Sustainable management of dipterocarp forests in the Philippines

by

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Abstract

Southeast Asian dipterocarp forests are highly diverse tropical forest communities, with the family Dipterocarpaceae alone comprising hundreds of species. Dipterocarps are remarkable in that they exhibit supra-annual mass flowering events, which occur in irregular intervals of two to ten years, normally involving several dipterocarp species and sometimes including many other plant families as well, a phenomenon known as general flowering event. Dipterocarps are a leading species group on 85% of Southeast Asia's forested land base and they are also commercially important, representing a quarter of global consumption of tropical timbers. Southeast Asia's dipterocarp forests are also one of the most threatened tropical ecosystems in the world. Only 16% of the total original forested area remains classified as primary forest in Southeast Asia and less than 6% in the Philippines, which is a focus of this thesis. The overall objective of this thesis is to support and improve reforestation and community forestry initiatives in dipterocarp forests by contributing ecological insight and compiling local community knowledge, and by addressing major impediments to successful forest restoration in Southeast Asia.

The first issue is the biology of dipterocarp reproduction that makes it hard to manage natural regeneration as well as reforestation. It is logistically difficult to mobilize resources to collect short lived dipterocarp seed without knowing in advance when flowering will occur. This thesis research investigates what environmental factors drive dipterocarp mass flowering and tests alternative resource accumulation and trigger models to predict mass flowering. Using a variety

of candidate predictor variables (precipitation, cloud cover, minimum temperature and El Niño indices) a plausible environmental trigger could not be found (median AUCs around 0.55 indicating near random predictions), while the best resource accumulation model had a median AUC of 0.70, which could be improved to 0.75 when the date of previous flowering was included in the model. Further, the analysis revealed that a simple resource accumulation by individual trees can cause inter- and intraspecific flowering synchronization leading to community-wide general flowering events.

The second issue is a lack of knowledge of which dipterocarps and which other native species are suitable for reforestation. Community based forest management programs typically use readily available exotic species that may not be desired by the local communities. Choosing species for reforestation programs or community forestry in species-rich tropical rainforest ecosystems is a complex task. Reforestation objectives, social preferences, and ecological attributes must be balanced to achieve landscape restoration, timber production, or community forestry objectives. In a case study for an upland tropical rainforest in the Philippines, socioeconomic preference in five forest-dependent communities were surveyed. In addition, ecological suitability of tree species for open-field plantations was inferred from growth rates, density and frequency of native tree species in long-term monitoring plots. Notably, ecological suitability indicators and socioeconomic preference ranks were generally negatively correlated, with few species being classified as both ecologically suitable and socioeconomically valuable. The results also highlight that reforestation species must be carefully chosen, and that species-rich tropical rainforests are not an easily renewable natural resource. Secondary and planted

forests do not serve socioeconomic needs of forest-dependent communities as do original native forests.

Finally, many communities rely on forests to supplement their livelihoods but these resources become increasingly limited because of deforestation. There is a need to protect the remaining dipterocarp forests for their ecological functions without restricting community use of natural resources and increasing poverty rates. A case study for the North Negros Natural Park investigates how land use, land use planning, and protected area management affects communities in and around a major forest reserve in the Negros Island in the Philippines. The analysis is first carried out at the provincial level using secondary socioeconomic data derived from government statistics. The second analysis was conducted at a local scale, where the socioeconomic status of communities close to a protected forest was compared to more distant communities. Generally, resources are more abundant in areas close to protected areas where population densities are low. While rural communities have fewer amenities and infrastructure than urban centers, communities close to a protected forest where non-destructive activities are allowed were better-off than other rural communities farther from protected areas.

Preface

Thesis chapter 2 has been submitted to a peer-reviewed journal under a title "Climatic drivers of dipterocarp mass flowering in Southeast Asia", authored by Mariya Chechina and Andreas Hamann. The study was conceived and designed by AH and MC. Data on flowering and fruiting was gathered and analyzed by MC with contribution of analysis tools and climate data by AH. First draft and revisions were conducted by MC and AH.

Thesis chapter 3 has been accepted for publication in the peer-reviewed journal *Ecosphere* under a title "Choosing species for reforestation in diverse forest communities: social preference versus ecological suitability", authored by Mariya Chechina and Andreas Hamann. The study was conceived and designed by AH and MC. Data from forest plots was gathered by AH (1995 and 2003) and Mariya Chechina (2012). Interview data was gathered by MC. MC performed the analysis and wrote the paper with revisions by AH. The fieldwork was conducted under a wildlife gratuitous permit no. R6-2012-10 through Philippine Biodiversity Conservation Project issued by the Department of Environment and Natural Resources in Iloilo City, on June 20, 2012 and expired in June, 20, 2013. The ethics approval (Pro00027508) for the questionnaire was obtained through the Human Research Ethics Review process at the University of Alberta Ethics Board, valid from February 28, 2012 to February 28, 2013.

Chapter 4 has been submitted to a peer-reviewed journal under the title "Effects of land use and protected area management on socioeconomic conditions of forest dependent communities: a case study in the North Negros Natural Park, Philippines", authored by Mariya Chechina, John R. Parkins, and Andreas Hamann. The study was conceived and designed by MC, JRP and AH. MC collected interview and census data, performed the analysis, and wrote the paper, with revisions from JRP and AH. The ethics approval (Pro00027508) for the questionnaire was obtained through the Human Research Ethics Review process at the University of Alberta Ethics Board, valid from February 28, 2012 to February 28, 2013.

Dedication

To my sister, Olya, who always believed in me

To Denali Sylvain

To the people of Patag

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Chapter 1 - Introduction and literature review

1.1. Dipterocarp forests of Southeast Asia

The family *Dipterocarpaceae* is the dominant forest type in Southeast Asia. Dipterocarp family contains 16 genera and 580 species that are found in Southeast Asia, 3 genera with 50 species are found in Africa and 1 genera containing 1 species in South America (Appanah et al. 1998). The Philippines ranks fourth in Southeast Asian dipterocarp species diversity with 65 species (Langenberger 2006), after Borneo (267 spp.), Peninsular Malaysia (155 spp.) and Sumatra (106 spp.). Dipterocarp forest is the most extensive type accounting for 85-90% of standing timber (Rojo 1993). More than 65 % of all tree species present in the Philippines are endemic (Langenberger 2006). Besides being used for their hardwood, dipterocarps also provide many non-timber uses from their seeds. Species of the *Shorea* genus produce seed kernels that are used for their fat as food by the locals or for export to be used in cosmetics and creams (Blicher-Mathiesen 1994). As a result, deforestation in Southeast Asia is attributed to the loss of dipterocarp forests; impacting local communities, national economies, and creating the need for reforestation and restoration of this ecosystem (FAO 2003). Dipterocarp timber, marketed as "Philippine Mahogany" comprised approximately three quarters of world trade in tropical timbers in first half of the 20th century (Kummer and Turner 1994) until the resource was virtually exhausted for economic purposes around the mid-80s and harvesting pressure shifted to neighboring Southeast Asian countries.

Dipterocarp forests in Asia usually occur in the lowlands ranging from 0 to 1200 meters in elevation and occupy the emergent stratum, although they are also found in the understory (Ashton 1988). They grow on different soil types but thrive on poor stony soils (Smitinand et al.

1980). Higher species richness of dipterocarps is found in the wet aseasonal tropical regions (Appanah 1985). Dipterocarps mainly occupy mature stages of primary forest and can also colonize secondary forests (Appanah et al. 1998). Dipterocarps rely on variety of pollinators for fertilization (Ashton 1988, Ashton et al. 1988) as they exhibit high outcross rate and high abortion rates of fruits that are self-fertilized (Ghazoul 1997).

Dipterocarps in the aseasonal regions flower in a highly synchronized manner every two to ten years, usually without a common interspecific pattern (Ashton 1988). These mass flowering events can occur over large areas and involve many species of dipterocarps and trees from related families (Ashton 1988, Appanah 1993). This phenomenon is particular to the aseasonal tropics of Southeast Asia and has not been recorded to occur in African or South American tropics (Appanah 1985). The mass flowering events require a lot of energy to produce big fruits in large quantities. As a result, tree growth is suspended substantially during mass flowering years (Ashton 1988). Mass flowering, even though energy intensive, serves a purpose of escaping seed predation by i) limiting predator abundance with non-flowering years and ii) saturating food source for predators during mass flowering and fruiting (Curran and Leighton 2000).

While the evolutionary purpose of mass flowering is well understood, the mechanism by which the flowering becomes synchronized is not entirely clear. For localized events, plant chemical signaling could provide a cue (Holopainen and Blande 2012), but would not explain general flowering that happen synchronously in different regions. Many studies focus on examining intreannual climatic cues (Ashton et al. 1988) point to different cues in different regions (Numata et al. 2003). The mass flowering phenomenon of dipterocarp forests makes them hard to manage. From a management perspective, unpredictable flowering patterns make natural regeneration of dipterocarps forests after harvest difficult. In additional, dipterocarps produce recalcitrant seeds that germinate within days after fruit fall occurs and the seeds cannot survive in cold-storage for more than a few weeks, adding to the logistical difficulties of reforestation with dipterocarps (Kettle et al. 2010, Kettle et al. 2011). Predicting mass flowering events could provide invaluable information to forest managers.

1.2. Deforestation of dipterocarp forests

Southeast Asia's dipterocarp forests are one of the most threatened tropical ecosystems in the world. In 2005, only 37% of total original primary forested area remained classified as forest cover in Southeast Asia, followed by Central America and Central Africa with 45%, and the Amazon basin with 56% forest cover remaining (FAO 2005). Primary forests are most threatened in Southeast Asia and Central America with 16% and 18% original forest cover remaining intact, respectively (FAO 2005). In more recent years, deforestation rates remain unsustainable with 1.2%/year of net forest loss between 1990 and 2005 in Central America, 0.7%/year net forest loss in Southeast Asia, and 0.5% per year in Central Africa (FAO 2005). As of 2005, several countries in Southeast Asia, for example the Philippines, Vietnam, Laos, and Cambodia, had less than 6% primary forest cover (FAO 2005). In these countries, land degradation, erosion, water supply concerns, and landslides have often become common problems (Sidle et al. 2006). According to another study, tropical forest ecosystems experienced a loss to gain ratio of 3.6 for the 2000-2012 period as compared to 1.6 loss to gain in temperate regions for the same period (Hansen et al. 2013). Within major geographic regions, countries with high accessibility to forests and high population densities have little primary forest left, often limited to mountainous regions.

Logging operations created roads and opened access to the valuable forest areas that are quickly converted to agriculture or other uses (Liu et al. 1993, Kummer and Turner 1994, Saastamoinen 1996). Old growth forests are important ecologically as they are high in biodiversity (Noss 1999) and form storage stocks for carbon (Luyssaert et al. 2008, Lewis et al. 2009). Increased deforestation also contributes to an increased number and magnitude of humanitarian disasters as cleared land in combination with frequent typhoons has been shown to result in massive landslides and flash floods (Vitug 1993). These disasters would not be so severe if the surrounding hills have sufficient forest cover that would have prevented massive landslides (Vitug 1993). In developing countries, such as the Philippines, the combination of resource accessibility, foreign corporations providing logging technology and a corrupt regime that pays political cronies with timber clearing licenses created conditions for large scale deforestation with little efforts for reforestation (Bautista 1990, Kummer 1992a, Kummer and Turner 1994, Broad 1995, Bankoff 2007).

1.3. Reforestation initiatives

The reforestation opportunities in Southeast Asia are typically on sites of marginal or abandoned farmlands with high level of degradation where growing of dipterocarp and other indigenous tree species can be difficult (Tolentino 2008). In many cases degraded sites are invaded by cogon grass (*Imperata cylindrica)*, quickly taking over disturbed areas and making it almost impossible for anything else to grow. Cogon grasslands are extremely prone to fire, preventing natural establishment and succession of native species (Ainuddin 2010). In such places many programs focus on planting fast growing exotic species. The exotic species are often preferred because of their high yields and tolerance to a variety of stresses, available superior germplasm for planting, and widespread research and development of propagation (Cruz 2001). However, exotic species may be prone to diseases and pests and they out-compete native species for moisture and nutrients (Hooper 2002). Further, exotic trees do not support the native fauna, which comprises many endangered species (Wingfield et al. 2001b).

Natural forest regeneration of degraded forest lands is possible but depends on ecological constraints and ecosystem resilience. Degraded forest land after excessive logging operation is able to regenerate if the area is not suitable for agriculture with low soil fertility and high pest and weed density, but containing a nearby forest to supply seed stock (Fujisaka 1991). Other programs focus on assisted natural regeneration that involve sites of medium degradation and use techniques that control fire, restrict grazing, and involving local communities in forest management. This provides an inexpensive way of allowing natural regeneration of the forest (FAO 2003). There are programs that focus on reforestation of severely degraded sites with indigenous species (Cruz 2001, Tolentino 2008). Many such projects have been unsuccessful in the past since they were using native species that are economically important but turn out to be late-successional and unable to grow on denuded sites. Late-successional trees need certain degree of shade for survival (Tolentino 2008). Projects that experiment with establishing successional analogs, where pioneer species are first planted and then mid and late-successional species are introduced to re-create natural forest regeneration have had success with reestablishing of native species (Goltenboth 2004, Langenberger 2005). Other studies focus on provenance trails to select species that can survive on denuded sites (Butterfield 1996, Hooper 2002). More studies are needed on site specific indigenous tree species matching to find early-successional tree species that can create initial canopy cover for the economically important late-successional species (Tolentino 2008).

1.4. Forest dependent communities

Deforestation has had a drastic negative impact on the rural poor. In many Southeast Asian countries, the welfare of rural communities is dependent on natural resources (McElwee 2008). For example, in the Philippines more than 30% of the population is living below poverty rates, most of them in rural places where they depend on natural resources to enhance their livelihoods (IFAD 2011) as exploitation of natural resources is seen as the only way for survival. When the resources are abundant they bring about increase in economic and human development at first, but with further deforestation, the benefits diminish and the municipalities which have exhausted their resources are worse off than the ones that are more conservative (Rodrigues 2009). Additionally, there is often a conflict between the needs of the rural population and national government agenda of profiting from logging (McAllister 2007). This situation is further worsened by weak formal institutions and the inability to enforce sustainable practices. As a result, this leads to unfair distribution of profits, lack of consideration of long-term ecological impacts of logging, or the sustainability of the industry (McAllister 2007). Studies show that areas where forests have been cleared are worse off in terms of human development (measured by literacy, human development index, standard of living, and life expectancy) (Okello et al. 2009).

In the recent years many developing countries focus on community based forestry programs that involve local people in managing the forests for commercial and non-commercial purposes including subsistence timber production, wild life conservation, and cultural significant of forest preservation (Johnson 1999). However, the governments retain most of the control of natural resources in community based forest management system (Gauld 2000, Dressler et al. 2006). Many of the benefits are known to go to community elites as participation is usually poor and decisions are made by the select few as the designed programs do not take into account historic or ancestral ties leading to weak community involvement (Okello et al. 2009). Studies show that seemingly homogenous regions contain diversity among the villages and hence different reactions to the projects which requires flexibility in the programs and in-depth analysis of socioeconomic factors to draw appropriate incentives (Walters et al. 1999). Better understanding of socioeconomic conditions of rural communities can lead to better management of deforested and protected areas and result in designing reforestation activities that are inclusive, introduce appropriate livelihood options and designing equitable land tenure (Okello et al. 2009).

1.5. Research objectives and thesis structure

The overall objective of this PhD research is to support and improve reforestation initiatives in dipterocarp forests through ecological and local knowledge by addressing major impediments to successful forest restoration in Southeast Asia. The first issue is the biology of dipterocarp reproduction that makes it hard to manage regeneration. It is logistically difficult to mobilize resources to collect short lived dipterocarp seed on short notice without knowing in advance when flowering will occur. Second, knowledge is lacking which native species in Southeast Asia's dipterocarp forests can be used in reforestation. Community based forest management programs typically use readily available exotic species that may not be desired by the local communities. And finally, many communities rely on forests to supplement their livelihoods but these resources become increasingly limited because of deforestation. There is a need to protect the remaining dipterocarp forests for their ecological functions without restricting community use of natural resources and increasing poverty rates. This research thesis aims to address these issues by providing guidelines for reforestation and restoration of degraded landscapes with the hope to contribute to sustainable forest management by increasing ecological and socioeconomic benefits to local communities.

In my **first research chapter**, I examine the possibility of predicting mass flowering events using climate cues and accumulated resources. Several studies were conducted to show that irregular flowering of dipterocarps is related to interannual changes in climate caused by El Niño Southern Oscillation (Ashton et al. 1988, Curran et al. 1999, Wich and van Schaik 2000). However, other studies did not find consensus on the particular climatic trigger, citing drop in temperature (Ashton et al. 1988) or drop in precipitation (Sakai et al. 2006, Brearley et al. 2007), or showing effects of El Niño in some areas or years and not others (Numata et al. 2003). In this study I investigate several environmental triggers using data from 11 regions and sites in Southeast Asia to see if any patterns become apparent. Further, to build on studies that describe dipterocarps seeds as energy intensive, I investigate the effect of resource availability on flowering induction. Finally, I use resource accumulation models to predict flowering events. This chapter answers three specific questions:

1. What are the environmental factors that control mass flowering in dipterocarp forests?
2. Can resource accumulation models successfully predict flowering events in dipterocarp forests?

In **my second research chapter,** I identify local tree species that can be used for reforestation in the Philippines. I use ecological characteristics to identify trees that are early-successional and are likely to survive open planting environments. I use information on species frequency, recruitment, and growth increments collected in three successional stands to infer ecological characteristics of native tree species. Secondly, I identify species that have socioeconomic importance to forest communities with the help of a questionnaire. I investigate the relationship between socioeconomic rank and ecological suitability of native tree species to propose species that are suitable for community forestry. The objectives of this chapter are to answer two questions:

3. Can easy to collect tree measurements from ecological plots be used to identify native species that are ecologically suitable for reforestation on open sites in order to bypass lengthy species test plantations?
4. Can we identify species that have both high ecological suitability for reforestation and high value for forest dependent communities?

In my **third research chapter,** I investigate the impact of land use planning, protected area planning, and actual land cover on socioeconomic conditions of communities. More specifically, I examine if land use planning achieved its purpose of designating areas for use and for protection by comparing planned land use with actual land cover based on satellite data. Further, I use government census data to infer socioeconomic conditions in areas of different land cover. Finally, I examine socioeconomic conditions of rural communities close to a protected area and further away areas located in agricultural settings in order to understand the impact that protected areas have on socioeconomic status. This chapter aims to answer the two following questions:

5. What effect do protected areas have on socioeconomic status of communities that live within or nearby areas with restrictions on resource extraction?

All research chapters, even though they cover different methodologies and different disciplines have a common aim: to aid reforestation activities in the tropics and contribute scientific knowledge for economic, social, and ecological benefits. The common ideas of these three research chapters are summarized in the conclusions at the end of the thesis.

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Chapter 2 - Climatic drivers of dipterocarp mass flowering in Southeast Asia

2.1. Summary

Dipterocarps, a dominant family of trees in Southeast Asian tropical forests, are remarkable in that they exhibit supra-annual mass flowering events. The flowering patterns are related to the El Niño Southern Oscillation, but the trigger that precipitates mass flowering is unknown. Here, we propose resource accumulation as alternative to a trigger mechanism, and we test the alternate hypotheses in a meta-analysis with published flowering records. Using a variety of candidate predictor variables (precipitation, cloud cover, minimum temperature and El Niño indices) we could not find a plausible environmental trigger (median AUCs around 0.55 indicating near random predictions), while the best resource accumulation model had a median AUC of 0.70, which could be improved to 0.75 when the date of previous flowering was included in the model. We further show that simple resource accumulation by individual trees can cause inter- and intraspecific flowering synchronization leading to community-wide flowering events.

2.2. Introduction

The family *Dipterocarpaceae* consists of 16 genera and nearly 600 species, most of which are found in Southeast Asia where they are the leading species group on 85% of the forested land base (Appanah et al. 1998). They are also commercially important, representing a quarter of global consumption of tropical timbers (Kettle 2010). Dipterocarps are remarkable in that they exhibit supra-annual mass flowering events, which occur in irregular intervals of two to ten years, normally involving several dipterocarp species (Ashton et al. 1988). Such mass flowering events have further been documented to involve numerous species unrelated to dipterocarps (Yap and Chan 1990, Hamann 2004, Sakai et al. 2006, Brearley et al. 2007, Cannon et al. 2007), a phenomenon that may involve up to three-quarters of all forest species, and which has been termed general flowering (Appanah 1985). Beside mass flowering events, it has been documented that a small proportion of dipterocarp trees flower unsynchronized almost every year, and notably small flowering events have often been observed to precede a mass flowering event in the following year (Curran et al. 1999, Hamann 2004, Sakai et al. 2006, Brearley et al. 2007).

Mast fruiting is thought to be a predator satiation strategy, which is well supported by empirical data showing that seedling survival rates increase with increased intensity of a masting, and that during low intensity flowering, all viable seeds are lost to predation (Curran and Leighton 2000, Sun et al. 2007). However, it has been proposed that several rather than a single factor have shaped the evolution of mass flowering including pollination success and outcrossing rates (Sakai 2002). This is empirically supported by several studies that observed high fruit abortion rates because of inbreeding in years of minimal flowering (Ghazoul et al. 1998, Maycock et al. 2005). Other studies found reduced survival of seedlings and recruitment of saplings that originate during non-masting years (Naito et al. 2008).

Another problem, distinct from the question of evolutionary causes, is what trigger mechanism precipitates mass flowering or general flowering events. Ashton et al. (1988) hypothesized that interannual climate anomalies would be a probable candidate for a trigger mechanism, and they were the first to suggest that a drop in minimum night-time temperature below 20°C for several days around six to nine weeks prior to mass flowering may be a plausible trigger based on their data from the Malaysian peninsula. Other studies at different sites have supported the relationship between flowering and drop in night-time temperature (Yasuda et al. 1999, Numata et al. 2003). However, subsequent studies found no associations with low-temperature events, instead pinpointing lack of rainfall as a plausible trigger with typical lead times between the trigger occurrence and flowering of several weeks to a few months (Sakai et al. 2006, Brearley et al. 2007).

Both low minimum temperatures and lack of rainfall can be linked to the irregular El Niño Southern Oscillation (ENSO), which brings dry air masses into Malesia, creating drought conditions and cloudless skies causing night-time temperatures to drop. The ENSO index in turn has been shown to correlate reasonably well with mass flowering events (Ashton et al. 1988, Appanah 1993, Curran et al. 1999, Wich and van Schaik 2000, Cannon et al. 2007). The positive association of mass flowering with El Niño events is not consistent throughout the region, however. Yasuda et al. (1999) and Numata et al. (2003) observed mass flowering events during the opposite La Niña anomalies, while others found no association (Numata et al. 2003, Hamann 2004). The strongest associations with El Niño are observed in eastern Malesia weakening westward (Wich and van Schaik 2000). In summary, there is significant spatial, temporal, inter- and intraspecific variability in dipterocarp mass flowering, in parallel to occasional general flowering events that involve many species over large geographic areas (Ashton et al. 1988, Appanah 1993).

We find it difficult to reconcile this mix of spatial and temporal variability, combined with occasional highly synchronized events across multiple species, with an evolved environmental trigger mechanism. The lack of flowering correlation with topography and minor flowering events preceding a large mass flowering in the following year point to a different mechanism that may precipitate flowering. We hypothesize that a resource tracking mechanism may play an important role in precipitating mass flowering events. Species-specific thresholds, where enough resources are accumulated for mass flowering, could explain different flowering intervals in dipterocarps, while community-wide events could be triggered by unusually high resource availability (e.g., during El Niño events). Primary productivity and mobilization of photosynthate in the production of large, energy rich seeds requires resource accumulation for several years (Ashton 1988, Ichie 2005a). These resources were found to be more abundant during drought conditions and higher irradiance (Ashton et al. 1988, Wich and van Schaik 2000, Newbery et al. 2006).

Understanding and predicting mass flowering in Southeast Asian dipterocarps has important practical applications. Dipterocarps produce recalcitrant short-lived seeds that pose difficult logistical challenges for reforestation and conservation of dipterocarp resources (Appanah et al. 1998, Kettle 2010). The most recent major general flowering event in Borneo, the first in 12 years, went unutilized for lack of time to mobilize resources for seed collection (Kettle et al. 2010). Better predictive models of mass flowering events are therefore also needed to improve forest resource management. Here, we re-examine potential environmental cues of mass flowering in different regions of Southeast Asia and contrast the classical trigger model with a straightforward resource tracking model of mass-flowering based on availability of resources (measured by climate variable proxies). We evaluate which model has better predictive capabilities, and further investigate whether a resource tracking model can in principle explain intra- and interspecies synchronization of flowering.

2.3. Materials and methods

2.3.1. Phenology data

We compiled flowering data from published literature at eleven study sites in Southeast Asia (Fig. 2-1, Table 2-1). Data sources include observations from five long-term ecological plots and flowering records from six general geographic areas in Peninsular Malaysia, Borneo and northern Sumatra. Few of these records were available for individual species, and we therefore perform our meta-analysis at the community level, focusing on dipterocarps only. We included studies that involve at least ten years of data and at least three mass flowering events. The average length of the eleven time series we compiled was 29 years, excluding flowering data prior 1950 to ensure high quality of climate data, which relies on a sufficiently dense network of weather stations.

The flowering data were reported in different units, including seed export in tons per year for various *Shorea* species used for food products, cosmetics, and lubricants (Blicher-Mathiesen 1994). Other regional data is reported as minor, major, or no flowering for each year. Data from long-term ecological plots were recorded either as percent of dipterocarp trees or species flowering each year. For our analysis, we converted all data to a binary response variable, focusing on major flowering events (top quartile when measured quantitatively). When data was reported as major/minor, we accepted those classifications. The database with original records is available in Appendix A.

2.3.2. Climate data

We use two sources of climate data for analysis, daily weather station data and monthly interpolated climate grids. Monthly variables for study sites and regions were extracted from the CRU 2.1 database (Mitchell 2004). We used precipitation (mm), average cloud cover (%), vapor pressure (hPa), and minimum temperature (°C) as putative predictors. Daily minimum night-time temperature was directly obtained from a global weather station database (NCDC 2013). For the long-term ecological plots, the following weather stations were used: Gunung Palung: USAF ID 966150, 965810 (the second station used to fill missing data), Barito Ulu: USAF ID 966550, Pasoh: USAF ID 486470, Lambir Hills: USAF ID 964490, and Ketambe: USAF ID 960350. For regional data, we used averages of all stations located within the region and that covered at least 60% of the data period (number of stations in parenthesis): East Borneo (6), West Borneo (7), East Peninsular Malaysia (6), West Peninsular Malaysia (9), Kalimantan (5), Sarawak (6).

Beside climate variables as predictors for flowering events, we also use El Niño indices. These indices may be measured with greater accuracy than the resulting climate patterns, and/or they may stand as proxies for climate variables that we do not measure. We use two indices, the Southern Oscillation Index (SOI), and the El Niño Region 3 anomaly index (NINO3) (Climate Prediction Center Internet Team 2012). SOI is computed using a standardized monthly sea level air pressure difference between Tahiti and Darwin (Bureau of Meteorology 2013). Values below –8 signify an El Niño phase and values above +8 signify a La Niña phase. The NINO3 index is calculated using the average monthly sea surface temperatures of the eastern equatorial Pacific region enclosed by the latitudes 5°N to 5°S and the longitudes 150°W to 90°W (Trenberth 1997). The periods where the anomalies for the running average over a period five months exceed +0.5°C are defined as an El Niño phase and an anomaly below –0.5°C is interpreted as La Niña phase (Trenberth 1997). We should note that all putative predictor variables for the trigger or resource tracking model have moderate to high collinearity and should be considered proxies for unknown climate factors that serve as biological triggers or resources.

2.3.3. Environmental trigger model

We assume a logistic rather than a linear relationship between the environmental trigger and the probability of a flowering event. The s-shaped relationship implies that a low trigger value is very unlikely to precipitate a flowering event, followed by a short transition of intermediate values to a high probability of a flowering at high trigger values (or vice versa for some candidate variables). Thus, our environmental trigger model conforms to the function: $y = 1/(1+e^{-(a+bx)})$, where y is the probability of flowering, x is the environmental predictor variable, the parameter a represents the intercept with y, and the parameter b represents the steepness of the transition from low to high probabilities.

Because the flowering data is recorded annually, and climate data is given as monthly or daily records, some temporal alignments are required. For seed data records, we assume that flowering occurred three months prior (based on observations by Hamann 2004, Sakai et al. 2006, Brearley et al. 2007). Flowering was assumed to occur in one of two flowering periods with peak times around April or May in northern and eastern sites and in September in southern Borneo sites (see Table 2-1 for flowering times), reflecting the periods where flowering was most frequent. To capture the cue of flowering, we analyze climate data for the nine months preceding the peak of flowering period at each site in each year. For example, at the West Borneo region, where flowering peaks in September, the environmental cue was assumed to occur between January and September. This ensures that even for early flowering species relative to the peak flowering period, we would capture a climate cue up to six months prior to the flowering event, and a flowering cue is not likely to occur more than six months prior to flowering induction (Ashton et al. 1988, Brearley et al. 2007). Daily data were screened for extreme minimum temperature events, and monthly data were screened for minimums in monthly precipitation, cloud cover, and average minimum temperature. As an alternate predictor, means rather than minimums of monthly data in the period leading up to the flowering event were also tested.

The quality of the predictive models were evaluated using the Area Under the receiver operating characteristic Curve (AUC) using the ROCR package (Sing et al. 2005) for the R programming environment (R Core Team 2014). Receiver operating characteristics evaluate true positives (where flowering predictions correspond to observed mass flowering events) versus false positives (Fawcett 2006). The true positive rate, or model sensitivity, is plotted over the false positive rate, corresponding to model specificity, with any model-generated probability between 0 and 1 tested as potential threshold to predict a flowering event. The AUC value is subsequently obtained by integrating the area under the curve. An AUC value of 1 implies a perfect model fit, an AUC value of 0.5 is a random predictor, and an AUC below 0.5 implies a model with more false positives than expected by random chance.

2.3.4. Resource accumulation model

Candidate predictors for the resource accumulation model were climate proxies of drought conditions and high irradiance: cloud cover, precipitation, NINO3 index values, and SOI index values (representing El Niño periods). We converted the variables to a standardized unit, where the average resource availability was represented by a value of one, with a standard deviation of one. Precipitation, SOI, and cloud cover negatively correlate with high resource availability (drought, El Niño event, and clear skies) and therefore the inverse of these values was used. The NINO3 index was not inverted as high values represent El Niño event. Monthly resources are added up to a preset flowering threshold, which precipitates a flowering event and resets resources to zero. The model was implemented in the R programming environment with the following algorithm:

```
for (i in 1:nrow(x)) { r[i+1] ← ifelse( r[i]<ft, (x[i+1]+r[i]), (x[i+1]) ) }
```

where, *x* is a vector of monthly resource availability as explained above, *r* is a vector of the resource accumulated over time, *ft* is the flowering threshold, and *i* represents a monthly counter.

We also developed a second model that makes predictions of the next mass flowering event including knowledge of when the previous mass flowering event occurred. This version differed from the algorithm above as it resets the resource to zero when an actual mass flowering event was observed, regardless of whether the modeled flowering threshold was reached:

```
for (i in 1:nrow(x)) {  r[i+1]←ifelse(f[i]==0, (r[i+1]+r[i]), (x[i+1]) ) }
```

where *f* is a vector of observed mass flowering events placed in the months of peak flowering according to Table 2-1.

For model evaluation, we are assuming the same logistic relationship between the value of the accumulated resource and the probability of flowering as was used in the trigger model above. The reasoning is similar: we assume that a minimal level of the accumulated resources implies a very low flowering probability followed by a rapid s-shaped transition to high flowering probabilities once a certain available resource threshold is exceeded.

2.3.5. Flowering synchronization model

Building on the resource accumulation model above, additional parameters could be varied, and we tested the sensitivity of these factors in an expanded model. This model is meant to explore if various qualitative observations (synchronization of trees of one species, synchronization of several species, general flowering, or small flowering events prior to large flowering events) could in principle be explained by a resource tracking model. This model has the following additional parameters: a randomly varying resource baseline (*rb*), an irregular resource anomaly (*a*) with an interval (*ai*) and an anomaly size (*as*) that is expressed as a multiple of the resource baseline. Further, the model is expanded to simulate a multi-species community with a user defined number of species (*ns*) and number of trees per species (*nt*), and species specific flowering thresholds (*ft*) that now is a vector with different thresholds for different species (*ftv*). The model remains exactly as explained above, but multiple trees and multiple species are now kept track of in matrices rather than a single time series vector. R scripts for the single and multi-species models can be found in the online supporting information (Appendix B and Appendix C).

2.4. Results

2.4.1. Regional climatology and flowering

To visualize regional variation in the effects of El Niño on climate drivers, we plot climate diagrams for all regions and sites for which we have flowering data (Fig. 2-1). El Niño phases reverse the Pacific trade winds creating a failure in the southeast summer monsoon in Southeast Asia (Fan et al. 2009), while La Niña intensifies the monsoon. Those effects vary regionally, and we can distinguish different influences on the northeast and southwest-facing study sites. In the regions facing the south Pacific (Fig. 2-1, WB, EB ,Kal), the divergence between El Niño and La Niña is largest during the second half of the year, with warmer temperatures and less precipitation during the El Niño phase, because of hot air masses being driven into the region during the summer monsoon. In contrast, the north and east-facing regions (Fig. 2-1, Keta, WPM, EPM, Sar) see the strongest influences of El Niño phases during the winter monsoon, when weakened trade winds originating in the South China Sea and the Bay of Bengal, cause reduced precipitation from January to April.

These differences in regional effects of El Niño are mirrored in the time of the year when dipterocarps flower in different regions (Table 2-1). In the southwest-facing regions and study sites (Table 2-1, WB, Kal, BU and GPNP), flowering is clustered around September, which coincides with the time of the year of El Niño driven low precipitation anomalies. In the east and north-facing regions and study sites, flowering occurs around April or May (Table 2-1, Keta, WPM, EPM, Pas, Sar, LH), which coincides with the time of the year when the El Niño effects are strongest in those regions as well. In summary, regional flowering times are synchronized with where and when El Niño effects are regionally strongest, implying that El Niño related climate phenomena may be a cue for flowering induction of dipterocarps in Southeast Asia.

While the seasonal association of flowering with El Niño is readily apparent, the association of interannual flowering events with El Niño is less obvious when visualizing the data for reproductive events (Fig. 2-2). In the monitoring site on the first line (Pas), flowering does not correspond to El Niño periods (in red), where all but one of the mass flowering events occurred during the normal or the La Niña periods. In fact, mass flowering throughout the northern and eastern monitoring sites and regions are very weakly associated with El Niño periods. Nevertheless, major flowering events observed across the entire study region occurred in

1957/58, 1968, 1976, 1981-82, 1997/98, with three out of five large events falling on or after El Niño years. Large flowering events across Southeast Asia happen every five to twelve years, but additional large and small regional events that are not synchronized with other locations are very common.

2.4.2. Trigger model versus tracking model

A statistical evaluation of the trigger and the resource models is summarized in Table 2-2. AUC values are based on the average probability of flowering for the six to nine months preceding flowering range reported in Table 2-1. The minimum temperature and NINO3 predictor variables for the trigger model yielded only slightly better predictions than expected by random chance (0.53-0.57). The alternate El Niño index (SOI) yielded the same AUC value as the NINO3 index. Further, the best predictor variable varied for different regions of the study area. For the Peninsular Malaysian (Pas, EPM, WPM) a drop in monthly minimum temperature yielded respectable AUC values between 0.7 and 0.8. For Borneo (BU, Sar, Kal, WB), and Sumatra (Keta) the El Niño index yields high AUC values between 0.6 and 0.8, while monthly minimum temperature regularly yields AUC values below 0.5.

The predictive ability of the resource tracking model is more accurate than the trigger model overall, with median AUC values of 0.6 for precipitation and 0.7 for the NINO3 index. Cloud cover AUCs were similar to precipitation as a predictor for resources. Sites and regions representing Borneo and Sumatra (Kal, EB, WB, Sar, Keta) yield the highest AUC values for the El Niño index as predictor. The Peninsular Malaysian sites (Pas, EPM, WPM) show better results for precipitation as a proxy for resources. When the date of previous flowering was included in the model, which would be relevant for practical applications, the model accuracy could be improved to 0.75.

In Fig. 2-3, we visualize selected model predictions to illustrate what the AUC statistics are based on. For the resource tracking model, we plot monthly probabilities of flowering based on a resource accumulation mechanism that uses the NINO3 index as a proxy for resource availability. For comparison, average monthly probabilities of the daily minimum temperature trigger model is indicated by horizontal bars (only one value is calculated based on the lowest daily temperature value in the preceding 6-9 month period). For the resource tracking model, probabilities increase as the resource is accumulated, and the model resets itself when a maximum resource threshold is reached, which is defined as the total resource accumulated over the study period divided by the number of intervals between flowering events (we confirmed that an equal interval flowering model has no predictive value, with an AUC=0.5 (± 0.06) across all sites and variables).

2.4.3. Intra- and interspecies synchronization

Although there is better statistical support for a resource tracking model than for an environmental trigger model, a resource tracking model does not obviously explain intraspecific synchronization of individual trees to flower in the same year. Each individual may be flowering and resetting the resource counter in its individual cycle. To explore the capacity of resource tracking mechanism to explain synchronization of mass flowering and general flowering across many species, we extend the model with additional parameters to simulate a group of trees in a single species (Appendix B) and further to include multiple species that may have different values for parameters for flowering thresholds, etc. (Appendix C).

The model as specified in Appendix B starts with 12 individual trees of a species that are initially completely unsynchronized, set in the model to the worst case of flowering sequentially in 12 consecutive years. The program first generates a randomly varying resource with a certain baseline value (here, set to *rb*=20) and a resource anomaly at most twice as large as the resource baseline (*as*=40) at irregular intervals of 1 to 12 years (*ai*=12). The results of the single species with a flowering threshold less than twice the maximum anomaly (*ft*=70) is shown in Fig. 2-4a. In this model, individual trees of a species quickly synchronize to flower in the same year because when resource anomaly occurs, all trees with resource levels close to the flowering threshold are triggered, and are subsequently on the same resource accumulation cycle. If the anomaly size is reduced, or the flowering threshold is raised relative to the size of the anomaly, small population flower more frequently out of synchronization (Fig. 2-4b, f*t*=80).

In the multi-species implementation of the same model (Appendix C), we simulate five species (*ns*=5), three with lower and two with higher flowering thresholds (specified with the user defined vector *ftv*=70, 71, 72,120,122), while other parameters remain the same (Fig 2-4c). As before, we start with all trees of all species being completely unsynchronized by setting their flowering times to consecutive years. It takes about 20 to 30 years for populations to self-synchronize and generate mass flowering events (species with similar thresholds), and occasional general flowering events (all species). The model output from iteration 40 to 100, after the initial self-synchronization has taken place (Fig 2-4c), resembles observed flowering patterns for dipterocarps in Southeast Asia (Fig. 2-2).

2.5. Discussion

In this meta-analysis we found slightly more statistical support for a resource tracking hypothesis than for the trigger model of mass flowering in dipterocarps. It should be noted, however that the two hypotheses are not mutually exclusive and that, for example, a trigger threshold could be modulated by a resource status. Other observations, however, point to a resource tracking mechanism being the primary driver of dipterocarp phenology in Southeast Asia. First, the widely documented occurrence of minor flowering events preceding mass flowering in dipterocarps (Curran et al. 1999, Hamann 2004, Sakai et al. 2006, Brearley et al. 2007) points to the resource tracking hypothesis, and is likely caused by some trees reaching the flowering threshold somewhat earlier than the main population. Second, there is the phenomenon of general flowering events that involves many species of dipterocarps as well as unrelated tree species (Yap and Chan 1990, Hamann 2004, Sakai et al. 2006, Brearley et al. 2007, Cannon et al. 2007). It seems unlikely that a large range of species would independently evolve a mass flowering strategy in response to the same proximate environmental cue, while a broad community response to resource availability appears more parsimonious.

Third, a key observation that emerged in our analysis is that seasonal flowering times in dipterocarps are determined by how the winter and summer monsoons mediate El Niño phases. Notably, the climate seasonality is similar throughout the study area with minimal precipitation in July and the highest temperatures around May when the monsoon directions reverse, and when the sun reaches the zenith for the first time during the year. Yet flowering times of dipterocarps do not show regional correspondence to this seasonality. Rather, they are associated with the time of the year that regionally shows the largest anomalies during El Niño phases. We interpret this observation as favoring the resource accumulation over the trigger hypothesis: a large anomaly relative to the flowering threshold would facilitate intra- and interspecies synchronization as shown in the model simulations (Fig 2-4).

Considering raising temperatures and increased droughts due to climate change (Dai 2011) dipterocarp reproduction may also be affected. Studies found mass flowering species that are synchronized during elevated temperatures to flower annually under changing climate conditions

(McKone et al. 1998). Under climate change, extreme weather events are likely to increase (Dai 2011) and frequency and intensity of El Nino Southern Oscilation (Timmermann et al. 1999). With these changes both the trigger and the resources involved in flowering will become more frequent and abundant (Butt et al. 2015), regardless of whether masting is cue or resource dependent.

A resource tracking model does offer a plausible explanation for an evolutionary path toward mass flowering and general flowering phenomena. Seasonal flowering is an evolutionary stable strategy, not easily invaded by an alternative (mutant) mass flowering strategy without the initial advantage of predator satiation. However, resource availability for fruit and seed production would have an immediate evolutionary advantage (documented by Isagi et al. 1997), and mutant populations would not be excluded from the general reproductive pool, because large anomalies would still re-synchronize all individuals of a species despite different flowering thresholds (as shown in Fig 2-4c).

In this paper, we tried to make the case for an alternative hypothesis that could explain the evolution of mass flowering in dipterocarps, the general flowering phenomenon, and within-species synchronization of individual trees. We should note, however, that our resource tracking model did not yield levels of predictive accuracy that would be required for local management. This is not unexpected, because our data compiled from the published literature includes many species and different record types of flowering events. While resource tracking appears to be an important component in controlling dipterocarp phenology, better predictive models could potentially be developed by fitting more complex resource-modulated trigger models to species-specific plot data. Building better models with accurate local data would have important applications in management and conservation of dipterocarp species (Kettle et al. 2011). If the probability of a flowering event could be estimated with reasonably high precision (AUC>0.8), forestry operations and seed collections for conservation purposes could be planned accordingly. In a recent letter to Science, (Kettle et al. 2010) pointed out that resource managers and conservation agency need to mobilize resources and personnel for seed collection.

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Table 2-1. Dipterocarp flowering and fruiting. This data are for the long-term ecological monitoring plots in Peninsular Malaysia (Pas), central Kalimantan (BU), West Kalimantan (GPNP), Sarawak (LH), North Sumatra (Keta) and regions (Kal, Sar, WPM, EPM, WB, EB) in Southeast Asia describing time period, data type, primary flowering times (although flowering may occur in April and September) and references of data origin. For sites with seed data and September flowering times (GPNP, Kal, WB), flowering was assumed to occur a year prior.

Code	Site, Region	Period	Data Type	Flowering Time	Reference
BU	Barito Ulu RA, Kalimantan	1990-1999	% Trees flowering	Sep (Jul-Dec)	(Brearley *et al*. 2007)
EB	East Borneo	1950-1983	Mass flowering (0,1,2)	May (Mar-Jul)	(Ashton *et al*. 1988)
EPM	Eastern Peninsular Malaysia	1950-1983	Mass flowering (0,1,2)	Apr (Feb-Jun)	(Ashton *et al*. 1988)
GPNP	Gunung P. NP, Kalimantan	1987-1999	Seed density (seeds/m^2)	Sep (Jul-Dec)	(Curran *et al*. 1999)
Kal	Kalimantan	1903-1999	Seed export (10^6 kg/year)	Sep (Jul-Dec)	(Blicher-Mathiesen 1994, Curran *et al*. 1999)
Keta	Ketambe Site, Sumatra	1971-1998	Mast fruiting (0,2)	Apr (Feb-May)	(Wich & van Schaik 2000)
LH	Lambir Hills NP, Sarawak	1993-2003	% Trees flowering	May (Feb-Aug)	(Sakai *et al*. 2006)
Pas	Pasoh FR Area, Malaysia	1973-2002	% Trees flowering	May (Feb-Aug)	(Yap & Chan 1990, Numata *et al.* 2003)
Sar	Sarawak	1903-1990	Seed export (10^6 kg/year)	May (Mar-Jul)	(Blicher-Mathiesen 1994)
WB	West Borneo	1950-1987	Mast fruiting (0,1,2)	Sep (Jul-Dec)	(Ashton *et al*. 1988, Wich & van Schaik 2000)
WPM	Western Peninsular Malaysia	1950-1987	Mass flowering (0,1,2)	Apr (Feb-Jun)	(Ashton *et al*. 1988, Wich & van Schaik 2000)

Table 2-2. AUC values for the trigger model and the resource tracking model of mass flowering. Predictor variables are the coldest monthly minimum temperature (Mmin), coldest daily minimum temperature (Dmin), and El Niño index (NINO3) and precipitation (Prec). The model version with reset (R) of the resource to zero after an observed flowering event indicates the potential predictive accuracy for management applications. Refer to Table 2-1 for the explanation of sites and regions.

	Trigger Model		Resource Tracking Model				
Code	Mmin	Dmin	NINO3	Prec	NINO3	Prec^{R}	NINO3^{R}
LH	0.33	0.55	0.57	-	-	-	-
Sar	0.34	0.52	0.66	0.54	0.73	0.79	0.80
WB	0.40	0.70	0.65	0.48	0.63	0.71	0.72
EB	0.51	0.53	0.45	0.55	0.78	0.65	0.73
Kal	0.52	0.55	0.59	0.56	0.73	0.68	0.68
BU	0.56	0.31	0.69	-	-	-	-
Keta	0.56	0.53	0.81	0.61	0.73	0.83	0.87
GPNP	0.63	0.37	0.47	-	-	-	-
Pas	0.70	0.53	0.27	0.76	0.67	0.86	0.87
WPM	0.74	0.43	0.46	0.61	0.50	0.72	0.76
EPM	0.79	0.67	0.31	0.64	0.61	0.70	0.65
Median	0.56	0.53	0.57	0.61	0.67	0.72	0.76

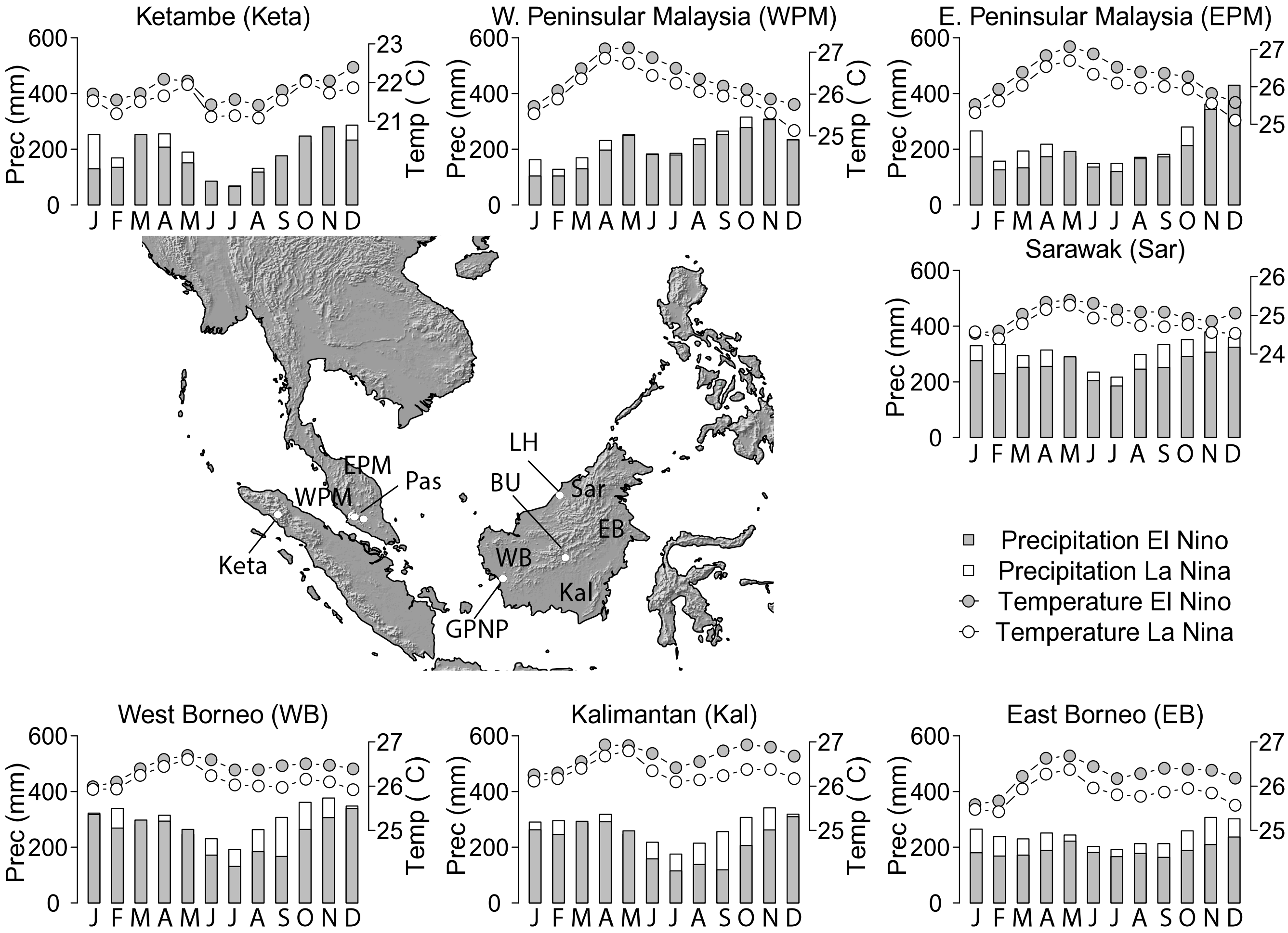

Figure 2-1. Effects of El Niño and La Niña on regional climate in Southeast Asia. The map shows long-term ecological monitoring plots and regions corresponding to data sources. Climate diagrams are for the 1950-2002 period with El Niño years in 1953, 1965, 1969, 1972, 1977, 1982-83, 1987, 1991-1994, 1997, 2002 and La Niña years in1950, 1955-56, 1962, 1964, 1971, 1973-75, 1988-89, 1996, 1999, 2000.

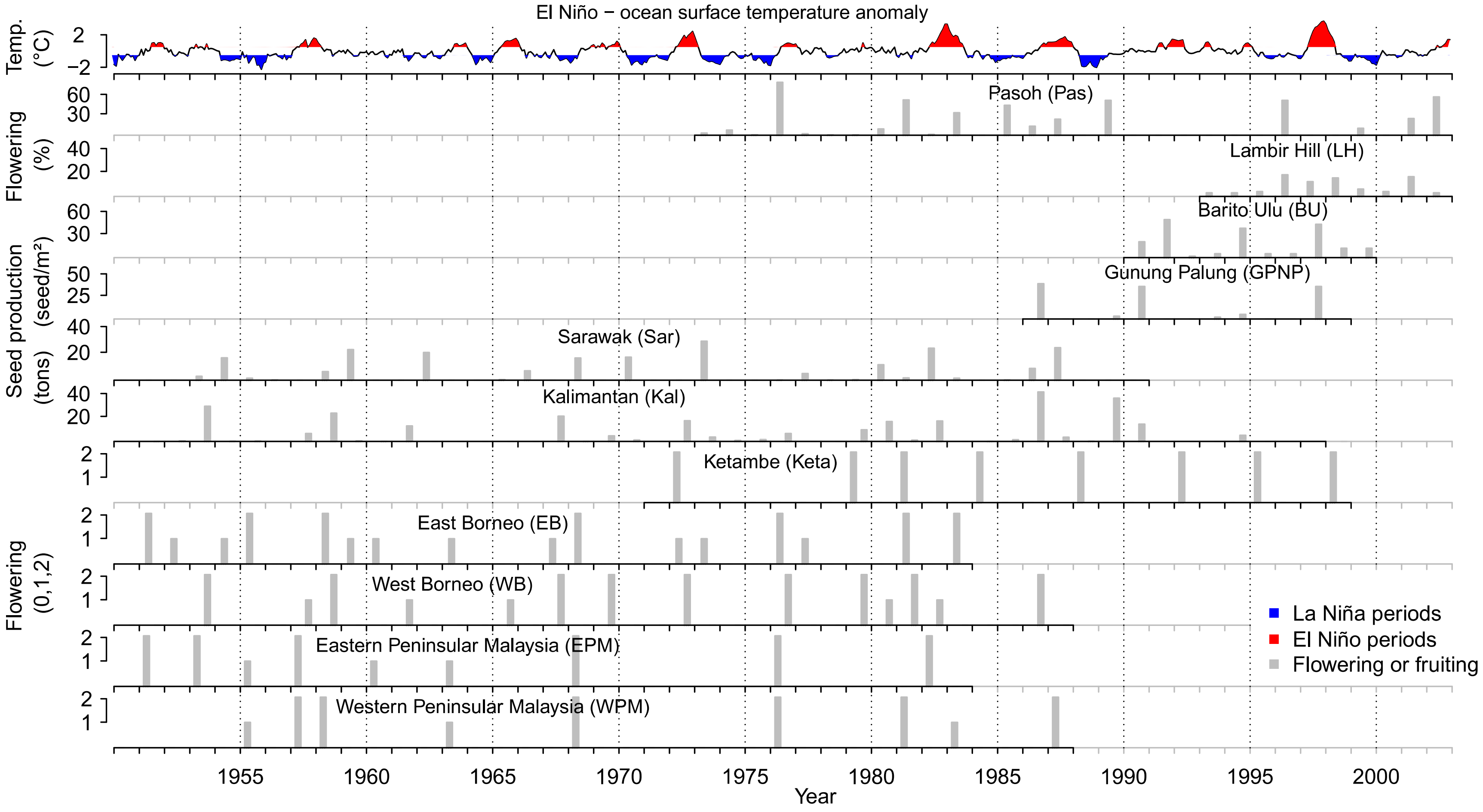

Figure 2-2. El Niño Southern Oscillation and dipterocarp flowering events. The raw data and source references are provided in Appendix A.

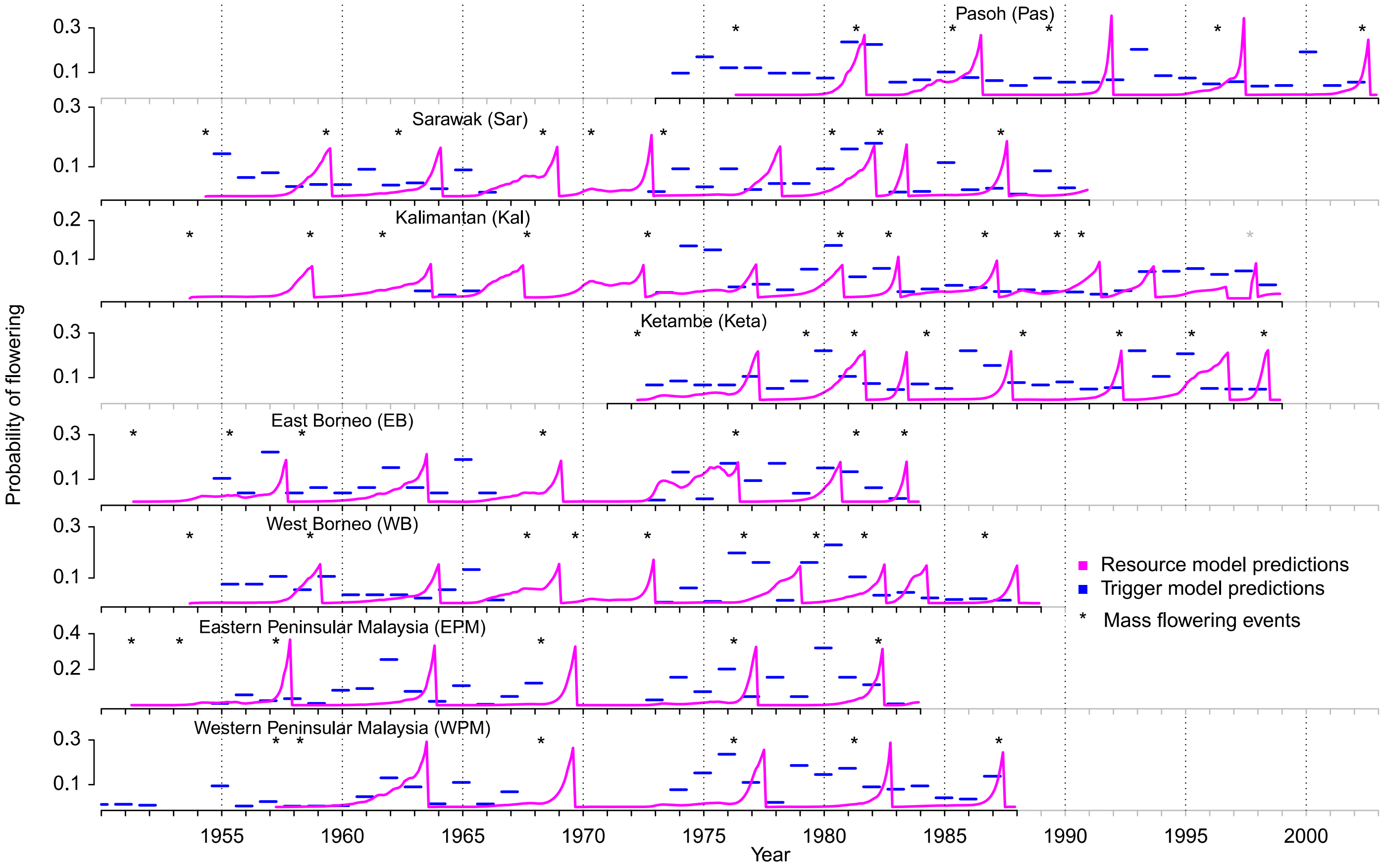

Figure 2-3. Trigger and resource tracking model predictions of dipterocarp mass flowering. For illustrating the resource tracking model, we plot monthly probabilities of flowering based on a resource accumulation mechanism that uses the NINO3 index as a proxy for resource availability. For illustrating the trigger model, we plot yearly probabilities based on daily minimum temperature preceding the flowering event (only one value is calculated based on the lowest daily temperature value in the preceding 6-9 month period).

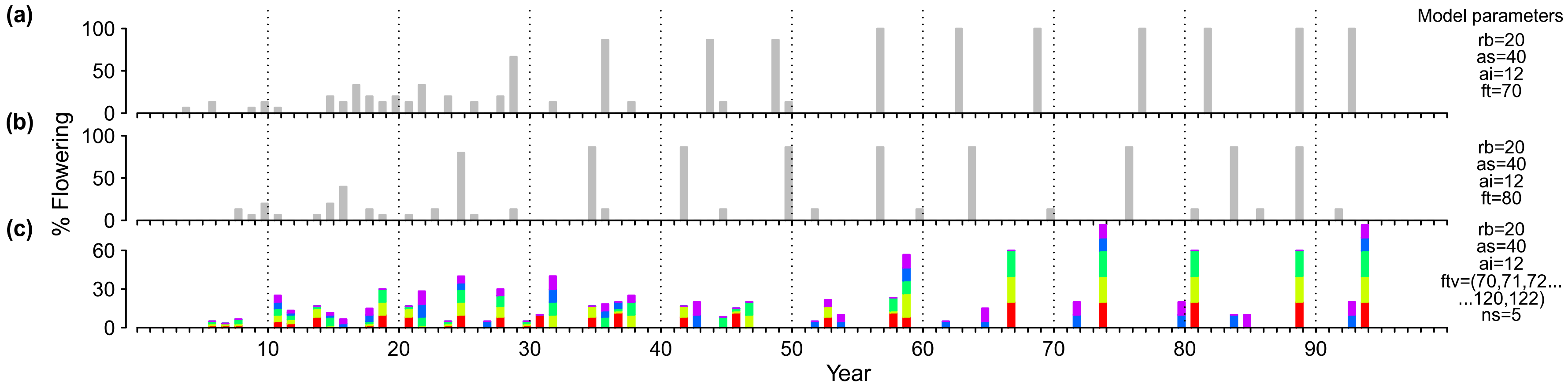

Figure 2-4. Resource tracking as explanation for intra- and interspecific synchronization of flowering. Synchronization of individuals of one species with a lower (a) and higher (b) flowering threshold relative to the anomaly, and interspecific synchronization of species (c) with different flowering thresholds. (The model parameters are resource baseline (*rb*), anomaly size (*as*), and anomaly interval (*ai*) and flowering threshold (*ft*), flowering threshold (*ftv*), and number of species (*ns*). (See Appendix B & C for details).

Chapter 3 - Choosing species for reforestation in diverse forest communities: social preference versus ecological suitability

3.1. Summary

Choosing species for reforestation programs or community forestry in species-rich tropical rainforest ecosystems is a complex task. Reforestation objectives, social preferences, and ecological attributes must be balanced to achieve landscape restoration, timber production, or community forestry objectives. Here we develop a method to make better species choices for reforestation programs with native species when limited silvicultural information is available. We conducted community surveys to determine social preference of tree species and inferred their ecological suitability for open-field plantations from growth rates and frequency in forest plots at different successional stages. Several species, for which silvicultural data was available, were correctly classified as promising or unsuitable for open-field reforestation. Notably, we found a strong negative correlation between ecological suitability indicators and socioeconomic preference ranks (r = - 0.51). Only a single outlier species ranked very high in both categories. This result highlights the difficulty of finding suitable native species for community forestry and offers an explanation why reforestation efforts with native species often fail. We concluded that the approach should be a useful first screening of species-rich forest communities for potential reforestation species. Our results also support the view that species-rich tropical rainforests are not an easily renewable natural resource in a sense that secondary forests will provide an equivalent resource value to local communities.

3.2. Introduction

Government reforestation and community forestry programs are important activities to offset the results of many decades of deforestation, which has left many countries and jurisdictions with an impoverished and degraded land base (Hansen et al. 2013). Tropical forest ecosystems have been particularly affected, and still experience a poor ratio of forest loss to forest gain (3.6:1 for the 2000-2010 period), despite promoting natural recovery after forest harvest and afforestation and reforestation efforts (Hansen et al. 2013). Loss of tropical forest cover by geographic regions is most severe in Southeast Asia (37% of total area remaining in 2005), followed by Central America and Central Africa with 45%, and the Amazon basin with 56% forest cover remaining (FAO 2005). Primary forest in Southeast Asia and Central America are most affected with 16% and 18% original forest cover remaining intact, respectively (FAO 2005). Deforestation rates also remain unsustainable with 1.2%/year of net forest loss between 1990 and 2005 in Central America, 0.7%/year net forest loss in Southeast Asia, and 0.5%/year in Central Africa and the Amazon region (FAO 2005).

Within major geographic regions, countries with high accessibility to forests and high population densities have little primary forest left, often limited to mountainous regions. For example, in Southeast Asia, the Philippines, Vietnam, Laos, and Cambodia had less than 6% primary forest cover left in 2005 (FAO 2005). In these countries, land degradation, erosion, water supply concerns, and landslides have often become common problems (Sidle et al. 2006). In recent decades, the occurrence of landslides in Southeast Asia increased fivefold from 1970 levels due to deforestation (Forbes and Broadhead 2013). With tropical cyclones making landfall five to seven times a year in the Philippines, disasters are common with 35.4 per million fatalities reported between 1950 and 2009, the second highest rate worldwide (Forbes and Broadhead 2013).

In response, government programs aimed at reforestation and protection of remaining forests were created. Many countries in Southeast Asia imposed logging bans on natural forest lands, phased out large timber operations and created large-scale government reforestation programs and community forestry programs (Tumaneng-Diete et al. 2005, McElwee 2009, Pulhin and Dressler 2009, Yonariza and Singzon 2010). Between 1985 and 1997 the percent of total protected areas increased from 4% to 9% worldwide (Zimmerer et al. 2004), and community forestry systems were widely adopted to produce locally required resources and reduce pressures on harvesting the remaining natural forests (Molnar et al. 2007). Community forestry programs worldwide increased by 70% to approximately 250 million hectares between 1985 and 2008 (Molnar et al. 2007). In addition, large-scale government reforestation program were created in many jurisdictions, including the Philippines, where government program reforested 560,000 hectares between 1990 and 2000 (Cruz 2001).

Although large-scale reforestation programs that aim to rehabilitate degraded forest lands and improve watershed properties are laudable, one potential concern is that most of these initiatives rely on exotic species (Tolentino 2008). Large-scale reforestation programs in Southeast Asia primarily plant *Gemelina arborea,* pine, *Swietenia* sp., *Eucalyptus* sp., *Tectona grandis* and *Acacia* sp., because of readily available seed from international seed banks and well established nursery protocols and silvicultural treatments for these species. As a result, exotics amount to 80% of all trees planted in tropical reforestation efforts (Cruz 2001, Tolentino 2008, Yonariza and Singzon 2010). Exotic species monocultures, however, may change natural habitat conditions, water balances, and nutrient cycles, typically in ways that do not support biodiversity of native species (Hooper 2002). Additionally, exotic species can sometimes be vulnerable to large outbreaks of insect pests and diseases, such as the Dothistroma Needle Blight in Central America and Asia (Watt et al. 2009) and the Rhizina root disease in Southern Africa that have set back reforestation efforts to the beginning (Wingfield et al. 2001a).

For large-scale reforestation and landscape restoration projects, native trees are therefore preferred (FAO 2014), although native tree planting is not easy to carry out for lack of silvicultural knowledge and regeneration ecology (Bautista 1990). Planting native species exclusively is not required for community reforestation projects with limited ecological impacts because of their small scale, but native species could also be beneficial here because the species may be well known and valued by local communities for their non-timber benefits (Peters et al. 1989, Lacuna-Richman 2002, Mangaoang and Pasa 2003, McElwee 2008). In the Philippines, community forestry with native species has shown some success in the form of “rainforestation farming” programs, which combine native species for reforestation with some exotic fruit trees to support local livelihoods (Goltenboth 2004, Vilei 2009, Schneider et al. 2014). Similar initiatives are sponsored throughout the world by regional governments and international agencies, including the Reducing Emissions from Deforestation and forest Degradation (REDD+) program, the largest global initiative to combat deforestation, foster sustainable forest management, and fight rural poverty (Agrawal and Angelsen 2009). As a result of these programs, Asian tropical regions are experiencing planted or spontaneous forest regrowth in more than 50% of sites of tropical forest degradation (Rudel 2012).

In support of such programs, this research aims to make better-informed species selections for reforestation programs based on the ecology and socioeconomic desirability of tree species. The primary objective of this paper is to develop a screening method for potential reforestation species when silvicultural information is lacking. To identify early-succession species, suitable for reforestation under open field conditions, we investigate if standard inventory plot data can yield the required ecological information. Our expectation is that high frequency and basal area in early-successional plots is associated with pioneer species and that will do well when planted in degraded sites. Our second objective is to assess social preferences and find species with early-successional ecological characteristics that are also highly valued as reforestation species by local communities. While our research is a regional case study, we aim at developing a simple, low-cost approach to screen species-rich communities for potential reforestation species that can be applied to other regions and forest communities.

3.3. Materials and methods

3.3.1. Study site and ecological data

Our study site is the North Negros Natural Park (NNNP), located in Silay City of Negros Occidental Province, Philippines (Fig 3-1). The NNNP was initially established in 1935 as a timber reserve (Hamann et al. 1999, Hamann 2002), but expanded in 2005 and declared a Nature Reserve to protect a high biodiversity, including many rare, endemic and endangered plant and animal species. Previous botanical assessment in 1999 found 92 different species of 39 families, with 50% of the trees coming from the *Dipterocarpaceae*, *Lauraceae*, *Sapotaceae*, *Burseraceae* and *Melastomaceae* families. The most common individuals were of the *Litsea luzonica, Canarium asperum, Platea excels* and *Palaquium* sp. species representing a sub-montane forest ecosystem without a pronounced dry season (Hamann et al. 1999).

The experimental study plots are on the northwestern slope of Mt. Mandalagan close to Mt. Silay (10°38.0' N, 123°13.0' E) at an average elevation of a 1000 m. The three experimental plots were chosen to represent early-, mid-, and late-successional stages. They include a 1.61 hectare plot in an early-successional forest area cleared for pasture and abandoned in 1975, a 1.71 hectare plot in a mid-successional forest destroyed through bombing during World War II and a 5.15 hectare area of old-growth forest not known to be cleared in the past 100 years. In the entire forested area truly undisturbed forest is difficult to find because of proximity to settlement. However, areas surveyed are on a rugged terrain and prevented logging industry from harvesting trees. Low intensity use by the community involves rattan, firewood and seedling collection. Some areas of the forest have been sub planted with coffee by the community.

The plot size was determined via species-area curves, so that further increases in plot size would not capture additional species (except very rare species). While this approach captures most species that would be considered for reforestation in a local area, the approach does not yield comprehensive information for a larger geographic region, for example other islands in the Philippines. Rather, our approach is meant to be easily repeatable to screen other species-rich communities for local reforestation programs elsewhere.

Tree measurements were made three times over an 18 year period in 1995, 2003 and 2012 at all three successional plots, where all trees more than 10 centimeters DBH were measured and permanently tagged. Measurements for these trees included DBH, crown position, survival. New recruiting trees recorded during the 2003 and 2012 measurements were also permanently tagged. Seed sizes used in the analysis in this paper were obtained from a previous botanical survey (Hamann et al 1999). Species diversity was evaluated based on frequency data using the Shannon–Weaver index (Shannon and Weaver 1948).

3.3.2. Survey of social preferences

A survey of local knowledge of native tree uses in North Negros Natural Park was conducted through questionnaires and picture guide developed from the botanical inventory effort. The questionnaire was designed with open ended questions about forest use and given to community members employed as forest guards of the protected area. Forest guards were a useful focus group because they were recruited from the portion of the population that was forest-dependent, in part to provide alternative livelihoods to forest extraction. While they may not represent a completely unbiased sample of the preferences of the local forest-dependent population, they provided an opportunity to efficiently assemble a local knowledge base of social preferences and uses of local tree species. The questionnaires were given at the general assembly and included forest guards from five municipalities around the North Negros Natural Park, Silay, Talisay, Victorias, E.B. Magalona and Sagay with 47 respondents. The forest guards filled out the questionnaire on their own with instructions given to the group prior to the start. All questionnaires from 3 general assemblies were analyzed as one group.

In compliance with Canadian research ethics requirements for community surveys, the involvement in the questionnaire was voluntary and the reasons and use of this data were explained to the respondents before giving the questionnaire. Participation could be terminated by the respondents at any time during or after the completion of the questionnaire and the incomplete questionnaires were not used in the analysis. Respondents had the right to request that their answers not be used in the study for a month after the completion of the questionnaire.

The questionnaire asked respondents to list tree species that the community uses as a source of food, lumber, firewood, medicine, and charcoal making or to the forest for general ecological observations (for example, use of fruit trees by wildlife). The responded listed local names of tree species they considered important to the community with their uses. Local names were later matched with a scientific name using a species guide developed during the first forest assessment that contains 92 species of trees found in the North Negros Natural Park in 1995 (Hamann et al. 1999). The importance was ranked by combining the number of uses noted by respondents and the number of respondents that mention the species. The highest-ranking trees are those that have several uses and were mentioned by many people as important to the community.

3.3.3. Statistical analysis of plot data

To analyze which native species have an early-successional life history syndrome, we derived three variables for each species at each successional plot. They are: 1) relative frequency, calculated as the number of trees per hectare for each species, 2) basal area, calculated as the total stem area per hectare of each species, and 3) growth increment among measurements in 1995, 2003, and 2012, calculated as the relative change in basal area. Growth rates were missing for some species because measurements were not available in one of the survey times. Growth data was not always available in one interval for rare species because of mortality, or became available in the second interval because of recruitment. To use this data, we estimated the growth rate for the missing year with a linear mixed model implemented with the *ASReml* (Butler et al. 2007) for R programming environment (R Core Team 2014). Rather than calculating means across multiple survey times, we use best linear unbiased estimates (BLUEs) from the linear mixed model. These estimates conform to straight means if there are no missing values, and otherwise provides an unbiased estimate of the true mean.

To visualize groups of putative early- and late-successional life history syndromes, we applied nonmetric multidimensional scaling (NMDS) of the nine variables, implemented with *nmds* function of the *ecodist* package (Goslee and Urban 2007) for the R programming environment. This scaling technique works by calculating a multivariate distance matrix based on the available measurements, and subsequently ordinate the observations based on this matrix (Kruskal 1964). Here, we used a Bray-Curtis distance matrix, suitable for presence absence and quantitative measurements in community ecology data to ordinates the observations with NMDS. One purpose of the NMDS ordination is to search for similarities in the multivariate dataset to find easy to measure variables, such as frequencies, that may stand for other, harder to get variables such as growth rates. The separation of data by successional syndrome was verified by plotting seed sizes of known species, with larger seed sizes usually belonging to species of late-successional syndromes (Budowski 1965, Bazzaz and Pickett 1980). The species with missing data estimated with best linear unbiased predictions with *ASReml* were left out of the original ordination and their NMDS scores were later determined with the *predict* function to avoid a biased ordination because of poor data quality. The NMDS scores of species corresponding to successional syndromes were plotted against social preference data from the questionnaire, and the relationship was quantified with a Pearson correlation analysis.

3.4. Results

3.4.1. Characteristics of experimental plots

The selected forest plots show largely typical characteristics of a successional series regarding basal area, tree density, and species diversity (Fig. 3-2). Notable is the reversal of expected density in the early- and mid-successional plot, because of many trees in the early-successional plot having several stems counted as a single tree. Before the second field season, a strong typhoon hit the area and many smaller trees fell or were destroyed by branches falling from large trees, causing the density of small trees (DBH 10-20cm) to decline. The basal area measurement, driven by large trees was not significantly reduced, however, with little mortality in the large diameter classes. Species diversity, measured by the Shannon–Weaver index, declined over time in the mature plot, while it increased in the successional plots. The early-successional plot's high diversity was driven by the evenness and not the richness of species. The early-successional plot has the fewest species but nearly half occur in high frequency (relative frequency > 5 trees/ha) compared with a third of species occurring in high frequency in mid- and late-successional plots.

3.4.2. Successional syndromes of tree species

The NMDS of ecological tree data (Fig. 3-3) explains 89% of the variance in the dataset in two dimensions. In NMDS, dimensions are not orthogonal, and therefore we do not report variance explained for individual dimensions. Vectors in the plot indicate how original measurements are related to NMDS dimension scores. The length reflects the strength of the correlation and the direction indicates positive or negative correlations between dimensions and original variables. Points in the plot represent tree species and their positions indicate associations with original measurements. For example, species toward the right have high frequency and basal area in the early-successional plot, and show high growth rates in the late-successional plot. Further, we scale the plots based on a life history trait often associated with a successional syndrome: seed size. For gray dots, no data on seed size was available.

Dimension 1 in Fig. 3-3 is strongly associated with an early-successional syndrome. High frequency and basal area in the early-successional plot are positively correlated with this axis, although the growth rate is negatively correlated, indicating that the species that fall on the right side of this plot have reached maturity in the early-successional plot. These species are also correlated with high growth relates on the late-successional plot, where this species group vigorously recruits in gaps after typhoon disturbances shortly after the plot series was established. Dimension 2 weakly separates late- and mid-successional species groups, with putative mid-successional species correlating primarily along the second dimension (species toward the top of the plot) and show little association with the first dimension. Putatively late-successional trees correlate with each dimension and are located toward the top and left areas of the plot. Large seed sizes are usually associated with mid- or late-successional species

(Budowski 1965, Bazzaz and Pickett 1980) and are grouping on the left and top area of the ordination.

3.4.3. Socioeconomic importance of tree species

Surveys of social tree uses resulted in 71 tree species mentioned by survey participants, of which three-quarters were native and one-quarter introduced species (Appendix D). Of the native species, 35 were found in the ecological monitoring plots evaluated in this study (Table 3-1). Plant families that ranked high in social importance (based on number of uses and number of participants that mentioned them) include *Dipterocarpaceae*, *Fabaceae* and *Lauraceae*, with five or more highly valued species each. *Dipterocarpaceae*, also known as Philippine mahogany are some of the most important tropical timber species. Dipterocarps are primarily used locally for construction and furniture making. *Fabaceae* contain fast growing, nitrogen fixing tree species with dense wood valued for charcoal production and lumber for construction. *Lauracea* are the second dominant tree species group in the upland forests of Southeast Asia, highly valued for their timber as well.

The most common uses mentioned across all species are traditional timber values. Construction, furniture making, and lumber constituted 48% of all mentioned use of native and non-native trees by the community in this study (Appendix D). Next in importance are the species used for charcoal and firewood, constituting 17% of all species-use combinations, although species used for firewood and charcoal can easily be substituted by any other tree species. Species that had mostly non-timber uses accounted for 28% of species-use combinations, with *Fabaceae*, *Guttiferae*, *Icainaceae*, *Moraceae, Lauraceae* and *Sapotaceae* families mentioned prominently for their use as fruit trees for wildlife and for human consumption. Medicinal use accounted for 5.6%, and included species of the families *Moraceae*, *Meliaceae*, *Lauraceae*, *Fagaceae*, and *Icainaceae*.

To identify species for reforestation that have economic uses and are suitable for open-field reforestation, we plotted the Dimension 1 from Fig. 3-3 where high scores indicated early-successional species as a function of their social preference from Table 3-1 (Fig. 3-4). The most desirable species should fall into the upper right quarter of this graph. Notably, we observed a negative correlation between inferred ecological suitability and social preference ranks (r=-0.41, p=0.015, df=42). If two outliers are excluded (Fig. 3-4, A.phi - *Agathis philippinensis* and S.con - *Shorea contorta*), the correlation is more pronounced (r=-0.51, p=0.003, df=42).

3.5. Discussion

3.5.1. Social preference versus ecological suitability

The negative association between ecological suitability indicators and social preference ranks highlights the difficulty of finding suitable native species for community forestry where social acceptability is essential and planting efforts are often restricted to abandoned farmland. Nevertheless, the realism of the proposed approach is highlighted particularly saliently by the two outlier species in Fig. 3-3, *Agathis philippinensis* and *Shorea contorta*.

Agathis philippinensis has a native range from Indonesia to the Philippines and related species are well regarded in Australia and New Zealand, where the species group is known as kauri, valued for a variety of products made from their resin (Lacuna-Richman 2006). *Agathis philippinensis* cannot survive in open-field plantations but can only be sub planted in secondary forest in enrichment efforts (Orwa et al. 2009). Our analysis correctly identified the species as late-successional species. *Shorea contorta* is a dipterocarp species native to Southeast Asian forests, known as white lauan in the Philippines and is marketed for its wood qualities worldwide. In this study, it had a high preference rank as well (Table 3-1 & Appendix D). In contrast with *Agathis* this species is known to grow well in full sunlight unlike most other dipterocarp species (Schneider et al. 2014) and was correctly classified in this study as an early-successional species.

Further, we note that there are many early-successional species in the category of medicinal uses. Thus, in this use category, the negative relationship between socioeconomic preference and ecological suitability is weaker and provides an opportunity for species selection. Some examples include the use of *Ficus septica* roots to cure boils and to eliminate access water, while the leaves have antirheumatic properties and relieve headaches (Lanting and Palaypayon 2002). The bark of *Shorea contorta* is used for cough or as an astringent while the wood decoction is used to treat tumors (Lanting and Palaypayon 2002). Another example is *Casuarina equisetifolia*, the leaves of which have antifungal and antibacterial properties and are used traditionally to treat gonorrhea and stomach infections (Doyle and Aalbersberg 1998). Species with medicinal uses are commonly found in the early-succesional group, which produce a host of secondary metabolites and allelochemicals as protection against herbivores and for other interspecies defenses (Bryant et al. 1983).

3.5.2. Native forests are not easily renewable

Our results indicate that species-rich Dipterocarp forests may not be an easily renewable natural resource with respect to their usefulness and desirability to forest-dependent communities. Late-successional tree species have more uses as timber and non-timber products, whereas, early-successional secondary forests and planted forests do not serve social needs of forest-dependent communities as original native forests. Previous research on economic valuation of mature tropical forest has shown that old-growth forest bring the highest economic benefits from non-timber forest products (Peters et al. 1989) and forest products from diverse forests can provide a diversity of marketable products to local communities (Pattanayak and Sills 2001). Studies from the Philippines also show that non-timber forest products, particularly the resin of *Agathis philippinensis*, can be a major source of livelihood for forest communities (Lacuna-Richman 2002). Alternative uses, such as plantation forestry or a cattle pasture compared unfavorably in valuations (Peters et al. 1989). Although this study found traditional timber uses rather than non-timber products to be the most important socioeconomic feature in contrast with Peters et al. (1989) those values were largely restricted to late-successional species. Other studies also find majority of households relying on timber uses for subsistence in forest dependent communities in Vietnam (McElwee 2008). Our results therefore confirm the importance of forest conservation and restraint in resource extraction from existing natural forests. Further research and trials in artificial propagation of old growth species is also needed as these species have limited seed sources, they are difficult to establish and are targets for extraction while they provide most of ecological and economic benefits, Our data also points to a potential explanation of why reforestation efforts with native species often fail. Typically, highly valued native trees used in local reforestation programs are late-successional species. However, late-successional species usually cannot survive the environmental stress associated with open-field plantation such as high insolation, dry and degraded soil conditions (Uhl et al. 1988). In the Philippines, for example, large-scale reforestation programs use a few native species, mostly dipterocarps, planted on small areas that show lower growth rates compared with exotics (Otsamo et al. 1997, Tolentino 2008). Other programs in the Philippines that focus on reforestation with a high diversity of native species, such as the "rainforestation farming" program, are small in scale and expensive to implement (Vilei 2009).

3.5.3. Reforestation compromises are necessary

The negative association between ecological suitability indicators and social preference ranks highlights the difficulty of finding suitable native species for community forestry where social acceptability is essential and planting efforts are often restricted to abandoned farmland. In our surveys we noted many replies for uses of exotic species in local communities, accounting for one-quarter of all species-use combinations. Including exotic species in small-scale, short rotation community forestry programs appears to be a sensible choice according to our surveys. Such programs should focus on high value and employ a variety of species for generating a reliable source of income (Lamb 1998, Leopold et al. 2001, Hartley 2002). The alternative choice of fast-growing, early-successional native species with few local uses may not meet the necessary socioeconomic criteria.

For large-scale afforestation projects to restore watershed services, and restore forest cover at the landscape level, we would discourage using exotic species. Although it is relatively easy to successfully establish large-scale monocultures of exotic trees with well-known propagation and planting techniques, the associated loss of biodiversity and the risk of unintended ecological impacts are well documented (Hooper 2002, Sayer et al. 2004, Carnus et al. 2006). In this case, planting a variety of native early-successional species would be the best first step to reestablish forest cover (Chazdon 2008). Such secondary forests will not provide immediate benefits to the communities, but over time natural regeneration (Lee et al. 2005, Piiroinen et al. 2015, Wolfe et al. 2015) or enrichment planting with highly valued native species (Paquette et al. 2009, Ashton et al. 2014) may restore the original economic and ecological benefits to local communities. Alternatively, high-value late-successional species may be established more rapidly by the initial use of early-successional species as nurse trees to shade out grasses and herbs and planting high valued canopy species after a few years (Ashton et al. 2001).

3.5.4. General guidelines for species selection

Our original expectation of being able to identify a suite of easy to measure, quantitative indicators for promising reforestation species from observation plots proved more difficult than expected. For example, growth rates of early-successional species in our study were highly dependent on stand history. In the early-successional plot used in this study early-successional species were already reaching maturity, resulting in low growth rates. In contrast, regeneration in forest gaps after typhoon damage in the late-successional plot resulted in rapid growth of early-successional species. Nevertheless, grouping species by similarity in their inventory plot measurements (Fig. 3-3), yielded interpretable ecological information and we would expect useful (while not necessarily identical) results if the approach was applied elsewhere in a different forest community with plots of somewhat different successional histories.

Information on suitability of species for reforestation under open-field conditions and degraded sites can, of course, most reliably be obtained from planting trials, but they involve an investment of time and resources if carried out for many species. Such trials can identify species with growth rates and survival that compares favorably with exotic species, as research in Costa Rica (Butterfield 1996), Panama (Hooper 2002, Wishnie et al. 2007), and Indonesia (Otsamo et al. 1997). The suitability of a range of native species for reforestation can further be enhanced by an initial cycle of genetic tree improvement (Leakey and Simons 1997). However, this involves long-term investments that can only be made for a few species.

A first round of screening of species-rich forest communities for suitable reforestation species therefore remains an important problem to solve. Life history attributes, such as small seed size, also corresponded well to successional syndromes inferred from inventory plot statistics. Data may be obtained for other regions from databases such as TRY (*http://www.try-db.org*). Similarly, plot inventory data may be obtained for analysis from international data collection efforts such as BIEN (*http://bien.nceas.ucsb.edu*). We propose that screening for such indicators of early-successional life history strategies should be an important first step in narrowing species choices for reforestation. Ecological indicators, combined with social surveys data can rapidly be generated to support new, large-scale reforestation programs such as those initiated under the Reducing Emissions from Deforestation and Forest Degradation (REDD+) programs that include objectives such as on biodiversity conservation and providing livelihoods for the local communities (Agrawal and Angelsen 2009).

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Table 3-1. Local uses of native tree species in the North Negros Natural Park. The table summarizes selected data from the questionnaires with 47 forest guards that listed species of socioeconomic importance and their uses. The uses are grouped into lumber/construction (Lum), varnish/resin (Var), charcoal/firewood (Cha), edible products (Food), ecological-food for animals (Eco), medicine (Med). The summary includes sum of total uses for each species (Tot), number of respondents that mention this species (N) and rank that combines the number of uses and the number of respondents. Local names could refer to several related species (reflected by multiple codes), although only one primary species name is given. Refer to Appendix D for a full list of local species uses.

Species (Code)	Family	Uses						Statistics		
		Lum	Var	Cha	Food	Eco	Med	Tot	N	Rank
Agathis philippinensis (A.phi)	Araucariaceae	2	3	1	0	0	0	6	37	43
Palaquium luzoniensis (P.luz)	Sapotaceae	4	0	0	1	0	0	5	20	25
Shorea contorta (S.con)	Dipterocarpaceae	4	0	1	0	0	1	6	17	23
Garcinia brevirostris (G.bre/sp2)	Guttiferae	4	0	0	0	1	0	5	15	20
Syzygium gracile (S.gra/S.sp2)	Myrtaceae	4	0	0	0	1	0	5	15	20
Cinnamomum mercadoi (C.mer)	Lauraceae	3	0	2	1	0	1	7	12	19
Platea excelsa (P.exe)	Icainaceae	0	0	0	1	3	1	5	11	16
Shorea polysperma (S.pol)	Dipterocarpaceae	3	0	0	0	0	0	3	12	15
Dillenia philippinesis (D.phi)	Dilleniaceae	0	0	0	1	1	0	2	12	14
Prunus fragrans (P.fra)	Rosaceae	2	0	0	1	2	0	5	9	14
Lithocarpus sp. (L.sp1)	Fagaceae	1	0	0	1	2	1	5	8	13
Memexylon brachybotris (M.bra)	Melastomataceae	2	0	0	0	1	0	3	9	12
Mallotus sp.(M.sp1)	Melastomataceae	2	0	0	0	1	0	3	9	12
Bischofia javanica (B.jav)	Euphorbiaceae	2	0	0	0	1	0	3	8	11
Garcinia binucao (G.bin/sp1)	Guttiferae	0	0	1	1	0	0	2	8	10
Arthrocarpus heterophyllus(A.het)	Moraceae	1	0	0	1	0	0	2	7	9
Actinodaphne sp. (A.sp1)	Lauraceae	0	0	2	0	1	0	3	6	9
Ficus crysolepis (F.chr/sp6/sp8)	Moraceae	0	0	0	0	3	0	3	6	9
Myrica esculenta (M.esc)	Myricaceae	0	0	0	0	1	0	1	8	9
Shorea almon (S.alm)	Dipterocarpaceae	3	0	1	0	0	0	4	5	9
Pometia pinnata (P.pin)	Sapindaceae	1	0	1	1	0	1	4	4	8
Trema orientalis (T.ori)	Urticaceae	1	0	0	0	1	0	2	5	7
Memexylon cumingii (M.cum)	Melastomataceae	0	0	1	0	0	0	1	6	7
Alphitonia excelsa (A.exc)	Rhamnaceae	1	1	0	0	0	0	2	3	5
Palaquium sp. (P.sp1)	Sapotaceae	1	0	0	0	0	0	1	4	5
Ficus benjamina (F.ben/F.sp4)	Moraceae	0	0	0	0	1	0	1	3	4
Litsea tomentosa (L.tom)	Lauraceae	1	0	0	0	0	0	1	3	4
Ficus septica (F.sep)	Moraceae	0	0	0	0	1	0	1	2	3
Canarium villosum (C.vil)	Burseraceae	1	0	0	0	0	0	1	1	2

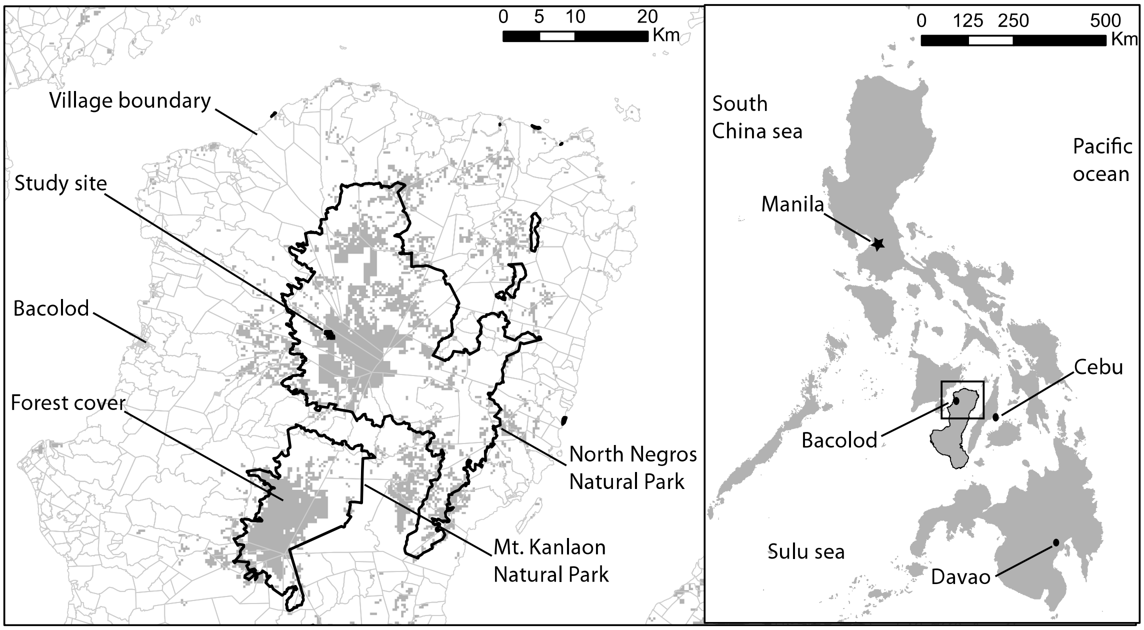

Figure 3-1. Map of the study area located within the North Negros Natural Park on Negros Island in the Philippines. The questionnaires were collected from local stakeholders that live in the communities around the park.

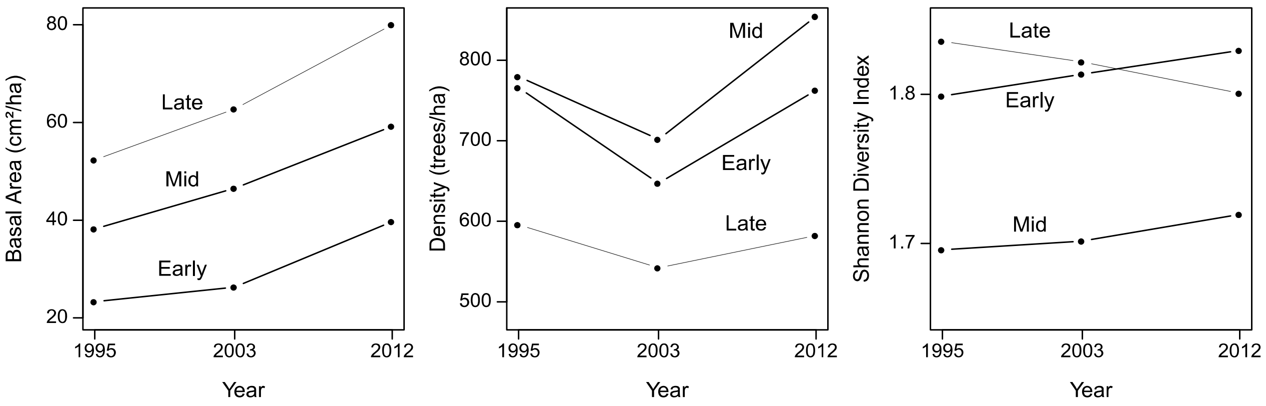

Figure 3-2. Tree size, density and diversity of trees with a diameter at breast height larger than 10 cm from the early, mid and late-successional study plots in the North Negros Natural Park. The graph shows the change in basal area, density and species diversity over a period of 18 years from three surveys.

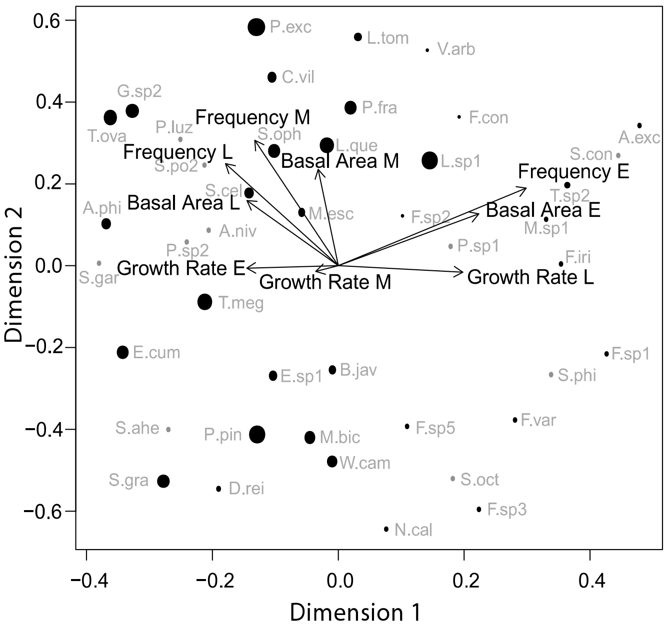

Figure 3-3. Ordination of tree species by similarity in frequency and growth rate in early (E), mid (M) and late-successional (L) study plots. The size of dots represents seed size (which was not used in the ordination), except when data was not available (gray dots). Dimension 1 of the non-metric multidimensional scaling (NMDS) procedure separates early successional species on the right from late-successional species on the left. Dimension 2 further separates putatively mid-successional species (high values) from late-successional species.

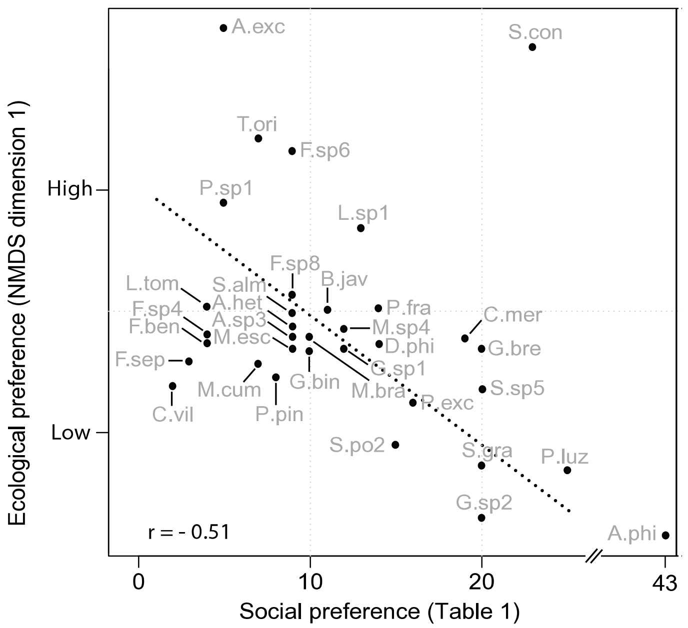

Figure 3-4. Scatter plot of native species ranked by socioeconomic importance (x-axis) and inferred ecological suitability for open field plantation (y-axis). The trend line and r-value exclude the outliers (A.phi and S.con). However, the relationship remains statistically significant and moderately strong (r = -0.41, outliers included or r=-0.51, outliers excluded).

Chapter 4 - Effects of land use and protected area management on socioeconomic conditions of forest dependent communities: a case study in the North Negros Natural Park, Philippines

4.1. Summary

Forest-dependent communities in the tropics typically rank lower in socioeconomic status than agricultural and urban communities, and improving the livelihood choices of forest-dependent communities while protecting forest resources can be a difficult task. Additional conflicts can arise where biodiversity conservation objectives further restrict resource access to forest dependent communities. In this study, we investigate how land use, land use planning, and protected area management affects communities in and around a major forest reserve on Negros island, Philippines. Using mixed methods we conduct a socioeconomic analysis at two scales: a provincial-level analysis relating land use to socioeconomic well-being, and a community-level analysis contrasting villages that are close versus distant to a protected area. While rural communities have fewer amenities and infrastructure than urban centers in general, the community-level analysis showed that socioeconomic well-being was higher in areas close to protected areas. The study provides an example of forest conservation that does not necessarily result in lower socioeconomic status for local communities.

4.2. Introduction

Forest dependent communities that rely on forest products for livelihoods are among the poorest in developing countries, including Southeast Asia (Kummer 1992a, Kummer 1992b, Kummer and Turner 1994, Saastamoinen 1996). This has not always been the case, with commercial logging being the largest source of revenue between 1960-80s (Tumaneng-Diete et al. 2005). The level of resource extraction has, however, often been unsustainable and causing deforestation and loss of livelihood options. Despite efforts to reclaim deforested lands, tropical forests are undergoing a net loss in recent years (Hansen et al. 2013). Primary forest in Southeast Asia and Central America are most affected with 16% and 18% original forest cover remaining intact, respectively by 2005 (FAO 2005, Koh 2007). In Southeast Asia, the Philippines experienced one of the worst rates of forest loss with 6% of primary forest left, mostly found on steep mountain tops and in the uplands (FAO 2005).

Rural communities in Southeast Asia rely on forest products for subsistence directly through logging and indirectly by collecting non-timber forest products for firewood, medicine, and food sources (Lacuna-Richman 2002, McElwee 2008, Cagalanan 2013, Chechina and Hamann 2015). About 20 million people in the Philippines, or 22% of the population live within public forest lands that are designated for forest use (Tumaneng-Diete et al. 2005). Close to 80% of these forest communities rely on farming as primary source of livelihood and most households have an income lower than the poverty threshold of one dollar a day per person (Pulhin and Inoue 2008). Overall the incidence of poverty, or percent of households below the poverty threshold, in the Philippines is 25% (Reyes et al. 2012). However, agricultural and forest-dependent households have a poverty rate of 57% compared with 17% in urban households (Reyes et al. 2012). On average, the incidence of poverty is strongly correlated with provinces that have larger forest cover in the Philippines (Tumaneng-Diete et al. 2005).

Poverty incidence may be an incomplete measure of well-being because many rural subsistence based communities do not have formal income. Other measures of well-being can include education levels, rates of malnutrition, among others. For example, rural remote households that have large family sizes and where the household head has little or no formal education are among the poorest and experience a high incidence of malnutrition (Reyes et al. 2012). There is less government investment in infrastructure in rural communities because of their remoteness and few households have running water or electricity (Reyes et al. 2012). Many poor communities are in the uplands, where the land is less suited to most forms of cultivation, aside from fruit trees, and where regulations prohibit farming on steep slopes. Most upland communities do not have secure land titles. Lack of land tenure negatively affects the behavior of its occupants encouraging indigenous and forest communities to practice unsustainable land management as they have no incentive to conserve the land and forest resources (Bautista 1990).

As a result of rising poverty in rural areas and diminishing natural resources, there is a need for government programs that increase forest cover for land and watershed protection, and at the same time give control of natural resources to communities and facilitate sustainable livelihoods (Reyes et al. 2012). Recently, the forest policy in the Philippines has taken account of these multifaceted problems with programs including Integrated Conservation and Development Projects (ICDP), Integrated Social Forestry (ISF, introduced in 1982), Communality Based Forestry Management (CBFM, introduced in 1995) and National Integrated Protected Areas System (NIPAS, introduced in 1992) (Bautista 1990, Cagalanan 2013). By the year 2000, these programs accomplished reforestation on 560,000 hectares (Cruz 2001), and currently 15% of total land area of the Philippines are managed under ISF or CBFM programs (Pulhin and Dressler 2009).

A central idea of community forestry programs is to give control of forest lands back to the communities, thereby encouraging them to manage their forest resources responsibly. However, ISF or CBFM programs in the Philippines are still under government control, imposing strict rules on how the land should be managed by the communities (Gauld 2000) with little to no local benefits (Pulhin and Dressler 2009). Several studies find that the livelihood opportunities from community forestry programs are not distributed equally and are received mostly by wealthy households (McElwee 2008, Cagalanan 2013). As little as 4% of the community may benefit from these forestry programs (Pulhin and Dressler 2009). However, positive examples exist as well, showing that strong local leadership, stakeholder involvement in management and strong social capital can improve outcomes of community forestry (Magno 2001).

For the most part, community forestry programs focus on planting fast-growing profitable, often exotic tree species, which do not serve biodiversity and conservation objectives (Cruz 2001, Tolentino 2008, Yonariza and Singzon 2010). On the other hand, protected natural area management focuses primarily on conserving the biodiversity and ecosystem functions, which can impose restrictions on the forest resources that communities rely on for subsistence (Cernea and Schmidt-Soltau 2008). Therefore, protected areas can exacerbate poverty by limiting access to land and imposing a change to livelihood activities that previously involved forest uses such as logging or charcoal production (McElwee 2008). On the other hand, protected areas can reduce poverty in communities where the focus is redirected towards using alternative, high-value forest products or focus on eco-tourism activities, and where the land under protection is not suitable for agriculture (Ferraro et al. 2011). As a result of the variable impact that protected areas and forest programs have on the poor, there is a need to understand socioeconomic conditions in these communities and to target the dual mandate of poverty alleviation and forest conservation (McElwee 2009, McCay and Jones 2011, Groom and Palmer 2012).

In our study, with a focus on the relationship between protected areas and the well-being of surrounding rural communities, we examine the land use of Negros Occidental in relation to socioeconomic outcomes. We conduct a provincial-level analysis for Negros Occidental to analyse the effects that land cover has on socioeconomic status of communities. Further, we conduct a village level analysis of communities from 11 municipalities around the protected area to gain additional insight into the socioeconomic conditions around protected areas. We compare the socioeconomic status of rural communities surrounding a protected natural park to other rural communities farther from the park as well as urban centers. Specifically, we compare the benefits and costs to communities associated with a protected forest reserve with rural communities outside the reserve. The analysis relies on several sources of secondary data supplemented with interview data to identify differences in socioeconomic status. Our aim is to quantify relationships between forest conservation and poverty alleviation in rural communities. Given the potential efficiencies in using mostly secondary sources for this assessment, we also evaluate the utility of secondary data sources in defining socioeconomic status in this region.

4.3. Materials and methods

4.3.1. Study region and area

We focus our research on the province of Negros Occidental in the West Visayan region of the Philippines. We analyze the socioeconomic indicators of rural barangays (villages) surrounding the North Negros Natural Park (Fig. 1). The study area includes 32 municipalities of Negros Occidental Province and 235 barangays that surround the protected areas, shown in gray in Fig. 1. Negros Occidental experienced deforestation and land degradation because of commercial logging. Commercial logging created roads that opened unsettled regions for settlement and agricultural development, mainly sugar plantations. As the sugar industry was booming in the 1950s, waves of migrant workers came in search for work. The migration creating increased demand for land (Mulkins 2000) putting stress on already diminishing resources. By 1996 only

3% of the original intact forest cover remained (Hamann 2002) that is far below the national average forest cover of 20% (Munez 1999). The majority of residents are substance farmers (Mulkins 2000). Similar to other forest communities in the Philippines (Lacuna-Richman 2002) they rely on forest resources to gather building materials, firewood, medicine and food. Deforestation has decreased the livelihood of many people in Negros Occidental (Garcia 2001).

The province has one of the few significant natural forests left in the central Visaya islands group of the Philippines (Hamann 2002). The forest was established as a timber reserve in 1946, but by 2005 became a protected area under the NIPAS act and named the North Negros National Park to preserve high diversity of species, many of which are rare, endemic and endangered (Hamann et al. 1999, Hamann 2002, Chechina and Hamann 2015). Although logging is not allowed in protected areas, illegal logging of old growth trees occasionally occurs in the park. Other resource extractions include rattan collection, gathering of non-timber products for food and medicine (such as fruits, seeds, bark and leaves), small tree removal for charcoal, and wildlife hunting (Hamann 2002). An ISF project was established in the park in 1995, and significant deforestation was curbed since the establishment of community forestry and protected area declaration.

4.3.2. Planned land use and actual land cover data

We collect land cover data because it can indicate potential for alternative livelihoods for a community and it also represents available natural resources. We compare land cover data to planned land use for Negros Occidental. Planned land use data (Fig. 2, left panel) was extracted from the Negros Occidental Land Use Plan 2012 prepared by the provincial planning office (PPDO 2011). Actual land cover (Fig. 2, right panel), was derived from Envisat MERIS Fine Resolution (300 m) data (Arino et al. 2012). Forest cover is composed of two layers from the remotely sensed data: the closed to open (>15%) broadleaved evergreen or semi-deciduous forest (>5m) and mosaic forest or shrub land (50-70%) / grassland (20-50%). Crop cover is composed of rain fed croplands and post-flooding or irrigated croplands (or aquatic) layers. Cropland often involves areas of large agricultural plantations, which is mostly sugarcane in Negros Occidental. Mosaic vegetation is composed of mosaic cropland (50-70%) / vegetation (grassland/shrubland/forest) (20-50%), mosaic vegetation (grassland/shrubland/forest) (50-70%) / cropland (20-50%), and closed to open (>15%) (broadleaved or needle leaved, evergreen or deciduous) shrubland (<5m) layers. Mosaic vegetation represents primarily subsistence farms scattered throughout the landscape among other vegetation. Mangrove is composed of closed to open (>15%) broadleaved forest regularly flooded (semi-permanently or temporarily) - fresh or brackish water and closed (>40%) broadleaved forest or shrubland permanently flooded - saline or brackish water layers. Finally, artificial surface class includes buildings, roads and similar areas (urban areas >50%) layer from the Envisat data.

Land cover summary statistics were extracted for each municipality in Negros Occidental and for each of 285 barangays of the study area using zonal statistics in ArcGIS (ESRI 2011). The planned land use data was obtained as vector data, which was converted to a 30 meter resolution raster, and then evaluated in the same way to obtain percent of planned land use for each municipality and 285 barangays. Municipality and barangay boundaries for the province were extracted from the Global Administrative Areas Database (Global Administrative Areas 2012). We calculate the percent of actual land cover in each planned land use category (forest land, protected area, sugarcane production, agriculture and rice, fisheries and mangrove, and build up areas) to analyze the degree to which the land use plan is being reflected by actual land cover.

4.3.3. Socioeconomic data

We base our methodology on several studies of forest dependent communities in Canada that use socioeconomic data from census subdivision to analyze community well-being, defined as social or economic benefits that communities gain (Patriquin et al. 2009, Stedman et al. 2011). We first look at municipality level data on socioeconomic indicators from 32 municipalities in Negros Occidental (Fig 4. municipal boundaries). Municipal level data on poverty, crime, home amenities (electricity, sanitation, potable water), number of facilities (heath centers and elementary schools), malnutrition, elementary school enrollment, population density, and total area was derived from the Negros Occidental Social and Economic Trends Report 2009 (PPDO 2011). This data is based on a 5% population sample and provides various measures of economic well-being in the municipality (PPDO 2011). We use data on investment in infrastructure and services, home amenities (electricity, potable water, and sanitation), elementary school enrollment, child malnutrition data and crime rates as indicators for social well-being of a community.

We conduct a Pearson's correlation analysis of land cover classes with socioeconomic indicators for municipalities of Negros Occidental to evaluate the well-being (Beckley et al. 2008). Correlation values (Carnus et al.) and significance (p-values) are given for this analysis. The significance values are adjusted for multiple inferences using the Holm method (Holm 1979) within each land class group.

In a second analysis we focus on 11 municipalities that surround the North Negros National Park and contain 285 barangays (Fig. 1, barangay boundaries). The barangay level socioeconomic indicators are different from the ones used for municipality level analysis for the lack of data availability on barangay level. For example, poverty or employment rates are not collected at the household or barangay level. For comparison, we collect indicators that are similar to the indicators at the municipality level and add others such as number of precincts, markets, daycare to substitute for lack of household amenities data. Barangay level demographic data on population density and data on facilities, such as number of markets, health stations, daycare centers, and elementary schools came from municipal planning department office for each barangay. Data on crime against women and children came from the women's desk at the Philippines National Police office of each municipality. Child malnutrition data was gathered at the municipal health centers. Number of registered and actual voters was gathered from the municipal commission on election office. Elementary school enrollment data was gathered from the Department of Education.

Based on the land use plan (PPDO 2011) we group the study area barangays into three groups as follows: (1) rural barangays in or close to the protected areas, (2) rural barangays outside of protected areas that are located in mostly agricultural settings and (3) urban barangays. We summarize land cover classes for 235 barangays of the 11 municipalities around the protected area (mean and standard deviation). Using the same groupings of barangays, we perform permutational analysis of variance of socioeconomic indicators (Anderson and Braak 2003) with the *aovp* function in *lmPerm* package in R programming environment (mean and standard errors reported). The pairwise comparison were performed using Wilcoxon signed-rank test (Wilcoxon 1950) and the results were adjusted for multiple inferences using the Holm method. These non-parametric methods were chosen because the socioeconomic data did not satisfy the assumption of normality in all cases.

4.3.4. Interview data

We conduct 15 interviews with residents of forest dependent households in North Negros to assess perception of community well-being. This group consisted of nine male and six female between the ages of 18 and 74 with average of 42. The interviews took place in the barangay Patag of Silay City, Negros Occidental. The barangay has 50% forest cover and is entirely located inside the boundary of the of the protected forest area. Participants were chosen using respondent driven sampling method (Salganik and Heckathorn 2004) to represent families living closest to the forest and that depend on forest resources. Interviews were also conducted with 38 residents of Patag that live farther from the forest but in the boundary of the protected area and are part of the ISF project that involves tree planting as part of the Japanese International Cooperation Agency (JICA) initiative in the area. This group consisted of 22 female and 14 male participants between the ages of 21 and 75 with an average age of 45.

The interviews were conducted with help of a local translator. The questionnaire was designed with close-ended questions focused on the source of income and livelihood, how respondents perceived their well-being, what trees and crops were planted by each household, how respondents supplement their livelihood, and their feeling about the protected forest. The involvement in the questionnaire was voluntary and the reasons and use of this data were explained to the respondents before administering the questionnaire. Ethics approval for interviews was obtained through the Human Research Ethics Review process at the University of Alberta Ethics Board and on February 28, 2012 with one year validity.

4.4. Results

4.4.1. Land use plan and actual land cover

In Table 1 we report how actual land cover corresponds to planned land use, corresponding to Fig. 2. Mosaic vegetation is the largest land cover in Negros Occidental and composes the majority of planned land use classes. Forest land use is composed of majority mosaic vegetation and protected areas also have majority mosaic vegetation and forest. Actual land cover shows that much of the forest has been deforested and converted to degraded lands with mixed subsistence farms (mosaic vegetation) and agricultural use (crops). Much of the cropland is found in designated sugarcane production and agricultural land. Although sugarcane is designated to be planted inland (Fig. 2, see sugarcane production in planned land use), most of it is planted in lowlands and in areas designated for agricultural production. Artificial areas are mostly located in areas designated for mangroves, mainly because of the coastal location.

4.4.2. Municipal-level socioeconomic analysis

The relationship between major land cover types (forest, crops, mosaic vegetation, mangrove, and artificial surfaces) and socioeconomic indicators of 32 municipalities in Negros Occidental is visualized in Fig. 3 with significant correlations reported in Table 2. The analysis reveals that municipalities that have large scale agriculture (crops) have lower poverty rates, more amenities such as homes with electricity and sanitation. Agricultural municipalities have more services which are represented by more health stations. Agricultural land is located closer to urban centers, which provided steady employment to surrounding communities and diversified livelihood. Sugarcane plantations that are located in crop land areas also provide additional seasonal work.

Areas that have mostly mosaic vegetation (areas of scattered farmland and mixed vegetation) or forest cover are highly correlated with poverty and negatively with amenities such as electricity and sanitation. These areas are mostly rural with subsistence type farming that does not provide many opportunities for employment. In places where subsistence farming is the main livelihood, families cannot afford to have children in school and many have children help on a farm instead of attending schools. These municipalities have somewhat higher rates of enrollment in elementary schools (not significant) than agriculture intensive areas.

4.4.3. Barangay-level socioeconomic analysis

Difference in land cover among rural barangays in or around the protected area, distant from the protected area and urban barangays are shown in Table 3. Barangays in protected areas have the lowest population density, are located further from the municipal center, have most of the forest land, larger total area and least of cropland. Other rural areas have most of the cropland and mosaic vegetation. Urban barangays have the highest population density, lowest total area, most of the mangrove areas, and large amount of cropland.

Difference in socioeconomic status of the three classes of barangays as above are shown in Table 4. Barangays in or around the protected forest area have higher rates of elementary school enrollment then other rural barangays. Even though there are fewer schools per capita, the enrollment rates are highest in urban centers, which could be explained by bigger schools inferred from the low student to teacher ratio. The schools are more accessible (data not shown) because of smaller distances and smaller total area in urban centers. Barangays outside the protected area are closer to urban centers and may have access to elementary schools where the enrollment rates may increase because of children from nearby rural centers attending these schools. It is likely that there is less need to build schools in rural centers if they are already in proximity to an urban center school. Malnutrition and crime rates were not significantly different between the two rural groups. Urban barangays have low child malnutrition rate (not significant) but higher crime. Barangays in protected areas have lowest voter registration rates than other rural communities and urban centers that could be associated with fewer registration precincts.

4.4.4. Socioeconomic characteristics of a forest dependent community based on interviews

Interviews of 15 participants in forest-dependent community from within the protected area revealed that they perceive themselves as relatively poor. Most of households have average earnings of 5,000 Philippine pesos per month (equivalent of 120 USD, less than one dollar per day per person for an average household of five). People in the community primarily rely on farming for subsistence. Half of respondents (7 of 15) noted farming as their only livelihood. A few people own small local stores, cater to local tourism or are employed by the municipality of Silay as forest guards or working in the ISF program. Interviewees noted that resources, power, and economic benefits are not equally distributed among the community with overwhelming majority (95%) not having access to these benefits and having no power for decision making.

The 38 respondents of the reference group that were part of the integrated social forestry (ISF) program had slightly higher income and more diversified livelihood options including community animal husbandry program established with the help of non-governmental development agencies. Of 38 respondents, more than half were involved in alternative livelihood activities to supplement farming. Only three of 38 participants worked seasonally on sugarcane plantations.

Respondents from both groups said that the forest reserve increases their well-being. The respondents from the ISF program also credited forest reserve as a source of additional livelihood option through community forestry and involvement of foreign NGO that provided them with funding. Others commented on the ecological benefits of the forest, mentioning that the weather is favorable for farming as the forest cover brings plenty of rain. Two participants attributed clean water to the remaining forest cover. Only one person in the Patag interview group said that the forest reserve increased poverty in the community. In general most community members interviewed agreed that the forest reserve positively affected their livelihood.

4.5. Discussion

4.5.1. Protected areas in Negros Occidental are not poverty traps

Negros Occidental is heavily deforested with 13 percent forest cover, of which 3 percent is old growth forest, and we perceived a strong drive to bring back the forest cover by the provincial government, non-governmental organizations, and local communities. In provincial land-use plans, significant amounts of land are designated for forest use, but most of this land is being used for large scale or subsistence agriculture. It is clearly challenging to convert agricultural land back to forest, especially in places that have a significant human population inside the designated protected areas. Strategies that focus on enforcing environmental protection usually either restrict forest use or relocate forest communities. Such restrictions and enforcements can exacerbate poverty in rural areas by limiting livelihood choices (McCay and Jones 2011, Groom and Palmer 2012).

Our results show that rural communities in or near protected areas do not necessarily have lower indicators of socioeconomic well-being, in agreement with a previous study by Ferraro et al. (2011). A more qualitative evaluation of living conditions via interviews revealed that protected area management in Patag limited original livelihoods of hunting and logging, but also created new livelihoods by employing former hunters and loggers as the local forest guards and allowing people to collect non-timber forest products. This strategy appears to minimize destructive forest while at the same time resulted in largely positive valuation of forest resources by local communities. Even though we did not formally evaluate how this affects the culture and traditions of the community, presently the traditional knowledge of local plants and animals is being utilized and maintained.

Our results show that urban and rural communities differ in socioeconomic status with more investment in services, infrastructure and commercial agriculture in urban areas. Furthermore, socioeconomic status is greater in areas that have commercial agriculture than in mostly forested and mosaic vegetation in municipal and barangay level. Even though agriculture related occupation are indicators of poverty in the Philippines (Reyes et al. 2012), in Negros Occidental, large-scale agriculture areas are located closer to urban centers and in the lowlands that makes them accessible to markets, job opportunities, and other sources of income such as fishing. There is more government investment in infrastructure and services in large-scale agricultural areas as these places also tend to bring more income from taxes then subsistence based areas. Most of these areas are dominated by large sugarcane plantations, owned by one family and provide seasonal, intermittent employment with inhabitants in need of finding other ways of subsistence during low seasons for sugarcane collection (Diprose and McGregor 2009). The statistics do not reflect the income disparity between plantation owners, people with permanent employment and sugarcane workers who are forced to look for alternative options in low season.

Further, our results show that rural communities in or near protected areas experience higher investment in infrastructure and services, show higher elementary school enrollment rates and provide more area per capita for subsistence farming than communities farther away from protected areas. Our study shows that protected areas do not necessarily cause or increase poverty for surrounding communities, in agreement with previous study by Ferraro et al. (2011). Therefore, there is evidence that NIPAS and CBFMP state programs are increasing livelihood conditions in rural communities near protected areas, even if the funding for these programs are fairly limited (Cagalanan 2013). Other studies have also shown that protected areas have alleviated poverty in forest reserves in Thailand and Costa Rica (Andam et al. 2010) and in long-term marine protected areas in Indonesia (Gurney et al. 2014). In all cases, the positive effects were associated with increased diversification of livelihoods as a result of government programs. A recent report by the International Union of Forest Research Organizations highlights the pivotal role that the forest resources play by providing alternative food sources and ecosystem services (IUFRO 2015). We therefore conclude that forest protection does not necessarily has to conflict with resource needs by forest dependent communities.

4.5.2. Guidelines for using secondary socioeconomic data to assess well-being in protected areas

Secondary data to assess the well-being of rural communities offers an efficient and cost-effective way to explore socioeconomic status over large areas. A socioeconomic account of the entire region is useful to observe general trends and associations of land use planning, actual land use, and socioeconomic well-being. However, additional more qualitative local information can complement the quantitative analysis by identifying the cause of the established relationships between protected areas and socioeconomic benefits or costs to the community using more in-depth household surveys and qualitative questionnaire methods (McElwee 2008). Focusing on census data could be a first step in a multi-staged approach that includes household surveys in targeted locations to establish the mechanisms by which protected area management may influence the socioeconomic status of forest-dependent communities.

One limitation of census data is related to issues of availability. In our study, employment records were not available for municipality or barangay level. Also poverty data were not available for the barangay level analysis and other data must be substituted to infer socioeconomic well-being. For example, poverty was highly correlated with head of households that didn't have school education, therefore enrollment in school is associated with more affluent families (Reyes et al. 2012). In fact, poverty indicators only reflect economic well-being in terms of earnings and are not sufficient in subsistence based communities where most of the population does not have steady employment or income. Distance of communities from administrative centers is an indicator of remoteness and is usually correlated with poverty as higher travel costs mean lower return on agriculture and forest products. On the other hand, remoteness is usually positively associated with income from tourism activities. As a result, the relationship between poverty and distance is not linear as shown in a study by Ferraro et al. (2011). Child malnutrition has a more straightforward positive relationship with poverty (Reyes et al. 2012). In summary, indicators of socioeconomic well-being need to be interpreted carefully and need to be viewed through analysis at different scales to understand both large scale associations of land use planning and socioeconomic well-being and the underlying causes of the relationships.

The results of this study offer several conclusions about protected areas management and poverty alleviation. First, ecological and socioeconomic goals can be achieved in protected areas management when community livelihoods are incorporated into planning. In particular, balance between curbing destructive activities while allowing non-destructive cultural use of the forest area and investment in infrastructure seems to encourage conservation while alleviating poverty conditions. Second, a variety of secondary socioeconomic data can show general status of forest communities at a large scale. In conclusions, this study offers conservation planners insight on using census data to evaluate community well-being.

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Table 4-1. Planned land use versus actual land cover for Negros Occidental. Refer to Fig. 4-2 for planned land use and actual land cover in Negros Occidental. Planned land use was derived from the Negros Occidental Social and Economic Trends 2009 report (PPDO 2011). Actual land cover classes were derived from 300 meter resolution Envisat MERIS remotely sensed data (Arino et al. 2012). See the methods section on land cover extraction and modifications.

	Actual land cover %				
Planned land use	Forest (13.2%)	Crops (27.8%)	Mosaic vegetation (57.7%)	Mangrove (0.2%)	Artificial surfaces (0.1%)
Forest land (30%)	21.1	20.5	58.5	0.00	0.00
Protected area (9%)	38.8	15.6	45.6	0.00	0.00
Sugarcane production (36%)	7.50	29.9	62.5	0.09	0.00
Agriculture and rice (21%)	1.86	36.2	61.7	0.19	0.00
Fishpond/mangrove (2%)	10.8	44.0	37.9	4.00	3.42
Urban area (2%)	0.37	47.9	48.3	0.43	2.98

Table 4-2. Correlation analysis between socioeconomic indicators and actual land cover in Negros Occidental. Spearman rank correlation coefficient and p-value are given for the actual land cover (Fig. 4-2, right) for each municipality (n=32) in Negros Occidental and socioeconomic indicators (see Fig 4-3 for visualization of correlations). The significance of the correlation was adjusted for multiple inferences using the Holm method within land cover groups.

	Actual land cover (%)				
Socioeconomic indicators	Forest	Crop	Mosaic vegetation	Mangrove	Artificial surface
Poverty rate 2003	0.16	-0.45~	0.57*	-0.12	-0.31
Homes with electricity 2003	-0.32	0.61**	-0.67***	0.16	0.41
Homes with sanitation 2003	-0.23	0.61**	-0.66***	0.05	0.37
Homes with potable water 2003	0.02	0.11	-0.31	0.38	0.02
Child malnutrition 2003	-0.08	-0.21	0.43~	-0.21	-0.15
Crime rate 2010	-0.16	0.01	-0.10	0.02	-0.15
No. health stations/ person	-0.46~	0.45~	-0.29	0.09	0.31
No. elem. schools/ person	0.10	-0.34	0.37	-0.03	0.11
Elem. school enrolment 2007	0.12	-0.47~	0.56*	-0.07	-0.02

~ $p<0.1$, * $p<0.05$, ** $p<0.01$, *** $p<0.001$

Table 4-3. Comparison of actual land use in barangays located the North Negros Natural Park and the ones outside the protected forest area. The mean and standard error (se) are given for rural barangays that are partially or fully located within a protected forest area (protected area, Fig. 4-2), all other rural barangays outside of protected area (other rural) and urban barangays (Goslee and Urban).

	Planned land use		
Actual land cover	Protected area mean (sd) (n=47)	Other rural mean (sd) (n=135)	Urban mean (sd) (n=53)
Population density (people/km^2)	2.45 (2.84)	22.0 (90.0)	111.5 (141.6)
Distance from municipal center (km)	20.6 (9.74)	8.39 (7.25)	1.75 (3.49)
Forest cover (%)	15.1 (14.8)	5.03 (8.86)	5.3 (12.77)
Crops cover (%)	0.25 (0.81)	33.4 (38.1)	31.2 (44.2)
Mosaic vegetation cover (%)	57.9 (15.9)	58.2 (25.4)	31.3 (31.2)
Mangrove cover (%)	0.00 (0.00)	4.84 (13.9)	58.3 (56.7)
Artificial surface cover (%)	0.06 (0.19)	1.13 (1.89)	7.12 (9.62)
Total area (100 HA)	28.5 (21.6)	11.3 (12.0)	2.86 (4.99)

Table 4-4. Comparison of socioeconomic indicators of barangays located the North Negros Natural Park and the ones outside the protected forest area. The mean and standard error (se) are given for rural barangays that are partially or fully located within a protected forest area (protected area, Fig. 4-2), all other rural barangays outside of protected area (other rural) and urban barangays (Goslee and Urban). Permutational analysis of variance was performed for the three groups of barangays and followed up with pair-wise Wilcoxon rank sum tests to identify significant differences between socioeconomic indicators (marked with different letters). The significance of the tests was adjusted for multiple inferences using the Holm method.

	Planned land use			
Socioeconomic indicators	Protected area mean (se) (n=47)	Other rural mean (se) (n=135)	Urban mean (se) (n=53)	p-value (perm. Anova)
% Child malnutrition	8.54 (0.89) a	7.83 (0.52) a	5.95 (0.83) a	0.1776
% Crime against women & children	0.09 (0.02) a	0.12 (0.02) a	0.28 (0.05) b	<0.0001
% Registered voters	51.2 (1.20) a	56.5 (1.11) b	62.4 (1.66) c	<0.0001
No. precincts/person *1000	2.90 (0.13) a	3.31 (0.11) a	3.87 (0.17) b	<0.0001
No. daycare/person*1000	1.00 (0.07) a	0.84 (0.06) b	0.42 (0.04) c	<0.0001
No. markets/ person *1000	0.08 (0.02) a	0.05 (0.01) a	0.06 (0.02) a	0.8150
No. health stations/person*1000	0.37 (0.04) a	0.35 (0.02) a	0.31 (0.04) a	0.8150
No. elem. schools/person *1000	0.59 (0.04) a	0.44 (0.03) b	0.25 (0.08) c	<0.0001
Student teacher ratio (elem. sch.)	37.3 (1.58) a	29.6 (1.48) b	15.7 (2.29) c	<0.0001
% Elem. school enrolment	18.7 (0.96) a	15.0 (1.55) b	21.3 (7.19) c	0.0489

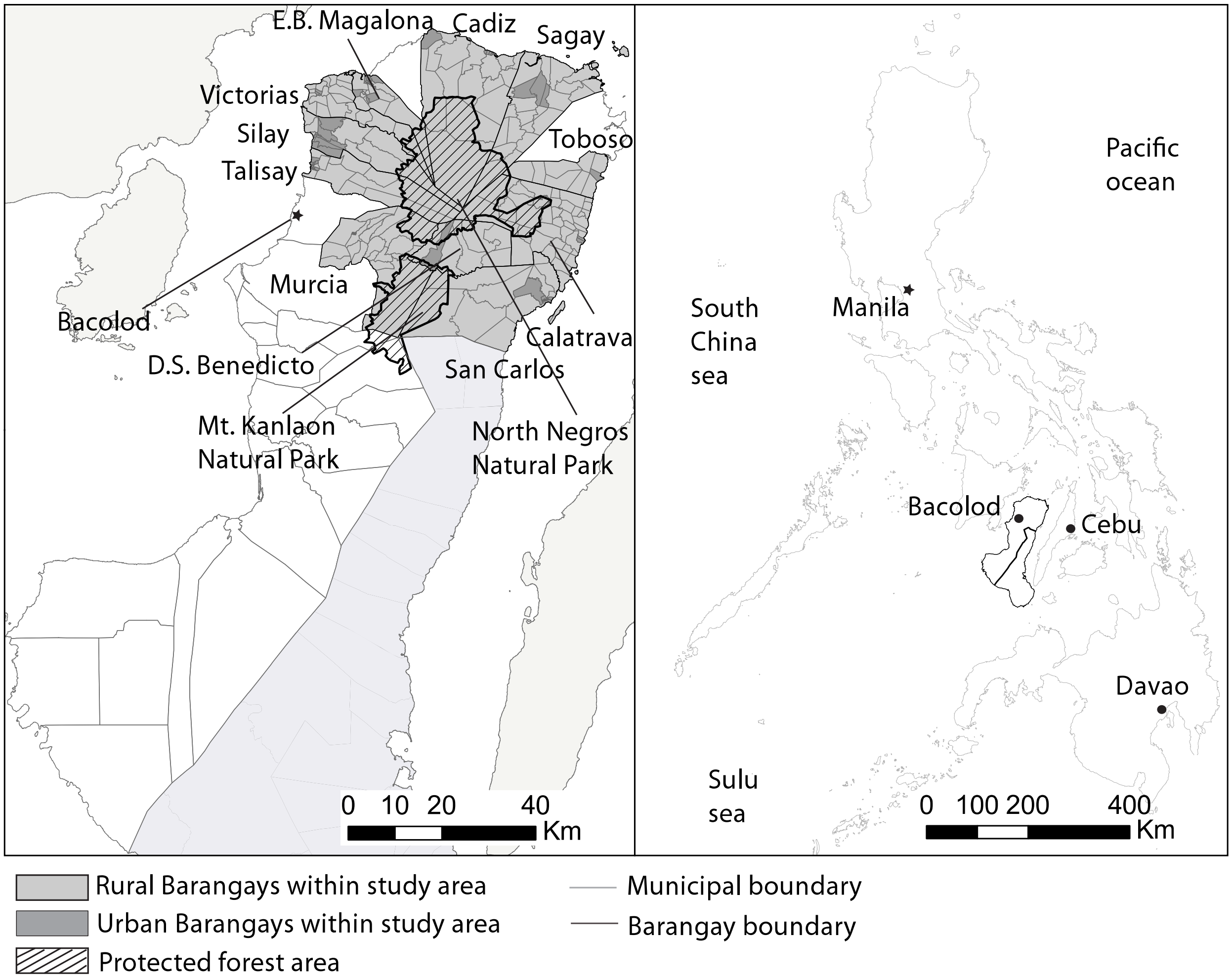

Figure 4-1. Municipalities and barnagays (villages) selected for analysis in this study. Thirty two municipalities of Negros Occidental, the northwestern part of Negros island were investigated (municipal boundaries are outlined in black). More detailed data was collected for 285 barangays of Northern Negros (shaded in gray), that surround North Negros Natural Park and Mt. Kanlaon Natural Park.

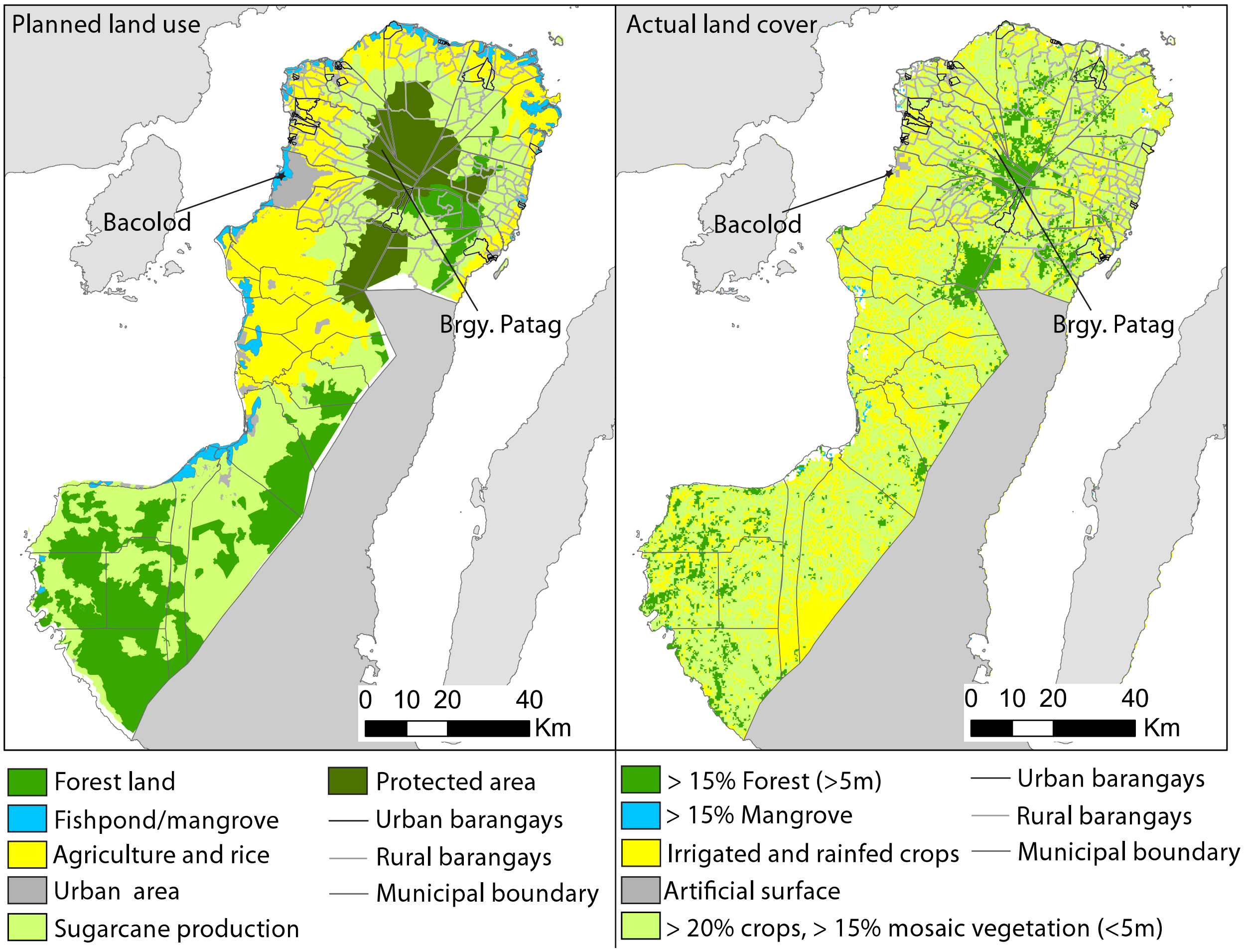

Figure 4-2. Planned land use (left) versus actual land cover (right) for the province of Negros Occidental, Philippines. The land use plan is according to the Negros Occidental Social and Economic Trends Report 2009 (PPDO 2011) and land cover was determined from 300m resolution Envisat MERIS remote sensing data (Arino et al. 2012)

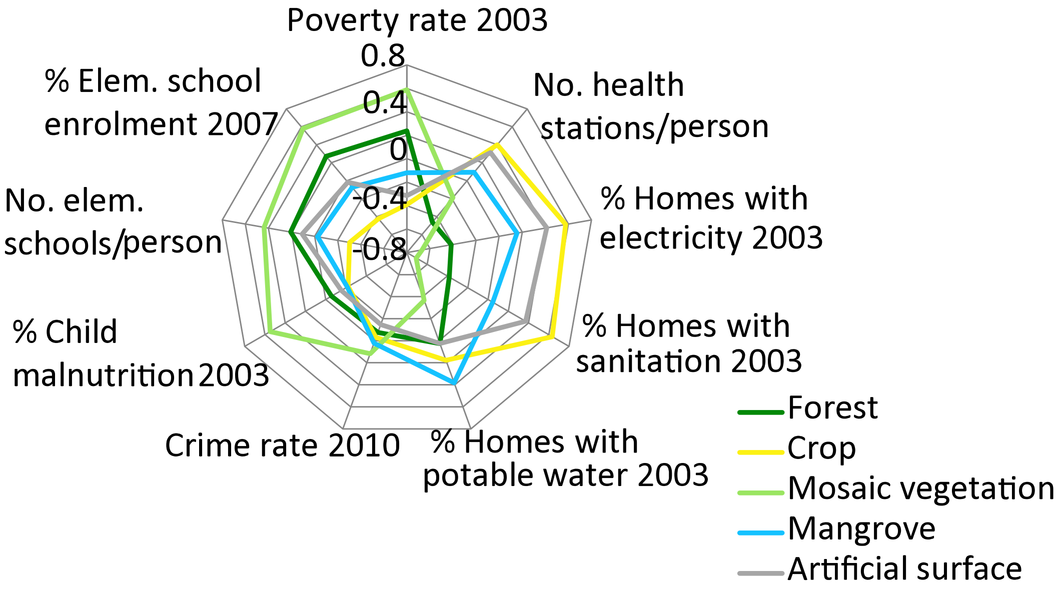

Figure 4-3. Correlation analysis between socioeconomic indicators and remotely sensed land cover percent across all municipalities of Negros Occidental (32 municipalities). Positive Spearman rank correlation coefficients indicate positive associations between land cover and socioeconomic indicators.

Chapter 5 - Conclusions

The objective of this PhD thesis was to improve tropical reforestation initiatives by addressing three major issues concerning recovery of dipterocarp forests in Southeast Asia. The first is the unique biology of the dipterocarps. The unpredictability of mass flowering events and recalcitrant seeds pose a serious problem to forest managers as mobilization of resources and people for large scale seed collection activities require significant time for preparation. The second issue with tropical forests in Southeast Asia is the lack of silvicultural knowledge of other native species that may be used for reforestation. The majority of the government reforestation and community forestry projects use exotic species that have few ecological and socioeconomic values. The third issue is that conservation and reforestation efforts have to include local communities, which are mostly poor and rely at least in part on forest resources for their livelihood. Community forestry programs often provide benefits only to a small group of the targeted community while protected areas can restrict use of forest resources. Understanding how protected area planning and land use planning affects local communities is therefore also important to effectively protect local resources.

Revisiting my original research questions outlined in the introductory chapter of my thesis work contributed the following knowledge to sustainable forest management, and to increasing ecological and socioeconomic benefits to local communities:

1. What are the environmental factors that control mass flowering in dipterocarp forests?

Using flowering and fruiting data from 11 sites and regions in Southeast Asia covering a time period of 53 years, I find that there are no environmental triggers of dipterocarp mass flowering common to all sites and regions. Model fit using environmental variables (precipitation, temperature, cloud cover, vapor pressure and El Niño) shows inconsistencies among regions, with precipitation and El Niño showing better predictability in Borneo sites and regions and temperature in Peninsular Malaysian sites and regions. El Niño is strongly associated with climate in Borneo, with the relationship weakening in Peninsular Malaysia. Further investigation showed that El Niño has variable effect on local climate by its interaction with the monsoons. South Pacific facing regions of Borneo experience warmer temperatures and drier weather from August to December because of hot air masses being driven into the region during the summer monsoon. On the contrary, the north and east-facing regions of Northern Borneo and Peninsular Malaysia see the strongest influences of El Niño phases during the winter monsoon causing reduced precipitation from January to April.

2. Can resource accumulation models successfully predict flowering events in dipterocarp forests?

Hypothesizing that resources are related to interannual variation of precipitation and El Niño, I test models that use climate variables as proxy for resources which accumulated over time to induce flowering in dipterocarps. The resource accumulation models show better predictive ability than environmental trigger models. To further explain interspecific flowering synchronization, this chapter explores a theoretical model with added parameters that include a resource baseline (rb), a resource anomaly (as) that varies on interval of 1 to 12 years (ai). The anomaly is modeled closely to the cycles of El Niño/La Nina that also occur irregularly every 1 to 12 years with an average of 7. The models show that a large anomaly is able to synchronize flowering in a community of trees of the same species independent of a flowering threshold and among related species that have different flowering thresholds. It seems likely for many related species of trees to evolve a response to resource availability rather than the same climate trigger independently and offers a possibility of predicting future flowering events.

3. Can easy to collect tree measurements from ecological plots be used to identify native species that are ecologically suitable for reforestation under open field conditions in order to bypass lengthy species test plantations?

The research in this chapter shows that easy to collect measurements such as tree frequency collected in regenerating and mature forest stands and seed size can be used successfully to classify species into general successional categories. This could be an important first step in narrowing down the number of species to be tested in provenance trails for growth rates on sites of different degree of degradation and provides an easily transferable technique. Identifying native species for reforestation is crucial for projects that focus on restoring ecological functions and habitat for native fauna. On the other hand, projects that want to incorporate local communities need to select species that can be useful to forest people and is the second goal of this chapter: This research highlighted superior ecological and economic importance of long lived old-successional trees that are more difficult to establish. Further research focusing on artificial propagation will be useful for regeneration of these species.

4. Can we identify species that have both high ecological suitability for reforestation and high value for forest dependent communities?

Using local knowledge of forest communities about native species, I found that the trees that grow in the mature forest are mostly preferred by the people for having more uses. The implication of this finding suggests that it is difficult to find species that have ecological suitability for forest regeneration and provide uses for communities. Therefore, compromises in forestry programs need to be made. This finding also implies that mature tropical forests are not easily renewable and protection of remaining natural forests should not be ignored in land use planning.

However, managing protected areas requires limiting access of forest communities to resources. Forest communities are generally the poorest and inhibitive land use planning further limits their livelihood options.

5. What effect do protected areas have on socioeconomic status of communities that live within or nearby areas with restrictions on resource extraction?

The analysis of census socioeconomic indicators and land use in this chapter finds that protected areas are not necessarily poverty traps. In this case study, rural communities around the protected area have higher socioeconomic status than communities in the lowlands that only rely on agriculture. Communities in the forest reserve have more natural resources and have better opportunities to supplement their livelihood by collecting forest products. There is also more government investment in infrastructure and services in communities around protected areas. This research shows that census data is useful for looking at entire regions around a protected forest to observe trends that are not accessible through other methods such as household surveys.

The research chapters cover different issues that plague reforestation success in Southeast Asia but with a common aim, to aid reforestation activities in the tropics with ecologic and socioeconomic knowledge. In particular, predicting mass flowering can help forest managers in seed collection efforts that are vital for reforestation with dipterocarps as seed propagation and storage are not viable options. Next, selecting species that are well suited for degraded environments can increase forest programs that use native species. Further, I outline the importance of selecting species that are useful for forest communities and the need for compromises between ecologic and socioeconomic factors when considering reforestation aims. Finally, land use planning and protected areas management need to incorporate socioeconomic conditions of forest dependent communities.

Further investigation should focus on building predictive models of dipterocarp mass flowering using site specific climate and flowering data. Using climate projections data would be the next step in predicting future mass flowering events and providing management recommendations of dipterocarp forests. Based on the results of the second chapter, the next steps would be to test growth rates of selected native species in provenance trails on degraded sites. Finally, a follow up to the results of the third chapter would focus on selecting several communities in the forest reserve and conducting household surveys to gain more insight into the impact that a protected area has on the community.

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Appendix A - Flowering and fruiting data from dipterocarp forests in several long-term ecological monitoring plots and regions in Southeast Asia. See notes and references corresponding to the numbers in the table heading below.

	Long-term ecological monitoring plots									Regions							
	Pasoh Forest Reserve Area					Ketambe	Barito Ulu	Lambir Hills	Gunung Palung	Kalimantan		Sarawak	W. Peninsular Malaysia	West Borneo		E. Peninsular Malaysia	East Borneo
Year	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17
1950											0	0	0	0		0	0
1951											0	0	0	0		2	2
1952											0.2	0	0	0		0	1
1953											0.6	2.8	0	2		2	0
1954											28	16	0	0		0	1
1955											0.2	1.5	1	0		1	2
1956											0.3	0.2	0	0		0	0
1957											0	0	2	1		2	0
1958											6.5	6.2	2	2		0	2
1959											23	22	0	0		0	1
1960											0.6	0	0	0		1	1
1961											0.1	0	0	1		0	0
1962											13	20	0	0		0	0
1963											0	0	1	0		1	1
1964											0	0	0	0		0	0
1965											0	0.5	0	1		0	0
1966											0	6.8	0	0		0	0
1967											0	0	0	2		0	1
1968										18	21	16	2	0	1	2	2
1969										2	0.1	0	0	2	0	0	0
1970										5	4.6	17	0	0	0	0	0
1971						0				2	1.2	0	0	0	0	0	0
1972						2				0	0	0	0	2	2	0	1
1973	3	6	2	0		0				18	17	28	0	0	0	0	1
1974	9	13	8	0		0				2	3.5	0	0	0	0	0	0
1975	2	0	1	0		0				1	0.9	0	0	0	2	0	0
1976	95	97	69	57	55	0				2	1.5	0.1	2	2	1	2	2
1977	0	6	5	2	0	0				7.5	6.5	4.8	0	0	0	0	1
1978	2	0	13	0	0	0				0	0	0.1	0	0	0	0	0
1979	2	0	7	0	0	2				0	0.1	0.4	0	2	0	0	0
1980	18	16	27	2	0	0				1	9.5	11	0	1	0	0	0
1981	24	45	72	69	65	2				6	16	1.6	2	2	1	0	2
1982	5	0	1	0	0	0				2	1.1	23	0	1	2	2	0
1983	82	18	24	14	15	0				11	17	1.5	1	0	0	0	2
1984					0	2				0	0	0	0		0		
1985					43	0				1	0.1	0.2	0		0		
1986					13	0				2	1.5	8.4	0		2		
1987					23	0			38	16	40	23	2		0		
1988					0	2			0	0	3.5	0					
1989					Mass	0			0	0	0.2	0					
1990					0	0	20		3	0	35	0					
1991					0	0	48		35	14							
1992					0	2	2		0	0							
1993					0	0	5	3	0	0							
1994					0	0	37	3	2	0							
1995					0	2	5	4	5	5							
1996					Mass	0	5	18	0	0							
1997					0	0	42	12	0	0							
1998					0	2	12	15	35	--							
1999					10		12	6	0	0							
2000					0			4									
2001					24			16									
2002					5			3									

Appendix A - continued from previous page.

Notes: Several site and regional data came from multiple sources and were combined for the analysis. Other sites close in proximity were also combined. Sites 1 through 5 were combined to form Pasoh Forest Reserve Area by averaging the flowering data by years. The same applies for Kalimantan (region 10-11) and West Borneo (region 14-15).

References: The numbers correspond to the table heading above.

1. Gombak Forest Reserve (Pas), Selangor, Malaysia. % of 16 species of *Shorea* trees flowering. Yap & Chan 1990, Fig. 3.3
2. Ampang Forest Reserve (Pas), Selangor, Malaysia. % of 16 species of *Shorea* trees flowering. Yap & Chan 1990, Fig. 3.3
3. Kepong Forest Research Site (Pas), Selangor, Malaysia. % of 16 species of *Shorea* trees flowering. Yap & Chan 1990, Fig. 3.3
4. Pasoh Forest Reserve (Pas), Negri Sembilan, Malaysia. % of 16 species of *Shorea* trees flowering. Yap & Chan 1990, Fig. 3.3
5. Pasoh Forest Reserve (Pas), Negri Sembilan, Malaysia. % of community wide dipterocarp trees flowering. Numata *et al*. 2003, Table 1
6. Ketambe site in the Gunung Leuser National Park (Keta), northern Sumatra, Indonesia. Community wide masting based on fruit fall on trial (no, major) converted to a scale of 0-2 for no and major events. Wich & van Schaik 2000, Table 2
7. Barito Ulu Research Area (BU), central Kalimantan, Indonesia. % of trees of 22 dipterocarps species flowering. Brearley *et al*. 2007, Fig. 4a
8. Lambir Hills National Park (LH), Sarawak, Malaysia. % of 40 species of dipterocarp trees flowering. Sakai *et al*. 2006, Fig. 1a
9. Gunung Palung National Park (GPNP), Kalimantan, Indonesia. Seed fall density (seeds/m^2). Curran *et al*. 1999, Fig. 1a
10. Kalimantan (Kal), Borneo, Indonesia. Sees export (10^6 kg/ year) of trees from the *Shorea* species. Curran *et al*. 1999, Fig. 1a
11. Kalimantan (Kal), Borneo, Indonesia. Seed export (10^6 kg/ year) of 18 species of *Shorea*. Blicher-Mathiesen 1994, Table 4
12. Sarawak (Sar), Borneo, Malaysia. Seed export (10^6 kg/ year) of 18 species of *Shorea*. Blicher-Mathiesen 1994, Table 4
13. West Peninsular Malaysia (WPM). Community wide mass flowering recorded by the forestry departments (no, minor, major flowering) converted to a scale of 0-2 for no, minor, major. Ashton *et al*. 1988, table 4, Wich & van Schaik 2000, Table 2
14. West Borneo (WB). Community wide mast fruiting based on seed export data (no, minor, major masting) converted to a scale of 0-2 for no, minor and major events, adjusted to reflect flowering a year prior. Ashton *et al*. 1988, Table 4
15. West Borneo (WB). Community wide mast fruiting based on seed export data (no, minor, major masting) converted to a scale of 0-2 for no, major and minor events, adjusted to reflect flowering times a year prior. Wich & van Schaik 2000, Table 2
16. East Peninsular Malaysia (EPM). Community wide mass flowering recorded by the forestry departments (no, minor, major flowering) converted to a scale of 0-2 for no, minor, major events. Ashton *et al*. 1988, Table 4
17. East Borneo (EB). Community wide mass flowering recorded by the forestry departments (no, minor, major flowering) converted to a scale of 0-2 for no, minor, major events. Ashton *et al*. 1988, Table 4

Appendix B - R program to simulate an extended resource tracking model with more parameters than the basic model used for evaluation of a resource tracking mechanism to trigger mass flowering. With the program below we explore if resource tracking can synchronize individual trees that track resources in different inter-annual cycles through the introduction of an irregular resource anomaly (equivalent to the El Niño Southern Oscillation). The code below can be pasted as a whole into the console window of the R programming environment, an open source software package available at: *http://www.r-project.org*.

```
### Variables that can be defined by the user:
ny <- 120    # Number of years (try 100 to 500)
nt <- 12     # Number of trees (try 5 to 20)
rb <- 20     # Resource baseline with random inter-annual variability
as <- 40     # Maximum anomaly size (try 2x to 5x the resource baseline value)
ai <- 12     # Maximum anomaly interval (try 5 to 15 years)
ft <- 80     # Resource threshold for flowering (try 2x to 5x the anomaly size value)
### Explanation of variables used below but not controlled by the user:
# i -  Counter for years
# j -  Counter for trees
# r -  Resource
# a -  Anomaly
# fm - Matrix that records flowering of each tree (years in rows, trees in columns)
# rm – Matrix that accumulates resources for each tree (years in rows, trees in
# columns)

### Create a table of a randomly varying resource as specified by rb, as, and ai above
r <- data.frame(y=1:ny, r=1:ny)
r$r <- (sample(1:rb, ny, replace=T))
y <- 1; while (y < ny-ai) { y <- y+sample(1:ai,1); r[y,2] <- r$r[y]+sample(1:as,1) }
r[1:20,]   # Outputs the first 20 rows of the resource data table for checking

### Create a an empty plot area to fill with results
graphics.off()
par(mar=c(0,0,0,0), oma=c(3,1,1,0), mfrow=c(nt+2,1))

### Resource tracking model with two loops that repeat the calculations for each tree
### (counter i), and for each year (counter j) and plots the resource status (rp) and
### the flowering events (fp) into a graphics window.
rm <- as.data.frame(matrix(nrow=ny, ncol=nt))
fm <- as.data.frame(matrix(nrow=ny, ncol=nt))

for (i in 1:nt) {
   for (j in i:ny){
      rm[i,i] <- 0
      rm[j+1,i] <- ifelse(rm[j,i]<ft,((r[j+1,2])+rm[j,i]),0)
      }
   rm <- rm[1:ny,]
   plot(rm[,i]~r$y,type="l", axes=F, ann=F, xlim=c(-15,ny))
   abline(v=seq(0,ny,ny/5),lty=2)
   text(-10,ft/2,paste("Tree",i))
   fm[,i] <- ifelse(rm[,i]>=ft,1,0)
   }
fm[is.na(fm)] <- 0    # replaces missing values with zeros for graphs
### plots the frequency of flowering events across all trees
plot(rowSums(fm[,1:nt])~r$y,type="l", lwd= 2, axes= F, ann=F, xlim=c(-15,ny))
abline(v=seq(0,ny,ny/5), lty=2)
text(-9,nt/2,"Flowering")

### plots the randomly varying resource for reference
plot(r$r~r$y, xlim=c(-15,ny), ylim=c(0,max(r$r)), axes=F,  ann=F, type="l")
text(-9,(as+rb)/2,"Resource")
axis(1, at=seq(0,ny,ny/5))
abline(v=seq(0,ny,ny/5),lty=2)
```

Appendix C - R program to simulate a multi-species extended resource tracking model with more parameters than the basic model used for evaluation of a resource tracking mechanism to trigger mass flowering. With the program below we explore if resource tracking can synchronize individual trees of different species that track resources in inter-annual cycle through the introduction of an irregular resource anomaly (equivalent to the El Niño Southern Oscillation). The code below can be pasted as a whole into the console window of the R programming environment, an open source software package available at: *http://www.r-project.org*.

```
### Variables that can be defined by the user:
ns <- 5      # Number of Species (try 2 to 10)
ny <- 120    # Number of years (try 100 to 200)
nt <- 12     # Number of trees (try 5 to 20)
rb <- 20     # Resource baseline with random inter-annual variability
as <- 40     # Maximum anomaly size (try 2x to 5x the resource baseline value)
ai <- 12     # Maximum anomaly interval (try 5 to 15 years)
ftv <- c(70,71,72,120,122)  # Vector of flowering thresholds (length must be ns)
### For additional explanations of variables not set by the user, see Appendix B.
### Create a table of a randomly varying resource as specified by rb, as, and ai above
r <- data.frame(y=1:ny, r=1:ny)
r$r <- (sample(1:rb, ny, replace=T))
y <- 1; while (y < ny-ai) { y <- y+sample(1:ai,1); r[y,2] <- r$r[y]+sample(1:as,1) }
r[1:20,]   # Prints the first 20 rows of the resource data table
fs=as.data.frame(matrix(nrow=ny, ncol=ns+1))
### Create a an empty plot area to fill with results
graphics.off()
par(mar=c(0,0,0,0), oma=c(3,1,1,0), mfrow=c(ns+2,1))
### Resource tracking model with two loops that repeat the calculations for each species
### (counter k), tree within species (counter i), and for each year (counter j) and plots
### the resource status (rp) and the flowering events (fp) into a graphics window.
for (k in 1:ns) {
    rm <- as.data.frame(matrix(nrow=ny, ncol=nt))
    fm <- as.data.frame(matrix(nrow=ny, ncol=nt))

    for (i in 1:nt){
         for (j in i:ny) {
               rm[i,i] <- 0
               rm[j+1,i] <- ifelse(rm[j,i]<ftv[k],((r[j+1,2])+rm[j,i]),0)
               }
       rm <- rm[1:ny,]
       fm[,i] <- ifelse(rm[,i]>=ftv[k],1,0)
       }

    fm[is.na(fm)] <- 0  # replaces missing values with zeros for graphs

    ### plot the frequency of flowering events across all trees
    plot(rowSums(fm[,1:nt])~r$y,type="l", col=rainbow(ns)[k], lwd=2,
         axes=F, ann=F, xlim=c(-15,ny))
    abline(v=seq(0,ny,ny/5),lty=2)
    text(-9,nt/(3+ftv[k]/max(ftv)),paste("Species",k))
    fs[,k]<-rowSums(fm[,1:nt])
    }
### plot the combined tree population of all species
xpos=barplot(t(fs[,c(1:(ns))]),col="white",border="white", axes=F,  ann=F,yaxt="n")
par(new=T)
x=xpos[2]-xpos[1]
barplot(t(fs[,c(1:(ns))]),col=rainbow(ns), axes=F,  ann=F,yaxt="n", xlim=c((-15*1.2+0.1),
xpos[ny]+0.6))
abline(v=seq(xpos[1]-0.6,xpos[ny]+0.6,(xpos[ny]-xpos[1]+1.2)/5),lty=2)
text(-9*1.2,(ns*nt)/3.5,"General")
text(-9*1.2,(ns*nt)/5,"Flowering")
### plots the randomly varying resource for reference
plot(r$r~r$y, xlim=c(-15,ny), ylim=c(0,max(r$r)), axes=F,  ann=F, type="l")
axis(1, at=seq(0,ny,ny/5))
abline(v=seq(0,ny,ny/5),lty=2)
text(-9,(as+rb)/2,"Resource")
```

Appendix D - Local uses of native tree species in the North Negros Natural Park. The table summarizes complete data from the questionnaires with 47 forest guards that listed species of socioeconomic and importance and their uses. The summary includes statistics on the total uses for each species (Tot), number of respondents that mention this species (N) and rank that combines the number of uses and the number of respondents. Local names may refer to two or more related species, and may therefore be listed more than once.

				Uses[1]																	Statistics		
Species	Family	Local Name	Native	Lum	Con	Fur	Pos	Var	Re	Rub	Sol	Cha	Fire	Food	Bat	Bird	Pig	Mon	Buf	Med	Tot	N	Rank
Agathis philippinensis	Araucariaceae	Almaciga	Yes	1	0	1	0	1	1	0	1	1	0	0	0	0	0	0	0	0	6	37	43
Swietenia mahogani	Meliaceae	Mahogany	No	1	1	1	0	0	0	0	0	1	1	0	0	0	0	0	0	0	5	30	35
Shorea negrosensis	Dipterocarpaceae	W. Laua-an	Yes	1	1	1	1	1	0	0	0	0	1	0	0	0	0	0	0	0	6	26	32
Pterocarpus indicus	Fabaceae	Narra	Nat	1	1	1	0	0	0	0	0	0	0	0	0	1	0	0	0	0	4	24	28
Gmelina arborea	Lamiaceae	Gemelina	No	1	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	3	22	25
Palaquium luzoniensis	Sapotaceae	Nato	Yes	1	1	1	1	0	0	0	0	0	0	1	0	0	0	0	0	0	5	20	25
Shorea contorta	Dipterocarpaceae	R. Laua-an	Yes	1	1	1	1	0	0	0	0	1	0	0	0	0	0	0	0	1	6	17	23
Leucaena glauca/I.bijuga	Fabaceae	Ipil-Ipil	No	0	1	0	0	0	0	0	0	1	1	0	0	1	0	0	1	0	5	17	22
Litsea philippinensis	Lauraceae	Bakan	Yes	1	1	1	0	0	0	0	0	0	1	1	0	1	0	0	0	0	6	14	20
Garcinia brevirostris/sp.	Guttiferae	Odling	Yes	1	1	1	1	0	0	0	0	0	0	0	1	0	0	0	0	0	5	15	20
Syzygium gracile/sp.	Myrtaceae	Odling	Yes	1	1	1	1	0	0	0	0	0	0	0	1	0	0	0	0	0	5	15	20
Cinnamomum mercadoi	Lauraceae	Kaningag	Yes	1	1	1	0	0	0	0	0	1	1	1	0	0	0	0	0	1	7	12	19
Platea excels	Icainaceae	Panobul	Yes	0	0	0	0	0	0	0	0	0	0	1	0	1	1	1	0	1	5	11	16
Shorea polysperma	Dipterocarpaceae	Tangile	Yes	1	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	3	12	15
Prunus fragrans	Rosaceae	Bagosantol	Yes	1	0	1	0	0	0	0	0	0	0	1	0	1	1	0	0	0	5	9	14
Eucalyptus camaldulensis	Myrtaceae	Eukaliptus	No	1	1	1	1	0	0	0	0	1	0	0	0	0	0	0	0	1	6	8	14
Dillenia philippinesis	Dilleniaceae	Katmon	Yes	0	0	0	0	0	0	0	0	0	0	1	0	0	0	1	0	0	2	12	14
Lithocarpus sp.	Fagaceae	Lintakuban	Yes	0	1	0	0	0	0	0	0	0	0	1	0	1	1	0	0	1	5	8	13
Unidentified		Bogos	Un	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	1	11	12
Memexylon brachybotris	Melastomataceae	Haras	Yes	0	1	0	1	0	0	0	0	0	0	0	0	1	0	0	0	0	3	9	12
Memexylon sp.	Melastomataceae	Haras	Yes	0	1	0	1	0	0	0	0	0	0	0	0	1	0	0	0	0	3	9	12
Gliricidia sepium	Fabaceae	Madre cacao	No	0	1	1	0	0	0	0	0	1	1	0	0	0	0	0	1	0	5	7	12
Bischofia javanica	Euphorbiaceae	Tuog	Yes	0	1	1	0	0	0	0	0	0	0	0	1	0	0	0	0	0	3	8	11
Mansoa alliacea	Bignoniaceae	AhosAhos	No	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	9	10
Garcinia binucao/sp.	Guttiferae	Batuan	Yes	0	0	0	0	0	0	0	0	0	1	1	0	0	0	0	0	0	2	8	10
Artocarpus nitidus/cumi.	Moraceae	Kubi	Yes	0	1	1	1	0	0	0	0	0	0	1	0	0	0	0	0	1	5	5	10
Shorea almon	Dipterocarpaceae	Almon	Yes	1	1	1	0	0	0	0	0	1	0	0	0	0	0	0	0	0	4	5	9
Chisocheton cumingianus	Meliaceae	Balukanag	Yes	0	0	0	1	0	0	0	0	0	1	0	0	0	0	0	0	0	2	7	9
Litsea perottetii	Lauraceae	Baticoling	Yes	1	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	3	6	9
Arthrocarpus heterophyl.	Moraceae	Lanka	Yes	0	0	1	0	0	0	0	0	0	0	1	0	0	0	0	0	0	2	7	9
Ficus crysolepis/sp.	Moraceae	Lunok	Yes	0	0	0	0	0	0	0	0	0	0	0	0	1	1	1	0	0	3	6	9
Myrica esculenta	Myricaceae	Makadulon	Yes	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	1	8	9
Actinodaphne sp.	Lauraceae	Payong	Yes	0	0	0	0	0	0	0	0	1	1	0	0	0	0	0	1	0	3	6	9
Saribus rotundifolius	Arecaceae	Anahaw	Yes	0	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	6	8
Pometia pinnata	Sapindaceae	Ibu	Yes	1	0	0	0	0	0	0	0	0	1	1	0	0	0	0	0	1	4	4	8

Appendix D - continued from previous page

				Uses[1]																	Statistics		
Species	Family	Local Name	Native	Lum	Con	Fur	Pos	Var	Re	Rub	Sol	Cha	Fire	Food	Bat	Bird	Pig	Mon	Buf	Med	Tot	N	Rank
Acacia mangium/holo.	Fabaceae	Acacia	No	0	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	5	7
Dipterocarpus grandif.	Dipterocarpaceae	Apitong	Yes	1	1	0	0	0	0	0	0	1	1	0	0	0	0	0	0	0	4	3	7
Memexylon cumingii	Melastomataceae	Bulobayabas	Yes	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	1	6	7
Trema orientalis	Urticaceae	Hinagdong	Yes	0	0	1	0	0	0	0	0	0	0	0	0	1	0	0	0	0	2	5	7
Unidentified		Wild pine	Yes	0	0	1	0	0	0	0	0	0	1	0	0	0	0	0	0	0	2	5	7
Calliandra calothyrsus	Fabaceae	Calliandra	Yes	0	0	1	0	0	0	0	0	1	1	0	0	0	0	0	1	0	4	2	6
Ficus minahassae	Moraceae	Hagimit	No	0	0	1	0	0	0	0	0	0	1	0	0	0	0	0	0	0	2	4	6
Aglaia elliptica	Meliaceae	Lambunau	Yes	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	2	4	6
Triplaris cumingiana	Polygonaceae	Palusanto	No	0	0	0	1	0	0	0	0	1	1	0	0	0	0	0	0	0	3	3	6
Sandoricum koetjape	Meliaceae	Santol	Nat	1	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	4	6
Calameae sp.	Arecaceae	Uway	Yes	1	0	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	3	3	6
Pometia pinnata	Sapindaceae	Alauihau	Yes	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	4	5
Citrus microcarpa/mitis	Rutaceae	Kalamansi	Yes	0	0	1	0	0	0	0	0	0	0	1	0	0	0	0	0	0	2	3	5
Palaquium sp.	Sapotaceae	Laku-laku	Yes	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	4	5
unidentified		Takenis	Yes	0	0	0	0	0	0	0	0	0	1	0	0	1	0	0	0	0	2	3	5
Alphitonia excelsa	Rhamnaceae	Taulay	Yes	0	0	0	1	0	0	1	0	0	0	0	0	0	0	0	0	0	2	3	5
Litsea tomentosa	Lauraceae	Bonsliak	Yes	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	3	4
Ficus benjamina/sp.	Moraceae	Dalakit	Yes	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	1	3	4
unidentified		Malabaga	Un	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	3	4
Azadirachta indica	Meliaceae	Neem	No	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	2	2	4
unidentified		Ronidalia	No	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	1	3	4
Casuarina equisetlifolia	Casuarinaceae	Agoho	Nat	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	1	2	1	3
unidentified		Black Berry	No	0	0	0	0	0	0	0	0	0	1	0	0	1	0	0	0	0	2	1	3
unidentified		Black Virg	Yes	0	0	0	0	0	0	0	0	0	1	1	0	0	0	0	0	0	2	1	3
unidentified		Ero-Ero	Yes	0	0	1	0	0	0	0	0	0	0	0	0	1	0	0	0	0	2	1	3
Ficus septica	Moraceae	Labnog	Yes	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	1	2	3
Artocarpus odoratissimus	Moraceae	Marang	Yes	0	0	1	0	0	0	0	0	0	0	1	0	0	0	0	0	0	2	1	3
unidentified		Taraple	Un	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	1	2	3
Pariti tiliaceum	Malvaceae	Balibago	No	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	1	1	2
unidentified		Cansilay	Yes	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	1	2
Coffea arabica	Rubiaceae	Coffee	No	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	1	1	2
Durio zibethinus	Malvaceae	Durian	No	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	1	1	2
unidentified		Hamogni	Yes	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	2
Pittosporum moluccanum	Pittosporaceae	Malabuyo	Yes	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	1	1	2
Mangifera indica	Anacardiaceae	Mango	No	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	1	2	3
Nephelium lappaceum	Sapindaceae	Rambutan	Nat	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	1	1	2
Canarium villosum	Burseraceae	Salong	Yes	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	2

[1] Uses: Lum-Lumber, Con-Construction, Fur-Furniture, Pos-Polls/posts, Var-Varnish, Re-Resin, Rub-Rubber, Sol-Paint Solvent, Cha-Charcoal, Fir-Firewood, Food-Edible fruits/seed, Bat-Food for bats, Bird-Food/nests for birds, Pig-Food for wild pig, Mon-Food for monkeys, Buf-Food for water buffalo, Med-Medicine.