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Restoring Rainforest Fragments: Survival of Mixed-Native Species Seedlings under Contrasting Site Conditions in the Western Ghats, India

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Abstract

Historical fragmentation and a current annual deforestation rate of 1.2% in the Western Ghats biodiversity hotspot have resulted in a human-dominated landscape of plantations, agriculture, and developed areas, with embedded rainforest fragments that form biodiversity refuges and animal corridors. On private lands in the Anamalai hills, India, we established restoration sites within three rainforest fragments (5, 19, and 100 ha) representing varying levels of degradation such as open meadow, highly degraded sites with dense *Lantana camara* invasion, abandoned exotic tree plantations (*Eucalyptus grandis* and *Maesopsis eminii*), and sites with mixed-native and exotic tree canopy. Between 2000 and 2004, we planted annually during the southwest monsoon 7,538 nursery-raised seedlings of around 127 species in nine sites (0.15–1.0 ha). Seedlings monitored at 6-monthly intervals showed higher mortality over the dry season than the wet season and survival rates over a 2-year period of between 34.4 and 90.3% under different site conditions. Seedling survival was higher in sites with complete weed removal as against partial removal along planting lines and higher in open meadow and under shade than in sites that earlier had dense weed invasion. Of 44 species examined, survival across sites after 24 months for a majority of species (27 species, 61.4%) was higher than 50%. Retaining regenerating native species during weed clearing operations was crucial for rapid reestablishment of a first layer of canopy to shade out weeds and enhance survival of shade-tolerant rainforest seedlings.

Key words: Anamalai hills, *Eucalyptus*, fragmentation, *Lantana camara*, *Maesopsis*, plantations, tropical wet evergreen forest.

Introduction

Restoring degraded areas and promoting benign forms of land use around conservation areas are increasingly recognized as important conservation activities worldwide, especially in highly diverse tropical forests (Laurance & Bierregaard 1997; Young 2000; Society for Ecological Restoration International Science and Policy Working Group 2004). The tropical forests of the Western Ghats mountain range along the west coast of India is a case in point. This region is a biodiversity hotspot and a Global 200 ecoregion, with very high human population densities (Olson & Dinerstein 1998; Kumar et al. 2004). Less than 20% of the original tropical forest remains, most of it fragmented and degraded by agriculture, plantations, hydroelectric projects, logging, developmental activities, and forest produce exploitation. The current annual deforestation rate remains at 1.2% (Menon & Bawa 1997; Jha et al. 2000; Kumar et al. 2004).

Of particular concern in the Western Ghats is the effect of fragmentation and degradation on tropical rainforests that contain high diversity and endemic plant and animal taxa (Pascal 1988; Kumar et al. 2004). Chronic human extraction of fuelwood and forest products is known to degrade evergreen forests, converting them to more open, deciduous, or secondary vegetation (Daniels et al. 1995), and the concomitant fragmentation and disturbance may bring about substantial changes in vegetation structure and composition (Muthuramkumar et al. 2006). In the Anamalai hills, past studies have shown that rainforest fragments act as refuges for native plants and animals and as corridors for landscape-level connectivity between patches (Mudappa & Raman 2007). These studies demonstrate the need for restoration of rainforest in degraded sites for maintaining or enhancing their conservation values.

In tropical forests characterized by high biological diversity, restoration usually requires planting or encouraging regrowth of a diversity of native species. Although this poses challenges for seed collection, germination, and planting and may involve high costs, recent work indicates an encouraging potential for mixed-native species plantings for restoration of degraded tropical forests (Lamb et al. 2005). For example, there have been recent studies on seed germination, viability, growth, phenology, and suitability for use in tropical forest restoration of a number of tree species (Knowles & Parrotta 1995; Rai 1999;

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Stewart & Balcar 1999; Blakesley et al. 2002; Sautu et al. 2006). In addition, successful field trials are also being reported, such as Parrotta and Knowles (1999, 2001) who used around 70 tree species to restore tropical forests on bauxite-mined lands in Amazonia and others who have used a variety of tree species to restore tropical forest vegetation (Goosem & Tucker 1995; Tucker & Murphy 1997; Azani et al. 2001; Soudre et al. 2001; Elliot et al. 2003; Lamb et al. 2005).

Experience gained from tropical forest restoration projects around the world have shown that rainforest restoration can be influenced by factors such as proximity to seed sources, presence of animal dispersal agents, and degree of control of invasive weeds (Goosem & Tucker 1995; Lamb et al. 1997, 2005; Holl et al. 2000; Elliott et al. 2003; McDonald et al. 2003; Harden et al. 2004). Most rainforest restoration projects have focused on the recovery of trees to reestablish canopy closure which allows the forest floor to stabilize under a more constant microclimate and facilitates germination of shade-loving rainforest species (Harden et al. 2004). Earlier studies have suggested planting pioneer species in highly degraded sites, followed by the introduction of mature forest species once the pioneers form a closed-canopy layer (Goosem & Tucker 1995). Sites with exotic tree plantations may also prove useful for restoration as nurse ecosystems for shade-tolerant rainforest species (Ashton et al. 1997; Parrotta et al. 1997; Lamb 1998). Understanding the variability in seeding survival for individual species under different site conditions will enable the selection of suitable species (e.g., framework tree species, Goosem & Tucker 1995) and thereby potentially improve restoration efforts.

There has, however, been little effort at devising, implementing, and documenting programs to restore degraded tropical rainforests of Western Ghats and south Asia (Stewart & Balcar 2003; Lamb et al. 2005; Mudappa & Raman 2007). Rai (1990) reported the results of early planting trials with 31 economically important species in the Western Ghats of Karnataka and found that although seral species showed better early growth, only evergreen species eventually persisted. Nair et al. (2002) provided cultivation, growth and 1-year survival data for nine economically important native tree species, including four species that grow in the Western Ghats rainforests. This article describes a rainforest restoration project implemented on the Valparai plateau, Anamalai hills, India, in partnership with plantation companies. We describe the restoration protocols and plant survival under varying site conditions. These results will be useful in restoring rainforests using mixed-native species plantings in other regions, especially in south and southeast Asia.

Study Area

The Western Ghats is a 1,600 km long chain of hills running along the west coast of the Indian Peninsula (from lat 8°N to lat 21°N). This unique biogeographic region has pronounced north–south, east–west, and elevational gradients, with most of the higher hills and wet evergreen forests located between lat 8°N and lat 13°N. These areas have been shown to support high species diversity, with numerous endemic rainforest plants and animals (Pascal 1988; Kumar et al. 2004). The Anamalai Ranges are a major conservation area in the southern Western Ghats with a number of protected areas, including the Indira Gandhi Wildlife Sanctuary (WS), Eravikulam National Park, Chinnar WS, Parambikulam WS, and Reserved Forests. Most of the mid-elevation tropical evergreen forest is found in the Indira Gandhi WS (958 km^2, lat 10°12′ N to lat 10°35′ N, long 76°49′ E to long 77°24′ E), but rainforest fragments (at least 35 ranging between 0.3 and 650 ha in area) are found on private lands on the Valparai plateau. The Valparai plateau contains a large area of tea, coffee, and cardamom estates, which occupy over 220 km^2 near the center of the conservation areas. Within the Valparai plateau, there is a small town (Valparai) and many estates maintained by private companies, with a human population of about 100,000 people. There is approximately 100 km^2 of tropical rainforest around the plateau, including about 10 km^2 of slightly to severely degraded forest fragments within tea and coffee plantations on private lands. More details of the study region are available in other publications (Mudappa & Raman 2007; Raman 2006).

The region receives 3,500 mm average annual rainfall, of which about 70% falls during the southwest monsoon (June–September). The natural vegetation of this region has been classified as mid-elevation tropical wet evergreen forest of the *Cullenia exarillata–Mesua ferrea–Palaquium ellipticum* type (Pascal 1988). These species comprise the top canopy along with *Syzygium densiflorum* and *Calophyllum austroindicum*. The vegetation is also characterized by a midstory layer with trees such as *Myristica dactyloides* and *Diospyros assimilis*; understory trees such as *Oreocnide integrifolia*, *Antidesma menasu*, and *Gomphandra coriacea*; lianas (woody vines) such as *Connarus sclerocarpus*, *Aganosma cymosa*, and *Zanthoxylum ovalifolium*; and understory plants such as *Elatostemma lineolatum*, *Bolbitis semicordata*, and *Psychotria nigra* (further details in Muthuramkumar et al. 2006).

Restoration Sites

Three restoration sites, with planting areas that ranged in size from 0.15 to 1 ha were planted between 2000 and 2004 (Table 1). The rainforest restoration and monitoring methods described herein were initially developed and standardized through work at multiple sites on two of the forest fragment study sites, Injipara (19 ha) and Stanmore (5 ha), located on land owned by Tea Estates India Ltd (formerly Hindustan Lever Limited) and protected since 2000 as Biodiversity Plots. Plantings occurred at these two sites in 2000, 2002, 2003, and 2004 (Tables 1 & 2). Details

Table 1. Description of restoration sites on the Valparai plateau, Anamalai hills.

Restoration Site	*Fragment*	*Area (ha)*	*Year Planted*	*Canopy*	*Understory*
IJ-00 (mixed canopy)	Injipara	0.60	2000	Mixed canopy with exotics (SC, EG) and native species (AH, VA)	Very dense invasion of *Lantana camara* and *Chromolaena odorata*
S1-02 (open meadow)	Stanmore	0.15	2002	Open with just two exotic trees (ME)	Meadow with a thick carpet of closely cropped grass
S2-02 (open weedy)	Stanmore	0.25	2002	Sparse and open with scattered trees (including EG)	Very dense invasion of *L. camara* with *Mikania* sp. and *C. odorata*
I1-03 (open weedy)	Injipara	0.25	2003	Sparse and open with few midstory trees	Moderately dense invasion of *L. camara* with *Mikania* sp. and *Polygonum chinense*
I2-03 (exotic)	Injipara	0.25	2003	Mostly exotic canopy of two species (EG, ME)	Very dense invasion of *L. camara* with *Mikania* sp. and *C. odorata*
I3-03 (high shade)	Injipara	0.25	2003	Dense canopy of exotic (SC) and native species	Dense with native plants, few *L. camara* and *Mikania* sp.
I1-04 (exotic)	Injipara	1.00	2004	Mostly exotic canopy (EG) with some native trees (AH, VA)	Dense *L. camara* and *E. glandulosum* and regenerating exotic trees (SC)
S1-04 (open weedy)	Stanmore	0.50	2004	Sparse and open with few trees	Dense *L. camara* with *Mikania* sp. and *C. odorata*
Iy-04 (low shade)	Iyerpadi top	0.40	2004	Sparse and open with some native trees (MT, MP, AP)	Dense *L. camara* and *C. odorata*

Tree species: AH, *Artocarpus heterophyllus*; AP, *Acronychia pedunculata*; EG, *Eucalyptus grandis*; ME, *Maesopsis eminii*; MP, *Macaranga peltata*; MT, *Mallotus tetracoccus*; SC, *Spathodea campanulata*; VA, *Vernonia arborea*.

Table 2. Rainforest tree seedlings planted at different restoration sites in the Anamalai hills, Western Ghats (2000–2004).

Site	*Canopy*	*Area (ha)*	*Year Planted*	*Number of Seedlings*	*Number of Species*
IJ-00	Mixed	0.6	2000	829	33
S1-02	Open	0.15	2002	268	27
S2-02	Sparse	0.25	2002	479	54
I1-03	Open	0.25	2003	609	44
I2-03	Exotic	0.25	2003	599	62
I3-03	Dense	0.25	2003	642	57
I1-04	Exotic	1	2004	1923 (262*)	82 (49*)
S1-04	Sparse	0.5	2004	1,015 (254*)	64 (44*)
Iy-04	Sparse	0.4	2004	1,174	56

* Sample of seedlings/species monitored within the site.

of the vegetation in Injipara in comparison with relatively undisturbed reference sites within continuous rainforest are available in Muthuramkumar et al. (2006). The third study site, Iyerpadi Top, is a forest fragment of about 100 ha belonging to Parry Agro Industries Ltd, where restoration planting has been conducted since 2004. A brief description of the sites where restoration plantings occurred between 2000 and 2004 is provided in Table 1.

Methods

Restoration Protocol

All three forest fragments were mapped, demarcated, and protected with barbed wire fences to prevent human-induced disturbance such as fuelwood collection and grazing. Within each of the fragments, square or rectangular areas along the edge of the fragment ranging in size from 0.15 to 1 ha were marked for planting. Planting was completed at the onset of the southwest monsoon (June–July) to ensure sufficient water for seedling establishment and growth. The seedlings were planted at a spacing of 1.5 to 2.5 m (to facilitate monitoring) based on the density of existing seedlings and saplings at the site and the size of the site. Seedlings were mostly raised in the nursery from seed or, to a lesser extent, from wildlings collected from forest edges and trails (mostly used for planting between 2000 and 2002). Seeds used in the nursery were collected from wide trails and roads through or along the edges of rainforest fragments to minimize disturbance to natural regeneration in the rainforest interior. Seeds were collected from multiple parent trees within the Valparai region.

At one site (IJ-00), planting was carried out along narrow (0.5 m wide) lines cut through weed (chiefly *Lantana camara*) undergrowth. These lines were maintained clear of weeds with three or four rounds of weeding (cutting, without uprooting, of *L. camara*) over a 15-month period, after which weeding was stopped. At all other sites, invasive weeds such as *L. camara*, *Chromolaena odorata*, and *Mikania* sp. were cut with machetes and all *L. camara* uprooted with mattocks 1–4 months prior to planting. Two practices were strictly followed during weed clearing. First, special care was taken not to cut any naturally regenerating native vegetation, including shrubs and climbers. Second, larger (>1 cm girth) woody stems of *L. camara* were cut into small lengths and removed from the restoration areas to provide fuelwood to people in local settlements. The finer stems and all green leafy material were left on the site to form a mat-like ground layer to avoid soil exposure and erosion.

Multiple rainforest trees (with some lianas and shrubs) representing a combination of the framework species and maximum diversity methods (Goosem & Tucker 1995) were planted. Planted seedlings were at least 45–60 cm tall and hardened in the nursery through regular exposure to direct sun and reduced watering for 3–4 months prior to planting. Seedlings were planted in pits of 10–15 cm diameter and 45 cm depth. Seedlings were fertilized with 100 g organic manure mixed with the soil and 25 g rock phosphate on the sides and bottom of the pit. A layer of litter mulch was placed around the base of each seedling. Post-planting, hand weeding was completed four to six times during the first year and two to three times in the second year after planting.

Monitoring Protocol

All planted species were identified based on prior research experience and field guides (Gamble & Fischer 1915–1935; Pascal & Ramesh 1997). Each seedling was marked with biodegradable flagging tape and tagged with thin metal tags with eyelets fastened to seedlings by metal wire with the species code and plant number impressed on the metal tag with a pen. The seedlings were monitored for survival at 6-month intervals for the first 2 years: in December–January after the plants' first wet season, in May–June after the first dry season, and likewise during the following year after the plants' second wet and dry seasons. For two of the larger sites planted in 2004 (I1-04, S1-04), approximately 250 seedlings were randomly selected from among all planted areas. Data were collected regarding state (dead or alive), condition (browsed, cut, insect herbivory), and growth of each seedling. In addition, photographic documentation and standard quadrat sampling were conducted every 6 months and 2 years, respectively (results not presented here; Fig. 1).

Analysis

Seedling survival was the primary parameter of interest, and percent survival was calculated as the percentage of initially planted seedlings still alive at the time of monitoring. Six-month survival was also calculated as the percentage of seedlings at a given monitoring period that were found to be alive 6 months later. This corresponded to survival over the wet and dry seasons. Suitability of a species was determined based on its survival rates across sites at the end of 2 years. It was classified as excellent when survival was 76–100%, good when 51–75%, moderate when 26–50%, and poor when less than or equal to 25% (modified after Elliott et al. 2003). Interspecies comparisons were also made for tree seedling survival at the three restoration sites within the Injipara rainforest fragment because they were planted in the same year (2003) with a similar mix of species and similar age and condition of seedlings. Survival rate comparisons were made using chi-square contingency table tests and Pearson's product-moment correlation coefficients (Zar 1999).

Results

Survival Monitoring

Between 2000 and 2004, a total of 7,538 seedlings of 127 plant species (between 27 and 82 species per site) were planted in nine restoration sites ranging in size between 0.15 and 1.0 ha within the three degraded rainforest fragments on the Valparai plateau (Table 2). Survival varied across the nine sites at each monitoring period: survival after 6 months (78 to 98%), 12 months (49 to 98%), 18 months (39 to 95%), and 24 months (34 to 90% across eight sites; survival at one site, IJ-00, could not be measured at 24 months as explained below). Lowest survival was recorded at weed clearing and line planting sites in Injipara (IJ-00) and a weed removal site in Stanmore (S2-02, Fig. 2), in both of which mostly wildlings were planted. Survival rate in the first year's (2000) planting along weed-cleared lines in IJ-00 was initially high (88.4% after the first wet season). The survival had declined to 59.2% by 15 months (Fig. 2). With the subsequent overgrowth of weeds (chiefly *Lantana camara*) following the cessation of weeding, the seedlings were untraceable at 24 months. During the study, the highest seedling survival was recorded in the open-meadow site (S1-02) that had no weed invasion and only a ground layer of short grasses at the time of planting (90.3% survival at the end of 2 years).

At other weed removal sites established after 2002, where largely seedlings raised from seed were used rather than wildlings, seedling survival was higher (Fig. 3). Compared with the low-shade (Iy-04) and open-weedy (I1-03) sites, the other four sites had higher seedling survival ranging 67 to 76% at the end of 2 years (Fig. 3).

Comparisons were made of seedling survival from three sites in Injipara fragment planted in 2003, with varying canopy closure ranging from open (I1-03), through exotic *Eucalyptus–Maesopsis* canopy (I2-03), to dense mixed canopy of native and exotic species (I3-03). Seedling survival varied significantly across the three sites at the end of 2 years and was highest in the high-shade site (I3-03), lowest in the open-weedy site (I1-03), and intermediate in the exotic canopy site (I2-03; $\chi^2 = 14.96$, $df = 2$, $p < 0.001$; Fig. 3). Seedling survival in the open-weedy site at the end

Figure 1. Tracking changes at restoration sites from photographic reference sites: (a) Meadow (S1-02) site immediately after planting and (b) 3 years later in June 2005; (c) Open weedy site (S2-02) after weed removal and just before planting: note uncut naturally regenerated native plants and mat of biomass covering soil; (d) Open weedy site (S2-02) 3 years later in June 2005; (e) *Eucalyptus–Maesopsis* canopy site (I2-03) just after planting; and (f) same site 2 years later in June 2005.

of 2 years was significantly lower than the exotic canopy site ($\chi^2 = 8.32$, $df = 1$, $p = 0.003$) and the high-shade site ($\chi^2 = 14.1$, $df = 1$, $p < 0.001$). The latter two sites did not differ in survival rates ($\chi^2 = 0.69$, $df = 1$, $p = 0.40$). The three restoration sites planted in 2004 showed high seedling survival over the first 6 months (83.5 to 94.9%). At the end of 2 years, however, survival was higher in the *Eucalyptus* (76%) and open-weedy sites (76%) than in the low-shade site (37%).

Seasonal variation in survival rates was compared among the three Injipara sites planted in 2003. At all three sites, 6-month plant survival was higher for wet seasons compared to dry seasons in both years of monitoring (Fig. 4). Comparison of plant survival at a given season between years revealed little difference between 6-month survival over the first and second wet seasons. However, 6-month survival over the second dry season was higher than over the first dry season in I1-03 and I2-03 (Fig. 4).

Variation Across Seedling Species

A total of 109 species (4,287 seedlings) were monitored for survival across the eight sites established during 2002–2004 where survival monitoring data are available for 24 months. Of these, percent survival was calculated at the end of 24 months for the 44 species that had over 20 individual seedlings planted in two or more of these sites. Nine species (20.5%) showed excellent survival (>75%), and an additional 18 species (40.9%) showed good survival (51–75%). Of the species planted, 16 (36.4%) showed moderate survival (26-50%), whereas one species *Litsea insignis* showed poor (<25%) survival (Table 3). Although all species planted in the open-meadow site (S1-02) had high survival, in most cases, survival varied substantially among sites and species. Percent survival was only weakly related to strata and successional habit of species. At the end of 24 months, understory pioneers tended to have higher average survival (82.1%, SE = 4.35) than

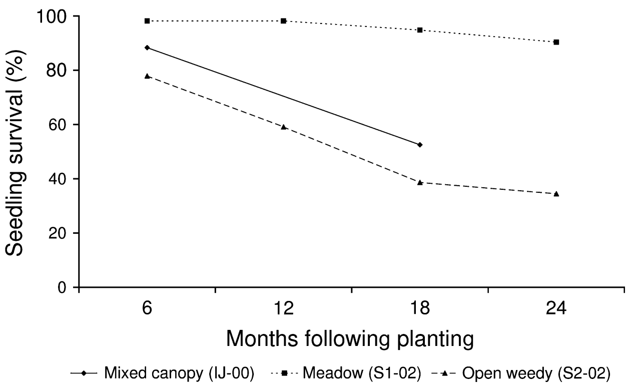


Figure 2. Survival of plants over a 2-year period at restoration sites planted in 2000 and 2002.

understory mature species (62.2%, SE = 6.2). Survival in the latter category was similar to the remaining categories (Table 3), specifically, topstory mature species (65.3%, SE = 4.08), midstory mature species (59.1%, SE = 3.71), and midstory pioneers (61.6%, SE = 6.48).

We also compared seedling survival for those species with at least 20 seedlings planted in 2003 at the three sites in Injipara fragment (I1-03, I2-03, and I3-03) to expected frequencies based on initial numbers planted in each site using chi-square tests. Of 10 species tested, *Palaquium ellipticum* showed a statistically significant difference in survival across sites ($\chi^2 = 6.14$, $df = 2$, $p < 0.05$). For this species, survival was lowest in the open-weedy site (I1-03), intermediate in the *Eucalyptus*-dominated site, and highest in the high-shade site (Table 4).

Six-month survival was higher over each wet season than the succeeding dry season in all species, with a few exceptions. Specifically, survival of *Mesua ferrea* appeared to be high across all seasons, survival of *Filicium decipiens* declined every 6-month interval, and *Trichilia connaroides* showed high but slightly varying survival unrelated to season (data not presented here).

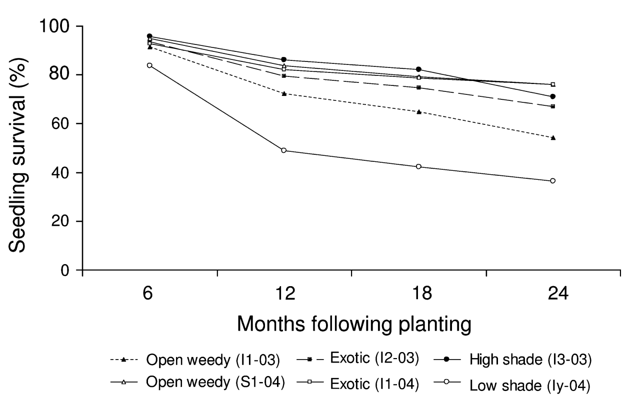


Figure 3. Survival of plants over a 2-year period at restoration sites planted in 2003 and 2004 in rainforest fragments in the Anamalai hills.

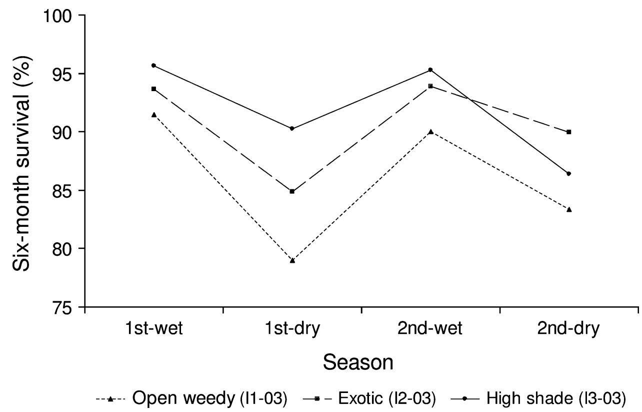


Figure 4. Six-monthly survival of plants over a 2-year period (2003–2005) at restoration sites planted in the Injipara rainforest fragment.

Relative to each other, species that survived better in one site also did better in other sites. Survival in I1-03 was significantly correlated to survival in I2-03 ($r = 0.683$, $df = 8$, $p = 0.030$) as well as I3-03 ($r = 0.761$, $df = 8$, $p = 0.011$). Survival in I2-03 was also highly positively correlated to survival in I3-03 ($r = 0.837$, $df = 8$, $p < 0.003$). Regression analysis of these data indicated that most species appeared to have lower survival in the open-weedy site (I1-03) than in the other two sites that had greater canopy cover (Fig. 5).

Discussion

Evaluation of Restoration Protocol

The restoration protocol used since 2003 in the present study evolved from the experiences gained from the initial planting at restoration sites in 2000 and 2002. In comparison to five other sites, seedlings planted in lines cut through the dense weed undergrowth in 2000 had low survival rates after 15 months. The seedlings planted in these lines were wildlings raised in the nursery prior to planting. During the monitoring at 15 months, few seedlings had grown to a height greater than 1 m. It is difficult to assess whether the poor survival and growth was due to the use of wildlings, continued proximity of dense weed growth, or other factors such as rainfall or incidents of grazing by stray cattle over those 2 years. However, the fact that just 6 months after weeding stopped, the seedlings were virtually untraceable in the dense tangle of weeds suggests that continuous weeding for more than 2 years is necessary to prevent seedlings from being overgrown or smothered by weeds such as *Lantana camara*, *Mikania* sp., and *Chromolaena odorata*.

The importance of controlling smothering weeds and vines through manual weeding or herbicide application has been noted in restoration of many Australian rainforest sites (Goosem & Tucker 1995; Lamb et al. 1997; Harden et al. 2004). The removal of invasive weeds reduces competition for space, light, and nutrients, improving survival of planted seedlings (Harden et al. 2004).

Table 3. Number of planted seedlings and percent survival (%) after 24 months in tropical rainforest restoration sites in the Anamalai hills (2002–2004).

Species	*Family*	*Strata*	*Status*	N	*Survival (%)*
Holigarna nigra	Anacardiaceae	Top	Mature	26	65.4
Nothopegia racemosa	Anacardiaceae	Understory	Mature	22	36.4
Semecarpus travancorica	Anacardiaceae	Middle	Mature	132	45.5
Bischofia javanica	Bischofiaceae	Top	Mature	110	63.6
Cullenia exarillata	Bombacaceae	Top	Mature	293	55.6
Bhesa indica	Celastraceae	Top	Mature	99	31.3
Euonymus angulatus	Celastraceae	Middle	Mature	44	72.7
Calophyllum austroindicum	Clusiaceae	Top	Mature	36	61.1
Mesua ferrea	Clusiaceae	Top	Mature	120	80.8
Vateria indica	Dipterocarpacee	Understory	Mature	24	79.2
Elaeocarpus munronii	Elaeocarpaceae	Middle	Mature	37	45.9
E. serratus	Elaeocarpaceae	Top	Mature	36	50
E. tuberculatus	Elaeocarpaceae	Top	Mature	126	56.3
Actinodaphne spp.	Lauraceae	Middle	Pioneer	152	63.2
Cinnamomum malabathrum	Lauraceae	Middle	Pioneer	83	68.7
Lauraceae sp.	Lauraceae	Middle	Mature	66	59.1
Litsea insignis	Lauraceae	Middle	Mature	46	19.6
L. oleoides	Lauraceae	Middle	Mature	147	65.3
L. stocksii	Lauraceae	Top	Mature	21	33.3
Persea macrantha	Lauraceae	Middle	Pioneer	162	48.1
Michelia champaca	Magnoliaceae	Middle	Mature	20	90
Aglaia exstipulata	Meliaceae	Middle	Mature	26	53.8
Trichilia connaroides	Meliaceae	Understory	Pioneer	173	79.2
Artocarpus heterophyllus	Moraceae	Middle	Mature	200	31
Ficus beddomei	Moraceae	Middle	Mature	21	90.5
F. nervosa	Moraceae	Top	Mature	24	54.2
Myristica dactyloides	Myristicaceae	Middle	Mature	188	50.5
Maesa indica	Myrsinaceae	Understory	Pioneer	30	86.7
Syzygium densiflorum	Myrtaceae	Top	Mature	62	40.3
S. gardneri	Myrtaceae	Top	Mature	73	84.9
S. racemosa	Myrtaceae	Understory	Mature	46	54.3
Syzygium spp.	Myrtaceae	Top	Mature	20	70
Olea dioica	Olea dioica	Middle	Pioneer	32	43.8
Ormosia travancorica	Papilionaceae	Middle	Mature	245	62.4
Prunus ceylanica	Rosaceae	Top	Mature	126	48.4
Atalantia racemosa	Rutaceae	Understory	Pioneer	60	78.3
Vepris bilocularis	Rutaceae	Middle	Mature	34	26.5
Dimocarpus longan	Sapindaceae	Middle	Mature	229	74.7
Filicium decipiens	Sapindaceae	Middle	Mature	56	46.4
Lepisanthes decipiens	Sapindaceae	Understory	Mature	139	66.2
Palaquium ellipticum	Sapotaceae	Top	Mature	221	28.1
Sterculia guttata	Sterculiaceae	Middle	Pioneer	31	35.5
Antidesma menasu	Stilaginaceae	Understory	Mature	42	50
Clerodendrum viscosum	Verbenaceae	Understory	Pioneer	27	92.6

Canopy position (strata) occupied by the tree species and successional status (mature forest *versus* pioneer) are also indicated.

In this study, survival rates in one of the first weed removal sites (S2-02) were also found to be low after 2 years (34.4%), possibly because wildlings were planted. In addition, the site also experienced low rainfall during the first wet season and an unusually long first dry season (nearly 7 months during which time the plants were watered twice).

Complete removal of invasive shrubby weeds (mainly *L. camara* and *C. odorata*) and vines (mainly *Mikania* sp. and *Aristolochia* sp.) was used as a standard protocol at all restoration sites since 2002. Although weed removal by cutting and uprooting was quite labor and time intensive, it proved more effective in the longer-term. This was because the sites could be maintained by weeding at regular intervals intensively in the first 2 years. By this time, rapid recovery of preexisting seedlings and shrubs (which were not cut during weeding operations), rapid regeneration of the pioneer shrub *Clerodendrum viscosum* (recording a 5-fold increase in density over a 2.5-year period, unpublished data), and growth of some of the planted seedlings formed the first layer of canopy closure. Our experience reveals that in most sites, little or no weeding

Table 4. Percent survival (%) at the end of 24 months for tree seedlings planted at three restoration sites with different canopy cover conditions in Injipara rainforest fragment, Anamalai hills.

		Number Planted			Survival After 24 Months (%)		
Species	Family	I1-03 Open Weedy	I2-03 Exotic Canopy	I3-03 High Shade	I1-03 Open Weedy	I2-03 Exotic Canopy	I3-03 High Shade
Semecarpus travancorica	Anacardiaceae	15	10	11	26.7	70	72.7
Bischofia javanica	Bischofiaceae	25	24	21	64	83.3	66.7
Cullenia exarillata	Bombacaceae	50	53	61	72	83	70.5
Mesua ferrea	Clusiaceae	20	30	43	100	80	93
Elaeocarpus serratus	Elaeocarpaceae	10	10	9	30	70	55.6
E. tuberculatus	Elaeocarpaceae	25	24	28	72	70.8	67.9
Litsea sp. (mostly *L. oleoides*)	Lauraceae	36	16	25	55.6	93.8	88
Trichilia connaroides	Meliaceae	48	35	21	58.3	82.9	95.2
Artocarpus heterophyllus	Moraceae	17	28	27	11.8	14.3	29.6
Myristica dactyloides	Myrsiticaceae	41	30	45	65.9	36.7	68.9
Syzygium gardneri	Myrtaceae	30	19	21	73.3	94.7	90.5
Ormosia travancorica	Papilionaceae	45	38	48	44.4	89.5	85.4
Prunus ceylanica	Rosaceae	27	24	22	51.9	70.8	68.2
Dimocarpus longan	Sapindaceae	33	25	14	63.6	72	78.6
Lepisanthes decipiens	Sapindaceae	25	14	16	60	78.6	68.8
Palaquium ellipticum	Sapotaceae	50	50	59	14	26	42.4

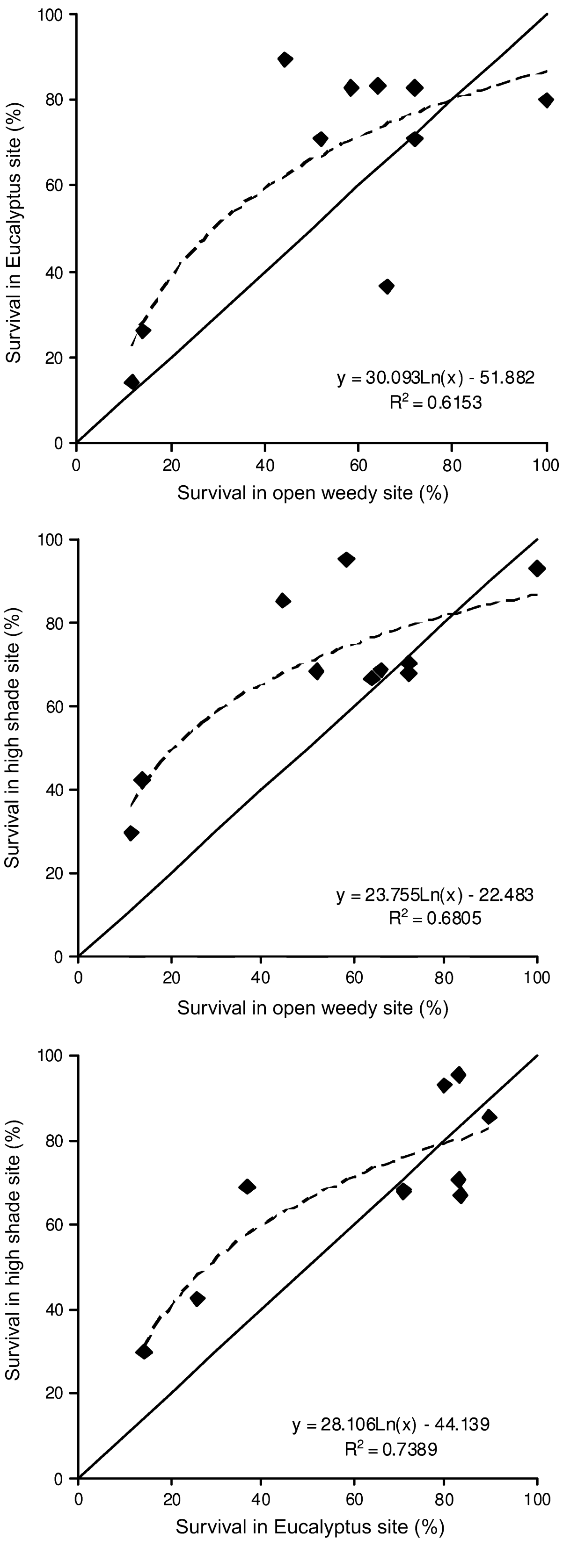
is required after the first 2–2.5 years (up to three growing seasons).

Seedling Survival

Few tropical forest restoration projects have reported percent survival by site, year, or species (Elliott et al. 2003; McDonald et al. 2003; Harden et al. 2004; Lamb et al. 2005). McDonald et al. (2003) reported average survival rates of 39 and 49% after 12 months and 21 and 20% after 42 months at ridge-top and valley-bottom locations, respectively, in a tropical montane forest of Jamaica. In a seasonally dry tropical forest in Thailand, survival rates at 17 months were 72.7 (15 species planted in 1998) and 52.4% (29 species planted in 1999, calculated from Table 2 of Elliott et al. 2003). Although vegetation type is different from that in the present study, survival rates reported here (49.1 to 98.1% after 12 months) are higher than those reported by McDonald et al. (2003). One reason may be the use of larger (45–60 cm height), hardened seedlings raised from seed in this study versus mostly wildlings uprooted and raised in a nursery to 15 cm height by McDonald et al. (2003). In this study, 18-month survival was more variable (38.6 to 94.8%) than survival reported by Elliott et al. (2003), although the survival rates from the 2003 sites were comparable possibly due to the similar protocol used. Another important factor facilitating plant survival may have been the protection accorded to the restoration sites from grazing and fuelwood collection in the present study.

A considerable amount of literature exists on seed and seedling dynamics in relation to microsite and canopy variability in relatively intact tropical rainforests, logged areas, and secondary forests (Uhl et al. 1988; Turner 1990; Rajesh et al. 1996; Richards 1996; Struhsaker 1997). Little is known, however, of species-specific survival of a wide variety of tropical rainforest tree species under field conditions in restoration projects (Holl et al. 2000; Khurana & Singh 2001; Elliott et al. 2003; Hau & Corlett 2003; McDonald et al. 2003). McDonald et al. (2003) reported survival rates of 26 tree species ranging from 0 to 85% after 1 year and from 0 to 50% after 42 months in tropical montane forest of Jamaica. Elliott et al. (2003) report 25 to 98.3% survival of 37 tree species after 17 months in tropical dry forest of Thailand. In the Peruvian Amazon, Soudre et al. (2001) noted survival rates between 3 and 99% after 13 months for six native species planted in degraded sites, with survival influenced by the dominant weed species in these sites. In one of the few studies that report survival rates over a relatively longer term of 5–6 years, Azani et al. (2001) reported survival rates between 10 and 96% for 18 species. Only two species did not survive in one of the four degraded sites planted. These levels of percent survival coupled with relatively good growth rates (mean annual height increment of 17 to 131 cm/yr and mean annual basal diameter increment of 0.31 to 2.88 cm/yr across species) indicate a high potential for the use of mixed-native species plantings for restoration of tropical rainforests.

Little information exists on seedling survival in native species plantings in rainforests of the Western Ghats. In field trials of nine economically important tree species, Nair et al. (2002) reported survival rates between 31 and 86% after 9 months for seven species. The remaining two evergreen forest species, *Dysoxylum malabaricum* and *Calophyllum polyanthum*, showed survival rates of 61 and 39%, respectively, after 12 months for seedlings that were less than 30 cm tall at the time of planting. Although the relatively low survival and slow growth rates of some



species may affect commercial viability of plantations, the survival rates indicate promise for ecological restoration and rehabilitation programs. These other studies are not directly comparable with the present study due to differing site conditions, methods, and the selection of particular subsets of the species found in an area. However, the 61.4% survival recorded after 2 years for the 44 species in this study indicates that multispecies planting with weed removal is a suitable protocol for restoration of degraded rainforest fragments. Even with a relatively high percent survival overall, there was considerable variation in survival across species and sites. The highest seedling survival was recorded in the open-meadow site (S1-02) in the Stanmore fragment, 90.3% of the seedlings surviving after 2 years. In contrast, Holl et al. (2000) report low percent survival, even for pioneer species, due to competition with pasture grasses in other areas. Possible reasons for higher survival include the use of large seedlings (>30–50 cm in height), selection of hardy species known to grow in disturbed areas and fragment edges, planting in deep pits (45 cm depth) beyond the range of root competition with grasses, and periodic weeding over the first 2 years.

Another notable result is the high seedling survival (67.1 and 76%) after 2 years at two sites with an overstory dominated by exotic timber species such as *Eucalyptus grandis* and *Maesopsis eminii*, possibly due to shade reducing competition from grasses and vines. The role of timber and monoculture plantations as foster ecosystems for colonization of native forest species, especially in plantations that are in close proximity of mature forests, has been highlighted in a number of studies (see Lamb 1998 for a review).

Comparisons of the three degraded rainforest sites in Injipara (I1-03, I2-03, and I3-03) reveal that seedling survival appears lower in more open conditions compared to sites with a denser canopy of exotics or mixed-native and exotic trees. This result in not surprising because most of the species that were planted are late-successional trees found in mature rainforest and included few pioneer species. Greater exposure to direct sunlight may have led to higher mortality of shade-tolerant mature forest species such as *Ormosia travancorica* and *Semecarpus travancorica*. Survival of species such as *Palaquium ellipticum*, *Artocarpus heterophyllus*, and *Syzygium gardneri* was lower in the more open site, partly due to browsing by barking deer (*Muntiacus muntjak*), browsing and trampling by Asian elephants (*Elephas maximus*), and drier soil due to high evaporation in open-canopy areas. But the seedlings of a few

Figure 5. Comparison of percent survival of rainforest tree seedlings 2 years after planting at three different restoration sites with different canopy types at Injipara, Anamalai hills. The dotted lines indicate nonlinear (logarithmic) regression lines fit to the data—regression equations (chosen based on higher r^2 value than linear fits) are indicated. The straight line indicates the line of equality ($y = x$); data points lying above this line indicate species whose survival was higher in the site represented on the y-axis.

species (*Canarium strictum*, *Bischofia javanica*, and *Elaeocarpus tuberculatus*) grew to over 2 m in height within 2 years at open-canopy site (I1-03). None of the seedlings, including the species mentioned above, in the high-shade site (13-03) exceeded this height. However, the reasons for these differences were not evaluated in this study.

As expected, seedling survival was lower over the dry season than the wet season. Seedlings that survived the first dry season were expected to survive the second dry season (as the seedlings that survived 1 year were expected to be hardier), and this was the case at two of the three sites. The causes for slightly lower survival in I3-03 over the second dry season are not known but may be related to cattle grazing.

A majority of species planted here were shade-tolerant mature forest species whose survival appeared to be consistent across sites. The species that performed poorly (*O. travancorica*, *P. ellipticum*, *A. heterophyllus*, and *S. travancorica*) were mature forest species that may require special efforts in restoration programs, such as provision of tree guards to prevent browsing, selection of appropriate microsites (e.g., shaded areas), and the use of older or hardened saplings that can better tolerate drought or browsing. The need to plant mature forest species for effective long-term restoration given constraints in seed dispersal and establishment of late-successional species in highly degraded sites has been emphasized by earlier authors (Parrotta and Knowles 1999, 2001; Lamb et al. 2005). The results of the present study also indicate that certain mature forest species are suitable for restoration of a rainforest sites ranging from open-weedy sites to those with a fairly dense native or exotic canopy.

Implications for Practice

- A high diversity of species can be used for tropical rainforest restoration in degraded sites, including under *Eucalyptus* canopy, protected from further disturbances.
- In sites with considerable weed invasion, complete weed removal including uprooting (particularly *Lantana camara*) is required, with care taken to retain all naturally regenerating native shrubs and seedlings.
- At least 2- to 3-year-old (60–75 cm tall) vigorous, hardened seedlings raised from seeds in a nursery need to be planted with regular monitoring and maintenance for 2 years.
- Under such conditions, high seedling survival (>50% after 2 years) can be expected, although survival and growth will vary depending on site conditions and species characteristics.
- At a broader level, restoration efforts require sustained support through incentives to landowners engaged in forest protection and restoration, provision of alternative fuels for local people dependent on forest fragments for fuelwood, and developing greater awareness regarding proper restoration methods.

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