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Will *Acacia* secondary forest become rainforest in the Australian Wet Tropics?

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ABSTRACT

Fragmentation presents a threat to tropical forest biodiversity and restoration can be expensive. Secondary forests regenerating on abandoned pasture are widespread, represent an opportunity to restore rainforest at minimal management cost, but can become arrested in a state dominated by a single tree species. Species richness and diversity was assessed from 26 sites in *Acacia* secondary forests in the Australian Wet Tropics of varying age since abandonment and the influence of rainfall and soils, and the context of remnant mature forest on succession were assessed. Stand structure indicated a lack of *Acacia* recruitment. Late successional species richness and diversity increased with age indicating recruitment under the *Acacia* canopy. The species richness of late successional tree species with fruit size 10 mm or larger also displayed an increasing trend with age, although it was statistically not significant. Forest succession progresses in *Acacia* secondary forest and large seeded tree species are able to recruit. The enhancement of rainforest succession with fertile geology, increased rainfall or with more remnant forest in the vicinity was not evident in this study. Secondary forest even when dominated by a single species, particularly a nitrogen fixing legume, represents a viable means of tropical forest restoration provided there is sufficient mature forest in the region to act as a seed source.

1. Introduction

Large areas of tropical forest have been cleared throughout the world, mainly for the establishment of pastures and agriculture land (Holl et al., 2000; ITTO, 2002; Paul et al., 2010), with dramatic consequences for the specialised and diverse biota (Laurance et al., 2011). The preservation of mature tropical forest alone is insufficient to sustain current biodiversity levels (Gardner et al., 2009), presenting an urgent need for their restoration. Ecological restoration of tropical forest through direct planting and appropriate maintenance is costly and not cost effective when measured against the net value of benefits provided by such ecosystem services (Birch et al., 2010). A recent estimate from Australia indicated that the cost of ecological restoration, including the subsequent maintenance of the forest plots can be in excess of AUD\$30,000 ha^{-1} (Kanowski et al., 2008b). Although there have been successful direct planting restoration programs, the cost and the need for an onerous and long-term maintenance program is prohibitive (Kanowski, 2010). Successful restoration plantings also represent a miniscule proportion of cleared land despite millions of dollars in investment in Australia alone (Kanowski et al., 2008c).

A passive approach to forest restoration utilises the natural regeneration process of secondary forests and represents a potentially viable and cost-effective solution (Kanowski et al., 2008c; Norden et al., 2009; Letcher and Chazdon, 2009; Birch et al., 2010; Chai and Tanner, 2011). The success of passive restoration can be measured by the time taken to recover the structure, composition, and function of the original forest. This seems to be dependent on the nature of land-use after clearance (Moran et al., 2000) and the landscape context (Kouki et al., 2011). Secondary forests can develop species richness and structural characteristics similar to mature forest in as short as 30–40 years (Aide et al., 2000; Letcher and Chazdon, 2009). However, forest succession can also be arrested for long periods or indefinitely, depending on various site conditions (Aide et al., 1995; Zahawi and Augspurger, 1999; Mesquita et al., 2001; Lugo, 2004; Wieland et al., 2011). Secondary forests affected by arrested succession are often dominated by a single species, but not all mono-dominant forests are arrested (Connell and Lowman, 1989).

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Long-lived mono-dominant early succession tree species that are able to regenerate under their own shade are particularly likely to result in arrested succession. Where a self-perpetuating mono-dominant forest inhibits the recruitment of late successional species the viability of a passive forest restoration strategy is precluded. An example of such arrested succession is the camphor laurel forest of the Big Scrub region in New South Wales, Australia. The exotic camphor laurel (*Cinnamomum camphora*) forms large areas of mono-dominant secondary forest and is able to recruit under its own canopy (Neilan et al., 2006; Kanowski et al., 2008c). Although it includes native rainforest species recruiting in its understorey (Neilan et al., 2006; Kanowski et al., 2008c), the longevity of the camphor laurels means that the recovery of forest with the structure and composition of undisturbed forest is not likely over multi-century time-scales (Neilan et al., 2006; Kanowski et al., 2008c). Other examples of succession that has been arrested by mono-dominant secondary forests includes *Baccharis trinervis* in Ecuador (Zahawi and Ausperger, 1999) and *Vismia* sp. in the Amazon Basin (Mesquita et al., 2001).

Secondary forest succession can be retarded by the context of surrounding land-use, and the climate and soil environment (Chazdon, 2003). Seed viability in tropical tree species is short and surrounding forest is an important source of propagules (Parrotta et al., 1997; Holl et al., 2000; Standish et al., 2007; Chazdon, 2008; Holl and Aide, 2011). Percentage of seed rain and animal dispersed seeds 5 m from any forest vegetation is drastically reduced compared to the forest edge (Holl, 1999; Peterson and Carson, 2008) and it is even more pronounced for larger, heavier seeds (Peterson and Carson, 2008).

The reliance on frugivorous seed dispersal by a majority of tropical forest tree species (Howe and Smallwood, 1982) means that fruit size in relation to the gape size of frugivores is an important dispersal limitation (Moran et al., 2004; Kanowski et al., 2008b). Frugivores with smaller gapes were found to be more likely to traverse between secondary forests and remnants while birds with larger gapes tend to be more restricted to remnant forest (Moran et al., 2004). It is expected that large fleshy-fruited forest species will be poorly represented in secondary forest.

Succession will be more rapid on fertile soils than infertile soils (Frye and Quinn, 1979; Tucker et al., 1998; Moran et al., 2000; Johnson et al., 2000) and where precipitation is high (Gentry, 1988a; Gentry, 1988b; Vazquezyanes and Orozcosegovia, 1993). However, although a fertile substrate leads to greater basal area increases than sites on less fertile substrate (e.g., volcanic), it also has an increase in exotic species colonisation (Johnson et al., 2000; Chazdon, 2008). Fertility is also an important determinant of which pioneer species recruit into a site (Chazdon, 2008).

Secondary forest dominated by *Acacia* spp. occurs over large areas of abandoned pastures in north-east Queensland on areas that were previously tropical forest. This study firstly aims to determine whether there is recruitment of a diverse tree flora under *Acacia* canopies, testing the alternate hypothesis that *Acacia* secondary forest represents an arrested succession. Arrested succession of *Acacia* secondary forest would be indicated by: (1) stand structure of the dominant *Acacia* suggesting continuous replacement, and (2) a floristic composition of *Acacia* forest stands unrelated to forest age.

A second objective is to examine the extent to which factors such as rainfall, soil fertility and landscape context affect regeneration dynamics. Explicitly we test the hypothesis that succession will be (1) more rapid at high rainfall sites on fertile soils; (2) the development of late successional species richness and diversity will be enhanced with proximity to remnant forest, and (3) the richness of species with larger seeds will increase with age of secondary forest.

2. Material and methods

2.1. Study area

The 18,000 km^2 Wet Tropics biogeographic region of tropical Queensland, Australia, comprises large tracts of tropical forest (or rainforest), which makes up about two-third of the region (Harrison et al., 2003; Florentine and Westbrooke, 2004). However, it also includes extensively modified agricultural landscapes such as the Atherton Tablelands where most of the tropical forest have been cleared, with approximately 900 km^2 of fragmented remnant forest remaining (Harrison et al., 2003; Florentine and Westbrooke, 2004). Secondary forest occurs on abandoned agricultural land with mono-dominant stands of *Acacia* being a common form (Florentine and Westbrooke, 2004; Erskine et al., 2007; Appendix A). *Acacia* is an effective coloniser because its seed remains viable in the soil for long periods compared to rainforest species, which generally have a viability of less than six months (Hopkins and Graham, 1987; Irlbeck and Hume, 2003). *Acacia* seeds are dispersed by frugivores which feed on its arils (Kanowski, 2008a).

All the sites for the current study are situated on the Atherton Tablelands in north-eastern Australia (17°21′S 145°42′E–17°07′S 145°36′E) and is representative of the range of environments were where *Acacia cincinnata* and *Acacia celsa* form secondary forest. The distribution of *A. celsa* extends 200 km north of the Atherton Tableland (Macdonald and Maslin, 2000). *A. cincinnata* is restricted to the rainforest margins in Atherton Tablelands, especially in areas with higher precipitation (CSIRO, 2001). Mean annual rainfall across the 120 km^2 area varies between 1500 and 2000 mm of rainfall per annum (Xu and Hutchinson, 2011). Metamorphics, granitic, and basaltic soils makes up the major geological parent material of the study area.

2.2. Site selection

Twenty-six accessible secondary forest sites with either *A. cincinnata* or *A. celsa* as the canopy dominants were selected to represent a range of ages since disturbance (Appendix A). Sites were selected by the image-pattern of mono-dominant *Acacia* forest canopy using Google Earth (Appendix A).

The age of the secondary forest was determined using historical aerial photographs dated 1943, 1949, 1952, 1961, 1965, 1971, 1974, 1976, 1978, 1983, 1986, 1992, 1997, 2000, 2003, 2007, and 2011. The age since abandonment was determined by taking the middle value between successive images showing pasture and then a developing forest. Sites abandoned earlier than 1943 and unable to be categorised to a specific age were assigned an age of 70 years and 100 years if the site had a closed canopy in the 1943 aerial photograph. All of the sites were abandoned pastures with the exception of two sites that were heavily logged, or cleared and immediately abandoned, and another site for which the status of the original clearing could not be determined (Appendix B).

2.3. Species composition survey

Trees were identified and their diameter at breast height (dbh) measured between April and June 2012 either side of a 50 m tape (with each end marked with a GPS) within three different plot dimensions: 50 × 10 m, large trees (>10 cm dbh); 50 × 5 m, small trees >2 m height, <10 cm dbh and 50 × 0.4 m, saplings >50 cm, <2 m height were identified and counted. In the narrow quadrat, only *Acacia* seedlings <50 cm height were counted and were only included in the size class structure analysis. Other species <50 cm in height were not included due to difficulties in identification.