in TTEBA, the recovery of soil properties and productivity as well as nitrogen mineralization would be investigated in this chapter. Ecological indicators that can be employed for the assessment of restoration progress have already been reviewed in Chapter 2. Below are a review on the general properties of soil and their effect on plant growth.

5.1.1 Plants and soil

The relationship between vegetation and soil is well documented in the literature (e.g., Brady, 1984; Trudgill, 1988; Wild, 1988; Foth, 1990; Miller and Donahue, 1990;

Wild, 1993). Soil provides habitat and supplies nutrients and water for plants to grow. Soil physical properties, such as texture, bulk density and porosity, affect the efficiency of nutrient absorption by plants. As a primary producer, plants transform carbon to organic matter through photosynthesis, and return the nutrients back to the soil via litter fall and root sloughing (Figure 5.1). The soil physical and chemical properties can then be improved by plant-root activities and decomposition of the litter materials (Dickinson, 1974).

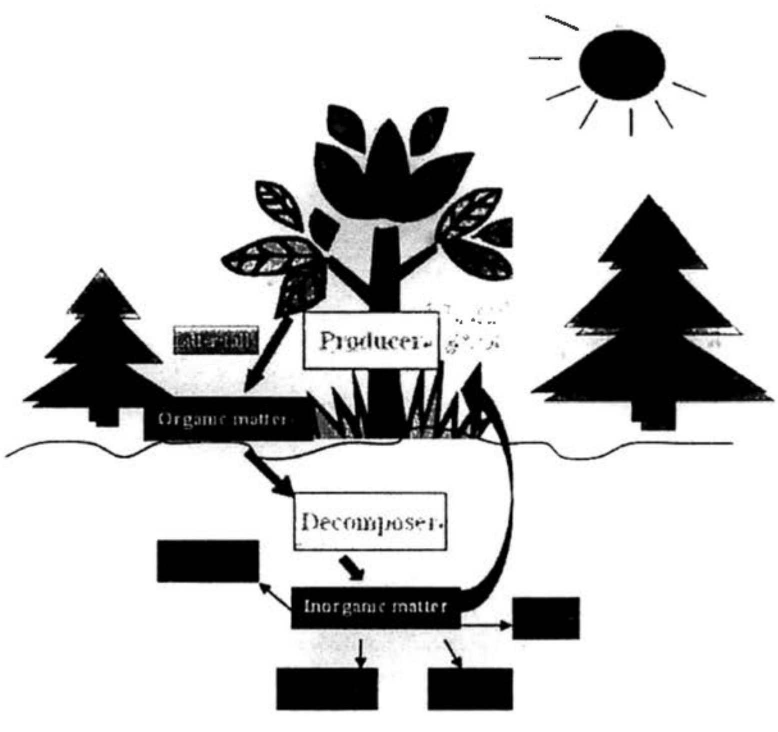


Figure 5.1 Nutrient cycling in the soil-plant subsystem

5.1.2 Soil physical properties

Soil physical properties refer to the air-water relationship, including infiltration and water-holding capacity, bulk density, aggregate stability and texture etc (Omuto, 2008). Degradation of soil physical properties affects soil available water content, permeability, drainage and resistance (Young, 1989), and ultimately soil productivity and environmental regulatory capacity (Doran and Parkin, 1994; Lal, 1993).

Hydrologic processes are not only affected by vegetation; bare soil surface has very different properties and induce different types of hydrologic behavior. Soil texture provides a preliminary estimate of soil permeability and water-holding capacity of the entire soil profile. The hydraulic conductivity of soils increases by one order of magnitude for each step in the texture series from clay to loam and to sand (Loomis and Connor, 1992).

Tree planting can modify soil bulk density. The roots of growing trees penetrate into soil and break up the agglomerate into pieces. The soil bulk density in Loess Plateau, China, has decreased after tree planting (Li *et al.*, 1993). Similarly, the bulk density of upland soils in Thailand increases from 1.0-1.2 gm/cc to 1.2-1.3 gm/cc immediately after deforestation (Lal, 1987). The key properties directly related to soil texture include porosity, permeability, infiltration, shrink-swell potential, water-holding capacity and erodibility.

Water-holding capacity is one of the most significant soil physical properties (Townend *et al.*, 2001). Field capacity (FC) is defined as the soil water content when drainage of gravitational water is negligible and moisture content is relatively stable (Kramer, 1969). It is recognized as the upper limit of water that is available to plants and is estimated at a soil matric potential of 0.33 bar (Kirkham, 2005). On the contrary, permanent wilting point (PWP) is regarded as the amount of water held by soil so tightly that plant roots cannot absorb and the plant may wilt (Kirkham, 2005). Wilting point is determined at soil matric potential of 15 bars (ADAS, 1982; Kirkham, 2005).

Available water capacity (AWC) or plant available water is defined as water available to plant between field capacity and permanent wilting point (ADAS, 1982; Kirkham, 2005).

In this study, soil texture, bulk density and water-holding capability are considered as potential indicators for measuring the recovery of soil productivity with time. In addition, soil temperature and moisture with a potential in affecting soil microbial activities are also investigated in this study.

5.1.3 Soil chemical properties

Forest degradation by natural and anthropogenic forces result in the deterioration of soil chemical properties, including extreme acidity, nutrient deficiency, lowered soil organic matter content, and modified soil organism population (Johnson and Bradshaw, 1979; Logan, 1992; Moffat and Buckley, 1995). The most commonly investigated soil chemical properties are organic matter (soil carbon), cation exchange capacity, and the nutrient contents of nitrogen (N), potassium (K) and phosphorus (P), etc.

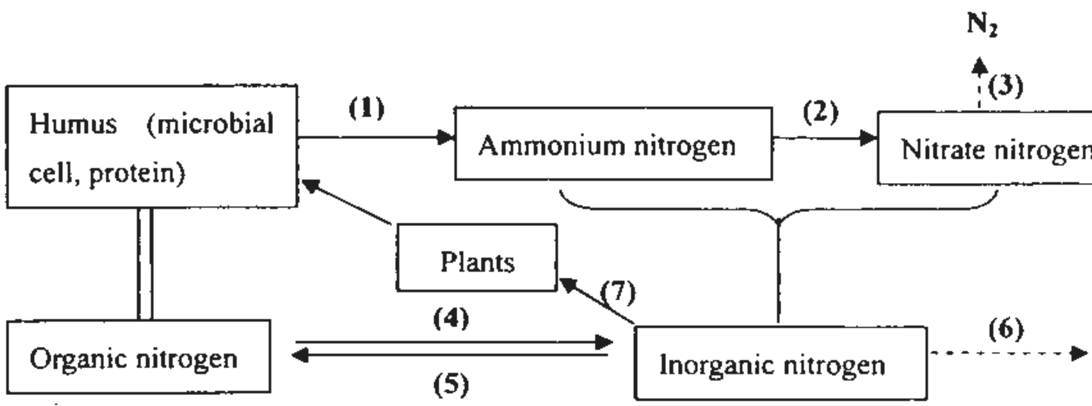
Organic matter and nitrogen are especially useful in soil studies, because they serve as indicators of energy flow (carbon) and nutrient cycle (nitrogen) (Craft, 2001). Nitrogen is a limiting factor to vegetation growth on derelict lands (Bradshaw, 1999). Ecosystem development on restored sites depends on the accumulation of plant biomass and nitrogen in the soil. Soil organic matter is another factor which supports secondary production by contributing detritus to heterotrophic organisms and labile carbon to fuel microbial processes, such as denitrification and nitrogen fixation in detritus-based ecosystems (Craft *et al.*, 1999; Thompson *et al.*, 1995; Piehler *et al.*, 1998). Soil organic matter is the fuel for running a soil's engine (Fisher and Binkley, 2000). Thus, soil organic matter plays a very important role in nutrient cycling, maintenance of soil porosity, hydraulic conductivity, bulk density, and soil detoxification processes.

Carbon-nitrogen ratio is an important index, which expresses the critical nitrogen levels in soil. The C:N ratio of soil organic matter is closely related to the processes of nitrogen immobilization and mineralization during organic matter decomposition by micro-organisms. The value decreases as decomposition proceeds; hence it is negatively correlated with net nitrogen mineralization (Yamakura and Sahunalu, 1990). The C:N ratio of 20:1 to 30:1 is the dividing line between nitrogen immobilization and mineralization. Ratios greater than 30:1 result in nitrogen immobilization while smaller than 20:1 favor mineralization (Alexander, 1977). At the nutrient level, the dividing line of nitrogen in the litter materials is 1.5%, below which the soil will be dominated by general purpose bacteria. Under this circumstance, C:N ratio of the soil will widen to 30:1 or above.

Phosphorous is second only to nitrogen in terms of quantity required by both plants and microorganisms. It plays an important role in the accumulation and release of energy during cellular metabolism. Therefore, organic carbon, nitrogen and phosphorous of the rehabilitated soils are investigated as potential indicators for the evaluation of restoration progress.

5.1.4 Nitrogen mineralization

Nutrient cycling is a key ecosystem function, in which soil nitrogen mineralization is a vital component. The demand of nitrogen by plants is highest among the macro-nutrients. Nitrogen is absorbed by plants almost entirely in the inorganic forms of ammonium and nitrate. In unfertilized system, nitrogen contained in the organic matter is present in the organic, complex and unavailable form. It has to be converted to inorganic, simple and available forms for plants to absorb. This conversion involves two processes: ammonification and nitrification. Ammonification is the process of converting organic N to ammonium N (NH_4-N) by heterotrophic micro-organisms, while nitrification is the subsequent oxidation of ammonium to nitrate (NO_3-N) by autotrophic bacteria including *Nitrosomonas* and *Nitrobacter* (Alexander, 1977). Ammonium nitrogen and nitrate nitrogen are known as mineral nitrogen that plants can absorb for the synthesis of amino acid. Nitrogen mineralization is, therefore, the sum of ammonification and nitrification (Figure 5.2).



(1) Ammonification (2) Nitrification (3) Denitrification
(4) Mineralization (5) Immobilization (6) Leaching (7) Uptaken by plants

Figure 5.2 Nitrogen transformation in soil

Mineralization can be affected by a wide range of factors, including soil pH, moisture, temperature, organic matter, texture, aeration and substrate availability (Frazer, *et al.*, 1990). Daniel *et al.* (2006) found that if water is sufficient, increased soil temperature after biomass removal will increase soil microbial activity and net N mineralization. Nitrification as a component of N mineralization is highest in the 1- to 3-year-old stand but lowest in the 80- to 100-year-old stand (Idol *et al.*, 2003).

Monitoring nutrient cycling is one of the measures recommended for the assessment of soil quality and restoration progress. The main N pools in forest ecosystem consist of plant biomass, forest floor and belowground litter, soil organic matter, soil microbial biomass, and inorganic N. The major processes involved in the movement of soil N include plant uptake, litter decomposition and humification, mineralization and immobilization, leaching and denitrification, atmospheric deposition, N_2-fixation, and plant uptake. N mineralization would be investigated in this chapter for several reasons. First, nitrogen is a limiting factor to plant growth in the subtropics, especially on soil destruction sites. N mineralization is, therefore, a measure of the element's fluxes and availability in the restored soils. Second, N mineralization depends on *inter alia* litter input, soil conditions and environmental conditions of the ecosystem. The intensity of N mineralization will likely reflect the extent of soil recovery in the restored sites. Third, N mineralization can be estimated from *in situ* incubation of the soil and the method is relatively straightforward.

Besides N mineralization, soil physical and chemical properties will also be investigated and compared among the uneven-aged plantations. The changes in soil properties are a direct measurement of the recovery of the impoverished soil 8-19 years after ecological rehabilitation. Through comprehensive analysis of the soil recovery data, useful indicators could be identified for the effective evaluation of restoration progress. Nevertheless, results obtained from this experiment can provide answers to the following specific questions.

1. After being rehabilitated for 8-19 years, has soil productivity in Tai Tong Borrow

Area been recovered?

2. Among the different soil parameters investigated in this experiment, which are suitable indicators for the assessment of restoration progress and under what conditions?

3. What is the successional status of the restored sites in terms of the recovery of soil productivity?

5.2 Materials and Methods

The present experiment consists of two parts: amelioration of the degraded soils by woodland plantations and soil N mineralization. This section describes the sampling of soils in the field, pre-treatment and procedures of laboratory analysis. N mineralization will be assayed by *in situ* incubation of the soils and periodic retrieval of paired sample cores. Procedures for statistical analysis of the data will also be addressed.

5.2.1 Experimental design and soil sampling

Soil sampling and pre-treatment

The present experiment was carried out on the same sites as the vegetation study; hence details of the sites will not be repeated here. One to two transects is demarcated in each of the sites for the sampling of soils, depending on site configurations. Fifteen sampling points were chosen systematically along the transect at 3-m intervals, and effort was made to avoid any edge effects. Soil samples were collected from each sampling point at three depths: 0-10 cm, 10-20 cm and 20-30 cm. We believe most of the soil activities will take place within the top 30 cm layer of the soil, including rooting, microbial activities, litter decomposition and humification. Because of this, any changes arising from ecological restoration can be detected. The soil samples were collected by an augar to the required depth, from November to December, 2006. About 0.5 kg of soil was collected from each depth.

The soil samples were returned to the laboratory, air dried at room temperature, ground, and passed through 2 mm and 0.25 mm sieves respectively for the analysis of reaction pH, texture, water-holding capacity, organic carbon (OC), total Kjeldahl nitrogen (TKN), and total phosphorous (TP).

Soil samples were also collected for the determination of bulk density in February, 2007. Ten samples were collected from each site systematically along the previous transect(s) at 3-m intervals. Aluminum cores measuring 10 cm in length and 5 cm in diameter were inserted into the soil for this purpose.

N mineralization

In situ soil-core technique described by Raison *et al.* (1987) was modified and employed to estimate N mineralization in the soil. This experiment was conducted in the field from November, 2007 to January, 2008, and the incubation tubes were retrieved at biweekly intervals. It involves the collection of ordinary soil samples at T_0, and paired samples, one open and the other covered, at T_1 for the analysis of mineral nitrogen (Figure 5.3). On each occasion, the tubes were incubated for 2 weeks before being retrieved for the determination of mineral nitrogen. This incubation technique can provide a wide array of information, including ammonification, nitrification, net N mineralization, leaching, immobilization and uptake of N by plants. As the primary objective of the present experiment is to ascertain the suitability of N mineralization process as a possible indicator of restoration progress, the study was only carried out in the dry season. Hence, N uptake is not a focus of the study.

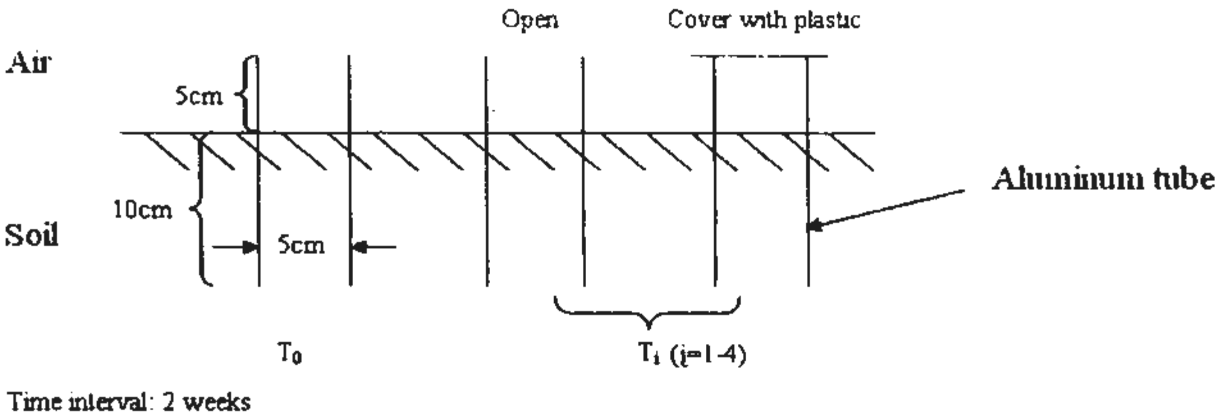


Figure 5.3 ***In situ* soil-core incubation method**

The incubation tubes used in this experiment are the same as those employed for the determination of soil bulk density. Ten quadrats, each measuring 0.5 m x 0.5 m, were randomly selected from each site (Figure 5.4). *In situ* incubation was carried out continuously on these quadrats during the study period. A total of 9 incubation tubes were inserted into each quadrat at T_0, to a depth of 10 cm. One of the tubes was immediately retrieved, which represents the ordinary soil sample. Mineral N contained in this ordinary sample represents reserve of the nutrient before start of the experiment (T_0). The eight tubes left in each quadrat were equally divided into two groups: open and covered. After two weeks of incubation (T_1), one each open and covered tube were retrieved from the quadrat. This process was repeated three more times (T_2, T_3 and T_4), again at biweekly intervals (Figure 5.4). There were altogether 360 samples for the four sites.

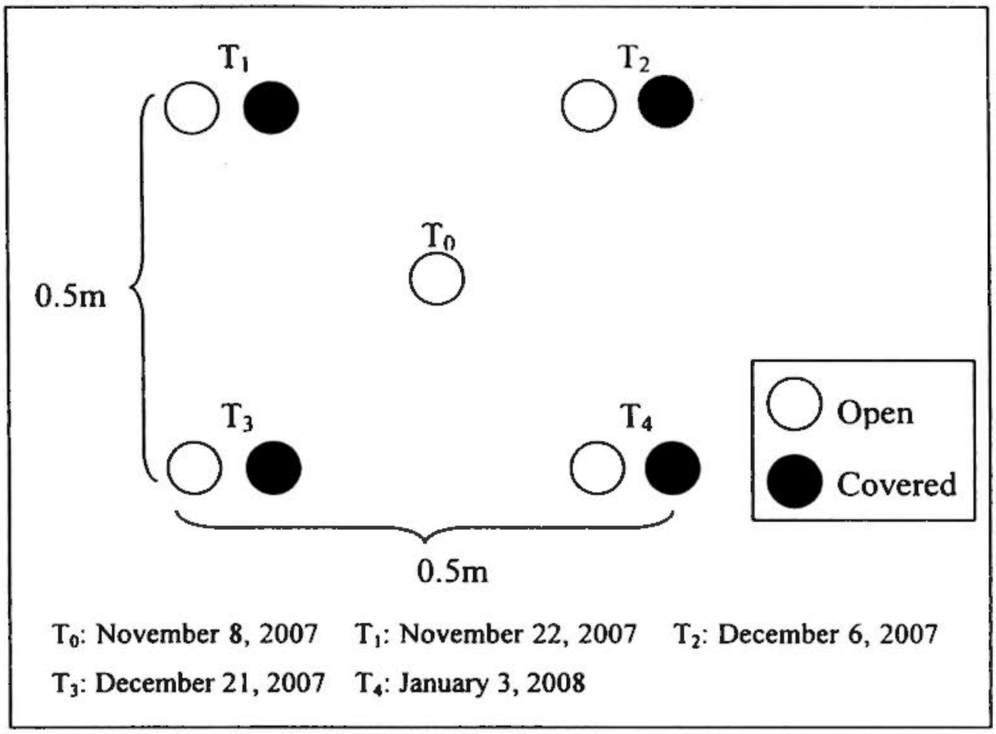


Figure 5.4 Layout of one set of soil tubes in a quadrat

N mineralization involves two processes, namely ammonification and nitrification. Net N mineralization and the leaching loss of mineral nitrogen were estimated from the two formulae below (Yau, 1996; Marafa, 1997).

- Net N mineralization ---(1)

$\Delta NH_4\text{-}N$ = net ammonification during incubation

$= NH_4\text{-}N(c_i) - NH_4\text{-}N(T_0)$

$\Delta NO_3\text{-}N$ = net nitrification during incubation

$= NO_3\text{-}N(c_i) - NO_3\text{-}N(T_0)$

Nmin = net mineralization during incubation

$= \Delta NH_4\text{-}N + \Delta NO_3\text{-}N$

where: NH_4-N(T_0) and NO_3-N(T_0) are the NH_4-N and NO_3-N content of bulk soil at the start (Time 0) of the incubation period. NH_4-N(c_i) and NO_3-N(c_i) are the NH_4-N and NO_3-N content of the covered (c*i*) at the time *i* of the incubation period.

- Leaching of N --- (2)

NH_4-N_{leach} = leaching of ammonium during incubation

= NH_4-N(c*i*) – NH_4-N(o*i*)

NO_3-N_{leach} = leaching of nitrate during incubation

= NO_3-N(c*i*) – NO_3-N(o*i*)

N_{leach} = leaching of mineral nitrogen during incubation

= NH_4-N_{leach} + NO_3-N_{leach}

where: NH_4-N(c*i*) and NO_3-N(c*i*) are the NH_4-N and NO_3-N content of the covered (c*i*) soil at the time *i* of the incubation period. NH_4-N(o*i*) and NO_3-N(o*i*) are the NH_4-N and NO_3-N content of the open (o*i*) soil at the time *i* of the incubation period.

5.2.2 Laboratory procedures

Texture

Soil texture was determined by the Bouyoucos hydrometer method, which measures the decrease in density of the suspension as particles settle. Fifty grams of <2 mm air dry soil was mixed with 5% Calgon solution (sodium hexametaphosphate). The mixture was stirred at high speed for 10 minutes and made up with water to 1,000 ml. Hydrometer readings were taken at 4 minutes 48 seconds to estimate silt and clay fractions, and at five hours to estimate clay fraction (Grimshaw, 1989). The results were expressed as percentages of sand, silt and clay. Percentages of sand (2.0-0.02 mm), slit (0.02-0.002 mm) and clay (<0.002 mm) were calculated after moisture correction while textural class was determined against the textural triangle of the International Society of Soil Science.

Bulk density

The aluminum cores containing the undisturbed soil samples were returned to the laboratory, dried to constant weight at 105°C, and weighed. Bulk density is the oven dry mass divided by the field volume of the sample (Landon 1991).

$$Bulkdensity(\mathrm{Mg}/m^3) = \frac{Soil \quad oven \quad dry \quad weight(\ \mathrm{Mg})}{Soil \quad core \quad volume(\ m^3)}$$

Water-holding capacity

Water-holding capacity of the soils was determined by the pressure plate extraction method at 0.3 and 15 bars tension (Klute, 1986). Moisture held at field capacity (FC) was obtained at 0.3 bar and the permanent wilting point (PWP) at 15 bar. The available water content (AWC) was estimated as the difference in moisture content between 0.3 bar and 15 bar. About 25g of the <2 mm air dry soil was placed in a soil sample retaining ring (about 1 cm high) on the ceramic plate. The samples were saturated with water overnight, put into the pressure plate extractor and separately subjected to the above suctions for 24 hours. Moisture content (weight percentage) was then determined gravimetrically after drying the samples at 105°C for 48 hours.

Reaction pH

Fifteen grams of <2 mm air dry soil was mixed with 37.5 ml of reverse osmosis water at a soil to water ratio of 1:2.5 (w/v). The slurry was shaken for 10 minutes and left to stand for another 30 minutes. A glass electrode coupled with the ORION Expandable ion Analyzer EA 940 was employed to detect pH in the supernatant liquid.

Total Kjeldahl Nitrogen (TKN)

The Kjeldahl oxidation method was used to determine total nitrogen in the soils. One gram of <0.25 mm air dry soil was digested at 370°C with 12 ml of concentrated sulfuric acid (H_2SO_4) and one Kjeltab catalyst tablet containing 3.5g K_2SO_4 and 0.4g $CuSO_4$ (Anderson and Ingram, 1993). The digest was then steam distilled by the Tecator Kjeltab 1026 automatic distillation unit where free ammonia was liberated in the presence of excess alkali. A receiver containing 25 ml boric acid was used to collect the distillate, which was then back titrated with 0.01*M* hydrochloric acid (HCl). TKN was calculated after moisture correction of the soil.

Soil organic carbon (SOC)

Soil organic carbon was determined by the Walkley-Black partial oxidation method (Walkley and Black, 1934). Ten ml 5% potassium dichromate solution and 20 ml concentrated sulfuric acid were used to oxidize the organic carbon contained in 1g of <0.25 mm air dry soil. The mixture was back titrated with 0.5*M* ferrous sulfate heptahydrate, using o-Phenanthroline-ferrous complex as indicator.

Carbon: nitrogen ratio was calculated by dividing organic carbon content by total Kjeldahl nitrogen content of the sample.

Total phosphorous (TP)

Mixed acid digestion was employed to determine total phosphorus (TP). Around 0.5 gram of <0.25 mm air dry soil was digested with 15 ml of mixed acids

(HNO_3:$HClO_4$:H_2SO_4 at the ratio of 10:2:1) at 240°C to obtain a clear to whitish solution. It was then filtered and diluted to 100 ml with deionized water. Total P was determined by Flow Injection Analysis 5000 Analyzer and expressed on an oven dry weight basis.

Mineral nitrogen (NH_4-N and NO_3-N)

The fresh soil samples collected from *in situ* incubation were immediately returned to the laboratory, passed through a 0.5 mm sieve, and stored in the refrigerator at 4±1°C. Ten grams of the <0.5 mm fresh soil were extracted by 100 ml 1.9 *M* KCl. The mixture was shaken for 30 minutes and filtered through Whatman No.5 filter paper. The filtrate was analyzed for mineral N by Flow Injection Analysis 5000 Analyzer. For the determination of NH_4-N, the filtrate was injected into a carrier stream of 1.9*M* KCl and mixed with 0.1*M* NaOH. NH_4-N was then detected colorimetrically at wavelength of 590 nm. For NO_3-N analysis, the filtrate was injected into a carrier stream of 5*M* NH_4Cl in 1.9*M* KCl solution. The mixture then passed through a cadmium reductor, where nitrate was reduced to nitrite. On the addition of acidic sulphanilamide and N-(1-Naphtyl)-ethylenediamine dihydrochloride from the merging streams, the resultant purple azo dye was detected colorimetrically at wavelength of 570 nm.

5.2.3 Soil temperature and moisture

Soil temperature and moisture at 10 cm depth were monitored in this study in an attempt to assess restoration progress in general and N mineralization in particular. It involved continuous monitoring of soil moisture (percent volume) by SMR110-5 and soil temperature by MicroTemp from April 13, 2007 to April 23, 2008 at each site. Automatic recording of the data was accomplished at half hourly intervals.

A hole was dug to a depth of 15 cm into the soil at roughly the centre of the study plot. It was then saturated with water before inserting the ECHO moisture probe at 10 cm depth (Figure 5.5). Soil was then backfilled to cover the moisture probe and firmed to ensure proper contact with the probe. The data logger was put inside a waterproof box, which was buried in the soil together with the Micro Temp data logger for soil temperature at 10 cm depth.

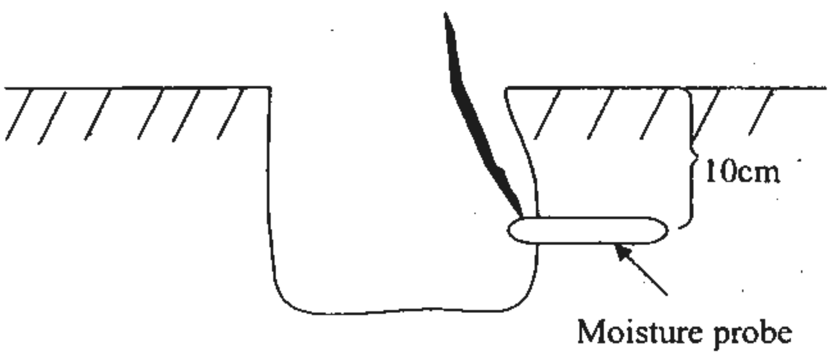


Figure 5.5 The installation of soil moisture data logger (SMR110-5)

5.2.4 Statistical analysis

All the raw data were entered into the spreadsheet software MS Excel for tabulation and calculation. The results were then transferred into the SPSS (for Windows) statistical package for numerical processing. The mean values and standard deviations were calculated. The differences between sites were tested by using Duncan's multiple range test, which is often employed to test the differences between group means. The significance level is set at $p<0.05$, unless otherwise specified. The differences in soil N mineralization between the open tubes and covered tubes were tested by the paired-sample T- test at 95% confidence level. Pearson correlation was performed to examine how soil temperature is related to ambient temperature inside a particular site, and ambient temperature at the nearby Shek Kong weather station. Pearson correlation analyses were also performed to investigate the relationship between N mineralization and different environmental factors.

5.3 Results and discussions

5.3.1 Amelioration of soil physical properties

Texture, water-holding capacity and bulk density of the restored soils are discussed in this section, with special emphasis on the effect of vegetation development. The results are also compared with that of the undisturbed soil at Tai Om.

Soil texture

Texturally, the soils at TTEBA belong to sandy loam or loamy sand, which are totally different from the clay loam to clay in Tai Om (Table 5.1; Appendix 1). The restored soils are coarse, containing a high percentage of sand (54.14-78.90%) and a low percentage of clay (5.28-17.46%). This composition is comparable to the original decomposed granite, which has a higher sand content (70.82%) than clay content (11.95%). Vegetation establishment on the sites, therefore, has no effects on soil texture. This is expected because according to the most conservative estimate, it would take 1 to 3 million years for tropical soils to form from weathering. These sandy soils, though good in aeration and water transmission, are relatively poor in water-holding capacity and organic matter content. Indeed, coarse sandy soils often support low-productivity stands (Fisher and Binkley, 2000). In contrast to this, the established soils in Tai Om are dominated by finer particles that have a greater affinity to water. Uninterrupted vegetation establishment can perhaps moderate the adverse effect of texture on soil properties. Unfortunately, the restored vegetation is only 8 to 19 years old and subsequent amelioration of soil organic matter is probably limited by the short history of restoration. It is perhaps worthwhile to retain the top soil for rehabilitation planting in borrow area, which contains some organic matter, nutrients and seed propagules. In terms of texture improvement, however, the top soil cannot meet this requirement because erosion in the past had already removed a large proportion of the finer particles.

Table 5.1 Soil texture of the sites

Site	Layer	Clay %	Silt %	Sand %	Textural class
DG		11.95	17.23	70.82	sandy loam
R00	0-10cm	12.15	25.33	62.51	sandy loam
	10-20cm	17.46	28.4	54.14	sandy loam
	20-30cm	15.41	26.08	58.51	sandy loam
R95	0-10cm	5.28	32.05	62.67	sandy loam
	10-20cm	6.18	14.92	78.9	loamy sand
	20-30cm	8.33	19.36	72.32	sandy loam
R89	0-10cm	8.29	19.04	72.67	sandy loam
	10-20cm	7.29	22.14	70.56	sandy loam
	20-30cm	6.22	21.21	72.57	sandy loam
TO	0-10cm	38.64	24.13	37.23	clay loam
	10-20cm	42.45	20.3	37.25	clay
	20-30cm	39.65	19.75	40.6	clay loam

DG (Decomposed granite): A typical growth substrate on degraded lands such as badlands, disused quarries and borrow areas in Hong Kong (Lam, 2008)

Water-holding capacity

Field capacity water (FC) tended to increase with age of the forest ecosystems, in the ascending order of R00 (15.60-16.30%), R95 (14.82-15.38%), R89 (17.24-18.27%), and TO (20.10-21.56%). This is expected because the soils in Tai Om contain much higher clay and organic carbon contents than the restored sites (see Table 5.1 and Figure 5.8). Having said this, the clay content of R00 is also higher than R95 and R89 and yet its FC is comparable to R95 but significantly lower than R89. This clearly shows that clay content is not the only factor affecting the adsorption of field capacity water. Instead, the accumulation of organic carbon with time, as in R89, is a more important factor than clay in elevating water held at 0.3 bar. In addition to the improvement of FC water 19 years after planting, intra-layer differences are not significantly different in the study sites.

Table 5.2 Soil water-holding capacity of the sites

Site	Layer	Water-holding capacity (weight %) Field capacity (0.3 bar)	Permanent wilting point (15 bar)	Available water content
R00	0-10cm	16.30^{abc} (0.41)	11.61^{e} (0.68)	4.69^{a} (0.89)
	10-20cm	15.71^{ab} (0.49)	11.05^{cd} (1.74)	4.67^{a} (1.42)
	20-30cm	15.60^{ab} (1.18)	10.82^{bcd} (1.35)	4.79^{a} (1.49)
R95	0-10cm	15.38^{a} (1.47)	9.14^{abc} (1.39)	6.24^{ab} (0.88)
	10-20cm	15.05^{a} (1.42)	8.44^{a} (2.03)	6.61^{bc} (1.44)
	20-30cm	14.82^{a} (1.86)	8.70^{ab} (2.35)	6.12^{ab} (0.96)
R89	0-10cm	18.27^{d} (0.72)	11.08^{cd} (0.54)	7.19^{bc} (0.50)
	10-20cm	17.69^{cd} (0.30)	11.46^{e} (1.10)	6.22^{ab} (1.26)
	20-30cm	17.24^{bcd} (1.96)	9.06^{abc} (1.86)	8.18^{c} (1.94)
TO	0-10cm	20.68^{e} (1.19)	9.82^{abcd} (1.29)	10.86^{d} (0.48)
	10-20cm	21.56^{e} (1.52)	10.94^{cd} (0.43)	10.62^{d} (1.50)
	20-30cm	20.10^{e} (0.84)	10.22^{abcd} (1.61)	9.88^{d} (1.16)

Column means sharing the same superscript were not significantly different at $p<0.05$ by Duncan's Multiple Range Test.
Values in parentheses represent the standard deviation.

Permanent wilting point (PWP) represents the relative easiness for water to be extracted from the soil at 15 bar. The higher the clay amount of a soil, the higher will be the percentage of water at PWP. In this regard, PWP is lowest in R95 implying that water is most easily extracted from this soil. For soils in R00, R89 and TO, there were no significant differences in PWP.

Available water content (AWC) is the difference between FC and PWP; it represents the amount of water available for plant growth. AWC seems to increase with age of the woodland plantations in TTEBA, in the ascending order of R00 (4.67-4.79%), R95 (6.12-6.61%) and R89 (6.22-8.18%), although not significantly different statistically (Table 5.2). This is probably caused by an increase in organic matter with time. Because of this, AWC is significantly higher in the feng shui woods (9.88-10.86%) than the restored sites.

The results clearly suggest that FC and AWC would increase with age of the plantations in TTEBA. Although the magnitude of increase after 8-19 years of planting is not sufficient for FC and AWC to surpass Tai Om, they are still considered useful indicators for the evaluation of restoration progress.

Soil bulk density

Soil bulk densities of the sites decreased significantly in the order of R95 (1.31 Mg/m^3) $\geq$ R00 (1.23 Mg/m^3) > R89 (1.12 Mg/m^3) > TO (0.90 Mg/m^3) (Figure 5.6). It clearly shows that bulk density of the soil improves with age of the vegetation. In TTEBA, the decomposed granite compacts easily when rain drops impact on the silt-laden soil. Subsequent to root growth, aggregation improvement and canopy interception of rain drops, bulk density of the soil is lowered after restoration. Soil density is, therefore, a reliable indicator for the assessment of restoration progress.

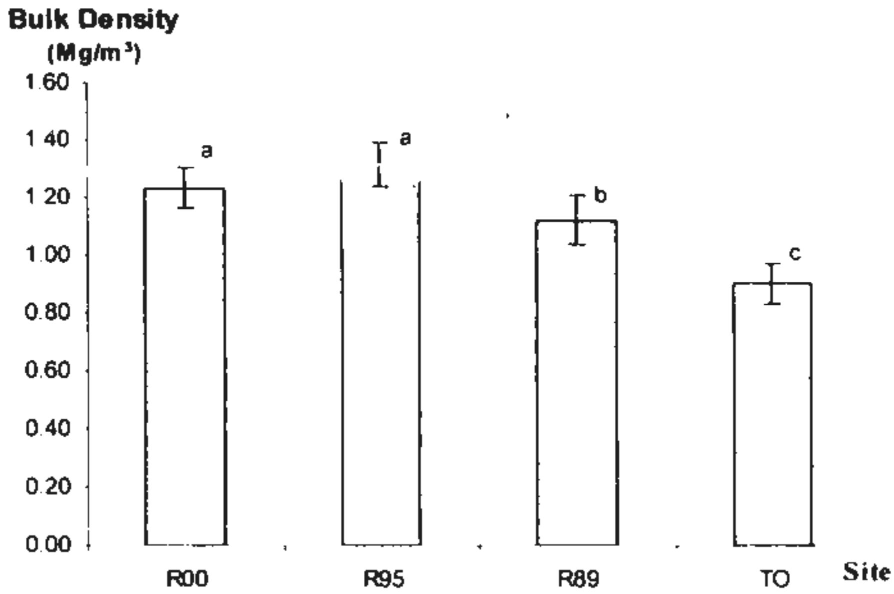


Figure 5.6 Soil bulk densities of the sites
Bar means sharing the same letters were not significantly different at $p<0.05$ by Duncan's Multiple Range Test. Error bars represent one standard deviation.

5.3.2 Amelioration of soil chemical properties

Soil reaction pH, total nitrogen, organic carbon, and total phosphorous are discussed in this section for the same reasons as the physical properties.

Reaction pH

The soils in all the sites are strongly acidic in reaction, pH averaging 4.05-4.64 (Figure 5.7). Among the restored sites, R89 seems to record lower, though statistically insignificant, pH than R95 and R00. Acidification of the soil as a result of vegetation establishment is caused by root respiration, microbial activities and uptake of the cation nutrients (Chau and Marafa, 1999). Because of this, pH is lowest in the undisturbed feng shui soil especially in the top 20 cm layer. Intra-layer differences in pH within a particular site are, however, not significantly different. In short, soil reaction is a potential indicator suitable for the evaluation of restoration progress in Hong Kong provided that the original growth substrate is slightly alkaline to neutral in reaction.

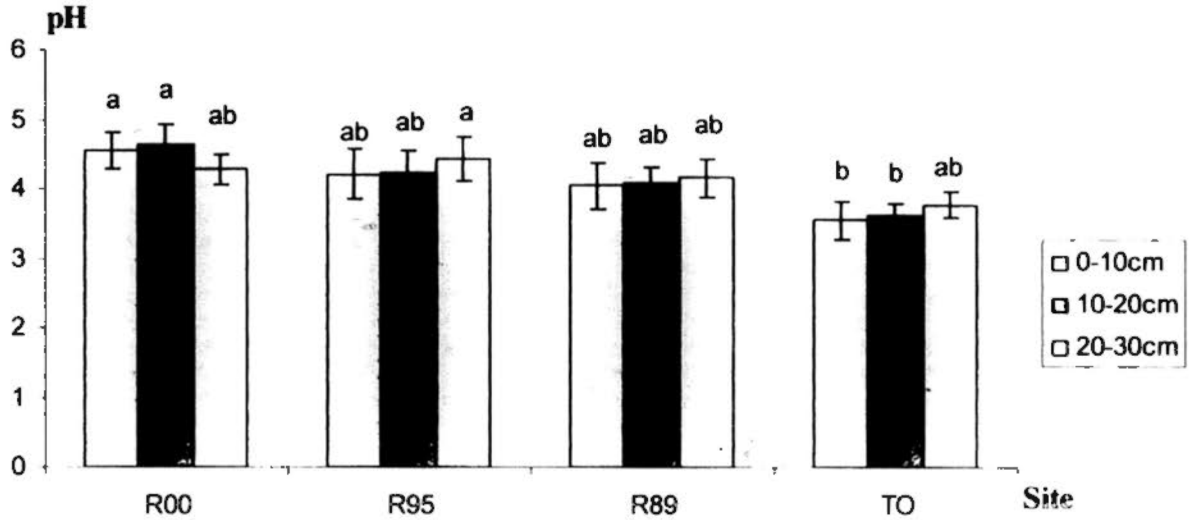


Figure 5.7 Soil pH of different soil layers of the sites
Bars sharing the same letters were not significantly different at $p<0.05$ by Duncan's Multiple Range Test. Error bars represent one standard deviation

Organic carbon

Soil organic carbon (SOC) fluctuated greatly among the sites (Figure 5.8). It increased with age of the vegetation but decreased progressively down the profile. This is expected because SOC is derived from vegetation and the decrease with depth is most conspicuous in R89 and TO. On these two sites, the vegetation has established long enough to cause a differentiation in SOC within the soil layers as a result of the humification of organic matter. There are two sources of organic matter input; namely, aboveground litter and root litter. Both are expected to increase with successional development of the vegetation. Indeed, SOC differentiation was not detected in the 0-10 cm (0.69%) and 10-20 cm (0.57%) layers of R00. After the site had been restored for 13 years (R95), however, SOC decreased significantly in the order of 0-10 cm layer (1.00%) > 10-20 cm layer (0.61%) ≥ 20-30 cm layer (0.44%). Intra-layer differences in SOC were further widened in R89 or 19 years after rehabilitation, a trend similar to TO. These findings clearly show that there is a gradual build-up of organic carbon in the restored soils, starting from the uppermost layer. This is made possible by the established vegetation that captures carbon dioxide through photosynthesis. The increase of SOC and its differentiation down the profile with age of the plantations is, therefore, an evidence of the recovery of soil productivity in TTEBA. In this connection, soil recovery occurred 13 years after revegetation.

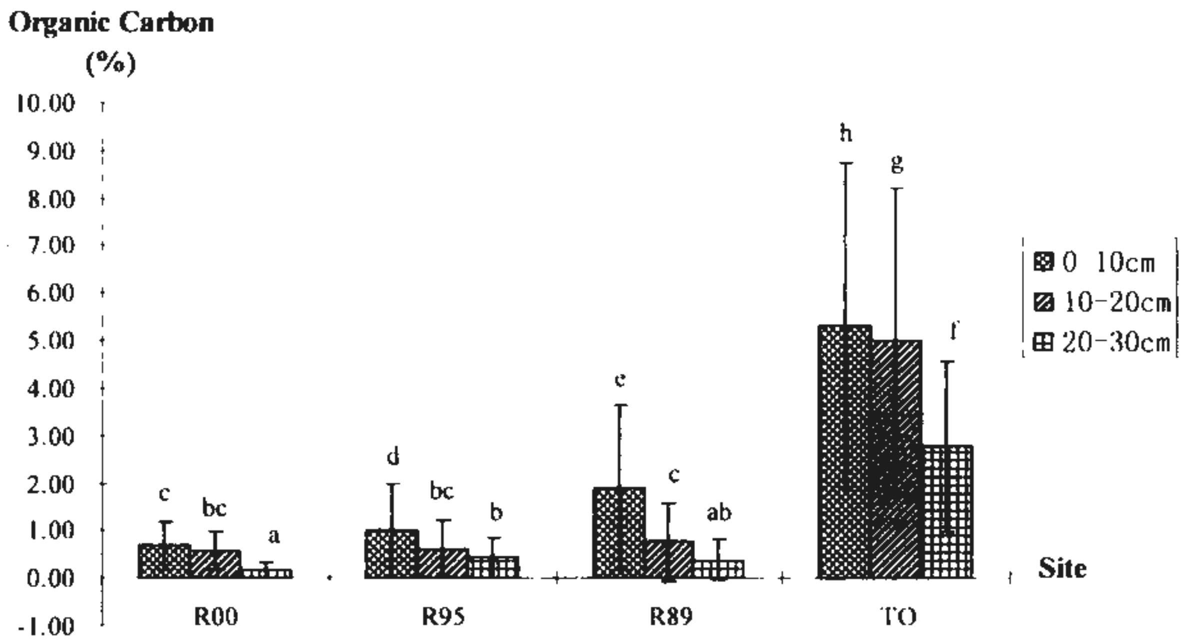


Figure 5.8 Soil organic carbon of the sites
Bar means sharing the same letters were not significantly different at $p<0.05$ by Duncan's Multiple Range Test. Error bars represent one standard deviation

SOC of the top 10-cm soil layers decreased significantly in the order of TO > R89 > R95 > R00. This trend is completely different to the distribution of standing litter on the forest floor, which is in the descending order of R00 > R95 > R89 ≥ TO (see Table 4.2). It is found that the higher the SOC, the lower will be the standing litter biomass. Litter decomposition and organic matter humification are, however, very complicated processes. As the restored sites are different from the feng shui woods in species composition, we cannot assume that the four sites are similar in litter quality. Besides litter quality, humification and SOC build-up is also affected by litterfall, environmental conditions and age of the forests. This issue will be further addressed in Chapters 6 and 7. In view of the importance of SOC on soil properties (Fisher and Binkley, 2000) and its build-up with age of the plantations, it is therefore a reliable indicator for the evaluation of restoration progress.

Total Kjeldhal Nitrogen (TKN)

According to the contract agreement with the then TDD, the landscaping contractor has to maintain the restored sites in TTEBA for a period of three years after planting, including weeding and fertilization. The feng shui woods in Tai Om is virtually untouched for over 300 years. At the time of investigation, therefore, the four sites are typical examples of unfertilized ecosystems, in which soil organic matter is the storehouse of nitrogen.

TKN of the soils followed closely the pattern of SOC, decreasing in the order of TO (0.13-0.24%) > R89 (0.04-0.12%) > R95 (0.04-0.08%) > R00 (0.02-0.04%) (Figure 5.9). Although the restored sites are dominated by exotic nitrogen-fixing legumes, the increase of TKN with time is relatively slow. This is expected because these sites are dry, especially R00 and R95, resulting in the accumulation of a thick layer of undecomposed litter on the forest floor. Nevertheless, intra-layer differentiation in TKN started to appear 13 years after rehabilitation (R95) and the magnitude was further widened in R89. TKN is, therefore, a potential indicator of restoration progress although its accumulation in the vegetation gradient does not seem to be linear.

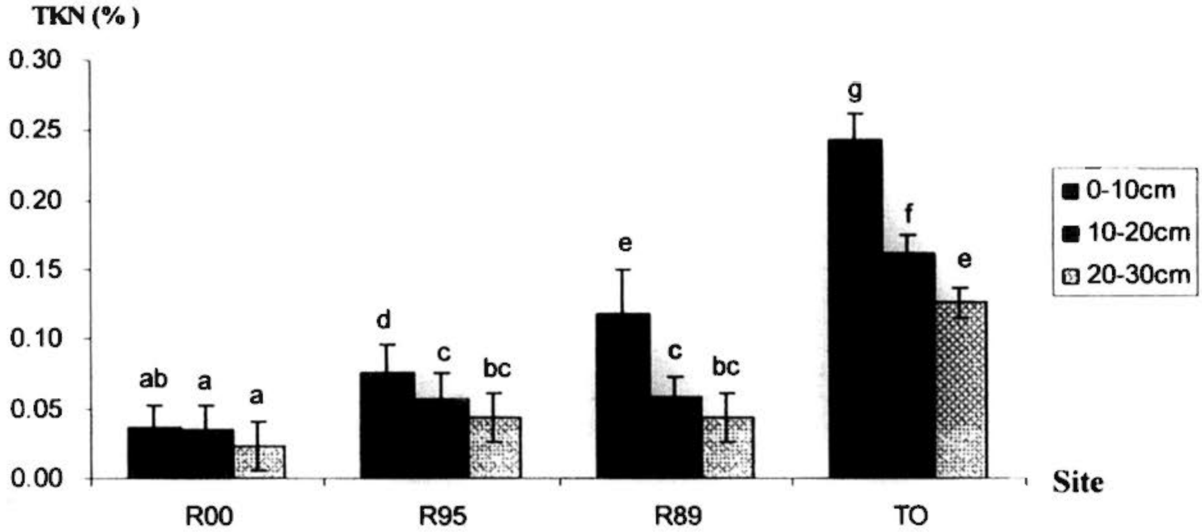


Figure 5.9 TKN in different soil layers of the sites
Bar means sharing the same letters were not significantly different at $p<0.05$ by Duncan's Multiple Range Test. Error bars represent one standard deviation.

<u>Carbon: Nitrogen ratio</u>

The C:N ratios of the soils ranged from 15.8-23.0 (R00) , to 10.5-13.1 (R95), 9.7-16.6 (R89) and 22.0-24.4 (TO) (Figure 5.10). There were no significant differences in C:N ratios among the three layers in both R95 and R89. The C:N ratios of the 0-10 cm and 10-20 cm soils in R00 were significantly higher than the corresponding layers of R95. The wider C:N ratios in R00 probably suggests a lower nitrogen mineralization rate than R95 and R89. In addition to this, C:N ratio of the established soil in Tai Om is much higher than the average soil of 10-12. This is unusual given the long establishment history of the feng shui woods.

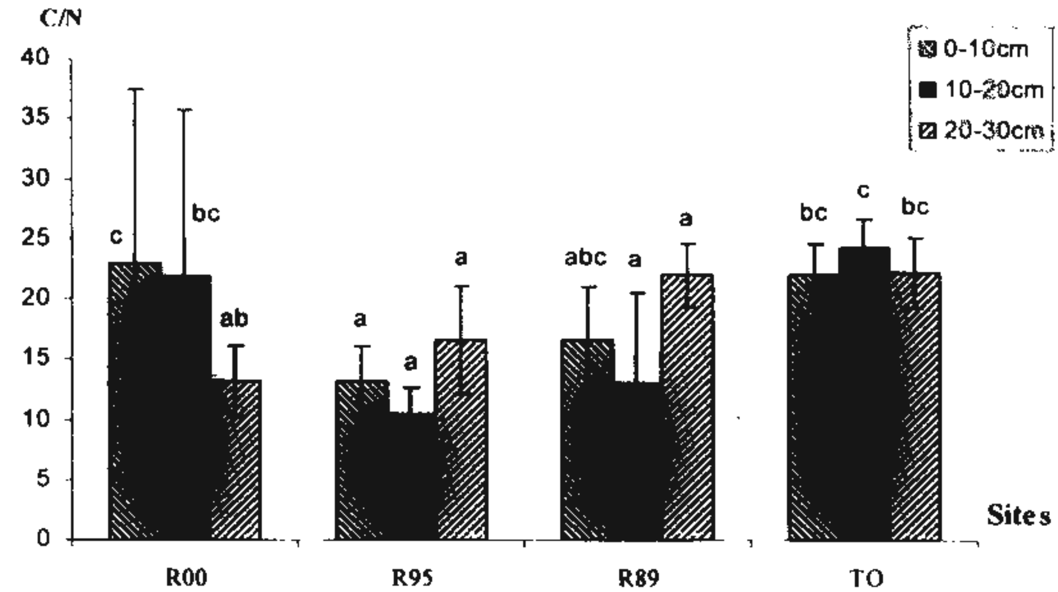


Figure 5.10 C:N ratios of the soils
Bar means sharing the same letters were not significantly different at $p<0.05$ by Duncan's Multiple Range Test. Error bars represent one standard deviation.

<u>Total phosphorous</u>

Phosphorous is another essential plant nutrient. The most important source of phosphorous in the soil is the mineral phosphorous pool (Fisher and Binkley, 2000). Because of this, there were no significant intra-layer differences in total phosphorus between R00 (0.32-0.34 g/kg), R95 (0.29- 0.32 g/kg) and R89 (0.41-0.43 g/kg) (Figure 5.11). Notwithstanding this, total phosphorus seemed to increase 19 years after restoration in TTEBA. As phosphorus is a relatively stable element in soil, most losses are associated with soil erosion instead of leaching (Stevenson and Cole, 1999). This is probably true in the restored sites, where total phosphorus is lower in the younger plantations (R00 and R95) than the older plantation (R89). As the canopy continues to develop with time, the soils are better protected against erosion by the canopy. In addition to this, total phosphorus is unequivocally higher in the lower horizons than the upper horizons. This clearly demonstrates a greater loss of the surface soil, hence total phosphorus, than the lower soil during runoff erosion. Other minor sources of phosphorus include the atmosphere and animals. Owing to its low sensitivity in the ecosystem, phosphorus is unlikely a suitable indicator for restoration assessment.

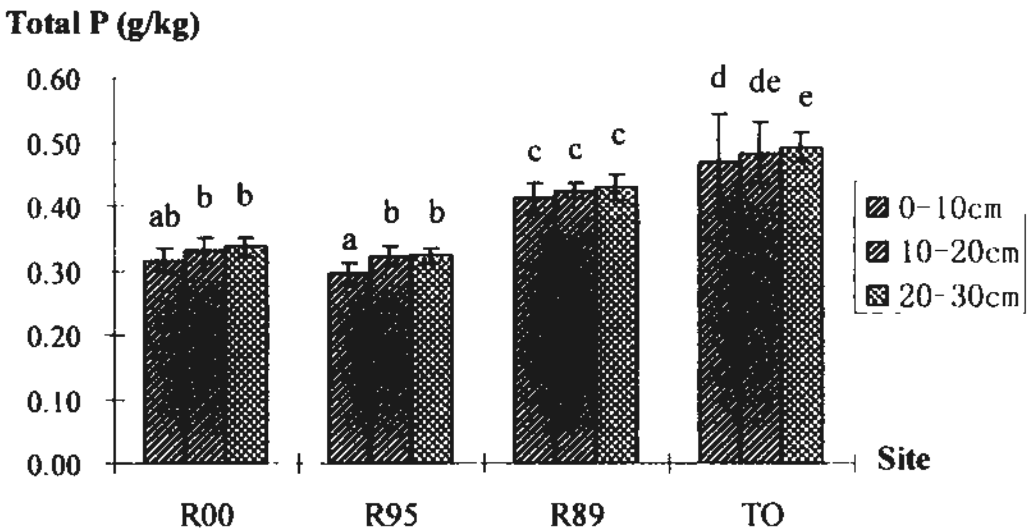


Figure 5.11 Total phosphorous in soils of the sites
Bar means sharing the same letters were not significantly different at $p<0.05$ by Duncan's Multiple Range Test. Error bars represent one standard deviation.

5.3.3 Soil temperature and moisture

Soil temperature

Although soil daily temperature at 10 cm depth fluctuated with seasons, the trend is similar among the four sites (Figure 5.12). The highest soil temperatures were recorded in July 2007 and the lowest in February 2008. The average soil temperatures throughout the study period decreased in the order of R89 > R95 > R00 > TO, although the differences appeared to be minimal (Table 5.3).

Table 5.3 Mean, minimum and maximum soil temperatures

Soil temperature	R00	R95	R89	TO
Mean (°C)	21.45	21.87	21.87	21.13
Standard deviation	4.23	4.21	4.42	4.40
Minimum (°C)	11.74	12.31	11.81	11.34
Maximum (°C)	27.16	27.80	27.77	27.00

The minimum and maximum soil temperatures of the sites followed a similar pattern of the mean soil temperatures, in reverse order with canopy closure (TO > R00 > R89 > R95) (see Table 4.3). Soil temperature, therefore, seems to decrease with canopy closure of the vegetation.

Table 5.4 The correlations of soil temperature, inner-forest and outer-forest temperature of each site

Pearson Correlation		R00 Soil	R00 Forest	R95 Soil	R95 Forest	R89 Soil	R89 Forest	TO Soil	TO Forest	SKW
R00	Soil	1	0.998**	NA	NA	NA	NA	NA	NA	0.958**
	Forest		1	NA	NA	NA	NA	NA	NA	0.954**
R95	Soil			1	0.949**	NA	NA	NA	NA	0.954**
	Forest				1	NA	NA	NA	NA	0.995**
R89	Soil					1	0.963**	NA	NA	0.964**
	Forest						1	NA	NA	0.996**
TO	Soil							1	0.963**	0.957**
	Forest								1	0.995**
SKW										1

** Correlation is significant at $p<0.01$ (2-tailed)
NA: not available
SKW: Shek Kong weather station

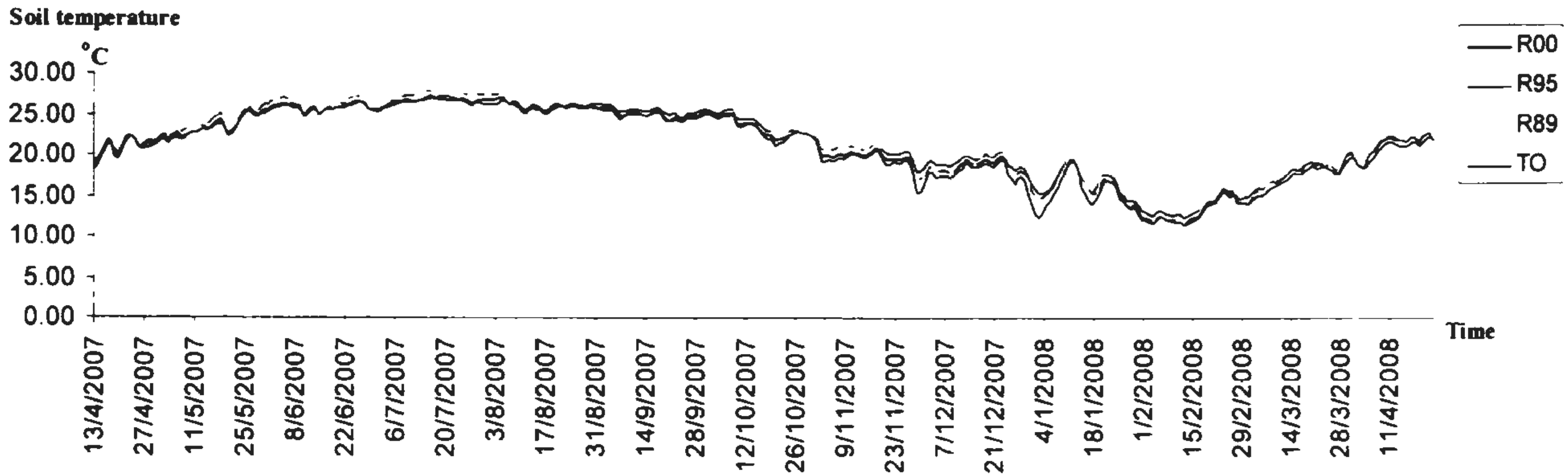


Figure 5.12 Soil temperatures at 10 cm depth, 13.4. 2007 to 11.4.2008

An attempt is made here to investigate the bivariate correlations between soil temperature, forest temperature and temperature at the nearest Shek Kong weather station (SKW). The results showed that soil temperature at 10 cm depth was positively correlated ($p<0.01$) with both forest temperature and temperature at SKW although there were slight differences in the correlation coefficients among the sites (Table 5.4). In this regard, correlation coefficients between soil temperature and forest temperature decreased in the order of R00>R89>TO>R95. In R00, the correlation coefficient between soil and forest temperature (0.998) was higher than that between soil and SKW (0.958). A reverse trend was found for R95, where the correlation coefficient between soil and forest temperature (0.949) was lower than that between soil and SKW (0.954). This clearly suggests that soil temperature in R00 was more determined by forest temperature than by ambient temperature outside the site. While the same result was repeated in TO, the soil temperature in R95 was more likely affected by ambient temperature outside the forest. Soil temperature is, therefore, related to canopy closure of the forests, which is higher for R00 and TO than R95. Thus, canopy closure is able to moderate the effect of ambient temperature leading to the formation of a unique forest environment.

Soil moisture

Soil moisture fluctuated greatly among the sites due to the influence of rainfall, soil and relief. Rainfall is expected to play a pivotal role in affecting soil moisture because none of the sites receive any irrigation, while soil and relief would play a secondary role in moderating the effect of rainfall. To illustrate the importance of rainfall, the changes of soil moisture is analyzed before and after a rainfall event that had lasted from June 26, 2007 to July 5, 2007. During this period, a total of 234 mm of precipitation was recorded at the Shek Kong weather station of which 117 mm was recorded on June 28, 2007 alone (Figure 5.13 & Appendix 5.2). After cessation of the rain on July 5, 2007, the weather remained fine for ten days before onset of another rainfall event after July 15 2007.

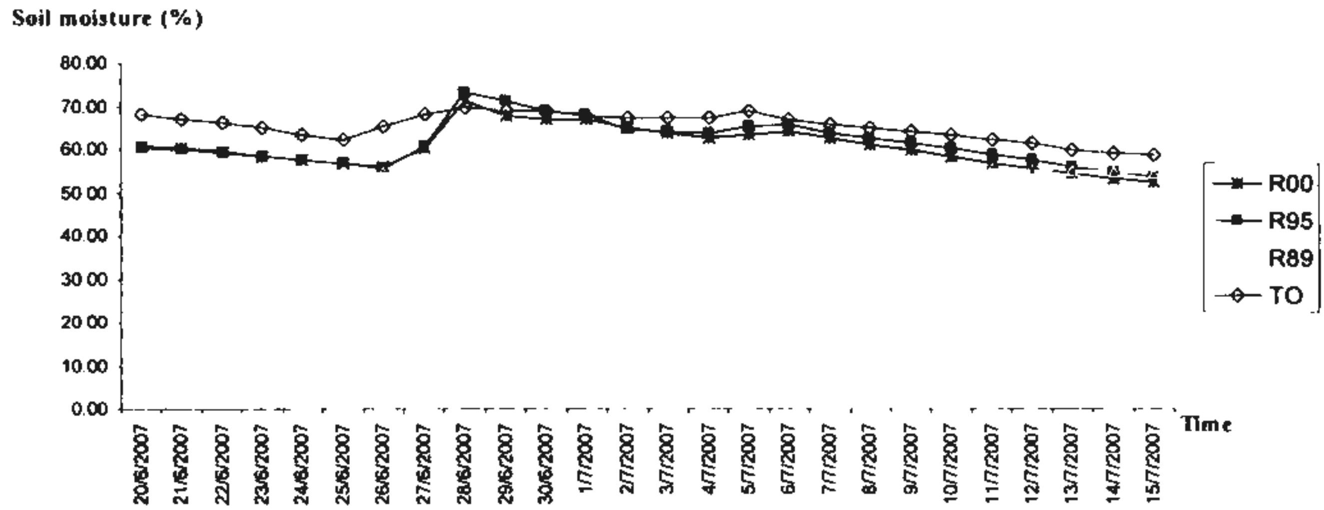


Figure 5.13 Soil moisture of the sites before and after the rain storm on 28.6.2007

Before occurrence of the rainfall event on June 26, 2007, soil moisture was in the order of TO > R95 > R00 > R89 (Figure 5.13). This is expected because the soil in TO has not been disturbed for some 300 years and it contains the highest organic carbon and clay contents (Table 5.1, Figures 5.8). No wonder its field capacity and available water contents are also higher than the restored sites in Tai Tong (Table 5.2). The increase of soil moisture content with age of the plantations is, however, inconspicuous in Tai Tong because it was lowest in R89. This is because the soil in R89 has the highest sand content (72.67%) among the restored sites (Table 5.1). For relatively young plantations, therefore, soil moisture is not a suitable indicator for the assessment of restoration progress. The influence of localized factor, such as texture in R89, is clearly demonstrated.

Soil moisture rose progressively with onset of the rainfall on June 26, 2007 and peaked on June 28, 2007 in all the sites. The rise in soil moisture was, however, more gradual in TO (2.49 % day^{-1}) and R89 (3.04 % day^{-1}) than in the restored sites of R00 (4.91 % day^{-1}) and R95 (5.50 % day^{-1}) (Table 5.5). Because of this, soil moisture in R95 and R00 briefly surpassed that of TO on June 28-29, 2007. It clearly demonstrates that the feng shui forest soil is better buffered against moisture fluctuations than the newly restored soils. The contributory factors to this property are probably related to the higher clay and organic matter contents in the TO soil.

Table 5.5 Changes of soil moisture before and after the rain storm (% day^{-1})

	R00	R95	R89	TO
MIR	4.91	5.50	3.04	2.49
MDR	1.11	1.15	0.57	0.65

MIR: Daily increase of soil moisture content; MDR: Daily decrease of soil moisture content

$$MIR = \frac{|M_{max} - M_{min1}|}{T_1(days)} \qquad MDR = \frac{|M_{max} - M_{min2}|}{T_2(days)}$$

Where, T_1: the number of days from commencement of rain to peak rainfall day; T_2: the number of days from peak rainfall day to the next rain event

M_{max}: soil moisture on the peak rainfall day (June 28.2007);

M_{min1}: soil moisture content before the rain event;

M_{min2}: soil moisture content before the next rain event.

Nevertheless, soil moisture content in all the sites decreased after the rainstorm on June 28, 2007 and reached the lowest level on July 15, 2007 (Figure 5.13). Again, the rate of decrease was faster in the younger plantations of R95 and R00 (1.11-1.15% day^{-1}) than R89 and TO (0.57-0.65 % day^{-1}) due to coarser texture and lower organic matter content of the DG (Table 5.1). TO regained its lead in soil moisture retention four days after the rainstorm. Since the newly restored sites are poorly buffered against moisture fluctuations, except R89, the need of irrigation during establishment period of the seedlings is further affirmed.

Table 5.6 Soil moisture content of the sites before and after the rain storm (%) (From 20 June 2007 to 15 July 2007)

	R00	R95	R89	TO
Mean	60.70	61.73	57.11	65.43
Standard deviation	4.74	5.12	2.31	3.16
Minimum	52.59	53.63	54.38	58.72
Maximum	71.44	73.25	64.05	69.83
Range	18.84	19.63	9.67	11.11

Although the soil moisture content of R89 was lowest among the restored sites, water availability is not necessarily most critical in this 18-year-old plantation. Indeed, AWC of R89 is highest among the restored sites and the accumulation of organic matter has obviously offset the drawback of coarse texture in water retention. The low standard deviation re-affirms a more stable supply of soil moisture in R89 than in R95 and R00 (Table 5.6). Given the high sand content, water is also more easily released from the soil in R89 resulting in lower risk of wilting. While ecological restoration has no direct bearing on absolute soil moisture content, it does regulate moisture behavior indirectly through the build-up of soil organic matter. This can be illustrated by the small standard deviations and range values of soil moisture content in R89 and TO (Table 5.5). The smaller the standard deviations and range values of soil moisture content, the higher will be the AWC.

5.3.4 Nitrogen Mineralization

Soil mineral-N contents

Ammonification and nitrification were detected in the top 10 cm soil throughout the incubation period. MH_4-N contents of the restored soils in Tai Tong averaged 5.62-8.63 mg/kg (T_0), 4.84-12.13 mg/kg (T_1), 10.14-15.07 mg/kg (T_2), 7.85-12.9 mg/kg (T_3) and 9.35-12.45mg/kg (T_4) (Table 5.7). Overall, NH_4-N contents were higher in R89 and R00 than R95 although significant differences between the restored sites were only found at incubation T_1 and T_3. Soil organic carbon, which decreased significantly in the order of R89 > R95 > R00, is the source of energy for ammonifiers and nitrifiers. Ammonification is therefore not consumerate with the increase of SOC during ecological succession. This can be further supported by even lower MH_4-N contents (1.01-2.56 mg/kg) in TO, which contains the highest level of SOC among the study sites (Figure 5.10). Under the existing experimental design, ammonification is unlikely a reliable indicator of restoration progress.

A different pattern is found for nitrification among the study sites, which is more active in TO than the restored sites. In TTEBA, NO_3-N ranged from 0.54-1.16 mg/kg (T_0) to 0.13-1.39 mg/kg (T_1), 0.46-0.86 mg/kg (T_2), 0.11-2.15 mg/kg (T_3) and 0.86-2.48 mg/kg (T_4) (Table 5.7). The highest NO_3-N content was associated with R89, with the exception at T_2, and the lowest with R00. This is expected because R89 also yielded higher levels of NH_4-N, which is the substrate needed for nitrification. Notwithstanding this, the differences in NO_3-N were not significant statistically.

It is thus clear that NH_4-N in the rehabilitated sites is substantially higher than the undisturbed feng shui forest while the reverse is true for NO_3-N. In other words, NH_4-N predominates over NO_3-N in the borrow areas that have been rehabilitated for 8 to 19 years. This finding agrees reasonably well with studies on fire-disturbed slopes in Hong Kong (Yau, 1996; Marafa, 1997) and mine site restoration in India (Singh *et al.*, 2004). While NH_4-N is the substrate needed for nitrification, plants prefer the absorption of ammonia to nitrate (Haynes, 1986) although nitrate is more easily leached from the soil than ammonium. Furthermore, NH_4-N could have been overestimated during incubation in dry season when plant uptake is minimal (Saynes *et al.*, 2005).

Nitrogen mineralization is mediated by microbial transformations and influenced by factors that affect microbial activity, such as temperature, moisture and pH (Stevenson and Cole 1999). Ammonification converts organic N to NH_3 by heterotrophic microorganisms, with minor participation of soil fauna. Nitrification is taken to mean the oxidation of NH_3 to NO_3^- through two groups of autotrophic bacteria, *Nitrosomonas* and *Nitrobacter* (Alexander, 1977). In this study, the rehabilitated sites in TTEBA are relatively similar in soil temperature, moisture, and pH. The accumulation of ammonium represents the quantity of substrate nitrogen in excess of the microbial demand (Alexander, 1977). The lack of sufficient autotrophic bacteria in the soil can also be a contributory factor to the predominance of NH_4-N over NO_3-N in the restored sites. This is, however, beyond the scope of the present study.

Total mineral nitrogen is the sum of NH_4-N and NO_3-N. With the exception of incubation at T_4, mineral nitrogen of TO is significantly higher than R00, R95 and R89 (Table 5.7). This is expected because among the factors influencing N mineralization, SOC plays the most important role. As the source of energy for microorganisms, SOC is significantly higher in TO than the other restored sites. Among the restored sites, SOC tends to increase with age of the plantations, resulting in significantly higher mineral nitrogen in R89 than R00 and R95 most of the time. Other factors being equal, total mineral nitrogen content is more suitable than NH_4-N or NO_3-N alone for the assessment of restoration progress.

Net nitrogen mineralization

Table 5.7 Net ammonification, nitrification and N mineralization, 8.11.2007 to 3.1.2008 ($kg \cdot ha^{-1} day^{-1}$)

	Site \ Time	2 weeks	4 weeks	6 weeks	8 weeks
Net ammonification	R00	-0.33	0.13	0.13	0.08
	R95	0.11	0.17	0.04	0.07
	R89	0.52	0.38	0.18	0.14
	TO	-0.10	0.00	-0.02	-0.02
Net nitrification	R00	-0.04	-0.01	-0.01	0.01
	R95	0.00	0.05	0.01	0.02
	R89	0.02	-0.01	0.03	0.03
	TO	0.06	0.21	0.14	0.06
Net N mineralization	R00	-0.37	0.13	0.11	0.08
	R95	0.11	0.22	0.05	0.08
	R89	0.54	0.37	0.21	0.16
	TO	-0.04	0.22	0.12	0.04

Negative values represent immobilization.

Table 5.8 Comparison of soil mineral nitrogen between the sites (mg/kg) (n=10)

Mineral-N	Site \ Time	T_0(Nov 8,2007)	T_{1c} (Nov 22, 2007)	T_{2c} (Dec 6, 2007)	T_{3c} (Dec 20,2007)	T_{4c} (Jan 3,2008)
NH_4-N	R00	8.63^{b}	4.84^{b}	11.70^{b}	12.9^{c}	12.21^{b}
	R95	6.53^{b}	7.72^{c}	10.14^{b}	7.85^{b}	9.35^{b}
	R89	5.62^{ab}	12.13^{d}	15.07^{b}	12.32^{c}	12.45^{b}
	TO	2.56^{a}	1.01^{a}	2.66^{a}	1.8^{a}	1.59^{a}
NO_3-N	R00	0.58^{a}	0.13^{a}	0.46^{a}	0.11^{a}	0.86^{a}
	R95	0.54^{a}	0.53^{a}	1.55^{a}	0.98^{ab}	1.24^{ab}
	R89	1.16^{a}	1.39^{a}	0.86^{a}	2.15^{b}	2.48^{b}
	TO	11.99^{b}	12.88^{b}	18.62^{b}	18.29^{c}	15.73^{c}
Total	R00	9.21^{a}	4.97^{a}	12.16^{a}	13.01^{ab}	13.07^{ab}
	R95	7.07^{a}	8.25^{b}	11.68^{a}	8.83^{a}	10.58^{a}
	R89	6.78^{a}	13.52^{c}	15.93^{a}	14.48^{b}	14.93^{ab}
	TO	14.54^{b}	13.88^{c}	21.27^{b}	20.09^{c}	17.31^{b}

T_{ic}: Mineral-N contents of the covered soil at time *i*.

Column means of ammonium-N, nitrate-N and total mineral-N sharing the same superscript were not significantly different at $p < 0.05$ by Duncan's Multiple Range Test.

Net ammonification, nitrification and N mineralization during the incubation period are summarized in Table 5.8. These microbial processes fluctuated greatly among the sites, possibly due to inherent characteristics of the activities as no two soils are the same. Another possible cause of the fluctuations could be the short period of incubation in the dry season only. Recognizing these limitations, several observations can still be generalized from the data. First, net ammonification rates were generally higher than net nitrification rates especially in the restored sites in TTEBA. Among the restored sites, net ammonification was higher in R89 (0.14-0.52 kg ha^{-1} day^{-1}) than R00 (0.08-0.13 kg ha^{-1} day^{-1}) and R95 (0.04-0.17 kg ha^{-1} day^{-1}). Second, immobilization of NH_4-N was more active in TO than the restored sites possibly due to rapid uptake by microorganisms in the site. Third, the immobilization of NO_3-N was generally more vigorous in the restored sites than the feng shui forest site, a trend totally different from the immobilization of NH_4-N. Indeed, net nitrification in TO amounted to 0.06-0.21 kg ha^{-1} day^{-1} compared to 0.00-0.06 kg ha^{-1} day^{-1} for the restored sites. While ammonification is typically associated with a waste-product overflow in microbial metabolism, nitrification is usually associated with the energy-yielding reactions in the metabolism of autotrophic bacteria. Therefore, nitrification is more important to enhance the efficiency of nitrogen cycling and more helpful to accelerate energy flow in the process of forest rehabilitation. With a lower net nitrification rate, all the restored sites in Tai Tong still lag behind TO in terms of the efficiency of nutrient cycling and energy flow. Fourth, net N mineralization was higher in R89 than R00 and R95 in TTEBA. More importantly, net N mineralization in TO seems to be lower than the restored sites of R89 and R95 during the 2-month incubation period. If this finding is not confined to the dry season, it suggests that mineral nitrogen is accumulating in the soils of Tai Tong that have been revegetated for 13-19 years. An improvement of soil productivity would pave the way for ecological succession.

An attempt is made here to find out if any relationship exists between N mineralization and precipitation, as well as soil temperature and moisture during the incubation period. . With few exceptions, the correlations between N mineralization and the three environmental parameters are not significant statistically (Table 5.9). It is not known if this finding is caused by the narrow range of soil temperature (18.09-20.73°C) and moisture (48.04-57.8%) during the short period of incubation in winter (Appendix 5.4). Alternatively, other factors such as soil pH and litter quality may play a more important role in accounting for the N mineralization of soils.

Table 5.9 Correlation between N-mineralization and precipitation, soil temperature and moisture

Pearson correlation coefficient		Ammonification	Nitrification	N Mineralization
R00	Soil temperature	-0.852	-0.995**	-0.862
	Soil moisture	-0.54	-0.01	-0.509
	Precipitation	-0.141	0.424	-0.112
R95	Soil temperature	0.371	-0.361	0.225
	Soil moisture	-0.143	-0.004	-0.174
	Precipitation	-0.221	-0.186	-0.284
R89	Soil temperature	0.927	-0.198	0.957*
	Soil moisture	-0.182	0.236	-0.188
	Precipitation	-0.309	0.439	-0.298
TO	Soil temperature	-0.832	-0.146	-0.429
	Soil moisture	-0.83	-0.237	-0.484
	Precipitation	-0.128	-0.739	-0.55

* $p<0.05$; ** $p<0.01$

Leaching of mineral N

Mineral nitrogen refers to NH_4-N and NO_3-N. The leaching of mineral N was estimated by the difference between the open tubes and covered tubes in the field incubation (Table 5.10).

During the first two (T_1) and four weeks (T_2) of incubation, mineral N did not differ between the open and covered tubes, suggesting no leaching of nitrogen from the soil. Some significant losses of mineral nitrogen, either NH_4-N, NO_3-N or both, were detected at T_3 and T_4. While there is no discernible pattern in the leaching loss of mineral nitrogen from the sites, losses were mostly associated with R89 and TO instead of R00 and R95. This seems to suggest that the leaching loss of mineral N from the sites is concentration-dependent at least during the cool dry season.

Table 5.10 Leaching loss of mineral N from November 2007 to January 2008 (mg/kg) (n=10)

Site	Mineral N	T_0 (November 8, 2007)	T_1 (November 22, 2007)			T_2 (December 6, 2007)			T_3 (December 20, 2007)			T_4 (January 3, 2008)		
			Covered	Open	T-test	Covered	Open	T-test	Covered	Open	T-test	Covered	Open	T-test
R00	NH_4-N	8.63	4.84	4.57	NS	11.70	10.17	NS	12.90	8.10	***	12.21	9.84	NS
	NO_3-N	0.58	0.13	0.11	NS	0.46	0.43	NS	0.11	0.09	NS	0.86	0.54	NS
	Total	9.21	4.97	4.68	NS	12.16	10.60	NS	13.01	8.19	***	13.07	10.38	NS
R95	NH_4-N	6.53	7.72	5.67	NS	10.14	9.42	NS	7.85	6.88	NS	9.35	10.26	NS
	NO_3-N	0.54	0.53	0.48	NS	1.55	0.98	NS	0.98	0.44	NS	1.24	1.21	NS
	Total	7.07	8.25	6.15	NS	11.68	10.40	NS	8.83	7.31	NS	10.58	11.47	NS
R89	NH_4-N	5.62	12.13	10.36	NS	15.07	14.60	NS	12.32	11.91	NS	12.45	7.08	**
	NO_3-N	1.16	1.39	0.97	NS	0.86	0.54	NS	2.15	0.82	*	2.48	0.95	*
	Total	6.78	13.52	11.33	NS	15.93	15.14	NS	14.48	12.73	NS	14.93	8.03	***
TO	NH_4-N	2.56	1.01	0.81	NS	2.66	1.95	NS	1.80	0.80	NS	1.59	1.43	NS
	NO_3-N	11.99	12.88	12.17	NS	18.62	16.09	NS	18.29	15.62	**	15.73	13.41	*
	Total	14.54	13.88	12.98	NS	21.27	18.04	NS	20.09	16.42	**	17.31	14.84	NS

"Covered" means the aluminum tubes were covered by a plastic wrap; "Open" means the aluminum columns were exposed to the air.
NS: Not significant; * $p<0.05$; ** $p<0.01$; *** $p<0.001$

5.4 Conclusions

The present experiment investigated a wide range of soil properties, processes and environmental parameters of three restored sites and one undisturbed forest site, with special emphasis on the recovery of soil productivity after land degradation. Based on the findings, the following conclusions can be drawn:

(1) Ecological rehabilitation with exotic species is capable of ameliorating the degraded soils, and the positive effects tend to vary with age of the plantations. While there is no change in the soil texture, ecological rehabilitation acidified the soil, lowered bulk density, elevated field capacity and available water contents, and lowered soil temperature at the 10 cm layer. After being restored for 19 years, R89 was better buffered against moisture fluctuation than R95 and R00..

There was a gradual build-up of organic carbon, especially in the top 10 cm soil after restoration, and it was more significant in R89 and R95 than R00. Intra-layer differences in SOC were widened in R89, too. A similar trend was found for TKN, which also increased albeit slowly with age of the plantations.

Ammonification and nitrification were detected in the restored soils. While NH_4-N predominated over NO_3-N in the restored soils, net N mineralization was higher in R89 than R95 and R00. Thus, there seems to be an accumulation of mineral nitrogen 13 (R95) to 19 (R89) years after restoration. No discernible pattern was found for the leaching loss of mineral nitrogen although it might be concentration-dependent.

(2) In view of the fact that planting can accelerate the recovery of soil productivity,

several soil parameters are considered suitable for the assessment of restoration progress. They include bulk density, field capacity and available water contents, SOC, TKN and net N mineralization rate. Equally useful is pH provided that the substrate prior to planting is neutral to alkaline in reaction.

(3) The restored sites in TTEBA had only been rehabilitated for 8-19 years; their soil quality is generally poor. With a distinctly lower net nitrification rate, the restored sites still lag behind the undisturbed forest in terms of the efficiency of nutrient cycling and energy flow.

CHAPTER 6

LITTERFALL AND NUTRIENT DYNAMICS

6.1 Introduction

Nutrient inputs from the atmosphere and rock weathering are important to the long-term development of soils and ecosystem, but on an annual basis, nutrient cycling within ecosystems provides the major source of nutrients for plant use (Fisher and Binkley, 2000). Biological nutrient cycling is a fundamental process in the functioning of forest ecosystem. The return by litterfall is considerably higher than the import by atmospheric deposition (Zimmermann, 2002). Nutrient return via litterfall represents a major biological pathway for element transfer from aboveground vegetation to the soil (Vitousek, 1982; Maguire, 1994; Clark *et al.*, 2001). It plays an important role in maintaining soil carbon and nutrient pools as well as fertility in a forest ecosystem.

Litterfall is a major material pathway from plants to soils. The quantity and quality of litterfall affect the amounts of nutrients in the soil and the rate of nutrient mineralization (Finzi *et al.*, 1998). Litterfall quantities and qualities are different among tree species (Binkley and Giardina, 1998). The spatial variations of litterfall at stand level are created by the composition and structure of canopies of different species (Hirabuki, 1991; Ferrari, 1999). The quantification of the rate of litterfall and its nutrient contents is still an important approach in understanding tropical or subtropical forest production and nutrient cycling. Meanwhile, seasonal variations in nutrient concentration and return are related to climatic fluctuations and/or changes in plant phenology, which in turn can affect the processes of decomposition, mineralization and immobilization (Liu *et al.*, 2004; Zimmermann *et al.*, 2002). As the results may reflect a given status of the bioelement cycle or a given development phase of the community, seasonal fluctuations in nutrient concentration are very important in relation to the timing of leaf sampling for analysis (Palma *et al.*, 2000). Therefore, it is essential to understand the forest ecosystem processes, including the amount of nutrients returned to soil and the seasonal fluctuations of nutrients contained in the litter.

In litterfall research, both the techniques and components of study may vary from place to place. Some report only leaf litterfall, while others report fine litterfall that includes leaves, twigs, flowers and fruits. Still others report total litterfall, which is defined as fine litterfall plus tree stems and large branches. On the average, leaf litter accounts for 70% of the fine litterfall (Bray and Gorham, 1964), and the woody component of total litterfall is too heterogeneous both spatially and temporally to allow any meaningful generalizations. The techniques used for litterfall collection can influence the results obtained. Ideally, litterfall should be collected at frequent intervals from well-replicated traps located above the soil surface (Proctor, 1983). Frequent collections are particularly important in lowland tropical areas, where rapid decomposition of litter in traps can lead to substantial underestimates of litterfall (Kunkel-Westphal and Kunkel, 1979).

The litter on the forest floor acts as input–output system of nutrient and the rate at which forest litter falls and subsequently decomposes contribute to the regulation of nutrient cycling and primary productivity, and to the maintenance of soil fertility in forest ecosystems (Singh *et al.*, 1999; Fioretto *et al.*, 2003; Onyekwelu *et al.*, 2006; Pandey *et* *al.*, 2007). Therefore, it is essential to understand the amount and pattern of litterfall in forest ecosystems. Although litterfall studies have been carried out on different types of woodland plantation in the subtropical region (e.g. Zhang *et al.*, 1993; Parrotta, 1999; Liao *et al.*, 2006; Pandey *et al.*, 2007), few attempts have been made to investigate litterfall in uneven-aged plantations of similar species composition. Furthermore, there exists a knowledge gap whether litterfall can be employed as an indicator of restoration progress in subtropical China. This is made possible by the availability of uneven-aged woodland plantations in Tai Tong East Borrow Area.

In the present study, leaf litter is found to account for 69-75% of the total litterfall. This value is comparable to other subtropical plantations investigated by Zhang *et al.* (1993) and Wang *et al.* (2007). While foliage litter is the most important component of litterfall in this part of the world, how will it change with age of the woodland plantations? Is it a reliable parameter for the assessment of restoration progress? The present experiment, therefore, investigates the seasonal pattern of litterfall and nutrient fluxes along the vegetation gradient. Owing to the constraints of manpower and time, only leaf litter would be investigated in the present study bearing in mind that litter components could change with age of the plantations. For instance, there is an increase in the percentage of fruit litters on the forest floor along the vegetation gradient (see Chapter 4). Nonetheless, results obtained from this experiment will provide answers to the following specific questions:

(1) Do the woodland plantations and feng shui forest differ in leaf litterfall production?

(2) What are the most important climatic factors affecting litterfall production?

(3) How much nutrients are returned to the forest floor via leaf litterfall and do they change with age of the plantations?

(4) Is leaf litterfall a reliable indicator of restoration progress?

6.2 Materials and Methods

This section describes the methods for litterfall collection, laboratory analysis of litterfall nutrients and the statistical analysis of data.

6.2.1 Litterfall collection

The litter trap method was employed for the collection of litterfall. Each litter trap has a surface area of 0.5 m^2 (0.71 m x 0.71 m) and the frame is made of galvanized iron. The mesh size of nylon net hooked to the metal frame is 2.5 mm, which is sufficient for the foliage litters in the present study. Each trap was positioned 0.5 m above ground level and the nylon net was set at 30 cm above ground (Plate 6.1). There were 15 traps per site and were placed systematically within a 15 m x 25 m plot demarcated for this purpose. As seen in Figure 6.1, each litter trap was placed at the middle of a 5 m x 5 m subplot within the study plot. There were altogether 720 litterfall samples within one year.

Litterfall was collected at monthly intervals from May 2007 to April 2008 with the use of plastic bags to avoid any contamination. The samples were returned to the laboratory and oven-dried to constant weight at 80°C. The leaf litter was then separated from the non-leaf litters of bark, flowers, fruits and small branches, weighed and ground to pass a 0.2 mm sieve before being stored in a sealed plastic bag for further chemical analysis.

Plate 6.1 The rectangular litter trap

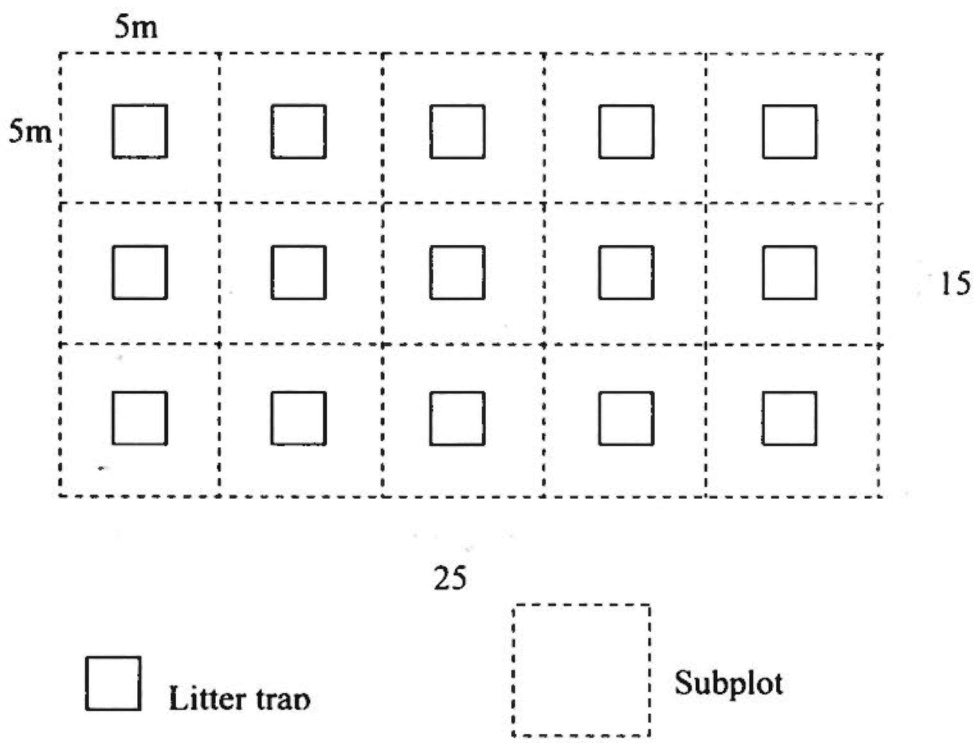


Figure 6.1 The location of litter traps at each site

6.2.2 Chemical analysis

The leaf litters were analyzed for total carbon (C), nitrogen (N), phosphorous (P), and sulfur (S). 1.5-2.5 mg litter subsamples were packed in a tin capsule, and then determined using a CHNS analyzer (Perkin Elmer Series II CHNS/O Analyzer 2400) for C, N and S.

Total P was extracted by the mixed acid digestion method. Around 0.3 gram of oven dry 0.2 mm foliage litter was digested with 15 ml of mixed acids (HNO_3:$HClO_4$:H_2SO_4 at the ratio of 10:2:1) at 240°C to obtain a clear to whitish solution. It was then filtered and diluted to 100 ml with deionized water. Total P was then determined by Flow Injection Analysis 5000 analyzer and expressed on an oven dry weight basis. Total C, N, S and P contents of the litterfall were estimated by multiplying their respective concentrations with the litter mass in each trap.

6.2.3 Statistical analysis

Mean monthly litterfall production at each site was obtained by averaging the 15 litter traps. Annual litter production and nutrient return was obtained by summing up the monthly litterfall and the nutrients contained therein. The results were transferred into the SPSS (for Windows) statistical package for numerical processing. The mean values and standard deviations were calculated. The differences in litterfall production and nutrient contents between sites were tested by using Duncan's multiple range tests and the significance level was set at $p<0.05$ unless otherwise specified. Pearson correlation analyses were also carried out to examine the relationship between litterfall production and climatological factors.

6.3 Results and discussion

This section describes the results obtained from this experiment, including litterfall production, annual return of nutrients from litterfall, as well as climatological factors that may affect litterfall production.

6.3.1 Leaf litterfall production

Litterfall varied with sites and seasons. In R00, monthly litterfall ranged from a minimum of 150.1 kg/ha in June 2007 to a maximum of 1,350.2 kg/ha in December 2007 (Figure 6.2a). The majority of the leaf litter was collected between July 2007 and January 2008, covering the summer, autumn and winter seasons. This is unexpected as peak litterfall usually coincides with the cool dry winter season in this part of the world. The monthly litterfall in R95 varied from 145.7 kg/ha in February 2008 to 854.1 kg/ha in August 2007 (Figure 6.2b). Litterfall production was again highest in the summer months of August and September 2007, as well as the spring months of March and April 2008. In R89, litterfall production varied from 257.6 kg/ha to 992.8 kg/ha (Figure 6.2c). Peak production was recorded in the winter months of December 2007 (872.5 kg/ha) and March 2008 (992.8 kg/ha), being followed closely by productions in August (847.5 kg/ha) and September 2007 (823.2 kg/ha). Litterfall in TO exhibited a pattern totally different from TTEBA. Peak production was found in March 2008 (2,914.9 kg/ha), which accounted for 30% of the total annual production (Figure 6.2d). Although productions in February and April 2008 were also relatively high, they were less than one third of the peak production. For rest of the months, litterfall production was substantially low especially between September 2007 and January 2008.

As noted in Chapter 4, the four study sites are different in species composition, diversity, evenness and status of successional development. While TO is dominated by native species, the restored sites are relatively species-poor and dominated by a few exotics only. Because of these inherent differences, they also differ in litterfall production. Total litter production decreased significantly in the order of TO (9,724.8 kg/ha) $\geq$ R00 (8,797.6 kg/ha) > R89 (7,329.2 kg/ha) $\geq$ R95 (6,663.1 kg/ha) (Table 6.1). This is expected because as a near climax community, TO has probably developed an efficient nutrient cycling pathway between the soil-plant subsystems. Litterfall is a storehouse of nutrients much needed to maintain ecosystem stability and resilience. Litterfall production in TO, therefore, topped the list and the amount is comparable to other subtropical regions (Bray and Gorham, 1964). The rapid cycling of nutrients in TO is further supported by the fact that this site has the lowest standing litter biomass (see Chapter 4). On the other hand, litterfall did not increase with age of the plantations in Tai Tong as expected, which was significantly higher in R00 than in R95 and R89. It is thus fairly safe to surmise that litterfall is not a suitable indicator in the assessment of restoration progress.

Litterfall production used to peak in the cool dry season in this part of the world, from December through March. This is true for sites R95, R89 and TO, where peak litterfall production coincided with March (Table 6.1). Litter production in R00, however, amounted to 525.9 kg/ha only, which is more than halved compared to peak production in October through December (1,087.1-1,350.2 kg/ha).

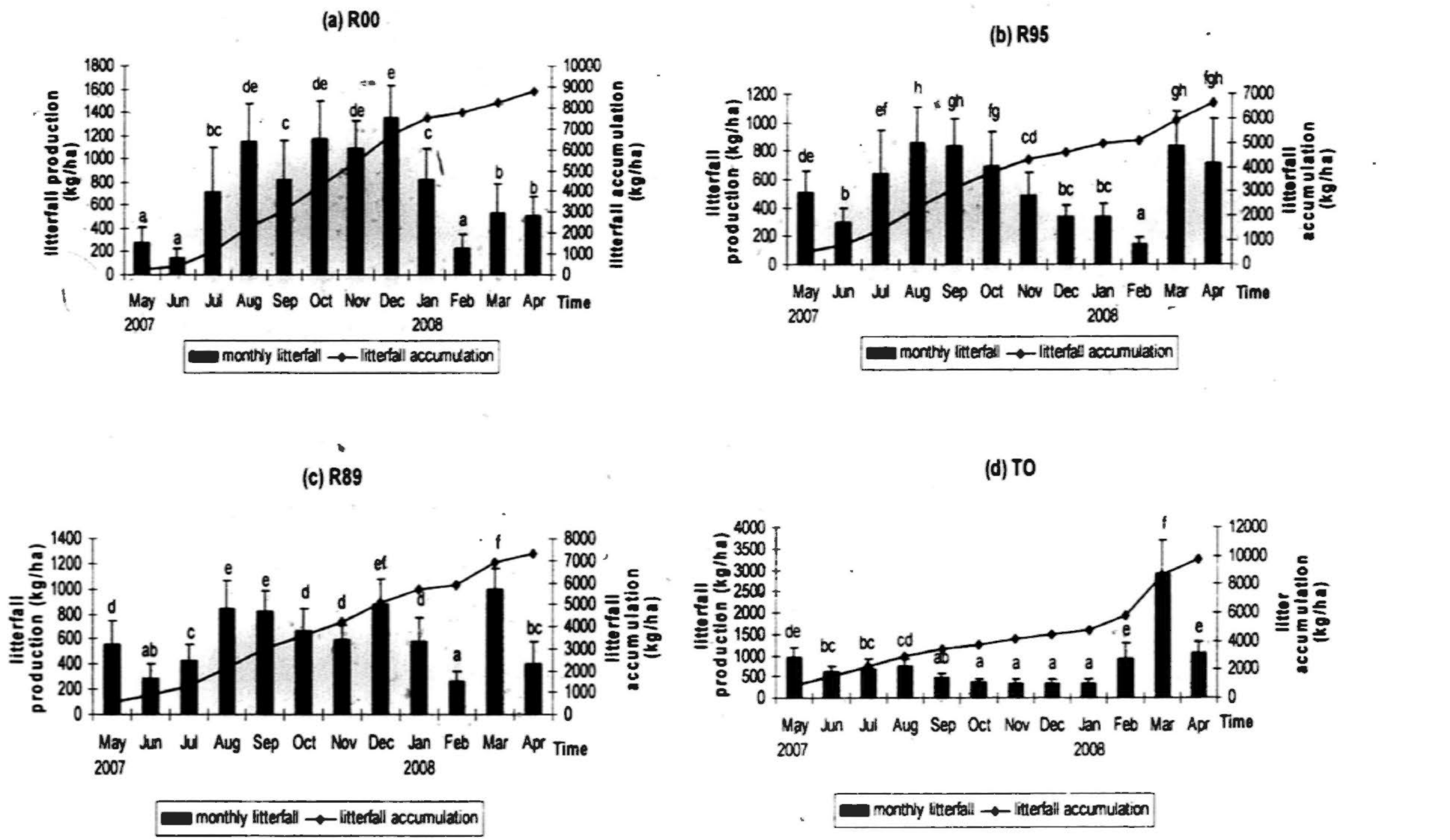


Figure 6.2 Monthly litterfall production (kg/ha) (n=15)
Bar means sharing the same letters were not significantly different at $p<0.05$ by Duncan's Multiple Range Test. Error bars represent one standard deviation.

Table 6.1 Monthly litterfall of the sites (kg/ha) (n=15)

Weight	May	June	July	Aug	Sep	Oct	Nov	Dec	Jan	Feb	Mar	Apr	Total
R00	274.9^{a}	150.1^{a}	715.7^{b}	1142.9^{b}	823.1^{b}	1170.2^{c}	1087.1^{c}	1350.2^{c}	820.9^{c}	225.7^{a}	525.9^{a}	510.9^{a}	8797.6^{b}
R95	505.5^{b}	299.3^{b}	644.5^{b}	854.1^{a}	835.1^{b}	686.9^{b}	484.3^{b}	331.0^{a}	330.9^{a}	145.7^{a}	$834.6a^{b}$	711.1^{b}	6663.1^{a}
R89	561.9^{b}	289.4^{b}	432.9^{a}	847.5^{a}	823.2^{b}	666.4^{b}	598.0^{b}	872.5^{b}	578.8^{b}	257.6^{a}	992.8^{b}	408.3^{a}	7329.2^{a}
TO	940.0^{c}	611.5^{c}	673.2^{b}	733.3^{a}	462.4^{a}	373.3^{a}	347.5^{a}	331.7^{a}	355.1^{a}	920.3^{b}	2914.9^{c}	1061.5^{c}	9724.8^{b}

Column means sharing the same superscript were not significantly different at $p < 0.05$ by Duncan's Multiple Range Test.

Table 6.2 Climatological information during the study period (Hong Kong Observatory)

Climatological information	May	Jun	July	Aug	Sep	Oct	Nov	Dec	Jan	Feb	Mar	Apr
Mean T (^{0}C)	25.5	28.6	29.4	29.4	27.4	27.2	20.8	18.8	15.9	13.1	19.0	23.0
Mean RH (%)	83.3	89.2	86.6	84.3	81.1	82.5	76.8	76.7	75.4	79.3	66.6	76.8
Total precipitation	243.0	243.5	262.0	178.5	114.0	55.0	26.0	31.0	23.5	44.0	31.0	180.5
Tropical cyclone	-	-	-	Pabuk	Francisco	-	-	-	-	-	-	Neoguri
Strong monsoon (hr)	-	2.00	-	9.75	4.75	57.50	15.50	24.25	7.00	44.08	26.42	13.00
Rainstorm (hr)	6.75	8.00*	2.25	2.58	-	-	-	-	-	-	-	4.83**

Pabuk: 32.67 hr alarm signal No.1; 14.42 hr alarm signal No.3; and 7.17 hr alarm signal No.8
Francisco: 32.58 hr alarm signal No.1
Neoguri: 28.42 hr alarm signal No.1; and 28.83 hr alarm signal No.3
* 1.5 hr red warning signal
** 2.08 hr red warning signal; 2.25 hr black warning signal; rest were amber warning signal

The temporal variation of litterfall is also different between the restored sites and fung shui forest site. All the restored sites are characterized by bimodal peaks of production whereas for TO, there is only one peak. In R00, for instance, litterfall production peaked at August 2007 and October through December 2007. The bimodal peaks of R95 and R89 are similar, coinciding with August 2007 and March 2008 while the only peak in TO occurred in March 2008.

There is a tendency for litterfall to increase or peak in August and September 2007, especially in the restored sites. Litter production in this month alone accounted for 13.0% (R00), 12.8% (R95), 11.6% (R89) and 7.5% (TO) of the total production. Hong Kong was attacked by tropical cyclones *Pabuk* and *Francisco* in August and September 2007, respectively. *Pabuk* was a particularly strong tropical cyclone, resulting in the hoist of typhoon signal No.8 for 7.17 hours (Table 6.2). The strong winds could have caused mechanical breakage of the foliage and branches, resulting in elevated litterfall for the month. This effect was more conspicuous in the restored sites, especially R00, than the feng shui forest site. Indeed, a visit to the site after the passage of *Pabuk* found many trees being uprooted, including principally *Acacia auriculiformis* and *A. mangium*. This problem is probably aggravated by shallow rootedness of the species and shallow depth of the substrate as a result of poor workmanship during re-contouring of the borrow area. Because of this, typhoons could probably help to thin out the dense stands in TTEBA. On the other hand, the established stand structure in Tai Om enables the forest to withstand the strong gale. As an endemic forest, the native species in TO must possess longer roots than the exotic legumes in TTEBA.

A correlation analysis was subsequently performed to find out if any relationship exists between annual litterfall and temperature, relative humidity, total precipitation, duration of strong monsoon and rainstorm. With few exceptions, such as litterfall vs. relative humidity in TO and litterfall vs. rainstorm duration in R00, the correlations were generally insignificant (Table 6.3). It is therefore safe to surmise that litterfall is more controlled by phenology of the species than environmental factors.

Table 6.3 Correlation between litterfall and climatological factors

Pearson Correlation	R00	R95	R89	TO
Temperature	0.089	0.564	0.051	-0.201
RH%	-0.142	-0.123	-0.496	-0.597*
Precipitation	-0.444	0.181	-0.399	-0.070
Strong monsoon duration	0.233	-0.113	0.078	0.097
Rainstorm duration	-0.600*	-0.084	-0.476	-0.001

* Correlation is significant at the 0.05 level (2-tailed).

The precipitation recorded from April to September accounted for 85.3% of the annual total (Table 6.2). An attempt is made here to differentiate litterfall production between the wet season (April-September) and the dry season (October-March). In this regard, leaf litterfall for R00 (5,179.2 kg/ha), R89 (3,966.1kg/ha) and TO (5,242.9 kg/ha) were greater in the wet season than the dry season. The only exception was site R95, in which litterfall was higher in the wet season (3,849.6 kg/ha) than the dry season (2,813.5kg/ha). Thus,litterfall in R95 was more easily affected by heavy rain or rainstorms than the other restored sites, possibly due to greater exposure of the site to wind.

Table 6.4 summarizes the relationship between litterfall and various vegetation characteristics of the sites. Among the five vegetation parameters, litterfall is only positively correlated with canopy closure of the stand (p<0.05). Other factors being equal, litterfall will increase with canopy closure or age of the forests. No wonder annual litterfall was higher in TO (9,724.8 kg/ha) than rest of the sites (6,663.1-8,797.6 kg/ha) (Table 6.1). This finding re-affirms that litterfall is mainly controlled by phenology of the species and unusual event of great magnitude, such as typhoon, elevates litterfall in a random manner only. With this in mind, the first peak of litterfall production for R89, R95 and R00 coincided with August and September, during which Hong Kong was struck by the tropical cyclones of *Pabuk* and *Francisco*. Peak production controlled by species phenology probably falls in March, especially for R95, R89 and TO (Table 6.1). For reasons unknown, the same peak for R00 occurred in December (1,350.2 kg/ha) instead of March (525.9 kg/ha). While litterfall is an important ecosystem function, the efficiency of nutrient cycling is not only controlled by its absolute quantity. Also important is the rate of breakdown, which will be investigated in Chapter 7.

Table 6.4 Correlation between litterfall and vegetation characteristics

Pearson correlation	Average DBH (cm)	Total transection area at breast height (cm^2/m^2)	Average height (m)	Total crown area (m^2)	Tree density (DBH>2cm) (ha^{-1})	Canopy closure (%)
Litterfall	0.15	0.65	-0.41	0.37	-0.10	0.98*

* Correlation is significant at the 0.05 level (2-tailed).

To sum up, leaf litterfall varied substantially with sites and seasons. The woodland plantations were characterized by bimodal peak production and the undisturbed forest by a single peak. Owing to the effect of tropical cyclones, which vary in force and time of occurrence, litterfall production is unlikely a suitable indicator for the assessment of restoration progress.

6.3.2 Nutrients in leaf litterfall

Carbon (C), nitrogen (N), sulfur (S) and phosphorus (P) are determined in this experiment because they are the major component elements of leaf litter. Their concentrations in litterfall are likely to reflect the abundance of their supply to the plant from the soil (Waring and Running, 1998). The results are summarized in Tables 6.5-6.8.

Carbon concentrations in the monthly litterfall varied from 49.81-52.07 % in R00, 49.66-51.60 % in R95, 49.38-51.53 % in R89, and 47.43-49.24 % in TO. Mean annual carbon concentrations decreased in the order of R00 > R95, R89 > TO. While the monthly concentrations of carbon remain fairly consistent among the sites, it tends to decrease with age of the plantations.

Nitrogen concentrations in the monthly litterfall averaged 1.05-2.38 % in R00, 0.85- 1.88 % in R95, 1.11-1.64 % in R89, and 1.04-1.36% in TO. Mean annual nitrogen concentrations of the leaf litter were in the descending order of R00 > R89 > R95 > TO. The higher percentages of nitrogen in litterfall at the restored sites can be attributed to the dominance of N_2-fixing legumes including *Acacia auriculiformis*, *A. confusa* and *A. mangium*. *Casuarina equisetifolia*, also planted in TTEBA, is a nitrogen-fixing non-legume. Other factors being equal, young and fast-growing legumes are usually more effective in nitrogenase activity than old and slower-growing counterparts. As the youngest plantation among the restored sites, R00 therefore yielded the highest nitrogen concentration in the leaf litter.

The concentrations of phosphorus in the litterfall were much lower, averaging 0.019-0.114 % in R00, 0.019-0.100 % in R95, 0.017-0.105 % in R89, and 0.045-0.121 % in TO. Mean annual phosphorus concentrations in the leaf litter decreased in the order of

TO > R00 > R89 > R95. The soils under investigation are strongly acidic in reaction, and there is P-fixation by aluminum and iron. Thus, the local soils are characterized by low levels of available P, which is perhaps reflected in the leaf litter. In this connection, the cycling of P may be more efficient in the near climax forest at Tai Om than the restored woodlands aged 8-19 years. The cycling of P in strongly acid soil can be enhanced by root fungus, which is however not the scope of the present study. Furthermore, the development of a closed canopy in TO can reduce soil loss and conserve the element within the ecosystem.

Sulfur concentrations in the monthly litterfall ranged from1.01-1.30 % in R00 to 0.95-1.29 % in R95, 0.94-1.31 % in R89, and 0.90-1.28 % in TO. Mean annual sulfur concentrations in the leaf litter were in the order of R00 > R89 > R95 > TO.

Overall, the mean bioelements concentration decreased in the order of C > N > S > P in the four study sites. This lyotropic series agrees reasonably well with those reported for subtropical China (Hou, 1982; Chan, 2002). Their concentrations, however, are closely related to species composition, age and reserves in the soil.

Table 6.5 Carbon concentrations in the leaf litterfall (n=15)

C %	May 2007	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Jan 2008	Feb	Mar	Apr	Annual mean
R00	51.64 (0.56)	51.78 (0.66)	52.07 (0.43)	51.25 (0.14)	50.94 (0.27)	50.95 (0.21)	51.05 (0.14)	51.44 (0.35)	50.48 (0.19)	50.58 (0.3)	49.81 (0.24)	50.39 (0.39)	51.03 (0.65)
R95	50.24 (0.39)	49.66 (0.62)	50.71 (0.49)	51.6 (0.26)	50.60 (0.34)	50.49 (0.21)	51.12 (0.31)	50.35 (0.23)	50.13 (0.34)	50.6 (0.39)	50.07 (0.16)	49.99 (0.38)	50.46 (0.52)
R89	49.49 (0.38)	49.77 (0.94)	49.45 (0.49)	51.11 (0.61)	51.39 (0.44)	51.01 (0.37)	51.53 (0.38)	51.19 (0.41)	50.31 (0.32)	50.77 (0.25)	50.13 (0.22)	49.38 (0.25)	50.46 (0.80)
TO	49.02 (1.01)	48.07 (0.73)	48.71 (0.48)	47.61 (1.08)	47.43 (0.69)	48.42 (0.18)	47.82 (0.88)	48.40 (0.20)	48.23 (0.36)	49.05 (0.28)	49.24 (0.40)	48.14 (0.36)	48.35 (0.58)

Values in brackets represent standard deviations.

Table 6.6 Nitrogen concentrations in the leaf litterfall (n=15)

N %	2007 May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Jan 2008	Feb	Mar	Apr	Annual mean
R00	1.64 (0.19)	2.21 (0.09)	1.60 (0.08)	1.56 (0.08)	1.53 (0.07)	1.35 (0.06)	1.46 (0.08)	1.52 (0.06)	1.05 (0.14)	2.38 (0.13)	2.03 (0.11)	2.05 (0.09)	1.70 (0.39)
R95	0.85 (0.09)	1.05 (0.12)	1.12 (0.17)	1.45 (0.05)	1.48 (0.17)	1.48 (0.1)	1.24 (0.05)	1.05 (0.05)	1.03 (0.08)	1.61 (0.07)	1.88 (0.28)	1.07 (0.04)	1.28 (0.30)
R89	1.22 (0.17)	1.31 (0.17)	1.23 (0.1)	1.17 (0.05)	1.36 (0.04)	1.37 (0.09)	1.31 (0.06)	1.28 (0.08)	1.19 (0.06)	1.64 (0.03)	1.57 (0.05)	1.11 (0.12)	1.31 (0.16)
TO	1.04 (0.06)	1.28 (0.1)	1.09 (0.06)	1.12 (0.12)	1.36 (0.06)	1.32 (0.09)	1.34 (0.11)	1.13 (0.07)	1.07 (0.13)	1.11 (0.07)	1.12 (0.07)	1.36 (0.09)	1.20 (0.12)

Values in brackets represent standard deviations.

Table 6.7 Phosphorus concentrations in the leaf litterfall (n=15)

P %	May 2007	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Jan 2008	Feb	Mar	Apr	Annual mean
R00	0.039 (0.003)	0.114 (0.015)	0.059 (0.017)	0.093 (0.013)	0.019 (0.004)	0.050 (0.025)	0.036 (0.011)	0.097 (0.015)	0.093 (0.034)	0.078 (0.016)	0.080 (0.021)	0.077 (0.047)	0.070 (0.029)
R95	0.042 (0.008)	0.085 (0.011)	0.055 (0.015)	0.085 (0.015)	0.045 (0.014)	0.036 (0.009)	0.031 (0.009)	0.046 (0.025)	0.100 (0.017)	0.064 (0.012)	0.019 (0.007)	0.036 (0.015)	0.054 (0.025)
R89	0.041 (0.007)	0.105 (0.016)	0.017 (0.007)	0.025 (0.018)	0.092 (0.022)	0.037 (0.02)	0.095 (0.017)	0.044 (0.018)	0.037 (0.026)	0.097 (0.015)	0.053 (0.017)	0.068 (0.017)	0.059 (0.031)
TO	0.083 (0.006)	0.087 (0.008)	0.085 (0.01)	0.055 (0.018)	0.111 (0.016)	0.088 (0.015)	0.121 (0.023)	0.098 (0.029)	0.09 (0.019)	0.045 (0.023)	0.062 (0.012)	0.069 (0.037)	0.083 (0.022)

Values in brackets represent standard deviations.

Table 6.8 Sulfur concentrations in the leaf litterfall (n=15)

S %	May 2007	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Jan 2008	Feb	Mar	Apr	Annual mean
R00	1.14 (0.1)	1.13 (0.04)	0.83 (0.04)	1.01 (0.02)	1.22 (0.05)	1.22 (0.06)	1.09 (0.04)	1.17 (0.05)	1.06 (0.03)	1.19 (0.08)	1.30 (0.05)	1.04 (0.02)	1.12 (0.12)
R95	1.20 (0.05)	1.03 (0.07)	0.77 (0.04)	0.95 (0.08)	1.11 (0.04)	1.29 (0.06)	1.18 (0.03)	1.10 (0.05)	1.13 (0.06)	1.22 (0.05)	1.21 (0.02)	0.98 (0.03)	1.10 (0.14)
R89	0.94 (0.06)	1.24 (0.09)	1.11 (0.04)	1.01 (0.05)	1.24 (0.06)	1.26 (0.04)	1.11 (0.05)	1.17 (0.06)	1.16 (0.06)	1.31 (0.05)	1.20 (0.10)	1.01 (0.04)	1.15 (0.11)
TO	1.28 (0.11)	1.23 (0.05)	1.15 (0.07)	0.90 (0.04)	1.12 (0.04)	1.26 (0.11)	1.09 (0.05)	1.03 (0.14)	1.12 (0.07)	1.27 (0.05)	1.15 (0.09)	1.04 (0.03)	1.14 (0.11)

Values in brackets represent standard deviations.

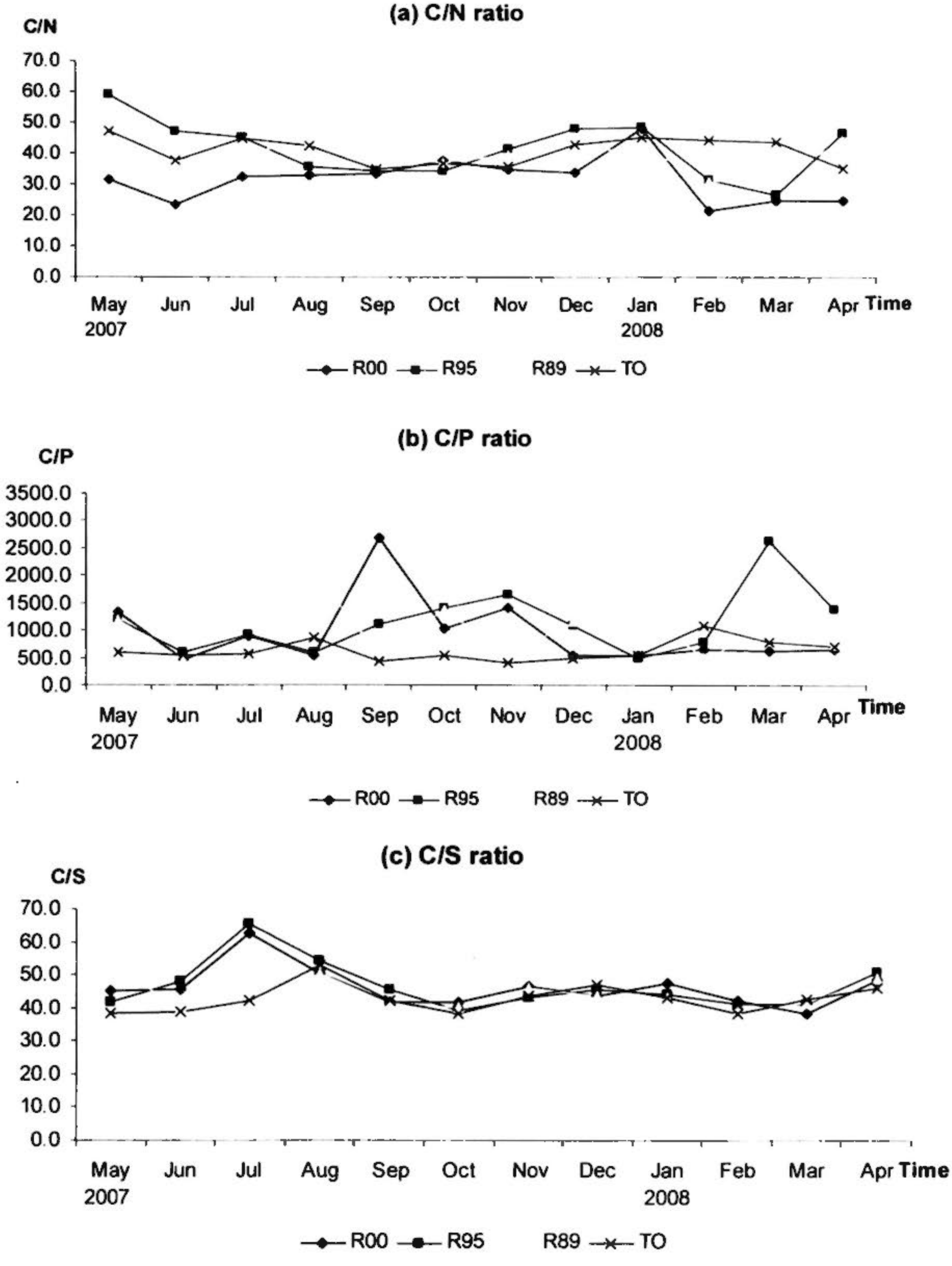


Figure 6.3 Litterfall C/N ratio (a), C/P ratio (b) and C/S ratio (c)

The C/N ratios of litter also varied with sites and seasons (Figure 6.3a). It averaged 21.25:1 to 48.08:1 in R00, 26.63:1 to 59.11:1 in R95, 30.96:1 to 44.49:1 in R89, and 34.88:1 to 47.13:1 in TO. Most of the C/N ratios were greater than 30:1 throughout the year, except for those in February through April and June at R00, and in March at R95. Mean C/N ratio of the litterfall decreased in the order of TO > R95 > R89 > R00. This pattern agrees with the fact that early successional forests are characterized by pioneer species with high litter nitrogen content (Descheemaeker *et al.*, 2006). Indeed, younger *Acacia mangium*, *A. auriculiformis* and *A. confusa* must have accumulated more nitrogen in their foliage than their older counterparts.

The C/P ratios of the leaf litterfall ranged between 454.2:1 and 2681.1:1 in R00, 501.3:1 and 2635.3:1 in R95, 474.0:1 and 2908.8:1 in R89, and 395.2:1 and 1,090.0:1 in TO. Mean C/P ratios of the litterfall decreased in the order of R89> R95 > R00 > TO (Table 6.9).

The C/S ratios of the litterfall averaged 38.32:1-62.73:1 in R00, 39.14:1-65.86:1 in R95, 38.76:1-52.65:1 in R89, and 38.30:1-52.90:1 in TO. Mean C/S ratios of the litterfall was in the order of R95 > R00 > R89 > TO (Table 6.9), suggesting a possible increase of S with age of the forests.

In general, net mineralization will occur at C/N ratios below 20:1, as well as at C/P and C/S ratios below 200:1. In contrast, net immobilization will become dominant at C/N ratios of 30:1, C:P ratios in excess of 300:1, and C:S ratios greater than 400:1 (Stevenson, 1986). Because of this, N and P are expected to decompose slowly in the leaf litter because the overall C/N ratios and C/P ratios are well above 30:1 and 300:1, respectively (Heal *et al.*, 1997; Wittich, 1961). With an average C/S ratio of less than 200:1, the leaf litter can decompose easily.

Table 6.9 Mean annual C/N, C/S and C/P ratios in the litterfall (n=15)

	R00	R95	R89	TO
C/N	31.71 (1.02)	41.81 (1.03)	39.02 (1.40)	41.12 (0.91)
C/S	46.40 (0.43)	46.96 (0.78)	44.53 (0.79)	43.09 (1.16)
C/P	967.9 (121.4)	1279.8 (163.4)	1409.4 (374.7)	632.19 (95.7)

Values in brackets represent standard deviations.

6.3.3 Nutrients return of leaf litter

Monthly return of C, N, P, and S is summarized in Figure 6.4. Carbon return of leaf litter in R00 was higher than all other sites from August to January. The highest carbon return, however, was associated with TO from February to June. Peak return of C via litterfall occurred in December for R00, August for R95, and March for both R89 and TO. The same trends were found for nitrogen, sulphur and phosphorus. It is thus clear that the amount of C returned to the soil is closely related to the quantity of litterfall.

The return of C, N, P and S was higher in the dry season than the wet season, a trend similar to the temporal variation of litterfall (Table 6.10). In R00, for instance, 61.2% of P was returned to the soil in the dry season compared to 38.8% in the wet season. We must, however, treat these findings with caution because litterfall in the wet season tends to fluctuate with species and the occurrence of tropical cyclones in Hong Kong. The only exception was R95 where nutrients return was higher in the wet season than the dry season as a result of the influence of tropical cyclones (Tables 6.1 and 6.10). Indeed, the amount of litterfall recorded for this site in August through October (2,376.1 kg/ha) already accounted for 35.7% of the total. Under normal circumstances, however, leaf abscission takes place as a result of water stress (Swift and Anderson, 1989) and because of the inability of the plant to supply water to leaves and to prevent desiccation and damage to plant structures (Holbrook *et al.*, 1995), the senile leaves are shed.

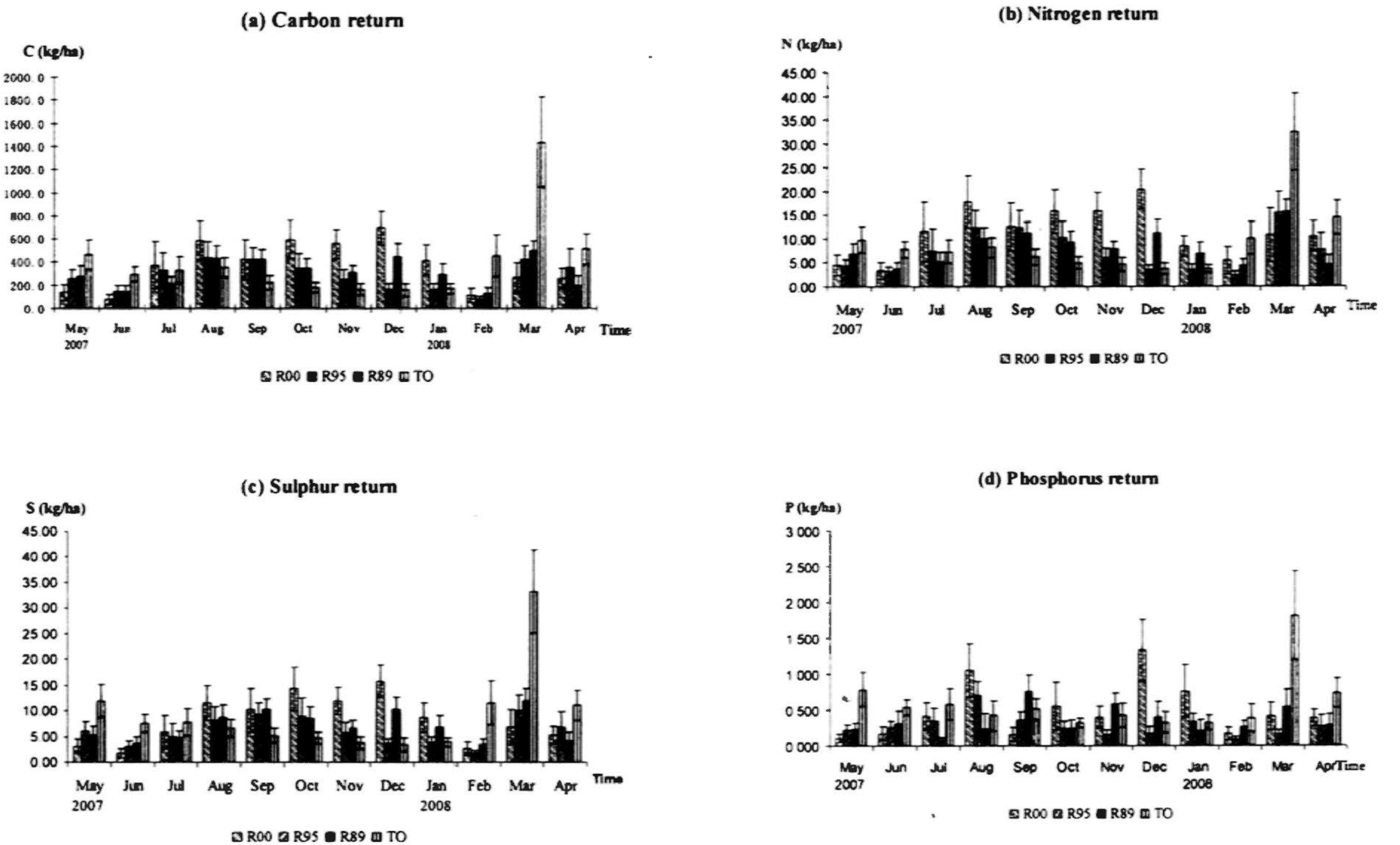


Figure 6.4 Monthly return of carbon (a), nitrogen (b), sulphur (c) and phosphorus (d) from leaf litterfall
Error bars represent one standard deviation.

Table 6.10 The percentage of nutrients returned between the wet and dry seasons (%)

	C return		N return		S return		P return	
	Wet season	Dry season	Wet season	Dry season	Wet season	Dry season	Wet season	Dry season
R00	41.3	58.7	43.9	56.1	38.5	61.5	38.8	61.2
R95	57.9	42.1	53.6	46.4	53.3	46.7	65.6	34.4
R89	45.7	54.3	43.1	56.9	43.5	56.5	46.0	54.0
TO	45.7	54.3	47.4	52.6	45.2	54.8	49.8	50.2

Wet season: April-September; Dry season: October-March

The annual return of C, N, P and S via leaf litterfall during the study period is shown in Table 6.11. The annual return of carbon was significantly higher in TO and R00 than R9[illegible] R89. It is perhaps surprising to find a huge difference in C return among the restored sites. For instance, the amount of C returned via leaf litter in R95 and R89 accounted for only 75% and 80% of that in R00. More or less a similar trend was found for N, P and S, which re-affirms the fact that the amount of nutrients returned via litterfall is dependent on quantity of the litter. It is equally interesting to note that leaf litter nitrogen was highest in R00 (136.8 kg/ha), being followed by TO (113.3 kg/ha), R89 (94.1 kg/ha) and R95 (88.1 kg/ha). The high density of nitrogen-fixing species in R00 must to a contributory factor to this observation. The return of sulphur and phosphorus from leaf litterfall exhibits the same order of TO > R00 > R89 > R95.

Thus, the amount of nutrients returned to soil via leaf litterfall decreased in the order of C > N> S > P for all the sites. This pattern is similar to the lyotropic series of nutrient concentrations.

Table 6.11 Annual nutrient returns from leaf litter (kg/ha) (n=15)

	C	N	S	P
R00	4490.0^b (836.5)	136.8^c (27.5)	97.8^b (18.1)	5.90^c (1.13)
R95	3367.1^a (586.3)	88.1^a (16.6)	72.4^a (12.0)	3.29^a (0.55)
R89	3610.7^a (603.6)	94.1^a (16.1)	81.7^a (13.8)	4.04^b (0.87)
TO	4728.8^b (890.6)	113.3^b (20.1)	111.0^c (19.8)	7.15^d (1.34)

Values in brackets represent standard deviations.
Column means sharing the same letters were not significantly different at p<0.05 by Duncan's Multiple Range Test.

6.4 Conclusions

Based on findings from the present experiment, the following conclusions can be summarized:

1. Litterfall tends to vary with sites and seasons. In the restored sites, especially R95 and R89, litterfall production was characterized by bimodal peaks and in TO by a single peak only. Peak production in August through September in Tai Tong was caused by mechanical breakage during the passage of tropical cyclones. In contrast, litterfall in TO was less easily affected by cyclones. Total litterfall production was in the order of TO $\geq$ R00 > R89 $\geq$ R95.

   With the exception of R95, leaf litterfall was higher in the dry season (October to March) than the wet season (April to September). Litterfall production was only positively correlated with canopy closure of the forests, but not other vegetation characteristics (p<0.05).

2. Under the existing experimental design, there were generally no significant correlations between leaf litterfall and temperature, relative humidity, total precipitation, and the duration of tropical cyclones and rainstorm. Instead,

litterfall in the study sites is mainly governed by phenology of the species while the impact of tropical cyclones is random only.

3. The quantities of carbon, nitrogen, phosphorus and sulfur contained in the litterfall were investigated, which decreased in the order of C > N > S > P. While nutrient return was higher in the dry season than the wet season, it is governed principally by the quantity of litterfall. The concentrations of these elements in litterfall, however, were also affected by species composition, age and soil reserve of a particular element. For instance, the return of nitrogen from leaf litterfall was highest in R00 dominated by young and fast-growing nitrogen-fixing species.

4. Litterfall is unlikely a suitable indicator for the assessment of restoration progress because in this part of the world, the occurrence of tropical cyclones can easily elevate litterfall production through mechanical breakage. In addition to this, the occurrence of typhoon during the wet season is quite unpredictable.

CHAPTER 7

LITTER DECOMPOSITION

7.1 Introduction

Cycling of nutrients is a fundamental component in the functioning of forest ecosystem. The key to the sustainability of any forest ecosystem lies in its nutrient cycling efficiency (Palma *et al.*, 2000). Vitousek (1984) pointed out that nutrient cycling in tropical forests is often characterized as "tight" or "efficient" with at least two distinct patterns. Firstly, relatively large amounts of organic matter could be fixed per unit of nutrient uptake. It could be characterized by high carbon/nutrient ratios in litterfall and other debris. Secondly, most of the nutrients released from trees were rapidly taken up by roots, mycorrhizae and decomposers, and retained within the ecosystem. It could be characterized by low nutrient losses from the system.

Accumulation of organic and inorganic matter, produced by litterfall and its decomposition, is an important factor in both soil formation and nutrient cycling (Van Wesemael, 1993; Palma *et al.*, 2000). Litterfall is the major pathway for the return of dead organic matter and nutrients from the aboveground plant community to the soil (Spain, 1984). The process of litter decomposition plays a crucial role in improving soil fertility and maintaining soil organic matter level through nutrient cycling (Dutta and Agrawal, 2001). The layer of dead and decomposing leaves, woody material, reproductive structures and other organic materials on the surface of the soil, formed as a result of different rates of litterfall and litterfall decomposition, is known as the litter layer (Spain, 1984).

The release of nutrients from litterfall is an important internal pathway of nutrient flux in a forest ecosystem. The biogeochemical cycling of organic matter and mineral elements between soil, vegetation and the surrounding environment are vital to any forest ecosystems. The decomposition of litter can be influenced by the chemical quality of litter, the nature and abundance of decomposers, as well as the environmental conditions in which litter decay takes place (Wang *et al.*, 2008).

The rate of leaf litter decay is often employed to represent litter decomposition. This is because leaf organ accounts for up to 50-80% of the total aboveground litter production (Sundarapandian and Swamy, 1999; Xuluc-Tolosa *et al.*, 2003). Leaf decomposition is an important ecosystem function that regulates nutrient cycling, contributes to soil formation, and provides an energy base for the detritus food web (Elliott *et al.*, 1993). Slow decomposition rates of leaf litter can lead to the accumulation of undecomposed organic matter, retarded rates of soil development, and limited forest productivity (Waring and Schlesinger, 1985; Walker and del Moral, 2003). Therefore, Knoepp (2000) considers leaf litter decomposition rate as one of the functional and biological indicators of ecosystem function, which involves the interplay of vegetation, soil nutrients, micro- and macro-fauna, and microbial population. Borders *et al.* (2006) compare leaf litter decomposition rates between restored riparian forests and mature forests in monitoring the development of ecosystem functions in restored forests. Litter decomposition rates can provide an accurate prediction of soil and site quality or productivity (Johansson, 1994).

On a global scale, litter decomposition rates are regulated by climate (Johansson *et al*, 1995). However, in a specified climatic region, litter chemistry best indicates the rate of leaf litter decay (Aerts, 1977). Moreover, litter species (Keenan, 1996) and litter quality (Taylor *et al.*, 1991) generally control litter decomposition rate, regardless of the environmental condition and microclimate differences. In order to compare the litter decomposition rates between sites, leaf litters of dominant species from TTEBA and Tai Om were selected for the present experiment.

It was found in Chapters 4, 5 and 6 that the restored sites are different from the mature forest site in soil, vegetation and litterfall dynamics. In this regard, several indicators have already been identified for the assessment of restoration progress. However, site quality evaluation of the restored borrow areas will not be complete without knowing their litter decomposition rates. The objectives of this experiment are twofold: (a) to compare the decomposition rates of litter between the restored forests and the mature forest; and (b) to assess the recovery of ecosystem functions in the restored forests. To accomplish these goals, both *in situ* and *ex situ* decomposition of the litter materials would be followed. In other words, site quality evaluation is assessed by comparing the decomposition rates of the same litter under different environment. By holding litter constant, we shall find out if decomposition in terms of percentages weight loss of the litter and nutrients as well as the decomposition constant (k), will vary with site conditions. Results obtained from this experiment shall provide answers to the following specific questions:

(1) How do foliage of *Acacia mangium* and *Schima superba* decompose under their own environment in terms of the percentage weight loss of litter and nutrients?

(2) How will the C/N, C/S and C/P ratios of the foliage litter of *Acacia mangium* and *Schima superba* change during decomposition?

(3) Will *Acacia mangium* litter decompose faster in its own environment than in the mature forest site? Likewise, what will be the response of *Schima superba* litter when left to decay in the restored sites?

(4) What are the decomposition constants (k) and half life (T_{50}) of *Acacia mangium* and *Schima superba* litter?

(5) What local factors will likely affect the decomposition of litter in TTEBA and Tai Om?

(6) How is quality of the restored sites, aged 8-19 years, different from the mature forest site?

(7) Is litter decomposition a reliable index suitable for the assessment of restoration progress?

7.2 Materials and Methods

This section describes the materials and methods employed for the study of litter decomposition in the restored and mature forest sites. These include the litter bag method, analytical procedures for litter loss, as well as the statistical analysis of data.

7.2.1 Materials

This experiment investigates two representative species from the study sites, namely *Acacia mangium* from TTEBA and *Schima superba* from Tai Om. Among the different components of litter, only foliage would be investigated in the present experiment because of its dominance in the litter. The selection of these two species is based on several considerations. First, *Acacia mangium* is an exotic N_2-fixing species that is found in R00, R95 and R89. It is extensively used as a pioneer species to revegetate barren slopes in Hong Kong and south China since the mid-1980s. As a pioneer species, it grows well on almost any degraded lands characterized by low fertility and extreme microclimate. Second, the mature forest in Tai Om has a rich flora that is highly endemic to Hong Kong. It has high species diversity but low species density. Because of this, it is difficult to find a truly dominant species in the near climax forest. *Schima superba* was chosen for this study partly because as a native species, comparison can be made against the exotic *Acacia mangium*. Furthermore, *Schima superba* is a species of the Theaceae family that grows particularly well on strongly acid soils characterized by low cation exchange capacity, P-fixation and high aluminum saturation. It is regarded as an acid soil indicator plant, which is well represented in Hong Kong's flora. In the feng shui forest at Tai Om, the species can be found at the tall- and low-tree layer as well as the understorey layer. This clearly suggests that *Schima superba* is a representative species in Hong Kong's mature to secondary native forests. Third, it is impossible to investigate too many species owing to replication requirement of the study, as well as the constraints of manpower and time. We therefore believe choosing the appropriate species is more important than increasing the number of species in the experiment.

Fresh litter of *Acacia mangium* and *Schima superba* are used in the present study and they were collected from their respective sites for two reasons. First, a large amount of the litter is required and it is impossible to collect enough quantity by the litter trap method. This is especially true for *Schima superba*, whose distribution in the feng shui forest is relatively scattered. Second, fresh litter is preferred to litter collected by litter trap as the latter might have undergone some degree of decomposition.

There are still drawbacks arising from the use of fresh litter in the decomposition study, most obvious being the overestimation of nutrient loss during litter breakdown. Nutrients contained in the fresh litter are expected to be higher than the senescent litter due to limited retranslocation. To minimize this problem, young leaves of current year's growth would not be collected.

The standard litter bag method is employed for the study of litter decomposition (Falconer *et al.*, 1933). The litter bags, each measuring 30 cm (L) x 25 cm (W), are made of heavy-duty nylon with an aperture opening of less than 2.5 mm. The use of nylon litter bag avoids contamination of the litter while the small aperture opening is considered small enough to prevent losses of litter due to breakage, and also large enough to permit the access of decomposers (Xuluc-Tolosa *et al.*, 2003; Borders *et al.*, 2006). These litter bags are readily available in the market, and each has a plastic zip to prevent litter loss.

7.2.2 Field methods

Fresh leaves of *Acacia mangium* and *Schima superba* were collected from their respective sites during March 20-30, 2007, which coincided with abscission season of the species. As noted in Chapter 6, peak litter production occurred in March for R95, R89, TO, and to a certain extent, R00 as well (Table 6.1). This peak is considered to be controlled by phenology of the species instead of mechanical breakage by tropical cyclone. It is, therefore, justifiable to collect the fresh litter in March for the decomposition study, which is expected not to differ greatly from the senescent litter. In the process of collection, only mature leaves of previous year's growth were picked in order to avoid the overestimation of nutrient loss. The fresh leaves were returned to the laboratory, air-dried for two weeks and thoroughly homogenized. Twelve sub-samples of each species were retained for the determination of initial oven-dried weight and chemical composition. Twenty grams each of the air-dried samples of the two species were separately placed into two litter bags. In order to compare the *in situ* and *ex situ* decomposition rates of the foliage, the two litter bags were put together as a paired sample in each of the site. This *in situ* and *ex situ* approach refers to the cross comparison of decomposition of the same litter under different forest environment. For instance, the leaf litter of *Schima superba* was not only placed on the forest floor of TO to decompose, it was also allowed to breakdown in R00, R95 and R89 separately. The same applies to the leaf litter of *Acacia mangium*, which was placed on the forest floor of TO, R00, R95 and R89. By adopting this approach, the decomposition rates of the same litter under different forest environment can be obtained and related to the influence of specific environmental factors. Furthermore, there seems to be abundant standing litter materials in the restored sites but not in the mature forest site. It is not known if litter breakdown in TTEBA is constrained by specific site conditions, such as drought or lack of decomposers, or poor quality of the litter. A cross comparison of the decomposition rates could provide answers to some of these questions and allow an objective analysis of the quality of the study sites, which can in turn highlight the progress of restoration in TTEBA.

Forty-eight pairs of litterbags were randomly placed on the forest floor of each site on April 26, 2007. The bags were secured to the forest floor by metal pins to prevent movement and to ensure proper contact between the bags and the soils. Twelve pairs of bags were harvested from each of the study site at tri-monthly intervals in July 25, 2007, October 25, 2007, January 25, 2008, and April 25, 2008. In other words, 48 pairs of litter lags were retrieved at each collection interval and there were altogether 384 litter bags in this experiment or 192 bags per species.

Litter bags retrieved from the field were returned to the laboratory. Residue litters in the bag were separated from other exogenous materials, such as soils, weeds and microfauna, either manually or washing with deionized water. It is sometimes necessary to wash the residue litters because the soil particles attached to them can easily spoil the findings. This was carried out by dipping instead of soaking the residue litters in deionized water. The residue litters were then oven-dried to constant weight at 80^0C for 48 hours, before being ground to pass through a 0.2 mm sieve for chemical analysis. The twelve sub-samples of oven-dried litter not used in decomposition study were similarly ground and sieved for the analysis of bioelements.

7.2.3 Chemical analysis

The litter materials before and after decomposition was analyzed for total carbon, nitrogen, phosphorous and sulfur in accordance with procedures described in the last chapter.

7.2.4 Statistical analysis

The percentage of litter weight loss was calculated by dividing the leaf litter weight loss by its original oven dry weight. The same equation was used for the calculation of the percentage weight loss of carbon, nitrogen, phosphorus and sulfur.

A single exponential decay model (Olson, 1963) was used to estimate the decay rate constants (*k*) of the litter, which describes the amount of foliage litter lost over time:

$$M_t = M_0 \cdot e^{-kt}$$

Where, M_t: the weight remaining at time t,

M_0: the original mass of litter,

e: the base of natural logarithm,

k: the decay rate constant, and

t: the time.

An annual decomposition rate constant was calculated for each type of litter in each study site by comparing the initial (April 26, 2007) and final weights (April 25, 2008). The differences in decay rate of the two species during the experimental period can then be obtained. The decay rate constants act as the measure of comparison between forest sites in the statistical analysis. The exponential decay model was fitted for each type of litter by the least-squares regression of the logarithm of percentage of mass remaining over time.

The means and standard deviations of the residue litters at each site were estimated by averaging litter mass in the 12 litter bags. The differences in nutrient concentrations between the two species within the same time interval were tested by paired-sample T test set at 95% confidence level. The differences in leaf litter percentage weight loss between sites within the same time interval were tested by Duncan's multiple range tests. The significance level is set at $p<0.05$, unless otherwise specified. Pearson correlation analyses were also carried out to examine the relationship between litter decomposition rate and litter chemical properties, and various climatological parameters.

7.3 Results and discussion

7.3.1 Percentage weight loss of leaf litter

The litters used in this experiment were thoroughly homogeneized before being weighed and placed in the litter bag. Twenty grams of air dried litter was used in the decomposition experiment, and the equivalent oven dried weights were 15.20 g for *Acacia mangium* and 16.41 g for *Schima superba* (Table 7.1). These ODWs were then used for the estimation of percentage weight loss of the litters retrieved during the experiment. In this connection, the ODW percentage of the *Schima superba* litter (82.1%) is slightly higher than that of *Acacia mangium* (76.0%).

Table 7.1 Oven dried weight percentage of fresh litter (n=12)

	ADW (g)	ODW (g)	ODW (%)
Acacia mangium	20.0	15.20 (0.28)	76.0
Schima superba	20.0	16.41 (0.11)	82.1

Values in brackets represent standard deviations.

The cumulative percentage weight loss of foliage litter and bioelements (C, N, S and P) of *Acacia mangium* and *Schima superba* from April 26, 2007 to April 25, 2008 were shown in Tables 7.2 and 7.3, respectively.

Acacia mangium

Acacia mangium grows in TTEBA but not in Tai Om. The percentage weight loss of its foliage litter differed greatly between sites and time of decomposition. After being left to decompose for 3 months in summer (T_1), the percentage weight loss decreased significantly in the order of TO (85.21%) > R89 (64.22%) ≥ R00 (47.63%) ≥ R95 (44.63%) (Table 7.2). Litter decomposition seemed to rise exponentially in the summer months of May through July, but decline substantially thereafter. The magnitude of decline with time was greater in TO than in the restored sites so that one year after decomposition (T_4), the cumulative weight loss of *Acacia mangium* changed to the order of TO (97.80%) ≥ R89 (90.52%) > R00 (72.5%) > R95 (60.59%). The rapid loss of leaf litter in T_1 is expected because litter breakdown is always faster in the hot wet season than the cool dry season, and water-soluble and palatable substances in the litter always decompose faster and sooner than water-insoluble substances. It is also clear that litter decomposition is faster in the mature forest site than the restored woodland sites even though *Acacia mangium* litter was alien to TO. With a closed canopy and high biodiversity, the microclimate in TO is probably more favorable to the growth of decomposers and hence litter decomposition. Indeed, the soil in TO contains higher percentage of soil moisture than the restored sites (see Chapter 5). Among the restored sites, *Acacia mangium* litter seemed to decompose faster in R89 than R00 and R95 so that its cumulative percentage weight loss had caught up with that of TO in T_3 and T_4 (p>0.05). The slower decomposition of leaf litter in R95 could have been caused by the influence of some localized factors, such as lower canopy closure, higher maximum soil temperature and greater fluctuation of soil moisture among the three restored sitese (see Chapters 4 & 5). Despite this exception among the restored sites, cumulative percent weight loss of litter seems to increase with age of the plantation. The main contributory factor could be improvement of the microclimate when litter type and quality are held constant.

Table 7.2 Cumulative percentage weight loss of *Acacia mangium* litter (May 2007-April 2008) (n=12)

Site \ Time	T_1	T_2	T_3	T_4
R00	47.63^a	63.89^a	70.90^b	72.51^b
R95	44.63^a	57.83^a	59.92^a	60.59^a
R89	64.22^b	81.47^b	88.12^c	90.52^c
TO	85.21^c	96.79^c	96.88^c	97.80^c

Column means sharing the same superscript were not significantly different at $p < 0.05$ by Duncan's Multiple Range Test.
T_1: May-July 2007; T_2: May - October 2007; T_3: May 2007 - January 2008; T_4: May 2007 – April 2008.

Schima superba

The percentage weight loss of *Schima superba* litter is comparable to that of *Acacia mangium*, rising exponentially in T_1 but declined gradually thereafter (Table 7.3).

After being left to decompose for one year (T_4), the cumulative percentage weight loss of litter decreased significantly in the order of TO (99.91%) > R89 (85.88%) ≥ R00 (81.69%) > R95 (57.23%). While decomposition was also found to be slowest in R95, there were no significant differences in cumulative percentage weight loss between R00 and R89 halfway the decomposition study. The leaf litter in TO had almost disappeared completely (99.91%) after one year, a trend similar to *Acacia mangium* (97.80%).

Table 7.3 Cumulative percentage weight loss of *Schima superba* foliage litter (May 2007- April 2008) (n=12)

Site \ Time	T_1	T_2	T_3	T_4
R00	44.19^b	71.78^b	78.54^b	81.69^b
R95	35.30^a	49.68^a	51.76^a	57.23^a
R89	54.49^c	81.34^c	85.13^b	85.88^b
TO	94.75^d	97.76^d	99.79^c	99.91^c

Column means sharing the same superscript were not significantly different at $p < 0.05$ by Duncan's Multiple Range Test.

Table 7.4 Comparison of percentage weight loss of litter and bioelements between *A. mangium* and *S.superba* (n=12)

Weight loss (%)	Time / Site	T_1			T_2			T_3			T_4		
		A.mangium	*S.superba*	T-test	*A.mangium*	*S.superba*	T-test	*A.mangium*	*S.superba*	T-test	*A.mangium*	*S.superba*	T-test
Leaf litter mass	R00	47.63	44.19	NS	63.89	71.78	**	70.90	78.54	*	72.51	81.69	**
	R95	44.63	35.30	*	57.83	49.68	*	59.92	51.76	NS	60.59	57.23	NS
	R89	64.22	54.49	NS	81.47	81.34	NS	88.12	85.13	NS	90.52	85.88	NS
	TO	85.21	94.75	**	96.79	97.76	NS	96.88	99.79	**	97.80	99.91	***
C	R00	48.77	44.36	NS	65.32	72.91	**	71.25	78.69	*	72.48	81.99	**
	R95	44.09	35.24	*	59.69	52.84	*	60.05	53.45	NS	62.85	59.57	NS
	R89	64.59	54.83	NS	84.06	82.77	NS	88.03	85.04	NS	91.39	87.21	NS
	TO	85.90	95.41	**	96.98	97.79	NS	97.12	100.00	**	98.01	100.00	***
N	R00	41.48	16.57	***	61.14	57.00	NS	69.56	72.36	NS	70.99	74.43	NS
	R95	39.75	20.18	*	52.07	33.70	***	57.36	55.48	NS	61.40	59.67	NS
	R89	67.64	51.70	*	82.59	76.28	NS	90.51	83.65	NS	90.81	84.40	NS
	TO	85.15	93.52	*	96.70	96.92	NS	96.94	100.00	**	97.59	100.00	***
S	R00	51.93	43.34	**	68.32	74.04	*	74.92	80.63	NS	78.24	85.66	**
	R95	45.51	37.85	*	60.74	53.96	NS	67.14	61.21	NS	69.69	67.07	NS
	R89	64.50	57.50	NS	83.69	84.13	NS	89.61	86.15	NS	92.95	89.30	NS
	TO	88.06	96.07	**	97.18	97.87	NS	97.71	100.00	**	98.28	100.00	***
P	R00	54.99	52.49	NS	70.28	75.15	NS	72.21	79.94	*	75.94	81.83	*
	R95	55.10	42.36	**	66.13	52.81	**	69.46	61.20	NS	71.63	63.41	NS
	R89	77.86	59.21	**	85.37	83.78	NS	90.00	86.87	NS	92.56	88.20	NS
	TO	89.00	95.38	*	97.86	97.85	NS	98.17	100.00	**	99.49	100.00	**

NS: not significant; * $p<0.05$; ** $p<0.01$; *** $p<0.001$

7.3.2 Percentage weight loss of C, N, S and P

The concentrations of C, N, S and P in the foliage litter of *Acacia mangium* and *Schima superba* are summarized in Table 7.5. The lyotropic series of both species decreases in the order of C > N > S > P. As a nitrogen-fixing legume, the foliage of *Acacia mangium* contains more nitrogen (2.60%) than *Schima superba* (1.46%). Besides nitrogen, the two species contain more or less than same carbon, sulfur and phosphorus contents. These baseline data would be used to estimate the percentage weight loss of the bioelements during decomposition.

Table 7.5 The concentrations of C, N, S and P in the foliage litter (ODW, %) (n=12)

	C	N	S	P
Acacia mangium	49.07 (0.27)	2.60 (0.13)	1.31 (0.06)	0.077 (0.002)
Schima superba	47.64 (0.20)	1.46 (0.08)	1.24 (0.04)	0.077 (0.009)

Values in brackets represent standard deviations.

The percentage weight loss of carbon followed exactly the pattern of litter weight loss. This is expected because carbon is a major constituent of the litter biomass. Again, there was an exponential rise in carbon loss three months into the experiment but the rate slowed down substantially thereafter. In T_1, for instance, the percentage weight loss of carbon for *Acacia mangium* decreased in the order of TO (85.90%), R89 (64.59%), R00 (48.77%) and R95 (44.09%) (Table 7.4). The percentage weight loss of carbon seemed to stabilize halfway into the decomposition experiment so that after 12 months of decay, it was in the descending order of TO (100%), R89 (87.21%), R00 (81.99%) and R95 (59.57%) for the *Schima superba* litter (Table 7.4). Again, the weight loss of carbon in the *Schima superba* litter was faster in its own site (TO) than in the restored sites

As a nitrogen-fixing legume, the leaf litter of *Acacia mangium* contains 2.60% nitrogen compared to 1.46% for *Schima superba* (Table 7.5). The percentage weight loss of nitrogen from the leaf litters differed considerably between sites and species. Three months into the decomposition experiment, for instance, nitrogen loss from *Acacia mangium* litter accounted for 39.75% (R95), 41.48% (R00), 67.64% (R89) and 85.15% (TO) of the total (Table 7.4). A slower loss of nitrogen was found for *Schima superba* during the same period of time, in the order of R00 (16.57%), R95 (20.18%), R89 (51.70%) and TO (93.52%). It is thus clear that the percentage weight loss of nitrogen was significantly higher for *Acacia mangium* than *Schima superba*, regardless of difference in site. This seems to suggest that nitrogen loss during decomposition is concentration-dependent. After the litter had been left to decay for one year, however, there were no significant differences in the cumulative loss of nitrogen between the two litters in the restored sites. Furthermore, the loss of nitrogen from the litter of *Schima superba* was generally faster in its own site (100%) than in the restored sites (69.67-84.40%) at the end of the experiment.

The percentage weight loss of sulfur from the foliage litters was similar to that of raw litter, carbon and nitrogen. The loss leveled off after an exponential rise three months into the experiment. The litter of *Acacia mangium* also decomposed faster in the undisturbed forest site (98.28%) than in the restored site (69.69-92.95%) after 12 months (Table 7.4). It re-affirms the fact that TO possesses a more favorable environment for the decomposition of litter. Because of this, sulfur was completely lost from the *Schima superba* litter when placed to decompose on its own forest floor for 12 months.

The percentage weight loss of phosphorus from the leaf litters was not different from the other bioelements. There were no significant differences in phosphorus loss between the two species at T_1 and T_2 in R00, yet the differences became significant in T_3 and T_4 (Table 7.4). In R95 and R89, the loss of phosphorus from the *Acacia mangium* litter was significantly higher than *Schima superba* at T_1 but this difference disappeared at T_3 and T_4. Lastly, phosphorus loss from *Schima superba* litter was significantly higher than *Acacia mangium* in TO for most of the time.

Thus, the cumulative percentages weight loss of C, N, P and S followed closely the pattern of litter weight loss regardless of species. When placed to decompose in the mature forest site (TO), the leaf litter of *Acacia mangium* tended to break down faster than in their own sites (R00, R95 and R89). This trend occurred throughout the study period although the difference was somewhat narrowed in R89, a site that had been restored for 19 years. On the contrary, the decomposition of *Schima superba* litter was suppressed when placed in the restored sites, being more apparent in R95 and R00 than R89. When placed to decompose in its own site (TO), however, the carbon, nitrogen, phosphorus and sulfur contained in the litter were completely lost after 12 months. There are several implications pertaining to this finding. First, litter decomposition in the restored sites is slower than in the mature forest site. The possible constraints include drought, heat, a poorly developed A horizon in the soil, and lack of decomposers. Second, the rapid decomposition of litter in TO explains why standing litter biomass is lowest in this mature forest site (see Chapter 4). More importantly, nutrient cycling is probably more efficient in TO than in the restored sites, especially R00 and R95.

7.3.3 C/N, C/S, and C/P ratios of leaf litter

Litter quality, especially the chemical property, has been considered an important factor controlling the rate of litter decomposition (Singh *et al.*, 1999; Ribeiro *et al.*, 2002; Xuluc-Tolosa *et al.*, 2003; Tateno *et al.*, 2007). In this section, the C/N, C/S and C/P ratios of the litters retrieved during the decomposition experiment would be discussed.

Table 7.6 Mean C/N, C/S, C/P ratios of the decomposing litter of *A. mangium* (Mean ± SD) (n=12)

	Sites	T_0	T_1	T_2	T_3	T_4
C/N	R00	18.9±0.97	16.51±0.46	16.81±0.38	17.9±3.00	17.94±2.35
	R95		17.53±0.43	16.42±2.41	18.02±2.73	18.44±2.32
	R89		20.86±1.11	17.20±0.65	24.1±2.34	17.69±0.55
	TO		18.16±0.97	17.38±1.92	18.3±0.93	15.21±2.28
C/S	R00	37.31±1.71	39.75±1.77	40.71±1.04	42.65±2.08	47.03±1.06
	R95		38.16±0.91	38.36±1.42	45.17±1.05	45.81±1.37
	R89		37.32±1.54	36.80±1.76	42.2±3.09	45.19±1.59
	TO		44.00±1.31	40.47±1.78	47.14±2.06	43.27±1.32
C/P	R00	63.88±1.56	73.62±10.19	76.83±17.83	67.57±15.00	74.09±10.88
	R95		89.19±35.18	81.52±18.96	86.1±18.46	89.61±17.63
	R89		102.1±17.30	77.7±33.10	76.9±4.22	74.35±6.09
	TO		96.30±32.6	100.6±45.4	103.6±19.54	322.3±194.0

The initial C/N ratios of *Acacia mangium* and *Schima superba* foliage were 18.9, and 32.81, respectively (Table 7.6). From T_1 to T_4, the C/N ratio of *Acacia mangium* had decreased from 18.9 to 16.51 and 17.94 in R00. However, the ratio fluctuated slightly in R95 and R89 so that after 12 months of decay, it stayed at 18.44 and 17.69, respectively (Table 7.6). The pattern for TO is similar to R00, recording a decrease from 18.9 at T_0 to 15.21 at T_4. Similarly, the C/N ratio of *Schima superba* litter averaged 20.59-25.62 in R00, 23.3-34.99 in R95, and 23.67-30.68 in R89 during the same period of time (Table 7.7). C/N ratios were not calculated for TO because the litter materials had almost decomposed completely at T_3 and T_4. It is thus clear that after 12 months of decay, the C/N ratios of *Acacia mangium* litter had not changed much compared to its initial value. A completely different trend is found in TO where C/N ratio of the legume litter decreased with time, which agrees reasonably well with the finding of Xuluc-Tolosa *et al.* (2003). Why do C/N ratios of the legume litter behave differently in the restored and mature forest sites during decomposition? It could be a result of the different site quality. We believe the restored sites in TTEBA are still inferior to Tai Om in terms of biodiversity, microclimate, soil chemical and physical properties, as well as microbial population. Litter decomposition during the first 12 months is probably dominated by general purpose bacteria instead of ammonifiers and nitrifiers. This would lead to a relatively slow decline of the C/N ratio of the litter. With a more favorable environment, ammonifiers and nitrifiers are likely more active than general purpose bacteria in the Tai Om soil. This explains why C/N ratio of the *Acacia mangium* litter is narrowed in TO but not as rapidly and conspicuously in Tai Tong. Although the data set is not complete for *Schima superba* litter in TO, its C/N ratio seemed to decrease with time of decomposition, too. This trend is repeated in R00 but is less conspicuous in R95 and R89 for the aforesaid reasons.

The C/S ratios of the litter increased during decomposition, regardless of species, site and season. For instance, the C/S ratio of *Acacia mangium* litter had increased from 37.31 to 47.03 (R00), 45.81 (R95), 45.19 (R89) and 43.27 (TO) after 12 months of decay (Table 7.6). The increase seemed to be faster towards the latter half of the decomposition experiment. The leaf litter of *Schima superba*, for instance, had an initial C/S ratio of 38.33. In R00, it increased gradually to 40.13 at T_2 and jumped to 48.21 at T_4 (Table 7.7).

Table 7.7 **Mean C/N, C/S, C/P ratios of the decomposing litter of *S. superba* (Mean ± SD) (n=12)**

	Sites	T_0	T_1	T_2	T_3	T_4
C/N	R00	32.81±1.81	21.85±0.8	20.59±0.56	25.62±4.6	23.26±2.22
	R95		26.65±1.52	23.3±0.64	34.99±5.31	32.77±3.51
	R89		30.68±1.05	23.67±0.85	27.32±6.08	28.41±4.36
	TO		23.03±0.52	24.42±0.69	ND	ND
C/S	R00	38.33±1.21	37.66±1.78	40.13±1.37	42.22±2.01	48.21±1.14
	R95		40.05±2.32	39.3±3.15	46.17±2.62	46.99±0.79
	R89		40.72±1.18	41.77±7.53	41.53±1.69	45.8±1.16
	TO		44.69±2.24	40.25±2.78	ND	ND
C/P	R00	62.42±8.49	76.79±24.04	68.27±5.67	66.56±6.82	61.12±7.69
	R95		69.31±5.03	61.96±3.22	80.24±28.35	69.15±6.50
	R89		69.89±13.9	64.96±6.21	68.8±3.69	67.39±2.50
	TO		61.29±3.18	64.49±9.08	ND	ND

ND: No data because the foliage litter had decomposed nearly completely.

The C/P ratios of the decomposing litters also widened with time, and the magnitude is greater for *Acacia mangium* than *Schima superba*. *Acacia mangium* has a C/P ratio of 63.88 before decomposition. When placed to decompose on its own sites, the ratio had increased to 74.09 (R00), 74.35 (R89) and 89.61 (R95) after 12 months. In TO, however, it had soared up to 322.3 (Table 7.6). In contrast to this, the C/P ratios of *Schima superba* litter had only changed from 62.42 to 61.12-69.15 in the restored sites and 67.39 in the mature forest site (Table 7.7).

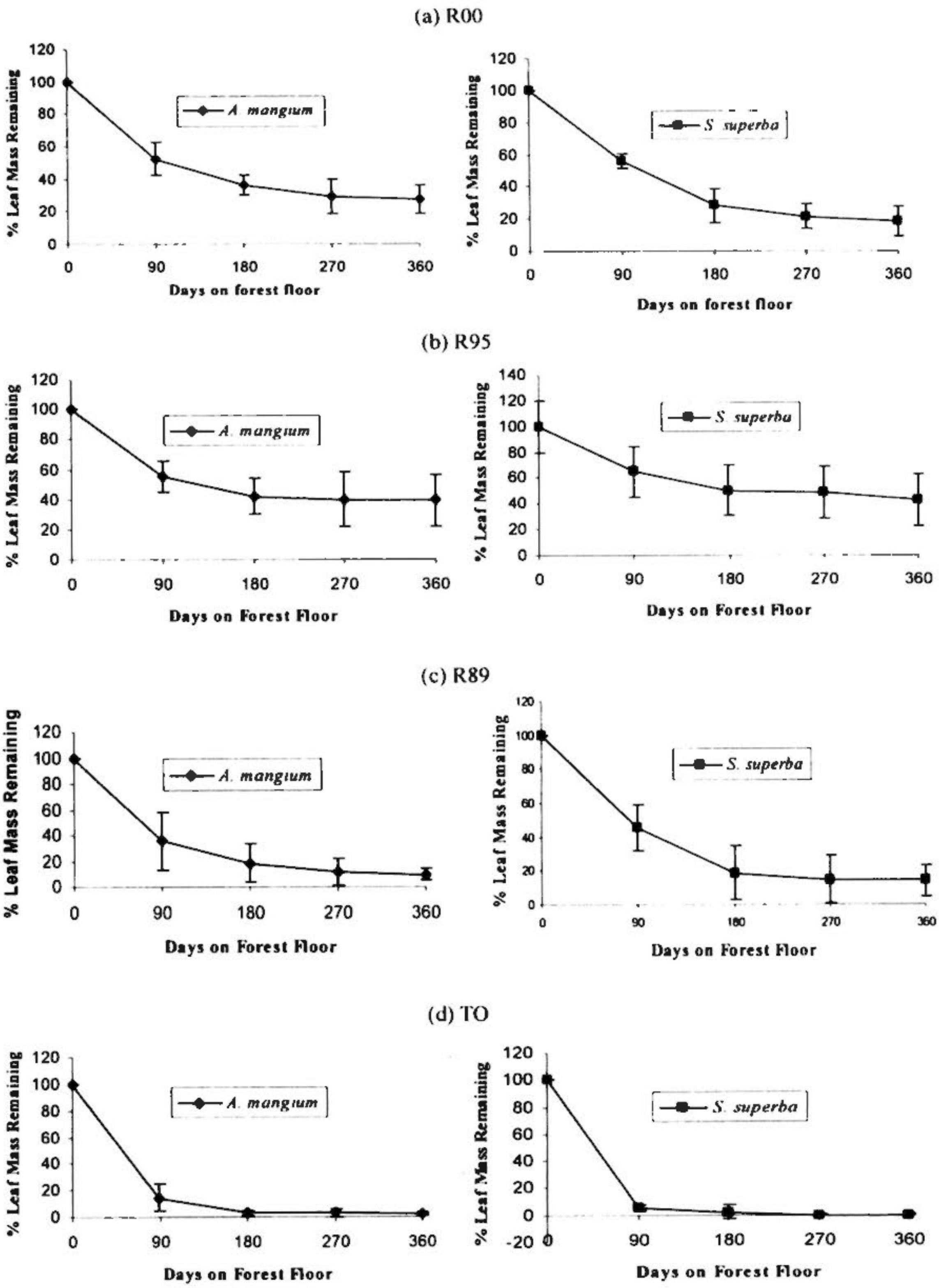


Figure 7.1 Mean percentages weight loss of foliage litter in the different sites (n=12, ±SD)

7.3.4 Litter decomposition dynamics

The percentages of foliage litter remaining on the forest floor over time are summarized in Figure 7.1. At the end of the experiment (T_4), the decomposition rate of *Schima superba* surpassed that of *Acacia mangium* in R00, although the latter's decay rate was faster three months into the experiment (T_1). A similar trend was found for R95 although the overall decay rates seemed to be faster for *Acacia mangium* than *Schima superba*. Overall, decay rates of the two litters were even faster in R89 than R95 and R00 due to the continuous improvement of microclimate and edaphic conditions during ecological succession. The decomposition rates of *Acacia mangium* and *Schima superba* litters were further accelerated in TO so that litter loss approached 100% after one year of decomposition.

Pearson correlation analysis was performed to find out how litter decomposition rate is affected by litter quality and different environmental parameters. Litter quality refers to the C/N, C/S and C/P ratios, while environmental parameters included in this analysis are soil temperature at 10-cm depth, air temperature and relative humidity measured about 2 m above ground level. It was found that the decomposition rates of both species were often negatively correlated with C/N, C/S and C/P ratios of the litter (Table 7.8). Although none of the correlations were significant, probably due to the small sample size, high carbon content in the litter is not favorable to decomposition. For instance, the C/N ratio of *Schima superba* litter (38.33%) is higher than the threshold level of 25-30 recommended for optimal litter decomposition. With a wide C/N ratio, there may not be sufficient nutrients for the decomposers resulting in the attraction of general purpose bacteria only.

On the other hand, litter decomposition rate was positively correlated with soil temperature, air temperature, relative humidity and precipitation (Table 7.8). Although the correlation coefficients were only significant in some of the litter traps, the findings clearly point to the importance of these parameters in enhancing the growth of decomposers or in the case of precipitation, the direct leaching of nutrients from the litter.

Table 7.8 Pearson's correlation coefficient between percentage weight loss of litter and litter quality, as well as environmental parameters

	Species decay rate	C/N	C/S	C/P	Soil Temp	Air Temp	Relative humidity	Precipitation
R00	*A.mangium*	-0.927	-0.938	0.220	0.909	0.915	0.978*	0.813
	S. superba	-0.760	-0.876	0.754	0.989*	0.975*	0.915	0.614
R95	*A.mangium*	-0.540	-0.912	-0.015	0.895	0.933	0.983*	0.929
	S. superba	-0.788	-0.840	-0.593	0.834	0.944	0.924	0.975*
R89	*A.mangium*	0.150	-0.915	0.851	0.906	0.869	0.976*	0.792
	S. superba	0.237	-0.761	0.178	0.995**	0.931	0.936	0.650
TO	*A.mangium*	0.165	-0.722	-0.349	0.914	0.980*	0.887	0.854
	S. superba	NA	NA	NA	0.153	0.085	0.373	0.309

*Correlation is significant at $p<0.05$.
** Correlation is significant at $p<0.01$.
NA: Not available due to complete decomposition of the litter

7.3.5 Litter decomposition rate

The annual decomposition rate constant (k) was calculated by the single exponential decay model. The constant for *Acacia mangium* decreased in the order of TO (1.28) > R89 (0.75) > R00 (0.43) > R95 (0.33) (Table 7.9). The pattern was similar for *Schima superba*, suggesting that decomposition rate is faster in the more established sites (TO and R89) than the less developed sites (R00 and R95). At the site level, the leaf decomposition rates of *Acacia mangium* were faster than *Schima superba* in R00 and TO while the reverse is true in R95 and R89.

The coefficients of determination (r^2) of the two species ranged between 0.637 and 0.946 (Table 7.9). The leaf litter of *Acacia mangium* left to decay in R89 has the shortest half life (T_{50}) of 92.4 days among the rehabilitated sites, followed closely by litter of *Schima superba* in R89 with a half life of 105.0 days. Although the rehabilitation history of R95 is longer than R00, some localized factors must have adversely affected litter decomposition in the site. In short, the decomposition rates of *Acacia mangium* and *Schima superba* litters were faster on the mature forest site than the restored sites due to differences in site quality.

Table 7.9 Litter decomposition rate and litter half-life

Site	Species	Annual decomposition rate constant (k) (yr^{-1})	Coefficients of determination (r^2)	Half life (T_{50}) (days)
R00	*A.mangium*	0.43	0.825	161.2
	S.superba	0.54	0.920	128.4
R95	*A.mangium*	0.33	0.637	210.0
	S.superba	0.27	0.798	256.7
R89	*A.mangium*	0.75	0.899	92.4
	S.superba	0.66	0.838	105.0
TO	*A.mangium*	1.28	0.755	54.2
	S.superba	2.11	0.946	32.9

Pearson correlation analysis was further performed to elucidate the relationship between the decomposition constant (k) and various soil and vegetation parameters. These soil and vegetation parameters, including SOC, TKN, total P, clay percentage, FC and AWC, bulk density and canopy coverage, have been already discussed in Chapters 4 and 5 of the thesis. The results of the Pearson correlation analysis are shown in Table 7.10

Overall, the correlation coefficients are high although not all of them are significant. The decomposition constant (k) of the foliage litter of *Acacia mangium* and *Schima superba* were positively and significantly correlated with SOC, TKN, total P, clay percentage and FC water of the soils. This finding reaffirms the importance of site quality in promoting litter decomposition. These parameters are related to the water- and nutrient-holding capacity of the soils over which the foliage litters were left to decay. They provide the habitat, nutrients and water much needed by the soil decomposers. In contrast to this finding, the decomposition constant of both species was negatively and significantly correlated with bulk density of the soil. High soil bulk density impedes drainage and reduces moisture supply, which will in turn suppress decomposition.

Table 7.10 Pearson's correlation coefficient between litter decay rate constant and soil properties and vegetation characteristics

	Soil pH	SOC	Soil TKN	Soil total P	Soil Clay	Soil field capacity (0.3bar)	Soil available water content	Soil bulk density	Canopy coverage
k_1	-0.905	0.971*	0.956*	0.973*	0.905	0.993**	0.938	-0.994**	0.809
k_2	-0.855	0.978*	0.930	0.878	0.986*	0.934	0.914	-0.960*	0.914

k_1: constant of *Acacia mangium* leaf litter decomposition rate;
k_2: constant of *Schima superba* leaf litter decomposition rate.
*Correlation is significant at $p< 0.05$;
** Correlation is significant at $p<0.01$.

Decomposition encompasses three processes, namely leaching, comminution and catabolism which may be temporally separated or superimposed (Swift *et al.*, 1979). It is well known that litter decomposition is regulated by the chemical composition of litter and climate at the regional scale. However, localized factors can also influence decomposition at the local, or even microscale levels (Berg *et al.*, 2008). These localized factors include soil characteristics, nutrient availability and cycling, plant community composition and structure, and soil fauna (Irmler, 2000; Dechaine *et al.*, 2005). The soil fauna largely controls the decomposition process by digestion and breakdown of litter and the stimulation of micro-organisms. Some of these factors exert their influence by modifying the microclimate. Other factors operate primarily through biochemical or nutritional influences on microbial metabolism. These site-specific factors not only influence microbial metabolism, they also alter the composition of the microbial community.

7.4 Conclusions

In this chapter, the decomposition rates of *Acacia mangium* and *Schima superba* foliage litters were investigated in terms of the percentages weight loss of litter biomass and nutrients with time. In addition to this, the decomposition constants (k), half-life (T_{50}) and factors governing decomposition of the two litters were also estimated. Based on the findings obtained from this experiment, the following conclusions can be drawn.

1. The decomposition of *Acacia mangium* and *Schima superba* foliage litters rose exponentially in May-July but slowed down thereafter. After being left to decompose for 12 months (T_4), the percentage weight loss of *Acacia mangium* litter decreased in the order of TO (97.80%) > R89 (90.52%) > R00 (72.50%) > R95 (60.59%). As for *Schima superba*, the percentage weight loss at the end of the experiment was in the order of TO (99.91%) > R89 (85.88%) ≥ R00 (81.69%) > R95 (57.23%). At the species level, therefore, the percentages weight loss of *Acacia mangium* and *Schima superba* litters were not distinctly different. At the site level, however, the litter of *Acacia mangium* decomposed significantly faster in R89 than in R00 and R95.

2. The C/N, C/S and C/P ratios of the *Acacia mangium* and *Schima superba* litter responded to decomposition differently. After one year of decomposition, the

C/N ratio of *Acacia mangium* had not changed much in the three restored sites but had narrowed down in the mature forest site. We believe the restored sites are dominated by general purpose bacteria while the mature forest site by ammonifiers and nitrifiers in the soil. On the other hand, the C/S and C/P ratios of the litters had widened during decomposition, regardless of species and sites.

3. Both litters of *Acacia mangium* and *Schima superba* decomposed faster in the mature forest site (TO) than in the restored sites (R89, R95 and R00). After one year of decomposition, for instance, the cumulative percentage weight loss of *Acacia mangium* reached 97.80% in Tai Om and 60.59-90.52% in TTEBA. The corresponding values for *Schima superba* were 99.91% in the mature forest site and 57.23-85.88% in the restored sites.

4. The annual decomposition constant (*k*) varied with species and site. At the site level, the decomposition constant of *Acacia mangium* decreased in the order of TO (1.28) > R89 (0.75) > R00 (0.43) > R95 (0.33). A somewhat similar pattern is found for *Schima superba*, in the descending order of TO (2.11) > R89 (0.66) > R00 (0.54) > R95 (0.27). The constant is positively and significantly correlated with SOC, TKN, TP, clay percentage and FC water of the soils but negatively and significantly with soil bulk density.

   The half life (T_{50}) of the decomposing litters followed exactly the same pattern, being shorter in the mature forest site (32.9-54.2 days) than the restored sites (92.4-256.7 days) regardless of species difference. It is thus clearly demonstrated that decomposition rate is faster in the established forest site, such as TO and R89, than the younger restored sites (R95 and R00).

5. The decomposition of foliage in TTEBA and Tai Om are likely affected by the C/N, C/S and C/P ratios of the foliage litter, as well as soil temperature, air temperature, relative humidity and precipitation of the site although not all the correlation coefficients are significant.

6. After being revegetated for 8-19 years, the quality of the restored sites is still inferior to the mature forest site in terms of stand conditions, microclimate, as well as soil chemical and physical properties. Given a slower decomposition rate, the efficiency of nutrient cycling is probably slower in the restored sites than the mature forest site.

7. Litter decomposition is likely a reliable indicator for the assessment of restoration progress. It includes three sub-indices, namely the cumulative percentage weight loss of foliage litter, annual decomposition constant (k), and half life of the decomposing litters. Overall, cumulative weight percentage and annual decomposition constant increased while half life of the decomposing litters decreased with age of the restored plantations. There is improvement in quality of the restored sites over time, being more conspicuous in R89 than R00 and R95.

CHAPTER 8

DISCUSSIONS AND CONCLUSIONS

8.1 Introduction

The thesis investigated the successional development of three restored subtropical forests in Hong Kong. Established on severely degraded borrow areas, the woodland plantations are dominated by exotic broad-leaved species, including nitrogen-fixing legumes and non-legumes. The borrow areas are classified as soil destruction sites, where the top 8 meters of soil and the aboveground vegetation were removed. Because of this, the soil destruction site is considered most difficult to restore in terms of the recovery of ecosystem function and structure. The growth substrate is coarse-textured, infertile and relatively unstable. The main objective of this thesis is ecological assessment of the established woodland plantations, involving the selection of indicators that can be used to assess restoration progress. These indicators can be related to ecosystem structure or function, including soil, vegetation and nutrient cycling etc. The study is prompted by the knowledge gap that ecological assessment of restored forests is virtually absent in the subtropical region and Asia. As the strategy of ecological restoration or rehabilitation has never been clearly defined in Hong Kong, we bench mark successional development of the three subtropical forests, aged 8-19 years, against a mature forest aged 300+ years. The study objectives, research questions and justifications for the selection of study sites have already been addressed in Chapter 1 of the thesis. The purposes of this chapter are fivefold: (a) to summarize the major findings of the thesis, (b) to develop an assessment system that can be employed for the assessment of restoration progress in the subtropical region, (c) to revisit the strategy of ecological rehabilitation in Hong Kong, (d) to discuss some limitations of the study, and (e) to suggest some tropics that are worthwhile for future research. It is hoped that the thesis could contribute to the theory of restoration ecology.

8.2 Summary on major findings of the thesis

This section summarizes some major findings of the thesis, with special emphasis on the characteristics and properties of the three restored woodlands in TTEBA and how they are compared with that of the mature forest in Tai Om. The findings shall cover the different aspects of the soil, vegetation and ecosystem functions, and equal weighting is given to the restored woodlands and the mature forest. Because of this, the section focuses more on the static aspect of these ecosystem attributes while the development of a comprehensive indicator system shall be dealt with in section 8.3.

8.2.1 Vegetation dynamics

The overstorey species in the restored sites are dominated by planted exotics of *Acacia confusa*, *A. mangium*, *Casuarina equisetifolia* and *Eucalyptus torelliana*; while planted native species are few and confined to *Cinnamomum camphora* and *Liquidambar formosana*. The number of woody species increases with age of the plantations, as is species diversity.

Tree density decreased in the order of R89 > R00 >R95. The percentage of trees with DBH>2 cm was higher than those with DBH≤2 cm in all the restored sites. The DBH, transaction areas, crown area and canopy closure seemed to increase with age of the woodland plantations. The restored sites are dominated by low trees of 3-12 m in height, yet their importance decreased with age of the plantations. Hence, there is a steady increase of canopy trees >12 m with time. Unlike the *feng shui* woods, shrubs are absent in the restored sites, and the major invading species, though few, include grass (*Miscanthus floridulus*), fern (*Dicranopteris linearis*), vine (*Embelia laeta*), climber (*Psychotria serpens*) and cellulose-decomposing fungus (*Ganoderma applanatum*). The latter three are more commonly found in R95 and R89 than R00.

Standing litter on the forest floor decreased in the order of R00 (23.99 t/ha) > R95 (15.87 t/ha) > R89 (7.93 t/ha) > TO (6.14 t/ha). The importance of foliage declines with age of the forests while the reverse is true for the fruits component, suggesting that seed bank in the restored sites is relatively small and unimportant. The restored sites lag behind the *feng shui* woods in terms of biodiversity (species number and life forms) and regeneration potential.

8.2.2 Soil properties and nitrogen mineralization

Ecological rehabilitation with exotic species is capable of ameliorating the degraded soils, and the positive effects tend to vary with age of the plantations. While there is no change in soil texture, ecological rehabilitation acidified the soil, lowered bulk density, elevated field capacity and available water contents, and lowered soil temperature at the top 10 cm layer. After being restored for 19 years, R89 was better buffered against moisture fluctuation than R95 and R00 probably due to increased soil organic matter content and reduced erosion of the finer soil particles.

There was a gradual build-up of organic carbon, especially in the top 10 cm soil after restoration, being more significant in R89 and R95 than R00. Intra-layer differences in SOC were widened in R89, too. A similar trend was found for TKN, which also increased, albeit slowly, with age of the plantations.

Ammonification and nitrification were detected in the restored soils. While NH_4-N predominated over NO_3-N in the restored soils, net N mineralization was higher in R89 than R95 and R00. Thus, there seems to be an accumulation of mineral nitrogen 13 (R95) to 19 (R89) years after restoration. No discernible pattern was found for the leaching loss of mineral nitrogen although it might be concentration-dependent.

The restored sites in TTEBA had only been rehabilitated for 8-19 years; their soil quality is generally poor compared to the mature forest. With a distinctly lower net nitrification rate, the restored sites still lag behind the undisturbed forest in the efficiency of nutrient cycling and energy flow.

8.2.3 Leaf litterfall production and nutrient fluxes

Litterfall tends to vary with sites and seasons. In the restored sites, especially R95 and R89, litterfall production was characterized by bimodal peaks and in TO by a single peak only. Peak production in August through September in Tai Tong was caused by mechanical breakage during the passage of tropical cyclones. In contrast, litterfall in TO was less easily affected by cyclones. Total litterfall production was in the order of TO $\geq$ R00 > R89 $\geq$ R95.

With the exception of R95, leaf litterfall was higher in the dry season (October to March) than the wet season (April to September). Litterfall production was only positively correlated with canopy closure of the forests, but not other vegetation characteristics ($p<0.05$).

Under the existing experimental design, there were generally no significant correlations between leaf litterfall and ambient temperature, relative humidity, total precipitation, and the duration of tropical cyclones and rainstorm. Instead, litterfall in the study sites is mainly governed by phenology of the species while the impact of tropical cyclones is random only.

The quantities of carbon, nitrogen, phosphorus and sulfur contained in the litterfall were investigated, which decreased in the order of C > N > S > P. While nutrient return was higher in the dry season than the wet season, it is governed principally by the quantity of litterfall. The concentrations of these elements in litterfall, however, were also affected by species composition, age and soil reserve of a particular element. For instance, the return of nitrogen from leaf litterfall was highest in R00 dominated by young and fast growing nitrogen-fixing species.

8.2.4 Leaf litter decomposition

The percentages weight and nutrients loss of *Acacia mangium* foliage litter differed greatly in its own environment (R89 > R00 > R95). The percentage weight loss of *Schima superba* in its own environment (TO) was significantly higher than that of *Acacia mangium* in the restored sites. Litter decomposition rose exponentially in the summer months of May through July, but slowed down substantially thereafter. The cumulative percentage weight loss of litter generally increased with age of the plantations.

The leaf litter of *Acacia mangium* tended to break down faster in the mature forest site (TO) than in their own sites (R00, R95 and R89). On the contrary, the decomposition of *Schima superba* litter was slightly suppressed when placed in the restored sites, being more apparent in R95 and R00 than in R89.

The decomposition constant (*k*) for *Acacia mangium* and *Schima superba* decreased in the order of TO > R89 > R00 > R95. In the restored sites, the shortest half life (T_{50}) of leaf litter was associated with *Acacia mangium* (92.4 days) and *Schima superba* (105.0 days) in R89. The T_{50} of leaf litter in R95 was 210.0 days for *Acacia mangium* and 256.7 days for *Schima superba*, which were much longer than their counterparts in R00. In the mature forest site, half life of the decomposing litters was reduced to 54.2 days (*Acacia mangium*) and 32.9 days (*Schima superba*).

Thus, litter decomposition was positively correlated with soil organic carbon (SOC), total Kjeldahl nitrogen (TKN), total phosphorous (TP), clay content and field capacity (FC) water of the soils, but negatively with soil bulk density. These attributes are closely related to the nutrient-holding and water-holding capacities of the soil.

After 8-19 years of restoration, there are considerable improvements in the vegetation performance, soil physical and chemical properties, and nutrient cycling efficiency of the restored sites. The improvements are more conspicuous in R89 than in R00 and R95. Notwithstanding this, overall qualities of the restored sites still lag behind the mature forest site.

8.3 Development of an assessment system for restored subtropical forest

According to findings of the thesis, I have developed an assessment system suitable for use in the evaluation of restored subtropical forests in this part of the world

(Figure 8.1). The assessment system consists of multiple indicators that cover both ecosystem structure and functions, static and dynamic attributes, as well as biotic and abiotic components of an ecosystem. Some of these indicators require longer-term monitoring, such as litter decomposition and nitrogen mineralization, while others require one-off sampling along the vegetation gradient only. The ultimate objective of this assessment system is to verify if a restored ecosystem is moving towards an appropriate trajectory of development in terms of ecosystem structure and functions.

Many parameters pertaining to forest structure and functions have been considered in this study. While some of them show signs of changes or improvements with age of the plantations, such as the build-up of SOC and TKN, others are not as consistent. It is undeniable that many of the indicators investigated in this study are not inherently suitable for the assessment of restoration progress. Besides this, micro-site variations might also have complicated the issue even though the three restored sites had been carefully chosen in terms of their age, geology, altitude and planted species etc. There are influences by localized factors unaccounted for in the thesis, as in site R95. This is always a problem in ecological studies. Recognizing this difficulty and limitation, specific indicators suitable for the assessment of restoration progress are summarized below.

In terms of vegetation dynamics, species diversity, standing litter biomass and its changing composition, transection areas, standard deviations of tree height, total crown area, saplings density and, to a certain extent, species invasion can be useful indicators of ecological assessment. The standard deviation of average tree height (DBH>2cm) should be a good indicator for the evaluation of changing complexity of the vegetation structure. This is true on the condition that the restored stands begin with seeds or seedlings. Timber transection area at breast height per unit area (TABH), which represents the biomass of plantation, is another indicator for vegetation growth performance. Biodiversity is one of the most popularly used indicators. Higher biodiversity like the mature ecosystem is one of the final objectives of ecological restoration, in order to achieve ecological, ethical and aesthetic values. Another important indicator, which is a symbol of the ability of natural regeneration of the plantations, is the size of the seed bank. It tends to increase with age of the restored ecosystem.

As a fundamental element of forest ecosystem, soil quality should not be ignored in the assessment of restoration progress. It helps to ensure that site productivity and soil functions are maintained. The potential indicators for ecological assessment related to soil quality must reflect the interactions between soil and plants, as well as soil and fauna. However, soil samplings and analyses are time-consuming, labor-intensive and somewhat disturbing environmentally. Thus, indicators that are easy to measure and sensitive to ecological succession in the long-term monitoring would be most useful. In view of the fact that planting can accelerate the recovery of soil productivity, several soil parameters are considered suitable for the assessment of restoration progress. They include bulk density, field capacity, available water contents, SOC and TKN. Equally useful is soil reaction pH, provided that the substrate prior to planting is neutral to alkaline in reaction. Other factors being equal, total mineral nitrogen content is more suitable than NH_4-N or NO_3-N alone for the assessment of restoration progress. Since nitrification is more important to enhance the efficiency of nitrogen cycling and more helpful to accelerate energy flow in the process of forest rehabilitation, net nitrification rate is another suitable indicator in terms of the efficiency of nutrient cycling and energy flow. Net N mineralization in the restored sites of R89 and R95 seems to be higher than TO during the 2-month incubation period. If this finding is not confined to the dry season, it suggests that mineral nitrogen is accumulating in the soils of Tai Tong that have been revegetated for 13-19 years. An improvement of soil productivity would pave the way for ecological succession.

Litter decomposition is likely a reliable indicator for the assessment of restoration progress. It includes three sub-indices, namely the cumulative percentage weight loss of foliage litter, annual decomposition constant (*k*), and half life of the decomposing litters.

Ecosystems are complex, and no two ecosystems are identical, at least not when examined in fine resolution (SER, 2004). For that reason, no restored ecosystem at a project site can ever be identical to any single reference. The number of ecosystem variables that can be used in an evaluation is too great for all to be measured within a reasonable period of time. The selection of which variables to assess and which to ignore requires pragmatism and value judgment by the evaluator.

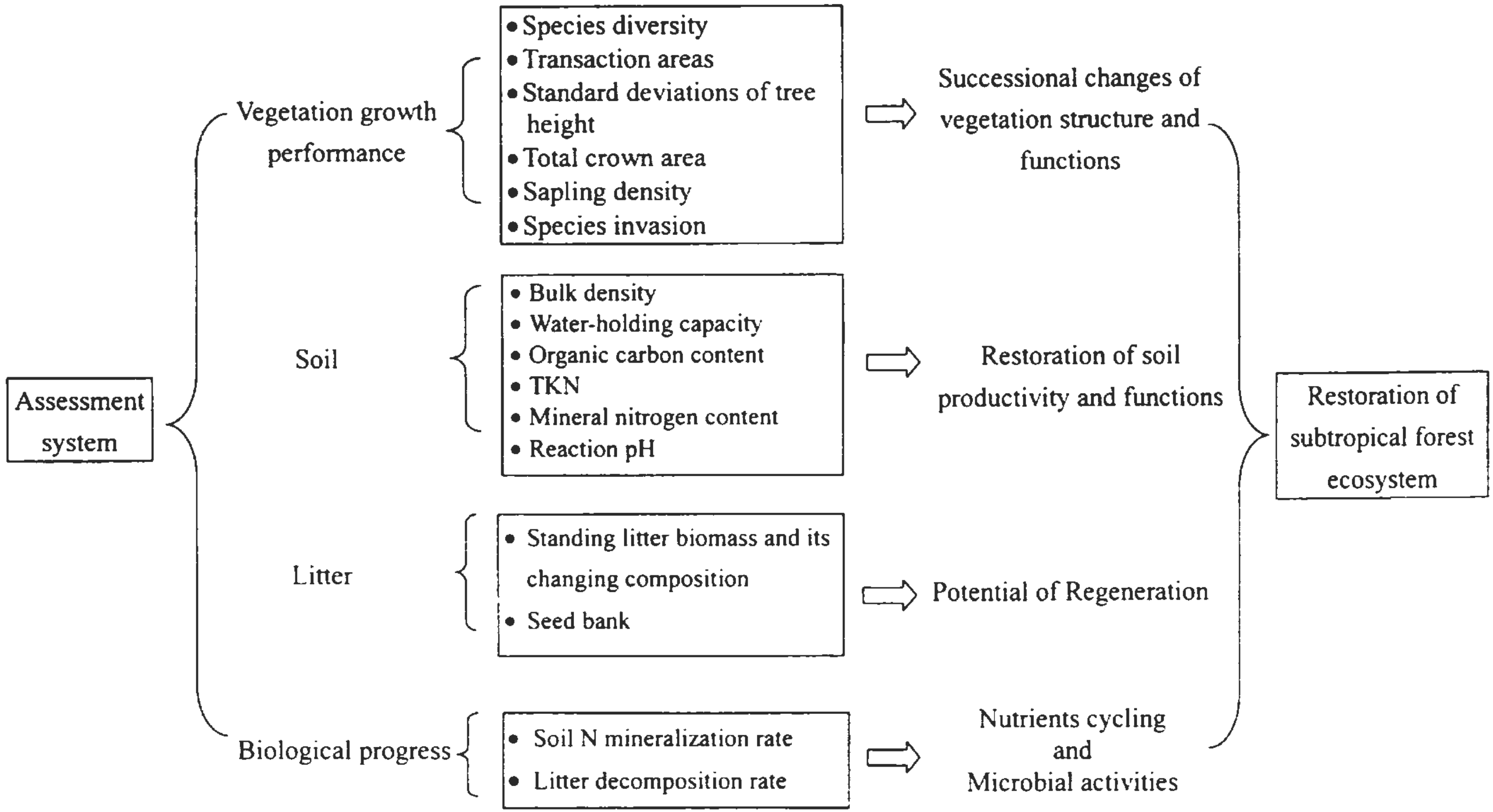


Figure 8.1 The framework of assessment system for restored subtropical forests

8.4 Implications of the study

8.4.1 Redefining HK's restoration goals

The objectives of reforestation have evolved continually. Taking the United Kingdom as an example, initially the aims were largely strategic (increasing timber reserves), then largely economic, followed by rapid change towards today's multiple function forests aiming to provide the public with environmental, social and economic benefits (Patternson, 2002). In Hong Kong, forest restoration goals are only vaguely defined on the website of AFCD although afforestation had been carried out as early as the 1870s. The major purposes of reforestation have shifted from visual amenity, erosion control, production of firewood, improvement of water supplies to the current interest in biodiversity enhancement and ecological restoration[1] .

Besides the borrow area, the abandoned quarry is another typical soil destruction site in Hong Kong, which receives most attention and prompt revegetation after excavation. The plan to rehabilitate quarries was formulated in 1989 as an outcome of the Metroplan Landscape Strategy for Urban Fringe and Coastal Areas, which has identified quarries as areas of degraded landscape requiring rehabilitation (GEO, 2007). The rehabilitation works involve major recontouring, mass tree and shrub planting and erosion control. Upon completion of the quarry rehabilitation works, attractive greened areas will be formed for a variety of uses beneficial to the community (GEO, 2007). In the course of quarry rehabilitation, the slopes are re-vegetated extensively with suitable vegetation with a long-term objective of creating anticipated climax vegetation

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[1] AFCD Website: http://www.afcd.gov.hk/english/conservation/con_flo/con_flo_enr/con_flo_enr.html

communities that will blend ecologically and aesthetically with the surrounding natural vegetation and providing favourable habitats for wildlife (CEDD, 2008). Since the public's concern of the ecological value of forest landscape is rising continuously, the near climax or climax forest with high biodiversity of native species is definitely the final objective of ecological restoration. On account of the final restoration goal, the feng shui woodland was selected as the best and most representative reference site.

The goal of ecological restoration is hereby redefined that the restored ecosystem should be resilient and self-sustaining with respect to structure, species composition and functions, as well as being able to integrate with the surrounding landscape and support sustainable livelihoods.

8.4.2 Strategies of ecological rehabilitation on soil destruction sites in subtropical region

There is still no explicit and comprehensive strategy of landscape revegetation in Hong Kong. From the General Specifications for Civil Engineering Works (2006) and my personal communication with staffs of CEDD and AFCD, the current strategy of borrow area revegetation in Hong Kong is summarized in Figure 8.2.

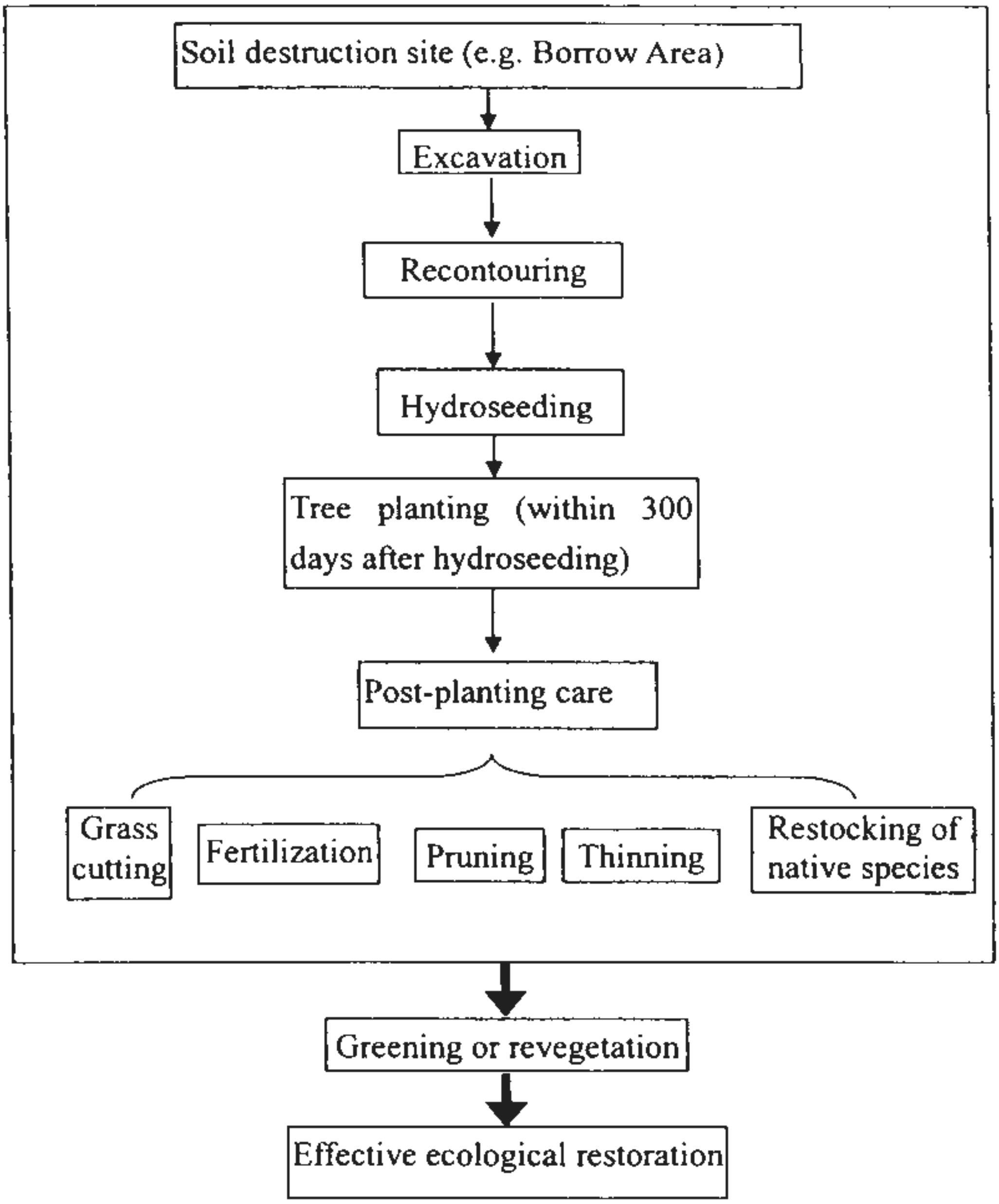

Figure 8.2 Current strategy of ecological restoration on soil destruction site in Hong Kong

After excavation, the borrow area is recontoured for later restoration works. The landscape rehabilitation or restoration starts with hydroseeding, which the grass seeds are mixed with water and fertilizers, and then sprayed on to the surface of the seedbed to form a temporary protective cover. After about 300 days, when the grass cover is well established, the grass cover is trimmed, followed by the planting of tree/shrub seedlings.

AFCD will take up the maintenance responsibility of the restored borrow area approximately 12 months after planting or upon the establishment of vegetative species by the landscaping contractor of CEDD. The post-planting care includes: (a) grass-cutting to reduce competition for soil nutrients; (b) application of fertilizers to replenish nutrients in the soil and promote growth of the plant; (c) pruning to improve the health and structure of the trees and the woodlands; (d) thinning to remove weak trees and to improve the growth rate and health of the remaining trees, and (e) enrichment planting of native species to increase biodiversity of the site (HKSAR, 1992). In general, fertilizers are applied during hydroseeding, planting of seedlings and also in the post-planting stage[2]. During hydroseeding, Nitrophoska 15:15:15 fertilizer or an approved equivalent is incorporated into the hydroseeding mixture. During the stage of planting of seedlings, Nitrophoska 15:9:15:2, a slow-release granular fertilizer is mixed into the backfill of the planting pits. In the post-planting stage, Nitrophoska 12:12:17:2 granular fertilizer or an approved equivalent is applied to the seedlings (HKSAR, 1992).

It is found in the present study that *Acacia mangium* and *A. confusa* are fast growing pioneer species on the borrow areas in Tai Tong. All the three degraded sites had finished the initial stage of forest rehabilitation or the rapid recovery of a vegetative cover. These *Acacia* species are considered ideal pioneer species in forest rehabilitation in Hong Kong. Although the three study sites had different pioneer species, they all achieved the foremost goal of green cover or vegetation re-establishment in a short period of time. With increasing concern of natural and biodiversity conservation, a green cover can no longer satisfy the desire of different stakeholders and ordinary people. The strategy for landscape restoration on soil destruction sites in Hong Kong should change

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[2] Personal communication with Samuel Lam, Nature Conversation Officer, AFCD on 25, March 2009.

with time. The recruitment of native tree or shrub species in the understory is the next key step.

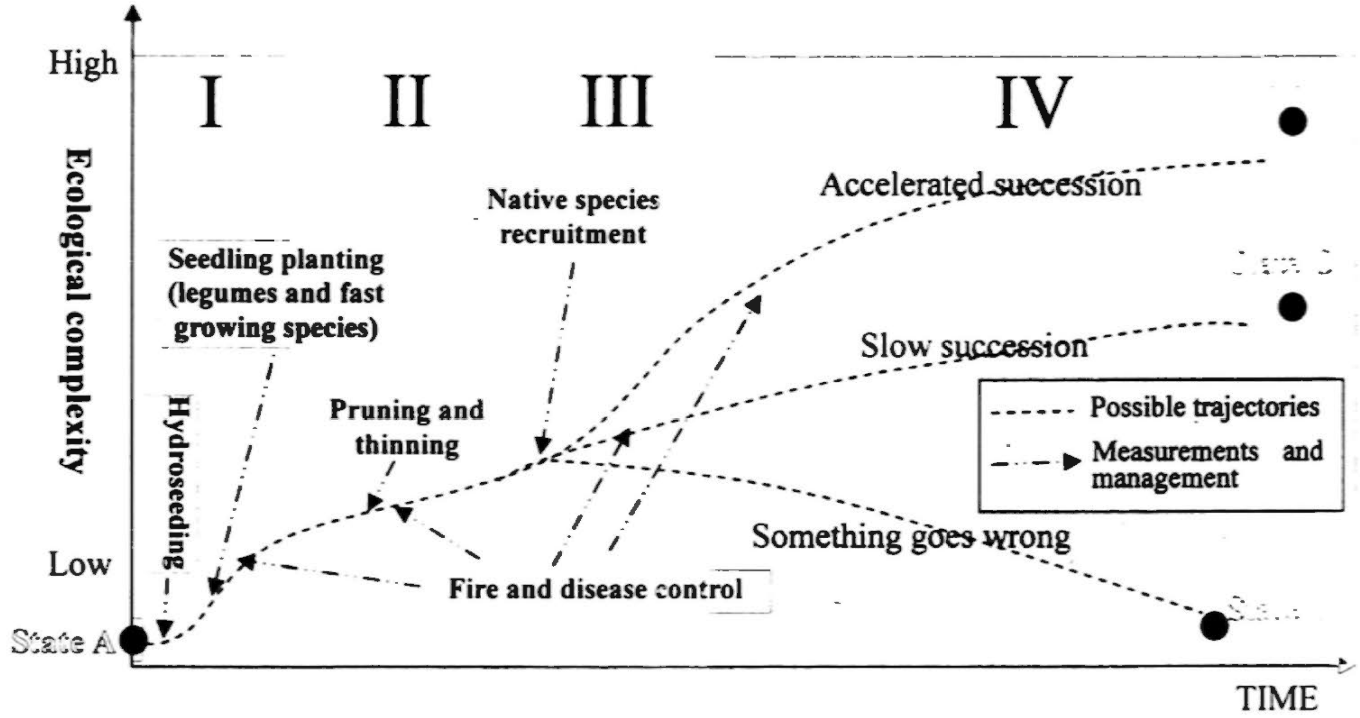
By planting nitrogen-fixing species, the physical, chemical and biological properties of soil can be ameliorated to some extent along the vegetation chronosequence. Yet, there are still huge differences in soil quality compared to the feng shui woodland. Soil nutrient contents can be enhanced by adding fertilizer and planting N-fixing species in the early stage of the forest rehabilitation. However, it is time-consuming to improve the soil physical properties, such as bulk density and hydraulic conductivity by roots and biotic processes. Soil chemical properties tended to be improved more easily than physical properties in the rehabilitation of soil destruction sites. Appropriate measures should be carried out to accelerate the process of soil rehabilitation, including the addition of soil amendment materials, and retaining the upper soils of the borrow area for planting use after excavation.

In this study, ecological succession of the restored vegetation, improvement of soil properties and nutrient cycling between the soil-pant subsystem are detected along the vegetation chronosequence aged 8-19 years. The result indicated that it is feasible to adopt the current strategy of forest recovery. However, there is always room for improvement. Firstly, it is advisable to stockpile the top 20 cm soil before excavation of the underground fill material, and to use it as a planting substrate later. There are pros and cons in this practice. It may increase the project cost but also accelerate ecological succession, especially the recovery of soil productivity. The top soil is not only the seed bank of native species, but is also rich in plant nutrients and organic matter. Moreover, the original top soil is better in soil texture and soil bulk density for flora and fauna growth than the skeletal DG. Secondly, it is better to use more native fast-growing species, such as species from the *Lauraceae* and *Theaceae* families (Chan, 2002), or specifically *Rhaphiolepis indica* and *Syzygium levinei* (Chan, 2007). Native species are valuable in establishing and enriching the local flora and fauna by successful seed dispersal and increasing the vegetation regeneration rate. According to Wang (1997), the diversity of vegetation, insects, birds and soil microbes are higher in mixed (native and exotic) forests than in exotic monoculture because the mixed forests are more frequently visited by wildlife. Thirdly, there is recruitment of native species seedlings by the understory strata of all the restored sites in TTEBA, being more apparent in R89 than in R95 and R00. In order to accelerate the rapid recovery of native woodlands, it is perhaps necessary to enrich the understorey with native seedlings. This process is known as enrichment planting. Besides an inventory study of the overstorey composition, we need to also find out the nutrient status of the soil and its physical properties. Taking R00 as an example, since it has a very high tree density and canopy coverage, selected clearfelling of the overstorey species may be required before enrichment planting. Fourthly, it seems that high soil bulk density restricts plant growth and soil fauna activities, as in site R95. It is thus necessary to loosen the surface soil before planting, although it is costly and labor-intensive. Finally, nutrient cycling via litterfall is an important ecosystem process closely linked to self-sustainability of restored vegetation. The thick litter layer on the restored sites acts as a barrier between the seeds and the soil, thus impairing seed germination. It also promotes wildfire. Therefore, prescribed burning of the litter floor is recommended during the early stage of forest succession, which can kill two birds with one stone. In addition, nutrients released from ash decomposition can further enhance ecosystem productivity.

Possible forest ecosystem trajectories are illustrated in Figure 8.3. In general, forest restoration is divided into four stages, including stand initiation, stem exclusion, understory reinitiation and later-succession. Forest resilience begins with the first stage of forest succession called "stand initiation". The typical feature of the ecosystem in this stage is the harsh soil conditions, especially the poor nutrient contents. This stage is expected to last for about 3 to 5 years and the relevant management inputs include hydroseeding, weeding, seedlings planting and fertilizing. When the canopy becomes closed, the second stage of "stem exclusion" begins. Trees begin to compete for light and space, and mortality rate increases. Pruning and selected logging can be implemented at this stage to enahnce ecosystem quality. After about 10 to 15 years, "understory reinitiation" begins as long as the soil conditions are improved and suitable for the growth of native tree and shrub species. The forest will now consist of two age classes and at the same time, there will be natural regeneration underneath the canopy so that age classes and species mortality will continue to increase spontaneously. Parallel to this, there will be continuous improvements in soil conditions, plant structure and ecosystem functions and soil fauna diversity while nutrient leaching from the forest ecosystem is expected to reduce. There are then no further auxiliary inputs to the restored forest, which becomes self-sustainable itself. This is a late-successional forest, approaching the climax state.

In view of a rising concern about the strategy of forest recovery in Hong Kong, the aim of restoration has shifted from merely providing a green cover to forest rehabilitation. To cope with this change, several new measures are thus recommended. This shall include the use of original soil cover in rehabilitation planting, use of native species with pioneer properties, genuine silvicultural practices, and enrichment planting of native species. While a good start is half the way done, continuous monitoring of each phase is also needed.

This study provides an example of how early or mid-successional stages of restored ecosystems on soil destruction sites can be evaluated and reported. It also provides a set of useful variables that contribute to restoration success or failure in the progress of restoration. Determining restoration outcomes at different stages along the trajectory provides the necessary information that can reduce project failure, improve the effectiveness and efficiency of woodland management inputs, and improve the usefulness of the trajectory model used in ecological restoration. This study, therefore, strengthens and improves our understanding of the importance and relevance of ecological assessment in the field of restoration science.



I: Stand initiation II: Stem exclusion III: Understory reinitiation IV: Late-succession

State A: Soil destruction land **State B:** Original state forest (Target for restoration)
State C: Alternative state (Target for rehabilitation) **State D:** Degraded ecosystem

Figure 8.3 Schematic portrayal of ecosystem trajectories of soil destruction site through time

8.5 Limitations of the study

The optimal method for evaluation of the progress of ecological restoration is to monitor the site condition long-term from time zero onwards after excavation. Unfortunately, there is a lack of detailed history records for the rehabilitated sites in Hong Kong. Therefore, in this study, we selected three rehabilitated sites of different ages (5 or 6 year time intervals) under similar management and control as the substituted method.

In this study, the oldest (R89) and youngest plantations (R00), as well as the middle-age plantation (R95) in TTEBA were selected as the research sites. They all belong to the middle-early stage of forest restoration. However, newly excavated borrow areas and sites with plantations over 25-years old were not found in Hong Kong. Instead, one native mature forest (TO *fengshui* woodland) was taken as the reference site. It would be better if we could find a secondary plantation over 50 years of age as another reference site to capture the dynamics and variability in natural systems. Moreover, because of the combination of TDD (Territory Development Department) and CED (Civil Engineering Department) involvement in 2004, the detailed records relating to the Tai Tong East Borrow Area were missing. The background information was obtained only from the former studies, communication with the staff or officers of CEDD and AFCD, and the General Specification for civil engineering works.

In this study, we tried to assess the progress of ecological restoration with regard to all aspects of the ecosystem, such as soil, vegetation and litter, in relation to ecosystem structure and function. However soil fauna and microbe diversities and populations, which have an important relationship to nutrient cycling and energy flow, were not mentioned in this study because of the limitations of time and knowledge background.

Due to time and labor force limitations, the assessment of the study sites, i.e. the vegetation growth performance, the soil condition and the litter nutrient cycling, was done only once in the year. The seasonal and annual variations of soil properties and vegetation performance cannot be detected given these limitations.

Restoration ecology is absolutely multi-disciplinary, including contributions from ecological engineering, silviculture, ecology, environmental science, soil science, botany, biology, microbiology, agrobiology, and so on. Restricted by my own background and understanding of these fields, what I have done in this research is only the tip of the iceberg. I hope to continue studying restoration ecology in the future, and contribute more to the field.

8.6 Suggestions for further study

More attention should be drawn to the assessment of diversity and populations of soil fauna, such as arthropods and invertebrates. These soil fauna may strongly affect the efficiency of matter circulation and energy flow in the soil-plant ecosystem. Bacteria and fungi should also be taken into consideration as potential indicators of successful ecological rehabilitation.

If possible, it is essential to build up an ecological research station to monitor the restored site long-term and to establish a data base for sharing. It is necessary to quantify the indicators, which can be the standards or criteria to differentiate stages of ecological restoration. The selection of pioneer species is still ambiguous in current strategy of restoration. It is suggested that the most effective species combination for initial introduction be figured out.

The common method of soil analysis involves the sampling and removal of soil from the sites to the laboratory. As huge columns of soil samples are dug out, here is obviously disturbance to the ground surface and interference with the soil fauna habitat. It is crucial to improve the sampling method and to reduce the need for columns of soil samples, such as using *in situ* assessment.

Future study on monitoring or assessment of ecological restoration of soil destruction sites is desperately needed. Apart from ecological assessment, social and economic assessment of ecological restoration should also be undertaken. They will benefit the management of forest ecological restoration and the establishment of a strategic forest restoration plan in the subtropical area.

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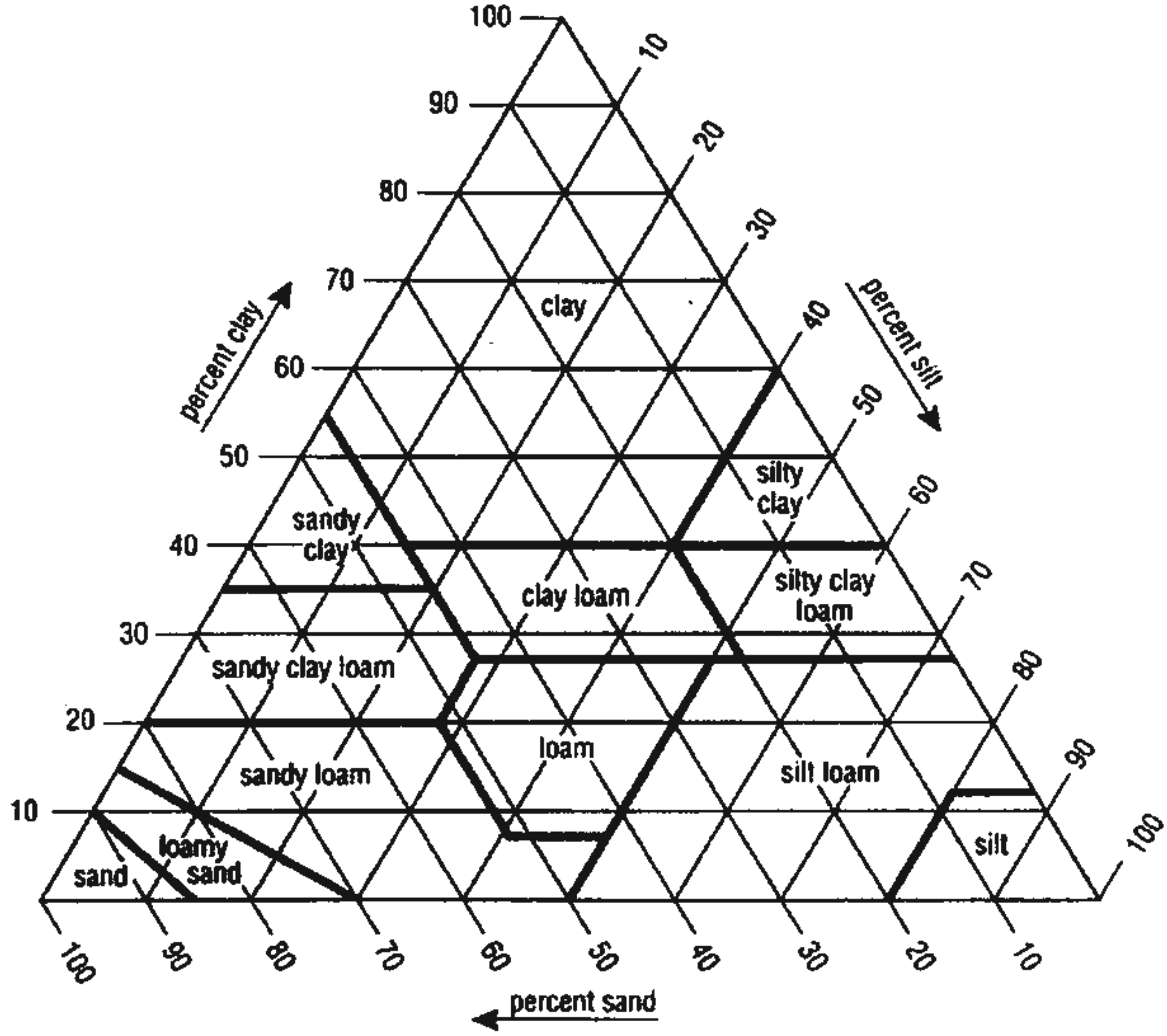
APPENDICES

Appendix 4.1 Vegetation growth performance of TO fengshui woods (DBH > 2cm) (10m x 10m)

Name	Genus	Species	DBH(cm)	Height(m)	Crown area (m^2)
九節	*Psychotria*	*asiatica*	2.80	3.10	1.23
銀柴	*Aporusa*	*dioica*	3.20	3.50	0.79
九節	*Psychotria*	*asiatica*	2.60	3.00	1.23
銀柴	*Aporusa*	*dioica*	3.60	4.50	3.98
九節	*Psychotria*	*asiatica*	3.80	3.00	2.41
木荷	*Schima*	*superba*	68.31	18.00	44.18
肉實樹	*Sarcosperma*	*laurinum*	7.50	6.00	4.52
潤楠屬 sp.	*Machilus*	*sp.*	9.90	10.00	0.20
魚骨木	*Canthium*	*dicoccum*	3.80	5.50	0.79
銀柴	*Aporusa*	*dioica*	2.80	2.50	0.79
中華杜英	*Elaeocarpus*	*chinensis*	5.50	3.50	2.84
橄欖	*Canarium*	*album*	2.40	3.00	0.20
魚骨木	*Canthium*	*dicoccum*	11.60	15.00	0.79
魚骨木	*Canthium*	*dicoccum*	3.20	4.00	0.33
中華潤楠	*Machilus*	*chinensis*	26.11	18.00	15.90
unknown 1			17.64	18.00	23.76
假蘋婆	*Sterculia*	*lanceolata*	3.20	3.00	1.23
肉實樹	*Sarcosperma*	*laurinum*	4.80	3.00	1.23
銀柴	*Aporusa*	*dioica*	2.80	2.00	0.10
銀柴	*Aporusa*	*dioica*	3.00	2.00	0.07
九節	*Psychotria*	*asiatica*	3.30	2.20	0.28
銀柴	*Aporusa*	*dioica*	4.20	3.20	0.20
九節	*Psychotria*	*asiatica*	4.20	2.00	0.33
肉實樹	*Sarcosperma*	*laurinum*	2.90	2.50	0.38
銀柴	*Aporusa*	*dioica*	4.00	2.00	0.16
肉實樹	*Sarcosperma*	*laurinum*	3.80	3.20	2.41
銀柴	*Aporusa*	*dioica*	2.50	2.70	0.10
中華潤楠	*Machilus*	*chinensis*	12.36	10.00	5.94
肉實樹	*Sarcosperma*	*laurinum*	2.60	2.20	0.57
木荷	*Schima*	*superba*	29.14	15.00	50.27
木荷	*Schima*	*superba*	99.68	20.00	86.59

Appendix 5.1 The textural triangle showing the percentages of sand, silt, and clay in the soil texture classes, according to U.S. Department of Agriculture Classification System.



Appendix 5.2 The dynamics of soil moistures of study sites before and after the strong precipitation

Date (year/month/day)	R00	R95	R89	TO	Precipitation (mm)
07/06/20	60.96	60.48	56.37	68.06	-
07/06/21	60.39	59.91	56.09	67.12	-
07/06/22	59.66	59.21	55.79	66.17	-
07/06/23	58.57	58.32	55.50	65.01	-
07/06/24	57.52	57.68	55.19	63.63	-
07/06/25	56.70	56.74	54.92	62.35	-
07/06/26	55.89	55.85	54.73	65.66	11.0
07/06/27	60.46	60.71	58.11	68.38	26.0
07/06/28	71.44	73.25	64.05	69.83	117.0
07/06/29	67.71	71.46	60.51	68.86	23.5
07/06/30	67.14	69.09	60.75	69.14	25.0
07/07/01	66.87	68.24	59.23	67.82	4.0
07/07/02	64.99	64.81	58.73	67.45	13.0
07/07/03	63.81	64.35	57.69	67.42	0.5
07/07/04	62.89	64.02	57.41	67.43	3.5
07/07/05	63.44	65.37	59.01	68.88	10.5
07/07/06	64.51	65.80	58.81	67.18	-
07/07/07	62.76	64.05	57.56	65.97	-
07/07/08	61.17	62.83	57.04	65.08	-
07/07/09	59.94	61.76	56.10	64.27	-
07/07/10	58.44	60.23	55.82	63.37	-
07/07/11	56.91	58.67	55.69	62.45	-
07/07/12	55.68	57.48	55.35	61.38	-
07/07/13	54.38	56.18	55.10	60.18	-
07/07/14	53.39	54.89	54.79	59.37	-
07/07/15	52.59	53.63	54.38	58.72	-

Appendix 5.3 Soil chemical properties of the sites (n=15)

Sample		pH	Organic C (%)	TKN (%)	C : N ratio	Total P (g/kg)
R00	0-10cm	4.55	0.69^{c} (0.51)	0.04^{ab} (0.02)	22.97^{c} (14.50)	0.32^{ab} (0.02)
	10-20cm	4.64	0.57^{bc} (0.41)	0.03a (0.02)	21.89^{bc} (13.85)	0.33^{b} (0.02)
	20-30cm	4.28	0.16^{a} (0.15)	0.02^{a} (0.02)	15.83^{ab} (18.09)	0.34^{b} (0.01)
R95	0-10cm	4.21	1.00^{d} (0.99)	0.08^{d} (0.02)	13.14^{a} (2.88)	0.29^{a} (0.02)
	10-20cm	4.22	0.61^{bc} (0.60)	0.06^{c} (0.02)	10.49^{a} (2.19)	0.32^{b} (0.02)
	20-30cm	4.43	0.44^{b} (0.43)	0.04^{bc} (0.02)	10.89^{a} (3.62)	0.32^{b} (0.01)
R89	0-10cm	4.05	1.89^{e} (1.73)	0.12^{e} (0.03)	16.60^{abc} (4.48)	0.41^{c} (0.02)
	10-20cm	4.10	0.76^{c} (0.84)	0.06^{c} (0.01)	12.93^{a} (7.65)	0.42^{c} (0.01)
	20-30cm	4.16	0.38^{ab} (0.43)	0.04^{bc} (0.02)	9.70^{a} (6.18)	0.43^{c} (0.02)
TO	0-10cm	3.56	5.32^{h} (3.42)	0.24^{g} (0.02)	22.02^{bc} (2.62)	0.47^{d} (0.08)
	10-20cm	3.62	5.00^{g} (3.22)	0.16^{f} (0.01)	31.08^{d} (3.66)	0.48^{de} (0.05)
	20-30cm	3.78	2.78^{f} (1.79)	0.13^{e} (0.01)	22.25^{bc} (2.93)	0.49^{e} (0.02)

Column means sharing the same superscript were not significantly different at $p < 0.05$ by Duncan's Multiple Range Test.

Appendix 5.4 Environmental parameters between Nov. 8, 2007 and Jan. 3, 2008

Parameters	Site \ Time	2 weeks	4 weeks	6 weeks	8 weeks
Average temperature (°C)	R00	20.12	19.35	19.22	18.79
	R95	20.73	20.02	19.87	19.42
	R89	20.61	19.68	19.53	19.06
	TO	19.8	18.77	18.66	18.09
Average moisture (%)	R00	50.13	49.70	49.48	50.24
	R95	48.27	48.19	48.04	48.85
	R89	49.51	49.36	49.12	49.94
	TO	57.8	57.20	56.84	56.86
Precipitation ($mm \cdot day^{-1}$)	Weather station	0.32	0.16	0.12	0.63