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Impact of Fire on Secondary Forest Succession in a Sub-Tropical Landscape

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Abstract: In Hong Kong, as in many tropical areas, grasslands are maintained by fire on disturbed and abandoned land. However, Hong Kong's native forests are regenerating in many areas, alongside frequent burning of the hillsides, and are in different stages of structural succession to closed canopy forest patches. Understanding the major determinants of secondary succession is a vital input to forest management policies. Given the importance of forests for biodiversity conservation, watershed protection and carbon cycling. This study examines the relationship between burning regimes and structural forest succession over 42 years from 1973 to 2015, using an archive of satellite images, aerial photographs and field plot data. Overlay of a fire frequency map with maps of forest structural classes at different dates indicates the number of fires undergone by each successional class as well as the time taken to progress from one class to another under different fire regimes. Results indicate that the native sub-tropical evergreen forests, which are naturally fire intolerant, can regenerate alongside moderate burning, and once the shrub stage is reached, succession to closed forest is relatively rapid and can occur within 13 years. More than one burn, however, is more destructive, and twice-burnt areas were seen to have only one-third of the woody biomass of once-burnt plots. The most frequent fires occurred in areas where mono-cultural plantations had been destroyed by disease in the 1960s and were subsequently invaded by grasslands. These former plantation areas remained in early successional stages of grass and open shrubland by 2015. Other plantations from the 1970s and 1980s remain as plantations today and have acted as a barrier to natural forest succession, attesting to the greater effectiveness of fire control over re-afforestation measures.

Keywords: hill fires; secondary forest; forest succession; degraded landscape; grasslands; NDVI; OBIA; Landsat; Hong Kong

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1. Introduction

Hong Kong is one of a growing number of tropical regions where the secondary forest has been regenerating naturally for several decades after cultivation abandonment. The large percentage (40%) of the land area designated for protection in Country Parks, coupled with a long archive of remotely sensed images enables investigation of the forest regeneration process alongside countryside management policies. The determinants of natural forest regeneration are still not fully understood and the rates and trajectories of succession can be highly variable [1]. Major determinants of natural succession have been recognized as seed availability and dispersal [2–4], plant pathogens [5,6], and the intensity of past and present land use, including fire [7,8].

In Hong Kong, human-induced fires occur during the dry season from September to April, and successive hill fires prevent the regrowth of shrubs and trees, leaving the hill slopes to grassland. The steeply sloping terrain in Hong Kong, coupled with almost complete deforestation in past centuries, as well as deforestation for timber and fuel during WW2 [9], have resulted in massive soil erosion on hill slopes as well as degradation of the natural vegetation. Therefore, forest regeneration in Hong Kong may differ from other tropical areas such as the neotropics, where the impacts of slash-and-burn agriculture have been studied or where a patchwork of forest remnants still exists [1,3,8]. While partial fire suppression commenced in the 1960s [9], authorities in Hong Kong still expend much effort to control hill fires. However, equal, if not more effort, has gone into afforestation planting over many decades.

As 40% of Hong Kong's land area is protected in Country Parks, which provide water catchments, wildlife habitats, recreation opportunities and carbon sinks, achieving a stable and diverse natural ecosystem is essential. Forest management in Hong Kong's country parks has operated for 140 years [9] under successive colonial governments. Over these 14 decades, the objectives of forest management have evolved from, initially, watershed protection, followed by production forestry, and more recently, wildlife conservation and recreation. Most recently, carbon sequestration by forests has become part of the Hong Kong government's "Zero Carbon Emissions" policy. As very few of Hong Kong's native fauna specialize in grassland, with maximum animal diversity being reached in shrubland and woodland, albeit now mostly extinct, fire-maintained grasslands are of little benefit ecologically.

Improved knowledge of the ecological response to fire and fire severity may provide a longer-term view of outcomes under different fire regime scenarios. To this end, two comprehensive datasets of aerial photographs and satellite images extending back to 1945 are available for examining the relationship between forest succession and fire incidence. From these datasets, two sets of time-sequential maps of Hong Kong's forest reserves were created, representing: (i) structural vegetation types in five time periods, 1945, 1963, 1989, 2001 and 2014, and (ii) fire incidence maps from 1973 to 2015.

2. Materials and Methods

2.1. Study Area

The study area comprises the Shing Mun and Tai Mo Shan Country Parks, totaling 2800 ha of rugged and steeply sloping terrain rising to 957 m at Tai Mo Shan, Hong Kong's tallest peak (Figure 1). Due to the long dry season, fire has been a major challenge, along with other natural forces such as tree disease [10] and climatic extremes [11,12]. Frequent fires occur on the steep slopes mainly due to the burning of joss sticks and incense at village graves during the Chinese Chung Yeung (mid-October) and Ching Ming (early April) festivals (Hong Kong Government, 1990). Thus, the upper slopes are covered by fire-maintained grasses while lower elevations support patches of shrub, forest and plantations. The natural vegetation which existed centuries ago is broad-leaved evergreen sub-tropical forest. Recent work [10,13,14] shows that forest cover is succeeding naturally, and independently of re-afforestation programs. However, although a few small patches of 'old growth' forest at least 100 to 150 years old remain, mainly in steep ravines, the 'new' forest is unlike the original native forest in several respects [14,15].

2.2. Data Used

A series of sequential habitat maps (Figure 2a) were available from a previous study, covering five time periods from 1945 to 2014 [10]. These habitat maps were developed by performing object-based image analysis on three sets of aerial photographs (1945, 1963 and 1989) and two sets of high-resolution satellite images (2001 and 2014) [13]. For the fire mapping, a total of 168 Landsat images between 1973 and 2015 (Table 1), were obtained from the United States Geological Survey (USGS)'s Earth Resources Observation and Science (EROS) Science Processing Architecture On Demand Interface (ESPA) [16]. The dataset was analyzed using the Normalised Difference Vegetation Index (NDVI) [17], and image segmentation using the Object-Based Image Analysis (OBIA) [18] was used to allocate pixels to either burned or unburned classes, as these areas were spectrally distinct. For example, Figure 3 shows a large fire which occurred in Tai Mo Shan Country Park during the Chung Yeung Festival on 19 October 1988, when a total of 97 hectares burned. Using the composition of bands 7, 4 and 2 on red, green and blue display, the burned area is shown as red, indicating the high reflectance in short wave infrared after burning. The NDVI rather than the Normalized Burn Ratio Index (NBRI) [19] was used for this study because the NBRI uses the short wave infra-red band which is not available in the earlier Landsat images. However, a comparison between NDVI and NBRI showed a similar ability to differentiate between burned and unburnt areas.

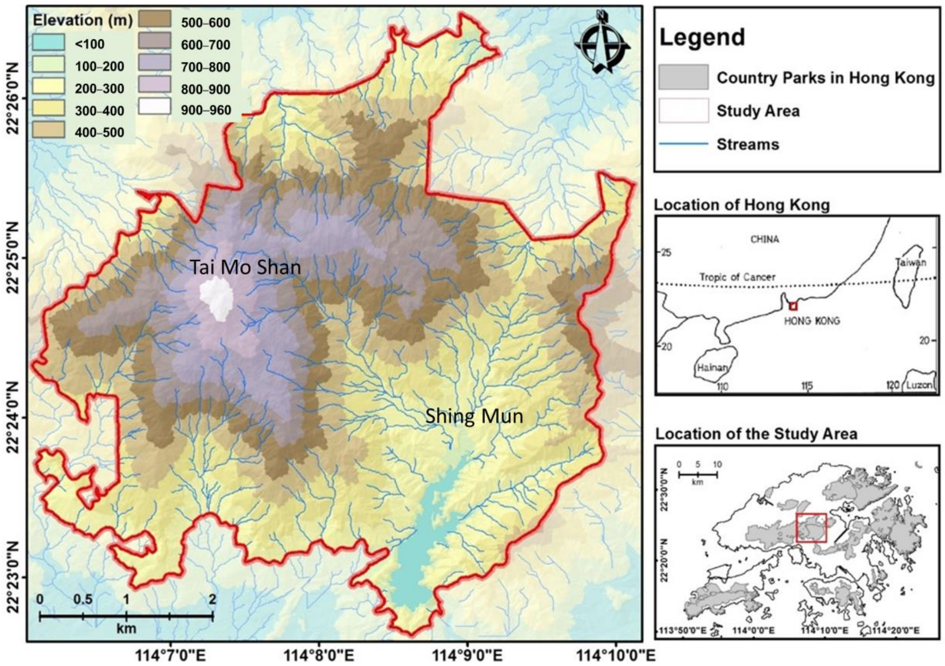

Figure 1. Study area with topography, and its location in Hong Kong and China.

2.3. Data Analysis

To perform analysis a fire frequency map was produced with eight classes corresponding to the number of times burnt, between 1973 and 2014 (Figure 2b). Then the fire frequency map was classified into burn periods corresponding to the reference years of the habitat maps, i.e., 1963, 1989, 2001 and 2014. These classes are given in Table 2. Since fire only occurs during the dry season, and it would be almost impossible for an area to burn more than once in any one dry season due to lack of fuel, the timing used for fire frequency corresponds to 'annual'. Field data from a previous study were available for 28 forest plots of 20 m $\times$ 20 m surveyed in 2014–2015 [14,15], including information on species richness and Basal Area (BA). The fire history of the plots was known by overlaying the fire frequency map.

For the long-term annual time series analysis of the vegetation succession, the linear mixture model technique [20] was applied to develop an NDVI-based fraction of the wood cover index, Equation (1). The existing nearby patches of well-established forest from the 1963 habitat map were taken as reference points to monitor the trajectory of vegetation growth from 1973 to 2016. The fraction of wood cover (or vegetation) index ranges from '0' to '1' which represents the gradient of vegetation growth from grasses or barren land to closed canopies of young secondary forest.

$$\text{Fraction of wood cover} = \frac{\text{NDVI}_{\text{i}} - \text{NDVI}_{\text{b}}}{\text{NDVI}_{\text{f63}} - \text{NDVI}_{\text{b}}} \quad (1)$$

where NDVI represents the Normalized Difference Vegetation Index, i is the image pixel, b is the accumulative temporal minimum value of the index over consistent bare patches, and f_{63} is an average temporal maximum value of the index over consistent dense forest patches from the 1963 habitat map.

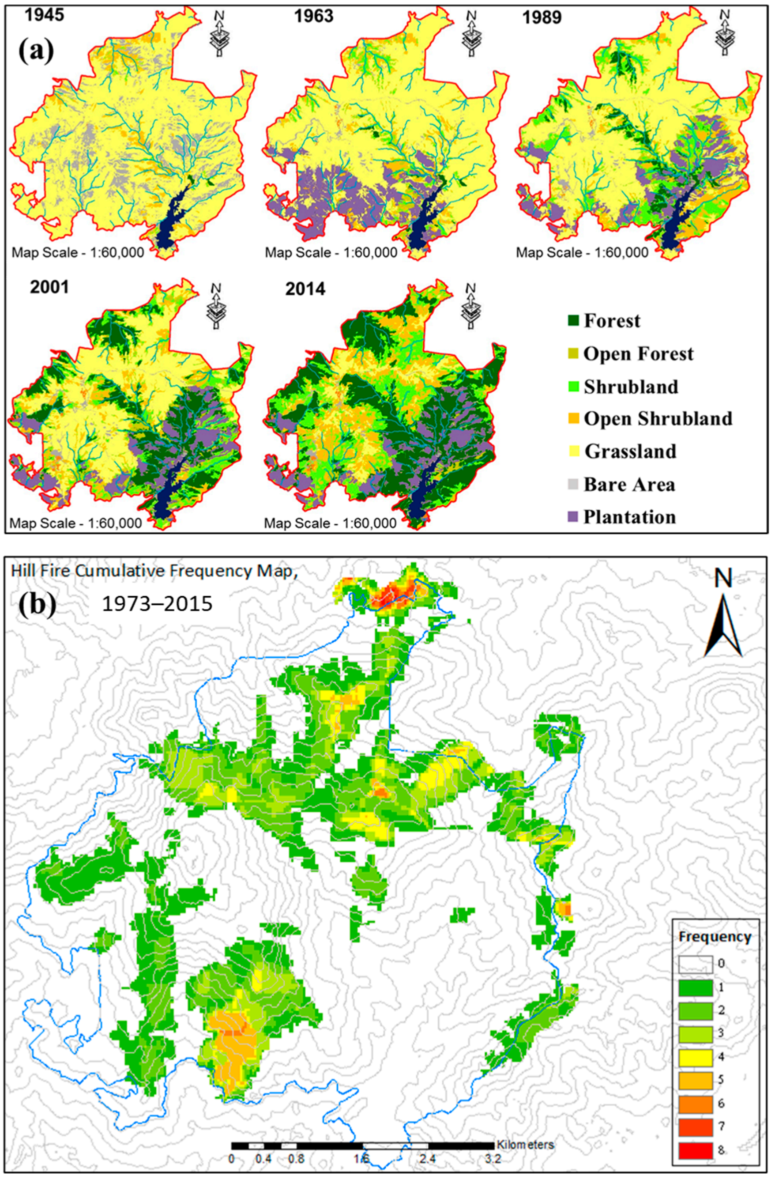

Figure 2. Sequential habitat maps (**a**), and fire frequency maps (**b**).

Table 1. Description of Landsat satellite images used for the burnt area mapping 1973–2015.

Year	No of Images	Day of Year
1973–1974	1	304
1978–1979	1	306
1979–1980	2	292, 310
1987–1988	1	342
1988–1989	5	329, 338, 354, 044, 196
1989–1990	1	231
1990–1991	5	343, 358, 113, 257, 266
1991–1992	1	282
1992–1993	1	285
1993–1994	1	278
1994–1995	1	297
1995–1996	6	293, 316, 031, 040, 063, 136

Table 1. *Cont.*

Year	No of Images	Day of Year	
1998–1999	2	232, 255	
1999–2000	16	287, 319, 328, 360, 002, 003, 019, 042, 107, 179, 195, 211, 242, 243, 258, 259	
2000–2001	13	291, 306, 307, 315, 363, 005, 020, 053, 060, 133, 220, 260, 261	
2001–2002	13	293, 324, 325, 356, 364, 365, 007, 008, 031, 048, 064, 240, 247	
2002–2003	10	311, 312, 010, 018, 019, 027, 058, 131, 187, 235	
2003–2004	10	290, 299, 331, 347, 021, 046, 069, 110, 165, 206	
2004–2005	15	270, 286, 293, 309, 325, 334, 341, 350, 007, 016, 023, 064, 112, 192, 224	
2005–2006	6	288, 295, 327, 042, 243, 266	
2006–2007	10	314, 355, 362, 013, 029, 038, 102, 214, 230, 262	
2007–2008	7	064, 137, 185, 208, 217, 233, 240	
2008–2009	10	272, 297, 329, 336, 011, 018, 034, 123, 139, 251,	
2009–2010	7	283, 299, 014, 030, 078, 085, 206	
2010–2011	7	302, 357, 008, 033, 072, 097, 152	
2011–2012	1	305	
2012–2013	1	221	
2013–2014	11	278, 294, 301, 333, 358, 365, 016, 153, 185, 249, 265	
2014–2015	3	281, 320, 329	
Summary			
Period	No. of Years	No of Years with Images during Dry Season	No. of Season with Fire Event
1973–1974 to 1988–1989	16	5	5
1989–1990 to 2000–2001	12	8	8
2001–2002 to 2014–2015	13	12	10
1973–1974 to 2014–2015	41	25	23

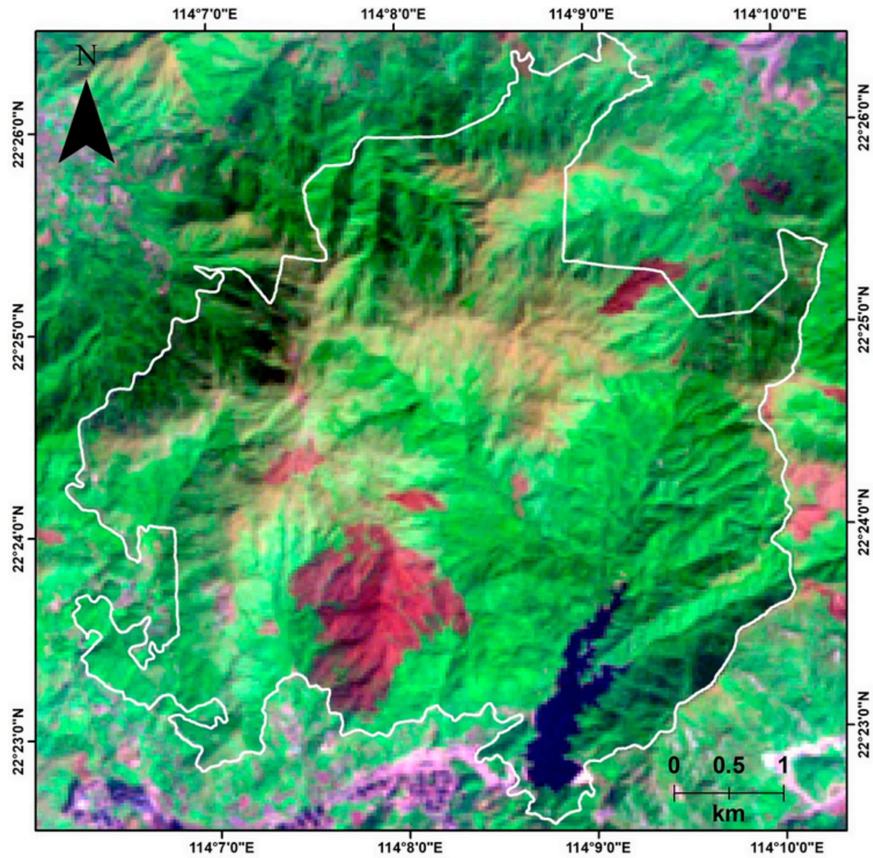

Figure 3. Extract of Landsat 5 Thematic Mapper image of 10 December 1988, RGB band composite 742 showing fire scar in red, from a fire on 19 October 1988.

Table 2. Burn categories according to reference years of habitat maps.

Class	Burn Frequency	Class Designation
1.	Area never burnt since the satellite record (since 1973)	No burn
2.	Area that burnt in all periods, i.e., consistently burnt	All
3.	Area that burnt before 1989 then did not burn after 1989	Early
4.	Area that burnt during 1989–2001	Mid
6.	Area that burnt during 2001–2014	Late
5.	Area that burnt before 1989 and during 1989–2001	Early and Mid
7.	Area that burnt before 1989 and during 2001–2014	Early and Late
8.	Area that burnt during 1989–2001 and after 2001	Mid and Late

3. Results

Table 3 shows that 24% of the study area was burnt at least once in the early period (1973–1989), 7% in the middle period (1989–2001), and 10.2% in the later period (2001–2014). The percentage of the study area burnt per year, however, was lowest (0.78% p.a.) in the middle period 1989–2001. This can be compared with data from an earlier study [10], which indicate that the regeneration of forest was highest in this period, with a very high rate of increase, at 11% per year. The annual percentage of the study area burnt increased again in the latest period after 2001, from 0.78% to 0.91%, which may seem surprising since the area of grassland providing available fuel for fires, had drastically decreased from 35% in 2001 to 8% in 2014.

Table 3. Area (ha) burnt in each of the three study periods, normalized for the number of years when images were unavailable. Percentages refer to % of the whole study area, i.e., 81,664 ha.

	1973/4–1988/9	1989/90–2000/1	2001/2–2013/4
The cumulative area burnt, including > 1 burn	26,932	7593	9525
Burnt at least once	19,392 (24%)	5780 (7%)	7978 (10.2%)
Burnt at least twice	2944 (3.61%)	844 (3.6%)	1586 (1.94%)
Burnt/year (av)	1683 (2.1%)	632 (0.78%)	732 (0.91%)

3.1. Impacts of Fire on Vegetation Growth

Changes in the phenology of burnt and unburnt patches were assessed from the time series analysis of NDVI. Figure 4 shows the phenology of a patch of grassland which was burned in the dry season of two years 2009 and 2011, but not in the intervening year 2010. The NDVI of the burned area fell to 0.3 during the dry season following the first burn but rose again to a normal growing season level of 0.7 for the following two rainy seasons. However, after a second burn the burned area NDVI reached only 0.65, suggesting some degradation of the available resources for growth. This is possibly due to repeated volatilization of soil nutrients, especially nitrogen, sulfur, and phosphorus, as well as organic matter, and a reduction in the cation exchange capacity (CEC) [21].

Overlay analysis of fire frequency maps, habitat maps and time series of the fraction of wood cover or closed forest, produced different trajectories of forest succession. Figure 5 shows the long-term annual fraction of well-established forest cover over the time trajectories for the eight fire categories given in Table 2. An additional class for areas which did not burn transitioning from grassland to forest and shrubland to forest is shown (Figure 5c). It is clear that a single burn (Figure 5a) is less damaging to the succession than two burns (Figure 5b). However, no burn (Figure 5c) gives the best outcome in terms of transition to forest cover, as 100% of woody cover, i.e., closed canopy forest is achieved after approximately 35 years of grass transitioning to forest.

Figure 6 shows the structural succession of an area which did not burn over the study period. The patch was classified as Grass in 1973, Shrub in 2001, Shrubby Grassland in 1989 and Forest in 2014. The slide suggests that if fire is excluded, it takes approximately

13 years to transition from shrub to forest, and 35 years to transition from grass to forest. The initial transition out of grassland to the shrub stage appears protracted, in this case, 28 years. Once the transition to shrub has occurred, the forest succession appears faster. Fluctuations in the graph may reflect a response to climatic variations and events such as a severe frost event in January 2016, which explains the dip in the curve in 2016.

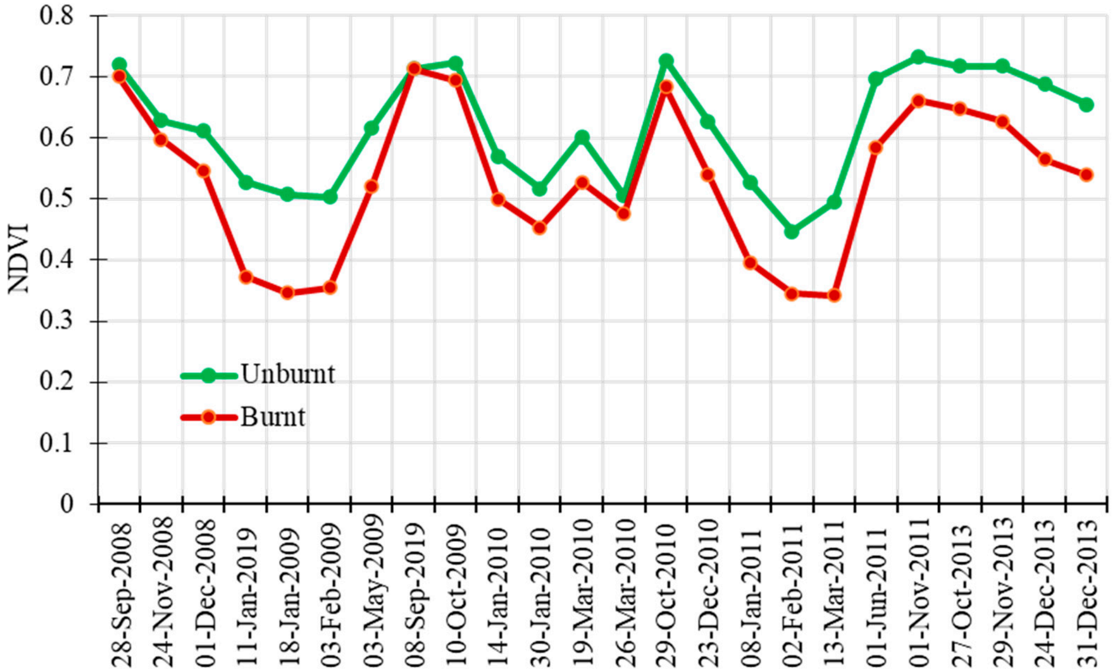

Figure 4. Normalized Difference Vegetation Index (NDVI) phenology over four years for grassland.

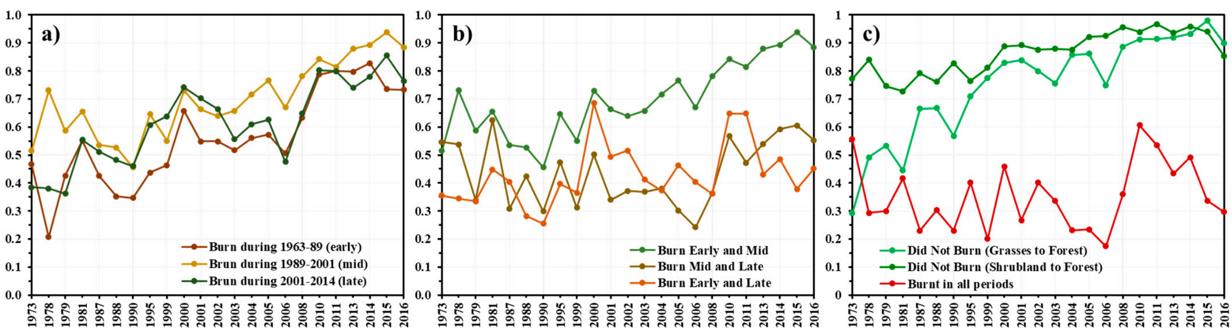

Figure 5. Time series graphs of a fraction of vegetation cover from 1973 to 2014 according to fire frequency and habitat maps: single burn (**a**), two burns (**b**), no burn (**c**). The Y-axis represents % of closed forests.

3.2. Impacts of Fire on Vegetation Successional Classes

Figure 7 shows the vegetation structural classes for the four reference dates, for areas which have undergone the eight fire regimes specified in Table 1. Figure 7d,e represent the extreme cases of areas which burned in all three intervening periods and areas which did not burn, respectively. By 2014, areas which burned consistently since 1973 (Figure 7d) are still in the early stage of structural succession of grassland and open shrubland. On the other hand, areas which did not burn over the years (Figure 7h) have gradually transformed from early to late structural succession stages, and now have 56% forest and 18% shrub or open shrubland.

Furthermore, areas undergoing a single burn, whether during early, mid, or late periods, fared better than those which burned more than once, and have approximately 25% of closed canopy forest. Nevertheless, once-burnt areas had less than half of the amount of forest by the end of the period, than unburnt areas. Given the destruction of the site and soil conditions by fire [21], and the greater nutrient demands of woody vegetation, it may seem surprising that twice burnt areas (Figure 7e–g) all show some signs of forest regeneration. Notably, graph Figure 7e (burnt both early and mid) had attained 7% of forest cover by 2014.

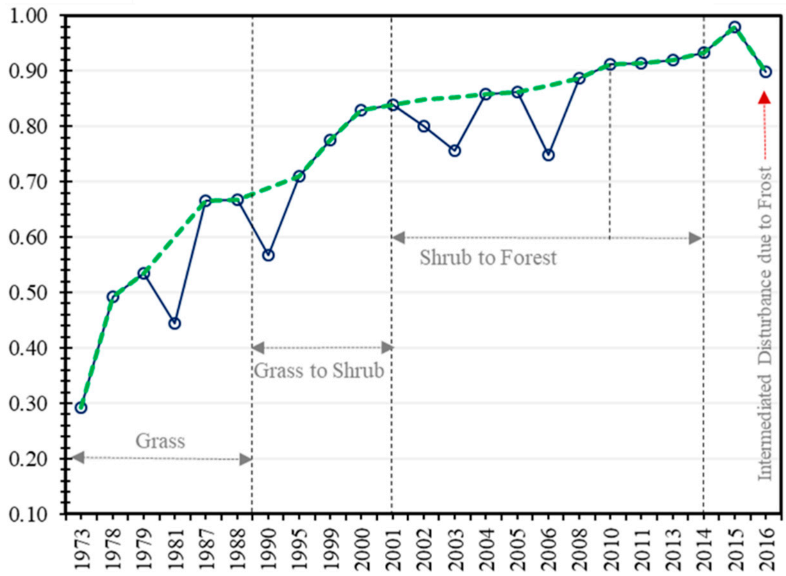

Figure 6. The structural succession of an area which did not burn between 1973 and 2016. The Y-axis represents % of closed forest, blue/solid line indicate NDVI values while the green/dash line connects the maximum value in moving window to indicate smooth transition.

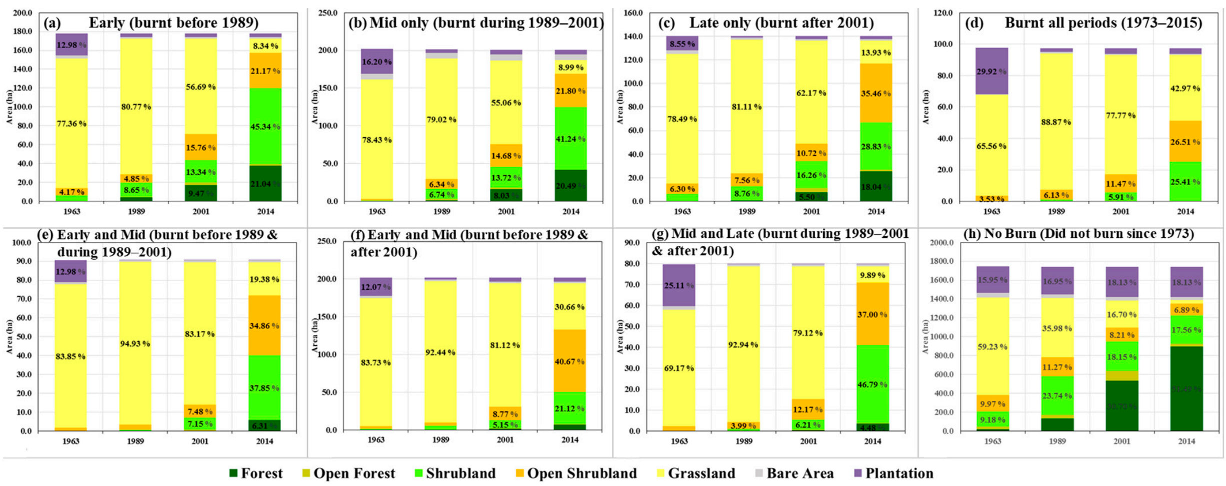

Figure 7. Vegetation structural classes at four reference dates, for areas which have undergone different fire regimes: (**a**) burnt before 1989 then did not burn, (**b**) burnt during 1989–2001, (**c**) burnt after 2001, (**d**) burnt in all periods, (**e**) burnt before 1989 and during 1989–2001, (**f**) burnt before 1989 and during 2001–2014, (**g**) burnt during 1989–2001 and after 2001, and (**h**) did not burn.

It should be noted that areas which did not burn (Figure 7h), had progressed to late successional stages, except for 18% of the area occupied by the Plantation class (purple), which had not changed in the last 50 years (Figure 2a). As plantation forest is closed canopy, it is difficult for fire to penetrate the cooler and more humid forest environment, and their poor understory makes plantations even less susceptible to fire. Additionally, after 1965 the Hong Kong authorities concentrated on fire-resistant species other than the native *Pinus massoniana* and Eucalypts [9] which are fire-susceptible. Thus, although plantation forestry may be effective for watershed protection and carbon cycling, in this case plantation forests have constituted a barrier to the natural forest succession. Furthermore, plantations of single or few species make them susceptible to disease and can be rapidly destroyed. This point is illustrated in Figure 2 which shows that extensive Plantation forests in the southwest of the study area (purple areas on the maps) had disappeared by 1989. This was due to a deadly pine nematode which infected the native *Pinus massoniana* plantations in the late 1970s. Thereafter the areas became subject to frequent dry season fires, and even by 2014, they were still largely grassland or in early successional stages of open shrubland. The fire frequency map (Figure 2b) indicates that these former plantation areas were the most frequently burnt. The large, burned patch on the Landsat image of December 1988 (Figure 3) corresponds almost exactly to the area of former plantations and the highest frequency of fires.

3.3. Plot Data

Out of the 28 forest plots surveyed in 2014, the old-growth forest plots have significantly higher biomass in terms of basal area, than the plot average (Table 4). Furthermore, the biomass increases from two burns to one burn to no burn to old growth, with two burns having only 1/3 biomass compared to a single burn, and 1/4 that of no burn. Therefore, the carbon implications of grassland fires are negative not only for the loss of herbaceous biomass but also for the greatly reduced woody biomass in the subsequent forest succession. An overall summary of area burnt and number of trees damaged during the three study periods is given in Table 5.

Table 4. The basal area representing biomass, and species richness indicates the number of species, in 28 forest plots surveyed in 2015, grouped according to fire history.

Fire History	Mean Basal Area (m^2)	Mean Species Richness
Old Growth Forest	67.3	19.5
Unburnt plots	45.9	35.6
Once burnt plots	30.3	30.4
Twice burnt plots	11.4	21.3

Table 5. Fire record data from the Hong Kong Agriculture, Conservation and Fisheries Department showing the total area burned and number of trees damaged during the three study periods.

Period	Area Burnt (ha)	No of Trees Damaged
1973–1988	30,423	19.5
1989–2000	45.9	35.6
2001–2014	30.3	30.4

Species richness appears inversely related to the number of burns, being higher for the unburnt plots and lower for the twice-burned plots, which is predictable, from the duration of time since the grassland phase. The old growth plots have the fewest species, but this may be explained by the intermediate disturbance hypothesis where species richness normally declines in the intermediate phase of succession due to a decline in the dominant competitors which gives way to more shade-tolerant mid-successional species [22].

4. Discussion

Fire control is an important factor in policies promoting large-scale recovery of secondary forests [23]. Contrary to historic notions about the destructiveness of fires, fire is now considered a fundamental ecological process which influences the structure of tropical vegetation [24–27]. The impacts of fire on secondary forests are regulated by a variety of factors including climate [25], the structural stage of succession [23,28], vegetation type [29], natural biodiversity [30], natural and anthropogenic disturbances [31,32], as well as the extent and intensity of the fire [27,33].

Hong Kong's native forest vegetation is not fire tolerant, but under current climatic conditions, natural fires would have been very rare in an undisturbed forested landscape. It may therefore seem surprising that forest has been able to regenerate naturally in many areas of the degraded landscape despite repeated burning. Climate change may exacerbate the susceptibility of tropical/subtropical landscapes to fire [34] and may influence ecosystem resilience [35] and post-fire recovery [36], depending upon the loss of other resources, such as seeds and their dispersion in the landscape [37]. In degraded landscapes, fire can greatly delay the structural succession from grass to shrub and/or forest [38].

The data presented here are deficient in details such as burn intensity, site condition before burning etc., so they do not allow to precisely judge the length of time to transition from one vegetation structural class to another. Newly developed Harmonized Landsat and Senintel-2 (HLS) images [39] will increase the availability of satellite imagery at improved spatial and temporal scales, which should enhance the methods to efficiently detect changes in vegetation structural stages. Nonetheless, these new integrated products lack historic temporal coverage. Lacouture et al. (2020) [40] also documented the ineffectiveness of daily Landsat NDVI composite images, due to cloud cover, to detect post-fire vegetation recovery in the frequently burnt habitat of subtropical pine savannas in the southeastern U.S.A [40]. However, they do indicate the possible minimum time for the succession, e.g., it is possible to transition from grass to forest in 30–40 years and shrub to forest in 15–20 years if fire is excluded in a degraded landscape such as Hong Kong. The relatively rapid succession of shrub to forest may be related to the fact that leaves and small twigs of shrubs are not combustible in the absence of grass as fuel, and can regenerate after light burning from buds lying below the bark [41]. The seed input into the grassland of the study area is high [42] with a low seed predation rate [43] due to loss of vertebrate fauna. After the first decade of structural succession, the grass and/or shrub-dominated landscape gradually transformed into light-demanding pioneer tree species [44]. Thus grassland can transform into shrubland if protected from fire for 10–15 years [41,44,45].

A further reason that the succession from shrub to forest is relatively fast, is that shrubs act as cover for birds such as bulbuls and small fruit bats which, given the disappearance of most native forest fauna, have become the major seed-dispersers between patches of the forest [2–4]. Without suitable perches, these dispersal agents will not move across treeless grassland areas, and our data indicate that successive fires prevent the establishment of woody species. In view of our observation (Figure 3) that the transition from grass to shrub stage can take up to three decades, even in the absence of fire, some measures to accelerate this stage of the succession may be recommended. This may include strategic planting of woody shrubs dispersed between patches of forest, at median seed dispersal distances of 40–130 m, depending on the fruit species, bird species and season of the year [46]. Our observations that repeated burning prevents shrub establishment also support observations from controlled experiments on Hong Kong's hillsides [41], where two-thirds of tree and shrub seedlings were unable to withstand four consecutive fires. Therefore, it is likely that the 23% of the study area (i.e., 640 ha) still under grass or open shrubland by 2014, has been subjected to repeated burning since at least 1973, the earliest date of Landsat images, and probably since 1945, the start of this study.

We found that the species richness and (woody) biomass accumulation decreased with increasing frequency of burn frequency (Table 4). It is noteworthy that post-fire structural recovery may be faster than the turnover of species composition and floristic diversity, compared with a pre-fire successional stage [37,47]. For instance, Slik et al. (2008) [48] studied the impacts of burn frequency in the eastern Bornean rainforest and found rapid recovery in structural attributes of the forest compared to a limited recovery in species richness [48]. Despite the post-fire rapid recovery of structural attributes in the recovering patches of secondary forest, the overall biomass was significantly reduced compared to those without burning [48,49]. This implies that frequent fires not only reduce the accumulation of woody biomass, but also delay the process of species accumulation and the ecological progression of early to mid and/or late successional species [38]. In moist subtropical ecosystems such as Hong Kong, frequent fires are less damaging to the structure and composition of later successional stages, than to the more open forest at early successional stages which comprises more shrubby understory [50]. Therefore, controlling fires at the early stage of forest succession is critical to assist the large-scale natural succession of secondary forests [51]. The transition of landscape patches from grass to forest decreases the fire-prone area which favors forest growth and canopy closure [24], and acts as a natural barrier to fire [23,51].

The topography of a landscape also plays a pivotal role in characterizing fire-prone areas and post-fire vegetation recovery [52]. Structural succession is relatively faster at the valley bottoms due to the suitable microclimate and availability of seeds [53]. The speed of recovery increases from valley bottoms upwards. The spatial pattern of forest recovery in the degraded landscape of Hong Kong also showed a delayed transition from the early (grass) to late (forest) structural stage at the mid and upper slopes [10,54]. This also explains the higher accumulation of species, determining species composition and diversity, at valley bottoms compared to mid or upper slopes [14].

Other major determinants of succession besides fire which are recognized in tropical secondary succession [1] have not been considered here. These include microhabitats associated with environmental conditions [55–58], local topography [59–62], seed dispersal and predation, natural disturbances, climatic extremes and edaphic factors [63–66]. Although it is known that woody species are more demanding of soil nutrients than grass, a single burn appears to have minimal impact on subsequent vegetation vigor, and even following two burns, soil conditions are evidently able to support the establishment of woody species after 15–20 years of no fire (Figure 7e). A study of the edaphic impacts of fire in Hong Kong [21] found that repeated fires reduced soil organic matter by 85%, the cation exchange capacity by 85%, nitrogen by 75%, phosphorus by 66%, as well as significant reductions in other essential cations. However, soil nutrients would be expected to be rapidly replenished in a hot humid climate such as Hong Kong, from annual dieback of grasses, soil weathering and atmospheric inputs. Hong Kong's sub-tropical climate is also subject to climatic extremes such as typhoons and frost events [11,12], both of which may interrupt the natural succession, by damaging or killing trees and shrubs. Although it is rare for woodland and shrubland to catch fire, large amounts of branches and leaves covering the forest floor following such events are deemed to pose a fire risk in the following dry season [11], especially in areas adjacent to grassland.

5. Conclusions

This study investigated the impacts of fire on a 70-year tropical/subtropical secondary forest succession by integrating the burn frequency, derived from Landsat time series 1973–2015, and four stages of the structural succession of secondary forest obtained from high-resolution habitat maps of 1945, 1963, 1989, 2001 and 2014. It is evident that in a rare site such as Hong Kong where forestry is actively regenerating on severely degraded land, even recovery to the shrub stage may take several decades, even in the absence of fire. (The Hong Kong case is different from many other studies, which are of slash and burn sites). Thus, we may deduce from the results that controlling fire could help to facilitate the succession of tropical/subtropical secondary forests in degraded landscapes where the natural forest is recovering after abandonment. Furthermore, our study has provided evidence that monocultural plantation forestry may have poorer outcomes than natural forest regeneration, ultimately leading to annual grassland fires.

Global warming has seen a vast increase in fire incidence worldwide in the last two decades, which may also partially account for the increase in fire incidence after 2001 observed here. It is also associated with an increase in extreme weather events, such as super-typhoons and severe frost events, which occurred in Hong Kong in 2018 and 2016, respectively. This resulted in dead trees and debris within the woodland, potentially providing an enhanced fuel supply for fire. This situation would lead to increased fire intensity, with greater ecological damage than grassland fires. This study was unable to test for fire intensity because we used the remote sensing indices only to measure the rate and magnitude of recovery after fire, which depend not only on fire intensity but on many other variables.

The areas subject to the highest frequency of fire in the study area correspond to former plantations which burned almost annually following destruction by disease. As plantations have lower ecological value than natural forests and are disease-sensitive, encouraging native species encroachment into existing plantations would provide long-term stability, given the short lifespan of many plantation species. This has already occurred to some extent, and planting in the last three decades has involved a mixture of species, both exotic and native.

Controlling fires in the remaining grassy hillsides should be much simpler henceforward, due to the much-reduced area of early successional grass and open shrubland, and the ability of spatial analysis to pinpoint the most likely sites. The almost total deforestation of the landscape before 1963 (Figure 2a), followed by gradual forest regeneration since 1963 suggests that the partial fire suppression by authorities since the 1960s has been effective overall. However, as observed in Section 3, the percentage of area burned annually had increased since 2001, even though the area of grassland providing fuel had decreased drastically from 35% to 8%. This may be due to increased human activities encroaching on the Country Park's land, such as around village settlements, fewer fire suppression activities by authorities in recent years, climatic factors such as higher temperatures, or a combination of these.

For the future, acceleration of the shrub establishment stage by strategic planting of woody species to enable the major seed dispersers to traverse open areas of grassland is recommended. Additionally, recent advances in drone technology for forest and wildlife monitoring [67] offer more efficient fire monitoring and may assist the acceleration of natural forest succession.

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