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Evaluating plantation forest versus natural forest regeneration for biodiversity enhancement in Hong Kong

Janet E. Nichol1 and Sawaid Abbas2

Abstract: Global trends predict a continuous increase in the proportion of forest occupied by plantations up to the end of the 21st century, while a dramatic loss of biodiversity is foreseen as a result of anthropogenic exploitation and climate change. This study compares the role and performance of plantation policies in Hong Kong, with natural regeneration of secondary forest, using detailed spatio-temporal data extracted from a previous study. The study extends over a 70-year period from 1945 to 2014 using aerial photographs and satellite images of five time periods to document spatio-temporal trends in plantation forestry and natural forest succession. Field data on species richness and woody biomass at different stages of forest succession are compared with available data from plantations in the same study area. Results indicate that plantation forests support relatively few native species in the understory, with much lower species richness than naturally regenerated forest, even after 6 to 7 decades. Time-sequential maps of habitat change show that natural forest succession from barren grassy hillsides, progressed at an annual rate of 7.8 %, from only 0.2 % of the forest area in 1945 to 2014.

Keywords: Plantations; Biodiversity; Secondary forest; Hong Kong

1. Introduction

Only 34 % of the world’s forests are now primary, with a decline of over 8 % since 1990. However, in east Asia only 6 % of forest is primary forest [1]. Although some of the primary forests cleared for farming are regenerating naturally following abandonment, these naturally regenerating forests have also declined worldwide by approximately 8 % since 1990. Naturally regenerating, or secondary forests differ from primary forests in their vastly simplified structure and species composition, but generally comprise a range of native woody species. On the other hand, planted forests, or tree plantations in east Asia (excluding oil palm) currently represent 36 % of forest compared with 7 % worldwide [2]. Global trends predict a continuous increase in the proportion of forest occupied by plantations up to the end of the 21st century [2,3].

Due to the importance of forest ecosystems in supplying a range of essential ecosystem services for human life, it is necessary to evaluate the ability of forest plantations to
provide these vital support functions. However, their ability to do so may be compro-
mised by the use of non-native and fast-growing species [4], the use of which increased
worldwide by 29% between 1990 and 2020, and by 138% in east Asia [2]. East Asia’s large
increase in forest plantations is largely due to massive afforestation programs in China
during the 2000-2010 decade [5], which mainly comprised fruit trees, rubber and eucalypt-
tus. Although such monoculture plantations may serve some of their intended purposes
such as carbon storage, economic productivity, and control of soil erosion, these often
come at the expense of other ecological functions [5]. Additionally, the use of single or few
species makes them less resistant than biodiverse natural forests to external disturbances
such as fire, disease and climatic fluctuations [6]. Biodiverse forests contain a wide range
of species enabling them to perform wide ranging ecological functions and adapt to
changing ecological conditions. Furthermore, studies have demonstrated a positive rela-
tionship between species richness in tropical forests and carbon storage [7–9].

There are two viewpoints about the role of plantation forestry’s ability to restore eco-
logical functions and biodiversity. Firstly that it is detrimental, and at best, its use repre-
sents a trade-off between socio-economic and ecological goals [5,6,10,11]. Secondly plant-
tations have sometimes been recommended as beneficial, as they can assist the early
stages of succession to a stable and biodiverse forest ecosystem [12–17].

Especially where land has become severely degraded, plantations are sometimes
viewed to have catalytic effects, by providing an understory, suitable microclimate and
build-up of a litter and humus layer, for native species to regenerate naturally [12,13,18].
Piirainen et al (2016) [17], for example, suggest that tree communities in plantations in the
Afro-tropics are moving towards old growth forest due to plantation trees suppressing
persistent grass invasion following reforestation. In tropical and sub-tropical Australia,
plantations, whether of single or mixed species, were considered to have strongly positive
consequences for restoring biodiversity in heavily cleared landscapes [19]. However, this
viewpoint is countered by another study in tropical Australia, where dung beetles, con-
sidered to be useful environmental indicators showed lower species diversity and num-
bers, in plantations than in native forest [20]. Also, in a summary of research papers on
south-east Asia, where plantation forests have increased by approximately 1.3% annually
since 2010 [1], all types of non-oil palm plantations were shown to be inferior to naturally
regenerating forest in species richness and abundance [21]. The lower richness and abun-
dance were attributed to the lack of preferred microhabitats and floristic diversity causing
reduced insect diversity, as well as lack of perennial fruits and food sources for birds.
Nevertheless, the authors [21] do agree with other studies that plantation provides a better
biodiversity alternative to other modified non-forest landscapes [16,22].

The question of whether plantations are detrimental or beneficial in forest regenera-
tion was addressed by a survey of regenerating forests worldwide [23] which suggests
that the previous land use before abandonment largely determines if forest could recover
naturally or would require interventions. Overall, biological processes recovered slower
following agriculture than logging or mining, and biodiversity recovery in tropical forest
was slower than in temperate forests. Although the objectives of plantation establishment
vary widely in different world regions, evaluation of their ecological and environmental
effects does not always consider these objectives [16]. If the primary objective is restora-
tion of a biodiverse and stable ecosystem, the high cost of preparation, planting and nur-
turing of tree seedlings, as well as subsequent fire protection, may make plantation an
inferior alternative to natural forest regeneration if the latter is feasible.

In the highly degraded hillsides of Hong Kong, planting programmes have been in
place since the 1870s, and Hong Kong is thought to be the earliest example of plantation
forestry in the tropics [13]. The main objectives have shifted from health and aesthetic
considerations in the 19th century, to preventing soil erosion and watershed protection in
the 20th century, and to ecological restoration today [13]. This has paralleled global con-
cerns surrounding loss of biodiversity in all continents due to habitat loss accompanied
by climate change [24]. Earlier work in Hong Kong by Lee et al [25] concludes that ‘few
studies have compared plantations with naturally regenerated forest of similar age. Natural succession with little or no intervention might have been a more effective rehabilitation method.’ More recent work in southern China and Taiwan appears to confirm this, for example, Liu et al [26] observed less than half the number of tree species seedlings in the understory of plantation compared to natural hardwood forest and Su et al [27] reported that, fir plantations showed low regeneration in terms of understory shrub cover and diversity compared to natural secondary forest.

The availability of a 7-decade long archive of aerial photographs and satellite images enables an evaluation of the relationships between plantations and natural succession in Hong Kong’s tropical secondary forests. The objectives of this study are therefore to evaluate the benefits of plantations in the process of natural forest regeneration in Hong Kong, by comparing the outcomes, of naturally regenerated forest, with areas which have undergone planting, over the last seven decades. Data for the naturally regenerating forested areas are derived from our previous studies [28,29], and data for plantations are taken from Lee et al [25]. This paper uses the term ‘regeneration’ to refer to the process of natural forest succession from grassland to secondary forest, on land where forest had existed at some stage in the past.

2. The study area

Hong Kong’s landscape is rugged and mountainous, with convex slopes rising from sea level to almost 1000 m, within a few kilometres of the coast (Figure 1). Scarcity of flat land has confined urban development to only 20 % of the land area, leaving over 80 % relatively undeveloped, supporting grassland, shrubland and forest. In 1976, 40 % of Hong Kong’s land area was designated for protection in Country Parks and these are administered by the Agriculture, Fisheries and Conservation Department (AFCD). Two of Hong Kong’s 24 Country Parks, Tai Mo Shan and Shing Mun in the New Territories, totalling almost 30 km² were selected for the study. Climatically, Hong Kong experiences cool dry winters and hot humid summers. Its position, at latitude 22.3 °N, on the edge of the tropics, supports a highly diverse flora and fauna belonging to both tropical and sub-tropical genotypes. However, the flora is recognised as tropical except for high altitude areas where some sub-tropical plants are reported. The main climatic limitations to plants are occasional frost events which limit tropical species to low altitudes, and typhoons. The fauna is recognised as tropical [30].

Figure 1. Location of study area within Hong Kong and China.

Corlett (1999) [13] states that it is likely Hong Kong’s original tropical evergreen broadleaf forests were removed in the 15th to 17th centuries, and planting by the colonial
government began around 1871 to make the, then barren, hillsides more attractive. Plantation efforts continued over the next 8 decades with a variety of species dominated by the native *Pinus massoniana*, as well as *Castanopsis fissa* and other exotic broadleaves including several species of *Eucalyptus*, *Lophostemon confertus*, *Acacia confusa*, *Leucaena leucocephala*, and *Melaleuca quinquervia*. Following almost complete devastation of Hong Kong’s wooded areas during the blockade years of WWII when trees were cut for timber and fuel, massive plantation forestry took place. The objectives were to control soil erosion on the barren hillsides, to improve water catchment functions, and for production forestry. Similar species, dominated by *Pinus massoniana*, were used as during pre-WWII [13]. Only after 1965 was conservation of wildlife seen as an objective of Hong Kong’s forestry policy, along with countryside recreation and protecting the water catchment areas. Thereafter, due to their fire susceptibility, *Pinus massoniana* and *Eucalyptus spp.* were less favoured, and a few easily propagating exotics have predominated, including *Lophostemon confertus*, *Pinus elliottii* and *Acacia confusa* [25].

3. Methods

The study compares the outcomes of forest plantation policies in Hong Kong, with those of natural forest succession, using data from two sources. Firstly, the present authors [28,29] utilised an image archive covering five time periods between 1945 and 2014, to track and map changes in land cover and habitat development (including both natural vegetation and plantation development) over the 7-decade period. The image archive is described in section 3.1 below.

The second source was a study of plantations by Lee et al (2005) [25] of the same study area in Hong Kong, including inventories of species richness and structural characteristics of plantations. Lee et al. used sequential aerial photographs to determine plantation ages, and conducted plot surveys on species richness and basal area, by plantation age and species type. Figure 2 (flowchart) summarises the steps of the study.

3.1. Remote sensing data

The image archive covered five time periods between 1945 and 2014, to track and map changes in land cover and habitat development over the 7-decade period. These comprised aerial photographs of 1945 at 1:40,000 scale, 1963 at 1:14,000, and 1989 at 1:20,000 scale, totalling 43 aerial photographs (18 for 1945, 12 for 1963 and 13 for 1989). For the two most recent dates, we obtained high resolution IKONOS satellite images of 2001 with 1 m. spatial resolution, and for 2014, WorldView-2 images with 0.5 m spatial resolution.

For accurate comparisons over time it was necessary to register all five sets of images to the same digital coordinate system. For this, the individual air photo prints were converted to orthophotos by scanning at a resolution of 1200 dpi, followed by orthorectification using 15 to 20 GCPs per photo. The X, Y and Z coordinates of the GCPs were obtained from a 0.5 m digital orthophoto of Hong Kong and a 2 m Digital Elevation Model (DEM). The multiple orthophotos were then clipped and mosaicked to form unified orthoimages. The satellite images were also converted to orthophotos using RPC (Rational Polynomial Coefficient) files, the Hong Kong orthophoto and the DEM. All the data sets were then reprojected into the Hong Kong 1980 Grid system. Co-registration of all datasets to the digital orthophoto produced an RMS error of 1 to 5 m, depending on scale and resolution of the dataset.
3.2. Habitat classification.

For vegetation classification, the study adopted an ‘a priori’ scheme from a previous habitat mapping project [31], as it was based on structural characteristics interpretable from remote sensing images. This scheme itself, is derived from a standard system, the Land Cover Classification System (LCCS) [32] (Table 1). The LCCS was devised by FAO and UNEP, to standardize data on land use/land cover, and is universally applicable [33]. The classes are easily identifiable on remote sensing images and are mutually exclusive and unambiguous.

Table 1. Description of habitat classes and structural stages of vegetation.

<table>
<thead>
<tr>
<th>Habitat Class (Structural Stage)</th>
<th>Description (based on LCCS definitions)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Forest</td>
<td>Broadleaved, evergreen trees, with &gt;60–70% canopy cover</td>
</tr>
<tr>
<td>Open Forest</td>
<td>Tree/shrub/grass mosaic: Open tree canopy (15-60%) with shrubland and/or grassland</td>
</tr>
<tr>
<td>Shrubland</td>
<td>Broadleaved, evergreen shrubland, with closed canopy &gt; 60–70 %</td>
</tr>
<tr>
<td>Open Shrubland (Shrubby Grassland)</td>
<td>Shrub/grass mosaic: Open canopy of shrubs (&gt;15-60%) with grassland</td>
</tr>
<tr>
<td>Grassland</td>
<td>Ground story of grasses (&gt; 70 %) as dominant vegetation form as well as &lt;15% shrubs/trees</td>
</tr>
</tbody>
</table>
Plantation Forest

Monocultural plantation stands- *Lophostemon confertus*, *Acacia confusa*, *Melaleuca quinquenervia*, as well as mixed spp.

3.3. Multi-scale image segmentation and habitat classification

Delineating habitat boundaries on the different types and scales of imagery from the five different time periods manually, is necessarily subjective. Therefore, an automated, object-based method was devised, which uses the spatial and tonal signatures of pixel groupings to segment the image into multi-pixel image objects using the multi-resolution segmentation approach. These defined objects maximize the variability between different objects and minimize that within objects, considering the expected size and scale of objects. Thus the segmentation process represents the automated digitizing of target boundaries. The segments, representing habitat patches were then assigned to a habitat type, while taking into account their respective spatial scales.

Habitat patches can be of any size and can grow or contract over time. Patchiness can also occur at finer scale, as each patch has internal structure, as well as at broader scale when similar patches are amalgamated. In this study, multi resolution segmentation was executed at three sizes of Minimum Mapping Unit: 1000 m² (for large patches of grassland, forest and plantations), 500 m² (for open shrubland and shrubland), and 50 m² (for isolated forest and shrubland patches). All the image interpretation, to classify the segments, was done by a single expert analyst to minimize interpretation errors. Plantation stands were recognised by homogeneity of tonal pattern, as well as multi-temporal analysis, as distinct plantation patterns were easier to recognize in the early stage. For simplicity in change analysis, a single class of plantation forest was created by merging the four sub-classes of plantation species *Lophostemon confertus*, *Acacia confusa*, *Melaleuca quinquenervia*, and mixed plantation. The mapping was validated using 352 field surveyed GPS points and 215 additional check points on very high resolution colour air photos, from a previous habitat mapping project [34,35].

The allocation of image objects delineated by multi-scale segmentation to a particular structural class was done manually, and is necessarily subjective. However, observation at multiple time steps was able to improve the mapping, as successional stages usually proceed logically. For example, it was difficult to distinguish between stands of mature plantation and forest on a particular image date, but reference to earlier growth stages (as young plantations are easy to identify and tend to precede mature plantations, and shrubland precedes forest) helped to reduce the subjectivity. Forest and plantation were identified with 98% and 100% accuracy respectively, and overall mapping accuracy exceeded 92%. The percentage rate of change for each class was calculated by using Equation 1 [36] which is not sensitive to difference in periods and provides a more meaningful estimation of annual change rate.

\[
R = \left( \frac{1}{T_2-T_1} \right) \times \left( \ln \frac{a_2}{a_1} \right) \times 100
\]

where \( R \) is the rate of change (% per year), \( a_1 \) and \( a_2 \) represent area corresponding to earlier time, \( T_1 \), and late time, \( T_2 \).

3.4. Field data collection

Field survey in the naturally regenerating (secondary) forest was conducted in 28 plots of 20X20 m (400 m²) from March 2015 to May 2016, to document and quantify the species composition and stand density of forest plots at different stages of succession [37]. This included species richness, ie. the total number of species occurring at each successional stage and basal area as a surrogate for total forest biomass. Plot selection considered the heterogeneity of the landscape and accessibility, to include a range of age classes, altitudes, aspects and slope. All trees having diameter at breast height (DBH) ≥ 1cm were identified and measured and the basal area for each plot was recorded. The age of a plot...
was estimated from sequential interpretation of vegetation cover, and plots were grouped into five median age classes: 7, 20, 39, 61 and >70 years. In total, the plots occupied 1.12 hectares at elevations between 205 m and 822 m. A total of 8,575 plants belonging to 229 species and 63 families were tagged, measured, identified and recorded, requiring 2800 man hours. The field data collected by forest age class enabled comparison between plantation forestry and natural forest succession, in their progress toward a biodiverse and stable ecosystem, over the several decades of the study. To enable comparisons with the ages of plantation, plot data for secondary forest were interpolated between the surveyed years using the annual rate of change and 2-sample t-test was applied to determine statistical significance of the differences.

4. Results

4.1. Landscape change over time

Table 2 shows the area in hectares of each habitat type at each of the five mapped years, for the four major habitat classes and Figure 3 shows the habitat map at five stages of succession. Over the whole 70-year period, the area of forest increased at a rapid rate of 7.8 % annually.

Table 2. Area statistics (ha. and %) for habitat classes 1945 to 2014 [29], where Forest class includes both open and closed canopy forest, and Shrubland class includes both open and closed canopy shrubland. Built areas and Bare areas are not included.

<table>
<thead>
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<tbody>
<tr>
<td></td>
<td>(ha)</td>
<td>(%)</td>
<td>(ha)</td>
<td>(%)</td>
<td>(ha)</td>
<td>(%)</td>
</tr>
<tr>
<td>Forest</td>
<td>4.82</td>
<td>0.17</td>
<td>19.85</td>
<td>0.71</td>
<td>140.03</td>
<td>5.01</td>
</tr>
<tr>
<td>Open Forest</td>
<td>0.22</td>
<td>0.01</td>
<td>27.69</td>
<td>0.99</td>
<td>38.09</td>
<td>1.36</td>
</tr>
</tbody>
</table>

Figure 3. Land cover classes at five time intervals from image interpretation.
In 1945, grassland comprised 79% of the landscape, the remainder being barren (12%), shrubland (7%) and only 0.17% of forest (totalling 5 ha.), which was mainly confined to Fung Shui woodlots adjacent to villages, and remote ravines. During the 18-year period from 1945 to 1962, natural succession was very slow, little change occurred in the landscape, and the area of forest only increased to 1.7% of the landscape, with over half of this being open forest. However, large-scale post-war afforestation programmes were mounted following WWII, and plantations increased from zero in 1945 to 15% (432 ha) by 1963, concentrating in the south-west of the study area (Figure 3). The native Pinus massoniana was the dominant species in this area.

The 26-year period 1963 to 1989, saw a greater increase in the percentage of the landscape occupied by secondary forest, from 0.17% in 1963 to 6.4% in 1989. Thus even after 44 years of natural succession following WWII, forest still only occupied 6.4% of the landscape, as the transition from grasslands through a shrubland stage, into open forest, then closed forest was slow. It is unfortunate therefore that the Pinus massoniana plantations established in the earlier period in the south-west, were destroyed by two diseases: the pine needle scale and a pinewood nematode [13], reducing the plantation areas from 15% in 1962 to 11% in 1989. The areas occupied by the plantations were seen to immediately revert to grassland by 1989, suggesting that no natural invasion by a tree or shrub understory had taken place during the plantation period.

The highest annual rate of increase in forested area, of 11% per year, was seen in the 12-year period from 1989 to 2001. During this period the proportion of the landscape occupied by secondary forest more than tripled, from 6.4% to 24.5% (from 179 to 787 ha.). This involved an increase along elevation as shrublands spread upwards from valley bottoms and transitioned into forest. Plantation areas increased only slightly, by 1%, to occupy approximately 12% of the landscape by 2001, as those areas of mixed species not destroyed by disease persisted. It is notable that the areas of the destroyed P. massoniana plantations were still, by 2001 largely covered by grassland.

Although the highest rate of change in forest was in the period up to 2001, the highest annual gain in forested area was from 2001 to 2014 (Table 2). By 2014, secondary forest occupied over 37% of the landscape, and covered most valley bottoms and steep-sided slopes, except for those areas covered by plantation forests, mainly in the Shing Mun Reservoir catchment (Figure 3). These plantations continued to occupy 12% of the landscape, with no change in the 13 years since 2001. By 2014, the area of shrubland occupied a similar percentage of the landscape to secondary forest, at 38% (orange and yellow on Figure 3), and grassland had now diminished to 8.5%, almost exclusively on mountain summits (pale green on Figure 3). It is notable that the areas occupied by the Pinus massoniana plantations established in the south-west of the study area post-WWII and destroyed by disease in the 1970s, supported only grassland and open shrubland by 2014 (Figure 3).

### Table 2

<table>
<thead>
<tr>
<th>Land Use</th>
<th>2001</th>
<th>2014</th>
<th>Gain</th>
</tr>
</thead>
<tbody>
<tr>
<td>Shrubland</td>
<td>24.03</td>
<td>26.26</td>
<td>2.23</td>
</tr>
<tr>
<td>Open Shrubland</td>
<td>170.99</td>
<td>181.26</td>
<td>10.27</td>
</tr>
<tr>
<td>Grassland</td>
<td>2203.17</td>
<td>2973.58</td>
<td>770.41</td>
</tr>
<tr>
<td>Bare Area</td>
<td>336.95</td>
<td>327.59</td>
<td>-9.36</td>
</tr>
<tr>
<td>Built-up Area</td>
<td>0.41</td>
<td>0.44</td>
<td>0.03</td>
</tr>
<tr>
<td>Water</td>
<td>57.21</td>
<td>57.21</td>
<td>0.00</td>
</tr>
<tr>
<td>Plantation Forest</td>
<td>0.00</td>
<td>0.00</td>
<td>0.00</td>
</tr>
<tr>
<td><strong>Total (ha)</strong></td>
<td>2798</td>
<td>3378</td>
<td>580</td>
</tr>
</tbody>
</table>

**4.2. Species richness and woody biomass in secondary forest and plantations**
Field plots in the regenerating secondary forest, showed an average of 26, 33, 37, 38 and 28 tree species in plots with median ages of 7, 20, 39, 61 and 70 years. In contrast, Lee et al (2005) [25], using the same sized (20 × 20 m) plots in plantation forests in Hong Kong, report an average of 15 tree species in A. confusa (median age 25 years) 11 in L. confertus (median age 33 years), and 25 in M. quinquervia plantations (median age 45 years) (Table 3). Thus even the youngest secondary forests contained more tree species than any of the plantations. Moreover, the forest plots contained more than double the number of species reported for plantation plots of similar age, except for the deciduous plantation species M. quinquervia, although this still contains considerably fewer species than secondary forest of similar age. The t-test results suggest that A. confusa and L. confertus stands were significantly different from similar aged secondary forest, but that M. quinquervia is less significant, which may be due to its age (45 years) or the species itself may support natural invasion. In terms of basal area (BA), which represents woody biomass, secondary forest has considerably higher BA in two out of three species of plantation, but lower than in the deciduous M. quinquervia plantations. Interestingly, these had reached comparable biomass to 70-year old secondary forest after only 45 years. Thus to some extent the results confirm the positive relationship between species richness and carbon storage observed in other tropical forests [7-9].

### Table 3. Mean (SD) values of tree species and basal area (m²/ha) in 20 m² plots in secondary forest, and plantations.

<table>
<thead>
<tr>
<th>Age (years)</th>
<th>n</th>
<th>No of Species</th>
<th>Basal Area</th>
<th>Monoculture Plantation Stand</th>
<th>n</th>
<th>No of Species</th>
<th>Basal Area</th>
<th>t-value</th>
<th>p-value</th>
</tr>
</thead>
<tbody>
<tr>
<td>7</td>
<td>5</td>
<td>26 (6.8)</td>
<td>25</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>20</td>
<td>6</td>
<td>33 (11.53)</td>
<td>36</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>25*</td>
<td>6</td>
<td>34 (11.53)</td>
<td>37</td>
<td><em>Acacia confusa</em></td>
<td>4</td>
<td>15 (3.6)</td>
<td>18</td>
<td>3.77</td>
<td>0.009</td>
</tr>
<tr>
<td>33*</td>
<td>8</td>
<td>36 (13.97)</td>
<td>39</td>
<td><em>Lophostemon confertus</em></td>
<td>4</td>
<td>11 (4.6)</td>
<td>25</td>
<td>4.59</td>
<td>0.001</td>
</tr>
<tr>
<td>39</td>
<td>8</td>
<td>37 (13.97)</td>
<td>40</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>45*</td>
<td>8</td>
<td>37 (13.97)</td>
<td>41</td>
<td><em>Melaleuca quinquervia</em></td>
<td>4</td>
<td>25 (8.7)</td>
<td>61</td>
<td>1.82</td>
<td>0.102</td>
</tr>
<tr>
<td>61</td>
<td>5</td>
<td>38 (6.16)</td>
<td>43</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>70</td>
<td>4</td>
<td>28 (10.72)</td>
<td>69</td>
<td></td>
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</tbody>
</table>

*Plot ages and values for secondary forest refer to interpolated ages and values*

Plantation data reported by Lee et al, 2005 [25].

### 5. Discussion

Naturally regenerated secondary forest is shown to be superior to forest plantations in terms of species richness, with natural forest containing approximately twice the number of tree species as in plantations of the same age reported by Lee et al (2005) [25] in Hong Kong. This is not to say that natural succession will continue accumulating species, and in the longer term, the limited species pool following centuries of deforestation and loss of native fauna will limit the future succession [30]. Thus intervention will probably be necessary to introduce some middle and late successional species.

There is also evidence [38,39], that forest plantations in Hong Kong are less able to withstand external disturbance, than native forest. In a severe typhoon in September 2018 [39], stands of exotic monoculture plantation (*Melaleuca quinquervia*, *Lophostemon confertus* and *Acacia confusa*) were the most seriously damaged by the typhoon, with more than 25% decrease in vegetation vigour, represented by the Normalised Difference Vegetation Index (NDVI). The canopy was almost completely destroyed and as there was no generation of young understory trees to replace those lost, very little standing vegetation remained in the plantation patches. The canopy of these plantation patches comprises tree species not adapted to a typhoon climate, unlike those of secondary forest which appear...
more adapted to storms. The extensive typhoon damage in plantations is potentially double-edged, as the dead trees and debris following a typhoon provide enhanced fuel supply for grassland fires which occur frequently in the dry season. Plantations normally show low fire susceptibility due to their poor understory. From a sustainability viewpoint therefore, natural forest is clearly preferable to plantations, quite apart from the barrier they impose to natural forest regeneration, as seen from the time sequence analysis presented here.

During the period of greatest natural forest advance (1989 to 2001) the main barrier to advance was apparently the plantations established in the 1970s to 80s, which impeded shrub colonisation and subsequent succession to forest. Even in the latest period 2001 to 2014, the plantations established earlier remain unchanged, and occupied 12 % of the landscape, on valley and interfluve sites which elsewhere had undergone natural succession to forest. By 2014 natural forest occupied 37.4 % of the landscape, and due to invasion of shrubland into grassland, shrublands occupied a similar proportion to forest ie. 38 % in 2014. Since the conversion of grassland to shrubland appears to be a much slower process than that of shrubland conversion to forest [29,40] this bodes well for the rapid succession of the remaining shrubland areas to forest, as the grassland to shrubland stage has been largely surpassed. The observation that the areas occupied by the Pinus massoniana plantations destroyed by disease in the 1970s, still supported only grassland and open shrubland by 2014, where other areas had succeeded naturally to forest, also suggests that plantations may impede natural succession. It counters arguments that plantations can promote understory native plant regeneration [41], or act as a nurse crop for diverse native seedlings [13]. It also counters arguments that plantations have strongly positive biodiversity benefits for restoration of heavily cleared landscapes [19]. Therefore the observation of several studies that plantation provides a better biodiversity alternative to other modified non-forest landscapes [16,21,22] is not supported by our study.

The other main counter-argument for afforestation versus natural regeneration in Hong Kong, is the considerable cost and effort required, which is difficult to estimate in monetary terms due to inflation over the decades. Corlett (1999) [13] describes huge labour costs over many decades starting in the 1870s including seed collection, sowing in the nursery and directly on hillsides, trials of different species, many of which failed, hand removal of parasites, application of chemical pesticides and fertilisers, thinning, brushwood cutting, and maintaining firebreaks. During the 1960s to 1990s decades for example, Hong Kong’s AFCD was hand-planting 300-350,000 seedlings a year. Even if the main objective for plantations is timber harvesting, the basal areas reported for plantations in our study area in Hong Kong [25] were lower than for secondary forest, for comparable aged stands.

Evidence from archived satellite images (Abbas et al, unpublished) indicates that frequent grassland fires during the 1980s, were a major impediment to shrub colonization in grasslands, which would explain the slow progress in the successional stage from grassland to shrubland noted above. It would also explain the faster progression from shrubland to forest, as shrubland is less susceptible to fire. An apparent decrease in hill fires observed on the satellite images during the 1990s (ibid) may result from more attention by the AFCD to fire control during the 1990s [13]. The latest figure reported here, for 37 % of the landscape occupied by forest in 2014, is remarkable, in view of Corlett and Turner’s (1997) [42] statement that the 14 % of the landscape occupied by forest (in 1997) was probably the largest area occupied by forest in Hong Kong for several centuries.

6. Conclusion

Over the 70-year period, plantation areas increased, then declined due to disease. Those areas where plantation species had been eliminated, were then completely lacking in woody species ie. a woody understory of native species had not been established, and they then reverted to grassland. Although these events occurred in the 1970s, these areas...
remain as grassland today, whereas other areas which were not afforested, have now proceeded naturally to shrubland or secondary forest. Thus planted forests have acted as a barrier to the establishment of a biodiverse natural forest ecosystem, although they may have satisfied the original objectives of forest policy in Hong Kong, of watershed protection and production forestry. Therefore the plantations established in the 1960s have left a negative legacy for current biodiversity. Furthermore, the 12% of the landscape currently occupied by plantations not destroyed by disease during the 1970s, exhibit poor colonisation by native species and they appear to have prevented natural forest succession in the study area. This limited native colonisation of plantations may result from limited seed dispersal mechanisms, given the vastly reduced native fauna, especially those capable of dispersing large seeds. Lee et al (2005) [25] state that only a few species of bird-dispersed shrubs were found in the understory of plantations.

The argument for plantations promoting native plant regeneration in the understory, or for plantations acting as a nurse crop, is not well supported by our findings. The limited lifespan of exotic plantation species, of 50 or 60 years, given increasing emphasis on conservation of biodiversity, is a major motivation for current policies of the AFCD, to artificially introduce native species into plantations. Given the high cost of afforestation in Hong Kong, and the observation that fire protection can promote fairly rapid shrub invasion which proceeds readily to secondary forest, control of hill fires would appear a better option for future forest conservation policies.

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