

ASSESSING A THREAT TO SUSTAINABLE NTFP HARVEST USING ECOLOGICAL DATA AND TRADITIONAL ECOLOGICAL KNOWLEDGE

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Abstract

Sustainable harvesting of non-timber forest products (NTFPs) and associated livelihood security can be compromised by changes in the broader ecosystem. This study investigates mistletoe infection of Amla, *Phyllanthus emblica* and *Phyllanthus indofischeri*, an NTFP of significant livelihood importance, in the Biligiri Rangaswamy Temple (BRT) Wildlife Sanctuary in southern India. Infection patterns of the Loranthaceae mistletoe, *Taxillus tomentosus*, were characterised across the Amla population and their underlying mechanisms explored. Two alternative management approaches were investigated, and traditional ecological knowledge documented and assessed for its concordance and additionality to scientific data.

A high prevalence of infection, and the apparent greater susceptibility of the more significant resource species, suggests that mistletoe infection has serious implications for Amla sustainability. Traditional knowledge provided novel information on infection spread, and highlighted failings in current population assessments, emphasising the importance of a multidisciplinary approach to assessing sustainability. Traditional approaches to management conflict with current institutional perspectives, but may be somewhat more effective and require further investigation. However, existing data, both from scientific studies and traditional knowledge, are insufficient to prescribe with certainty the best approach to mistletoe control. The participatory implementation of an active adaptive management programme may offer benefits over alternative management options for securing Amla as a viable NTFP harvesting system.

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Declaration

All the work presented in this thesis is my own

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CHAPTER 1

INTRODUCTION

1.1 Thesis context

1.1.1 Non-timber forest products (NTFPs)

Debate over the definition of the term non timber forests product (NTFP) has continued since the term was coined by De Beer & McDermott (1989). The expressions ‘*non-timber forest product*’, ‘*non-wood forest product*’, and ‘*minor forest product*’, have also been used interchangeably (Arnold & Ruiz-Perez, 2001), with variously wide limits as to what they include. The appropriateness of including woody plant products, and additionally, forest ecosystem services such as carbon sequestration, nutrient cycling or amelioration of water flows, have been two areas of dispute (FAO, 1999). In fact, NTFPs have been defined more often by what they are not (Neumann & Hirsch, 2000). The definition used by Arnold & Ruiz-Perez (2001) “*any product other than timber, dependent on a forest environment*”, when restricted to material products and their derivatives, is characteristic of what is now widely accepted (Belcher, 2003).

The NTFP term, in many respects, is not particularly helpful. These resources are as diverse as the socioeconomic contexts in which they are harvested. Property rights, harvesting practices, management strategies and the ecological effects associated with their harvesting are hugely variable. Many are local subsistence products, while others such as Brazil nuts or Ginseng, are harvested for sale on international markets. The one thing they all have in common is their association with the idea that forests will remain more or less biologically intact under their repeated harvesting (Neumann & Hirsch, 2000). The most commonly used NTFPs are often wild fruits, bushmeat, fuel wood, medicinal plants and weaving fibres. Other examples also include oils, seeds,

resins, gums, spices, dyes, ornamental plants, raw materials such as bamboo and rattan, as well as non-food wildlife such as reptiles and butterflies (Belcher, 2003).

1.1.2 NTFPs: an integrated conservation-development strategy

Forest management in the colonial period often included detailed provisions for the management of NTFPs (J. Sayer, personal communication). However, a later focus on the productive capacity of tropical forests, including the threats posed by industrial logging, resulted in the oversight of NTFPs up until the late eighties and early nineties. The ‘NTFP concept’, the paradigm of conserving tropical forests and improving rural livelihoods through the commercialisation of NTFPs (Allegretti (1990); Nepstad & Schwartzman (1992); Ruiz Prez & Arnold (1996); Wollenberg & Ingles (1998)), emerged at this time, a consequence of both the realisation of the importance of forest resources to local livelihoods, and from the growing trend of linking conservation and development objectives. It has since risen to become a leading focus in both international forestry research and conservation (e.g. Peres & Lake (2003); Alves & Filho (2007); Ndangalasia et al. (2007)). This concept, epitomised by Dove (1993) as the “rainforest crunch thesis”, after one of the better known internationally marketed rainforest products that emerged in this period, linked several ideas. First, many tropical forests have a greater long-term economic value if left standing; second, that if local communities receive direct economic benefits from harvesting forest resources they will manage that forest sustainably; and third, that if poverty, as both a cause and result of deforestation, can be alleviated through the harvesting of forest resources, then deforestation pressures will be reduced. The idea that NTFP production is less ecologically destructive than other forest uses, and has the potential to supply local people with sufficient income to provide incentives to maintain tropical forest cover, was fuelled by several now widely cited articles, most famously Peters et al. (1989). This paper in *Nature* proposed that the long-term financial return from the harvest of NTFPs found in a hectare of Amazonian rain forest far outweighed the net benefits of either timber production, or agricultural conversion of the same area of land.

The idea that forest conservation and rural development objectives could be linked attracted a vast amount of interest from international donor agencies. A proliferation of NTFP initiatives and research followed, not just in the tropics, but also in North America (e.g. Pilz & Molina (2002); Peck (2006)), and Europe (e.g. Emery et al. (2006)). However, this was closely followed by a realisation that the potential for ‘conservation by commercialisation’ was not as great as previously believed (Arnold & Ruiz-Perez (2001); Sheil & Wunder (2002); Belcher & Schreckenberg (2007)). Economic cycles and socio-economic characteristics (Crook & Clapp (1998); Pendelton (1992); Kline et al. (2000); Shackleton (2001)), the push towards domestication of NTFP species (Homma, 1996) and unsustainable harvest levels (Vasquez & Gentry (1989); Soehartono & Newton (2002)) all emerged as threats to the long-term viability of NTFP harvesting as an integrated conservation-development strategy.

In reality, for many forest communities, market barriers such as transportation difficulties, low resource density, high variability in fruit production, perishable products and lack of market expertise are problematic (Shanley et al. (2002); Salafsky et al. (1993); Pendelton (1992)). Economic cycles leading to ‘boom and bust’, variable demand (Rai, 2003), product substitution, domestication and exclusion of harvesters by more powerful actors, further limit opportunities (Dove (1993); Homma (1996); Richards (1993)).

Harvesting itself can have significant implications for the viability of NTFP commercialisation. The ecological impacts of NTFP harvesting have received considerable attention in the literature, although with a focus towards certain products (Ticktin, 2004). Harvesting affects the survival, growth and reproduction of individuals (Pinard, 1993), influences demographic and genetic patterns of populations (e.g. Gaoue & Ticktin (2007); Uma Shaanker et al. (2001)), and can alter community- and ecosystem-level processes (Peck, 2006). Tolerance to harvest varies according to the particular product, or part of an individual, that is harvested and its life history (e.g. Fa et al. (1995); Ticktin (2004)). Moreover, the effects of harvest for any one species are mediated by variation in environmental conditions over space and time, and by human management practices. The population impacts of bushmeat hunting compared to collection of wild fruits are very different, and the implications of fruit

collection via cutting of individual trees as opposed to collection from branches are considerably different. Harvesting methods may also change as a consequence of a shift from subsistence to commercial harvesting, increased value providing incentives for less sustainable practices (e.g. Clay (1997)).

Several review papers have documented disappointing outcomes for NTFP commercialisation projects, and analyses of factors influencing commercialisation success suggest a rather specific range of conditions are necessary to ensure a reasonable chance of successful commercialisation (Marshall et al. (2003); Neumann & Hirsch (2000)). The potential for conservation of tropical forests by commercialisation of forest products, may be limited. Research efforts have since focused more critically on the biological, social and economic aspects of NTFP use in an attempt to inform harvesting and management. In particular, significant effort has been devoted to the study of species-specific harvesting systems, including demographic modelling and experimental harvesting studies (Ticktin et al. (2002); Endress et al. (2006); Gaoue & Ticktin (2007)).

However, many factors can impact upon the sustainability of resource harvesting besides the fundamental interplay between the demographics of the harvested species and the rate, or nature, of its harvesting; particularly in forests subject to considerable human influence. Sustainable harvesting may be affected by many additional factors (Milner-Gulland, 2008). Many resource assessments focus on the impact of harvesting itself overlooking wider threats to the target of ecologically and economically sustainable harvesting (except see Sinha & Bawa (2002); Menton (2003)). These wider threats may have significant implications for the status of exploited populations and the viability of both subsistence and commercial harvesting.

1.1.3 The wider livelihood context of NTFPs

The ecological (Taylor et al. (2006); Peres & Palacios (2007)), livelihood (Ambrose-Oji (2003); de Merode et al. (2004)), and economic (Iqbal, 1993) importance of Non-timber forest products has been well documented. However, the focus on NTFPs as a opportunity for livelihood enhancement through commercialisation has been some-

what misplaced (Lawrence (2003); Shanley et al. (2002)), diverting attention from existing local livelihood, resource access and subsistence issues. NTFP management has also been inappropriately influenced by timber management concepts, with too little attention given to the differences in economic scales and actors (J.Sayer, personal communication).

Regardless of their sustainable development merits, NTFPs are the main component to many rural subsistence strategies. NTFP extraction often provides for daily needs and makes up shortfalls in income rather than providing a path to socioeconomic advancement (Neumann & Hirsch, 2000). Thus NTFP extraction is often less about ‘poverty alleviation’ and more about basic survival (Tewari, 1993). However, in many forest areas the availability of these resources is increasingly undermined. Forest disturbances have significant implications for the use of NTFPs (Menton (2003); Plowden et al. (2003), Forest Monitor (2001)). There is little information on shifting NTFP dependancy following land use change, but as logging, fragmentation, fire, and the spread of invasive species become greater problems, an increasing number of rural communities may rely upon degraded forests for NTFP extraction. In addition, the access of many communities to forest resources is increasingly restricted (Hegde & Enters (2000); Rainforest Foundation (2007)), or resources threatened by changing harvesting patterns (Larsen & Olsen, 2007). The potential impacts of these changes in NTFP availability on the sustainability of existing livelihoods has received little attention, this failure may have substantial consequences for livelihoods and conservation. Where forest degradation and loss exacerbate poverty, further declines in standards of living are likely to result in increased forest degradation (Geist & Lambin (2002); Geist & Lambin (2003)).

1.1.4 NTFPs in India

In India, possibly more so than anywhere else, harvesting of NTFPs is both a fundamental livelihood activity (Kaushal & Melkani, 2005), and a conservation issue. The vast majority of India’s forests are inhabited, and high rural population density and limited agricultural land mean that these communities have few other options

but forest dependence. It is estimated that 50 million tribal people are largely dependant on NTFPs (Poffenberger (1996), Shiva (1993)), while a further 200 to 300 million non-tribals depend on NTFPs to a lesser degree (Shiva, 1993). For example, NTFPs contribute up to 60% of the cash income of tribal communities in Karnataka (Uma Shaanker et al., 2000), and studies indicate a similar dependence in other states (Ministry of Environment and Forests, 1998). The Forest Survey of India estimated that there are over 170,000 villages adjoining or inside forested areas (Forest Survey of India, 1999). Many of these villages have only marginal holdings, the land is low in productivity and water is scarce (Tewari, 1994). Furthermore, wildlife damage to crops is very high and dairying not very remunerative as transporting milk is costly and time consuming (Kaushal & Melkani, 2005). Inhabitants of these villages are largely scheduled castes and tribes, amongst the most marginal in Indian society. Of the nearly 68 million tribal population in India, a large proportion depend entirely on forests for their livelihood requirements including food, fodder, firewood and health-care. Most are in a state of extreme deprivation with regard to health, nutrition, and education. Against this backdrop, the role of India's forests in development assumes great importance.

Forest degradation in India is a matter of serious concern. Although the area under plantation forestry has increased, natural forest cover declined by 32% in the period 1990-2000 (FAO, 2003). Encroachment, the transfer of forest lands to other land uses (e.g. mining), fire, spread of invasive species, grazing and overexploitation are all contributory factors. Over 7500 plant species are used for medicinal purposes in India (Ved, 2003), with over 90% of this material collected from the wild (Uniyal et al., 2000). There is a growing indication of declining yields and unsustainable harvesting in several important NTFPs (e.g Uma Shaanker et al. (2001); Bhattacharya & Hayat (2004)). The need for assessing harvesting sustainability is certainly great, but of possibly greater urgency is halting the forest degradation which threatens the longer term ability of forests to provide NTFP resources.

1.1.5 Amla (*Phyllanthus emblica* and *Phyllanthus indofischeri*)

Known locally as ‘Nelli kai’, Amla fruits are collected from two conspecific species *Phyllanthus emblica* Linn. and *Phyllanthus indofischeri* Bennet. (Euphorbiaceae) (formerly *Emblica officinalis* Gaertner.) (Figure 1.1). *Phyllanthus emblica* is indigenous to tropical South East Asia and occurs mainly in the dry, or moist deciduous forests of central and south India (Uma Shaanker & Ganeshaiah, 1997). *Phyllanthus indofischeri* is an endemic, restricted to the Deccan Plateau including the Eastern Ghats of southern India (Ganesan, 2003). The two are found in distinct forest types; *P. emblica* dominates in deciduous forest reaching heights of up to 13 metres (although heights much larger have been reported (Morton, 1987)), and *P. indofischeri* in scrub forest growing up to five metres.

Amla is an important medicinal plant species, its fruit being used extensively in the traditional Indian medicine system, Ayurveda. The Amla fruits are also used for making pickles, jams and cosmetics and are an important food resource for a number of ungulate species (Prasad et al., 2004). Trade in the south of India is largely from wild extraction (Balachander, 2002) but several cultivars are grown in commercial plantations in Uttar Pradesh, Maharashtra, Gujarat, Rajasthan, Andhra Pradesh, and also in some areas of Tamil Nadu and Karnataka (Revathy, 2003). The area under Amla cultivation is expanding rapidly, from 3,000 ha in the early 80s to 50,000 ha in 2001 (Revathy, 2003). The economic value of Amla fruits (including processed products) was estimated at between 200 and 250 million Rupees (US\$5-6.25 million) in 1996, more recent figures are not available, but market potential is considered to be much higher (Balachander, 2002).

1.1.6 The Biligiri Rangaswamy Temple (BRT) Wildlife Sanctuary

The Biligiri Rangaswamy Temple (BRT) Wildlife Sanctuary (11.40° to 12.09° N, 77.05° to 77.15° E) is located in the state of Karnataka, on the easternmost ridge of the Western Ghats, a global hotspot of biodiversity (Figure 1.2). The 540 km² protected area ranges in altitude from 600 to 1800 m and consequently has a rich diversity of vegetation types ranging from scrub to evergreen forest (Ramesh (1989);



Figure 1.1: *Phyllanthus emblica* and *Phyllanthus indofischeri*
 Clockwise from top left, Amla fruits from *Phyllanthus indofischeri*, *Phyllanthus emblica* tree, harvested ittu Nelli kai *Phyllanthus indofischeri*.

Ganeshaiah & Uma Shaanker (1998)) (Appendix A). Dry deciduous and scrub forests predominate, together constituting almost 90% of the total area of the sanctuary (Ganesan & Setty, 2004). These two forest types contain a high proportion of commercially important species (Shanker et al., 1998) (e.g. *Terminalia chebula*, *Acacia sinuata*, and *Sapindus laurifolia*).

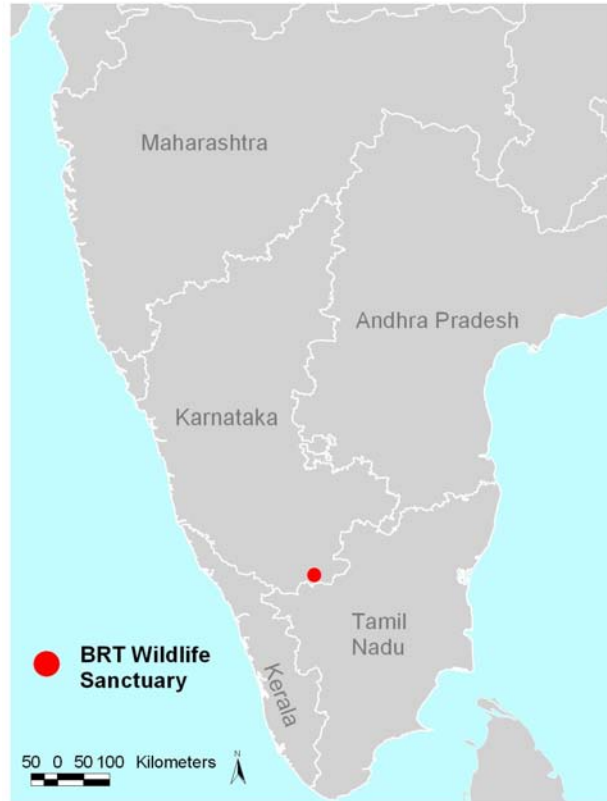


Figure 1.2: *The Biligiri Rangaswamy Temple (BRT) Wildlife sanctuary in Karnataka* (Source: ATREE)

The BRT sanctuary is home to the indigenous Soliga tribal community (Figure 1.3), as well as a smaller non-tribal population. Traditionally semi-nomadic, the Soligas practiced shifting cultivation (*Jhum*) but were settled into podus or villages and allotted land for agriculture when the area was declared a wildlife sanctuary in 1976. There are 57 tribal settlements within the limits or on the borders of the Sanctuary, a total human population numbering about 7500 (Hegde et al., 1996). In

addition to wage labour and limited agricultural production, these communities make a living through commercial extraction of a wide variety of NTFPs including fruits, honey, and lichens. Soliga dependence on NTFPs for household income is extensive, ranging from over 60% in the interior villages to about 30% in the fringe villages (Hegde et al. (1996); Uma Shaanker et al. (2002)).



Figure 1.3: *The Soligas and the BRT Wildlife Sanctuary*
Clockwise from top left, Soligas girls, Gombegallu podu (village), Amla harvesters, Malki betta ('hill') viewed from Jotti betta

1.1.7 Amla in the BRT Wildlife sanctuary

Eight NTFPs are harvested commercially within the sanctuary (Hegde et al., 1996), one of these is Amla. The Amla harvest takes place between December and February and lasts for about two weeks, with harvesters using a variety of techniques to collect fruit. Collection and sale is organised by a cooperative society (1.1.8) with family-based groups harvesting from pre-selected areas and returning to a central location at the end of the day for weighing and collection of fruits. The Soligas prefer the larger ‘Ittu’ Nelli (*P. indofischeri*), but fruits from both species are pooled for trading. Amla contributes a significant proportion of cash income of the resident Soligas (Hegde et al., 1996) and has been subject to livelihood development initiatives (Kaushal & Melkani, 2005) including post-harvest processing and product marketing (1.1.8). Between 104 and 1492 tons are collected per year, earning more than 42% of the annual USD 112,000 of NTFP income in BRT (Setty, 2004). Between 39 and 52% of total fruit production is harvested annually (Ganesan & Setty, 2004).

Assessments using estimated harvest rates and data from demographic studies indicate that current collection levels are sustainable (Setty et al. (2008); S Setty personal communication). However, many other factors impact the Amla population status at BRT. Ganesan & Setty (2004) inferred regeneration status from assessments of population structure. Finding a dearth of saplings and small adult classes of *P. emblica*, they considered disturbance, including fire, a likely cause of higher sapling mortality in *P. emblica*. Fire has also been suggested as having a detrimental impact on fruiting and therefore regeneration (Sinha & Brault, 2005), and there are indications that the prevalence of *L. camara* may have additional negative implications for Amla regeneration (Hiremath & Sundaram, 2005).

Many Amla trees are heavily infested by the mistletoe *Taxillus tomentosus* (Roth.) Var. Tiegh (Loranthaceae). Trees suffering from heavy infestation show defoliation and death of branches distal to the infection site, and many trees die as a consequence of these infections (Appendix B). Previous studies have documented the impacts of mistletoe infection on Amla including significant reductions in growth (Sinha & Bawa, 2002) and productivity (Sinha & Bawa (2002); Setty (2004)) and increased tree mor-

tality (Setty, 2004). These studies have not, however, attempted to establish the full extent, or severity, of the problem within the sanctuary. The livelihood implications and the effectiveness of existing, or potential future, management responses has also not been investigated. Amla harvesters believe that this mistletoe has undergone a recent population expansion due to change in the fire regime. Mistletoe infection has also be identified as a management issue at other sites (Uma Shaanker et al., 2002). *Taxillus tomentosus*, referred to locally as Uppilu, is found on several other tree species in BRT and more widley is distributed across India and Sri Lanka (Keeble, 1896). *Phyllanthus* trees also suffer from the effects of a bark eating caterpillar (*Indarbela* sp), which is known to affect several important fruit trees in the tropics (Morton, 1987).

1.1.8 Management history and institutions

Many formal and informal institutions influence the use of Amla in BRT, and the management of the sanctuary more generally. The Soligas only became economically dependent on NTFP sale when the BRT sanctuary was established and their traditional practices of shifting agriculture and hunting prohibited. Therefore, monitoring systems and self-regulation of extraction levels have largely given way to harvesting regulated by market demand for products Setty et al. (2008). In terms of formal organisations, the BRT sanctuary is state owned forest land managed by the Karnataka Forest Department (KFD) according to the National Forest Policy of India 1988. Current management is mainly for conservation purposes but retains some emphasis on revenue generation. Two NGOs; Vivekananda Girijana Kalyana Kendra (VGKK) a tribal welfare and development organisation, and the Ashoka Trust for Research in Ecology and Environment (ATREE), additionally have significant direct and indirect influences on the areas management.

The commercial collection of NTFPs has been permitted since 1976 solely as a consequence of its livelihood importance. This occurs under the agency of tribal co-operatives, called Large-Scale Adivasi Multipurpose Societies (LAMPS), established in India in the 1970s in states with significant tribal populations. The Karnataka

Government established LAMPS, including three associated with the BRT wildlife sanctuary, with the purpose of ensuring stable remuneration for the NTFP collection by Soligas. Additionally, in 1995, VGKK and ATREE were involved in an NTFP enterprise which aimed to increase NTFP income and thereby raise the Soligas economic stake in the sanctuaries biotic resources Setty et al. (2008). Participatory resource monitoring activities linked to this enterprise evolved over a 10 year period to include pre- and post-harvest meetings, participatory estimation of productivity and extraction rates, and assessment of regeneration status for several NTFPs including Amla. Although the participatory monitoring has demonstrated success Setty et al. (2008) the livelihood benefits are yet to be demonstrated.

Although the Soligas have held usufruct rights for NTFP collection within the sanctuary, these rights have become less certain in recent years, a consequence of the implementation by the Karnataka state forest department of an amendment to the Wild Life Protection Act 1972. This 2002 amendment prohibited extraction of forest resources for ‘commercial use and consequently in recent years an NTFP harvesting ban has been imposed in several Indian states including Karnataka. The ban was strictly enforced in BRT only after April 2006 but initial assessments suggest its socio-economic impact has been significant (Kalpavriksh, 2007a). In March 2007 several serious incidents of fire occurred within the sanctuary. The forest department claimed these occurrences were the result of Soligas retaliation against the NTFP collection ban. The soligas denied these accusations and cited the lack of preventative measures taken by the forest department (Kalpavriksh, 2007a). These incidents generated considerable conflict and tension and although the exact causes and the damage that resulted are still yet to be established, they highlight the need to reconsider current management approaches in the area.

This site currently faces complex and multiple management challenges, specifically significant recent ecological change within a complex social and institutional setting. Both the invasive shrub *Lantana camara* (L.), which has become widespread and now dominates the forest understory (Murali & Setty, 2001), and the current fire regime, threaten forest structure and function (Barve et al. (2005); Kodandapani et al. (2004)) with implications for both conservation and local livelihoods (Ganesan

& Setty, 2004). Local and institutional opinions conflict over the causal factors behind these and other recent changes and the appropriate management responses.

1.2 Aim and objectives

To investigate the threat posed by mistletoe infection to the sustainable harvesting of Amla and to examine how this threat might be tackled in management.

This aim will be addressed through the following objectives:

- Assess the prevalence and intensity of mistletoe infection within the Amla population of BRT
- Analyse the mechanisms behind these infection patterns and consider their implications for the implementation of mistletoe control measures
- Consider the implications of observed infection patterns for the Amla population and for harvesters
- Assess the effectiveness of local and institutional management approaches to mistletoe infection of Amla
- Assess the traditional ecological knowledge of Amla harvesters in terms of additivity and cost effectiveness, and its possible role in management
- Evaluate the potential for participatory active adaptive management of mistletoe infection of Amla

1.3 Thesis outline

- Chapter two: *Ecology and management of mistletoe infection in a non-timber forest resource* (published in *Forest Ecology and Management*)

This chapter describes the prevalence and intensity of infection of the mistletoe

Taxillus tomentosus across fourteen host species in the BRT wildlife sanctuary. It identifies *Phyllanthus emblica* and *Phyllanthus indofischeri* as the main host species, demonstrating that mistletoe infection is a serious threat to the sustainable harvesting of Amla. It experimentally investigates two methods of managing mistletoe infection, highlights deficiencies in each, and makes suggestions for developing more effective approaches.

- Chapter three: *Seed deposition and the distribution of mistletoe on Phyllanthus emblica and Phyllanthus indofischeri*

This chapter investigates the spatial distribution patterns of mistletoe at three scales; tree, patch and landscape. It looks at the roles of variation in seed dispersal and host susceptibility, as well as the role of forest fire and infection-induced host mortality, in generating these patterns. The findings are considered in the context of their implications for management.

- Chapter four: *Assessing the contribution of traditional ecological knowledge in the management of mistletoe infection as a threat to Amla*

This chapter assesses the concordance between traditional ecological knowledge and data from ecological studies, highlighting the additional understanding that TEK provides with respect to mistletoe infection. It investigates the impact of infection on the Amla harvest in recent years, comparing official harvest records with harvester accounts of resource abundance. Finally, this chapter highlights that monitoring of yields in isolation is insufficient to assess the sustainability of harvested resources.

- Chapter five: *Participatory Active Adaptive Management, is it feasible? An NTFP case study*

This chapter reviews the theory of adaptive management, considers patterns in its application to natural resource management, and examines the barriers identified to its implementation. The current uncertainties involved in managing mistletoe infection are outlined, using a conceptual model to bring together the findings of the previous three chapters with information from published stud-

ies. The potential for resolving these uncertainties, through the participatory implementation of an active adaptive management programme is explored.

- Chapter six: *Discussion*

This last chapter places the main findings of the thesis in a wider context, outlining the increasing challenges of managing natural resources from both the ecological and social perspectives.

CHAPTER 2

ECOLOGY AND MANAGEMENT OF MISTLETOE INFECTION IN A NON-TIMBER FOREST RESOURCE

2.1 Outline

Sustainable harvesting of non-timber forest products (NTFPs) and associated livelihood security can be affected by forest disturbances additional to harvesting. This chapter examines the infection of *Phyllanthus emblica* and *Phyllanthus indofischeri* by a hemiparasitic mistletoe and investigates two alternative management approaches. Forest surveys, mistletoe removal experiments and mistletoe seed deposition surveys on cut and intact branches were employed to assess the prevalence of mistletoe infection, characteristics of infection in relation to resource value and the appropriateness of local versus institutional management approaches. Results suggest that mistletoe infection is widespread in the BRT wildlife sanctuary, affecting over half of the Amla population, and in particular, those reproductive trees important for population persistence. Infection characteristics and resource values differ between the two *Phyllanthus* species, having significant implications for Amla collectors. Institutional perspectives on the management of this species conflict with local practice. Cutting of branches by collectors has previously been considered destructive, but may in fact have management benefits in terms of increased productivity through coppicing and reduced risk of mistletoe infection. However, neither mistletoe removal by hand, nor branch cutting appear to offer a viable control strategy in isolation. New approaches to management are needed to safeguard the role of this resource in local livelihoods.

2.2 Introduction

The impacts of over-harvesting on non-timber forest product (NTFP) sustainability have received considerable attention (Boot & Gullison (1995); Bernal (1998); Peres & Lake (2003); Ticktin (2004); Endress et al. (2006)), but despite growing threats to forest habitats and increasing human disturbance in many exploited forests (e.g. Laurance (2006); FAO (1997); Muller-Landau (2007)) resource susceptibility to natural or semi-natural threats such as fire, diseases and invasive species are rarely integrated with assessments of over-harvesting or other human impacts (for exceptions see Sinha & Brault (2005); Ticktin et al. (2006)). Studies also still lack an integrated perspective with regard to other forest uses. For example, activities such as timber harvesting may influence resource availability and management outcomes, of particular importance where the same species has both timber and non-timber uses (Shanley et al. (2002); Menton (2003)).

In the BRT sanctuary, many Amla trees are heavily infested by the mistletoe *Taxillus tomentosus*. Trees suffering from heavy infestation show defoliation and death of branches distal to the infection site, and many trees die as a consequence of these infections. Previous studies have documented the impacts of mistletoe infection on Amla including significant reductions in growth (Sinha & Bawa, 2002) and productivity (Sinha & Bawa (2002); Setty (2004)) and increased tree mortality (Setty, 2004). These studies have not, however, attempted to establish the full extent or severity of the problem within the sanctuary and its implications for livelihoods, or to investigate the effectiveness of current management responses.

The Karnataka Forest Department (KFD) advocates removal of mistletoes by hand, and has promoted this strategy to local harvesters. Although targeted at the annual Amla harvest, the KFD does not offer collectors any payment for the time or effort invested in mistletoe control. Harvesters consider removal by hand to be both impractical and ineffective. Hand removal is physically difficult and time intensive, and mistletoes frequently re-sprout from tissue remaining within the infected branch (personal observation).

Many fruit collectors cut branches while harvesting and claim that, in addition

to facilitating collection, this strategy has management benefits. Collectors cite enhanced productivity in future years through coppicing and reduced risk of mistletoe infection from the removal of outer branches, the preferred perching sites for seed dispersing birds. Indeed, some harvesters specifically cut branches to remove mistletoe infections, both during the harvest period and from high yielding trees at other times (personal communication with Amla harvester). The KFD has strongly discouraged the cutting of branches during the Amla harvest, believing that this practice leads to tree mortality, consequently tensions have arisen over this issue. Previous studies have suggested that this harvesting method is, indeed, destructive, and is motivated by a desire to maximise short term economic gains in terms of income from fruit collection (Sinha & Bawa (2002); but see Setty (2004)). There has, however, been little consideration of alternative motivations for this behaviour. This harvesting technique has also been suggested to increase the risk of mistletoe infection by providing sites for germination and establishment of mistletoe seeds on the cut surfaces of branches (Sinha & Bawa, 2002).

The prevalence and intensity of mistletoe infection across fourteen host species is described, and the distribution of mistletoe infection in relation to host species, and additional for individual tree attributes in Amla assessed. Experimental removal of mistletoes, surveys of mistletoe seed deposition on cut and uncut branches, and assessment of coppicing rates of cut branches were used to explore the following questions; (1) are *P. emblica* and *P. indofischeri* the main hosts of *T. tomentosus*? (2) Is mistletoe infection equally prevalent and intense in these two species? (3) How does infection differ with height and age of trees? (4) Is hand removal of mistletoes effective? (5) Does branch cutting reduce risk of infection and is this an improvement on removal by hand? In answering these questions, the implications of observed infection patterns for the Amla resource are considered, specifically for population regeneration, and a clearer understanding of the limitations of current management approaches is developed.

2.3 Methodology

2.3.1 Host-mistletoe surveys

Between September 2005 and April 2006, sixty 500 m² (10 x 50 m) plots were surveyed for tree abundance and infection characteristics, thirty in deciduous, and thirty in scrub forest. Plot locations were selected using a staggered hierarchical sampling regime in a nested forest survey (Pettitt & McBratney, 1993) (this design being necessary for the investigation of spatial variance in infection, Chapter 3, Section ??). All trees greater than 4 cm diameter at breast height (DBH) within each plot were sampled. Four centimetres was deemed to be a reasonable cut-off point following a preliminary survey of parasitised trees, as mistletoes were not observed on trees smaller than 4.2 cm DBH. Positive infection status was assigned to individuals bearing one or more mistletoes, and the number and size of mistletoes were recorded. Mistletoes were counted by thoroughly inspecting tree branches with binoculars, and were classified by sight into four, size-related, morphological groups, following (Norton & Ladley, 1997): juvenile (one or two small shoots or twigs), small (several twigs present but no branches >one cm diameter), medium (at least one prominent branch present with smaller side branches), and large (several large axial branches with numerous side branches). Height of each tree was measured with a clinometer, or estimated visually where forest structure required.

Infection characteristics were compared across host species. Relative host infection (the number of infected trees of each host species out of all infected trees, of all hosts) was compared with relative host abundance (the proportion of each host species out of all hosts) using a Fisher test (Crawley, 2007). Prevalence of infection (i.e. proportion of the population of each host species infected) was compared across species using a binomial proportions test (Crawley, 2007).

Probability and intensity (number of mistletoes) of infection in *P. emblica* and *P. indofischeri* were investigated using generalized linear mixed effects models with Binomial and Poisson error structures respectively and maximum likelihood. Mixed effects models were used to take account of the spatial non-independence of trees

within plots. Models were run within the statistical software R, version 2.4.1., using the lmer function from the Matrix package (R Development Core Team, 2007). Probability and intensity were modelled as a function of species (a two-level factor), height (m) and age (m/DBH) as fixed effects with plot specified as a random effect. Because many trees have been cut, height does not provide a reliable indicator of tree age, therefore DBH was used as a proxy for age. Multi-stem individuals (59 out of 227 trees) were excluded as DBH could not be used as a reliable indicator of age in these trees (K. Kirby, personal communication). Main effects and interactions between explanatory variables (fixed effects) were considered and quadratic terms fitted for height and DBH. Model simplification was conducted using analysis of deviance with Chi-squared goodness of fit tests, deleting the least significant terms. Model checking was undertaken with plot fitted as a fixed effect using the binned plot function from the arm library (R Development Core Team, 2007) for the binomial model and standard checks of residual deviance for the count model. Model deviance components from the generalized linear models were partitioned using ANOVA to assess the relative influence of individual and area based characteristics on variance in probability and intensity of infection. The number of dead mistletoes, as a proportion of all mistletoes sampled, was compared between the two species using a binomial proportions test (Crawley, 2007).

2.3.2 Experimental removal of *Taxillus tomentosus*

Between March and May 2006, a total of 100 mistletoes were removed by hand from 11 *P. indofischeri* and seven *P. emblica* trees to study regrowth following removal. Trees were selected opportunistically in two sites, simulating the behaviour that might be expected of collectors, one site in scrub forest (*P. indofischeri*) and one in deciduous forest, (*P. emblica*). These two locations were chosen on the basis of mistletoe abundance and accessibility for repeated monitoring. Only some of the mistletoes from each tree were removed; those chosen for removal were selected to represent all four size classes and were distributed throughout the canopy with respect to height and aspect. Removal was carried out by hand by Soliga field assistants in such a

manner as to simulate removal by Amla harvesters and the mistletoe scar was tagged for monitoring.

At monthly intervals for 12 months, the occurrence and extent of regrowth of the mistletoe from its removal site was recorded. The condition of the haustorial connection was noted as dead or alive, and regrowth measured (number and length of branches) where it had occurred. Condition of the host branch (both distal and proximal to the infection site) was scored for ‘health’ (i.e. dead, defoliated, bearing leaves, or bearing flowers/fruits i.e. reproductive) so that any improvement in health following removal could be identified. Diameter of the host branch proximal to the site of infection was also recorded and re-measured at monthly intervals.

Mistletoe regrowth and branch recovery were modelled as binary responses using generalized linear mixed effects models as before (Section 2.3.1). Models were fitted with tree species (*P. emblica* or *P. indofischeri*), size class of the mistletoe (one to four) and tree branch diameter (cm) as explanatory variables and tree as a random effect, using Binomial error structures and maximum likelihood following the collapsing of factor levels for mistletoe size class from four to two levels. Model checking was carried out with tree as a fixed effect using the ‘binned plot’ function from the *arm* library (R Development Core Team, 2007).

2.3.3 Mistletoe seed deposition and Amla coppicing rates

During February 2007, 80 *Phyllanthus* trees (43 *P. emblica* and 37 *P. indofischeri*) that had been recently cut (i.e. in the current harvesting season or that of the previous year) were selected and mistletoe seeds counted on branches both proximal to the cutting site (i.e. the former ‘stump’) and on the arising coppice. Less recent cutting was also found on some of the selected trees. Fieldwork was conducted with a Soliga field assistant (also an Amla collector) allowing us to obtain estimates of time since cutting (0, 1, 2 or 3 years) which, along with measurement of coppice length, provided estimates of coppicing rates. The field assistant’s estimates had been previously validated with those of other collectors. Seed counts were compared between the cut and coppiced portions of branches using pair-wise comparisons in a

Wilcoxon signed rank test with a continuity correction (Crawley, 2007) and between *P. emblica* and *P. indofischeri* using analysis of variance on mean counts.

2.4 Results

2.4.1 Host species, mistletoe distribution and infection characteristics

Fourteen different species were identified as host trees (Table 2.1). Although prevalence of infection was significantly different among species ($X^2 = 1130$, d.f. = 13, $p = < 0.001$.), relative host infection did not reflect relative host abundance (Fishers exact test, d.f. = 13, $p = < 0.001$). *P. emblica* and *P. indofischeri* were both infected at considerably higher frequencies than would be expected from their abundance within the forest community (Table 2.1). Over half of all sampled *Phyllanthus* bore mistletoes, suggesting infection is widespread within the Amla population of the BRT. Probability and intensity of infection were significantly greater in *P. emblica* (Table 2.2).

Probability and intensity of infection also increased with height for both species, although less markedly in *P. emblica*. Intensity of infection increased with DBH for all but the largest trees (Table 2.2). Partitioned deviance components from the generalized linear models indicated that the plot in which a tree was located had a greater influence on its probability and intensity of infection (explaining 69.2% and 66.3% of deviance, respectively) than characteristics of the tree itself i.e. species, height or DBH. A tree found in a plot where prevalence of infection was particularly high was more likely to be infected and to bear larger numbers of mistletoes, irrespective of its size, age or species. The number of dead mistletoes found, as a proportion of the total sampled, was similar between both *Phyllanthus* species.

2.4.2 Effects of experimental removal of *T. tomentosus*

Hand removal led to branch recovery in only 14% of instances, and in all cases recovery was proximal to the site of infection. Of the mistletoes removed, 49% showed signs of regrowth after 12 months, suggesting hand-removal benefits are temporary as

Table 2.1: Host species of *Taxillus tomentosus* within BRT Sanctuary

N , number of sampled trees; N_i , number of infected trees; RHA, relative host abundance (the proportion of each host species out of all host trees available); RHI, relative host infection (the number of infected trees of each host species, out of all infected trees, of all hosts), Prop, proportion of total sampled *Taxillus tomentosus* population on host species.

Species (local name)	Family	N	N_i	Prevalence	Intensity (mean $\pm$ S.D.)	RHA	RHI	Prop
<i>Anogeissus latifolia</i> (Beija)	Combretaceae	1233	14	0.011	1.5 $\pm$ 1.31	42.2	8.6	0.041
<i>Randia dumetorum</i> (Karre)	Rubiaceae	721	3	0.004	1.33 $\pm$ 0.43	24.6	1.8	0.008
<i>Terminalia crenulata</i> (Matti)	Combretaceae	261	2	0.008	2.0 $\pm$ 4.1	8.9	1.2	0.008
<i>Pterocarpus marsupium</i> (Honne)	Fabaceae	148	3	0.020	2.33 $\pm$ 1.51	5.1	1.8	0.014
<i>Grewia tiliifolia</i> (Daddasalu)	Tiliaceae	137	3	0.022	1.0 $\pm$ 0	4.7	1.8	0.010
<i>Phyllanthus indofischeri</i> (Ittu nelli)	Euphorbiaceae	118	45	0.381	3.07 $\pm$ 2.47	4.0	27.6	0.270
<i>Phyllanthus emblica</i> (Nai nelli)	Euphorbiaceae	109	70	0.642	4.01 $\pm$ 4.57	3.7	43.9	0.490
<i>Bridelia retusa</i> (Sirome)	Euphorbiaceae	45	1	0.022	2.0	1.5	0.61	0.004
<i>Diospyros melanoxylon</i> (Tubare)	Ebenaceae	43	1	0.023	1.0	1.5	0.61	0.002
<i>Stereospermum personatum</i> (Padure)	Bignoniaceae	38	1	0.026	1.0	1.3	0.61	0.002
<i>Dalbergia lanceolaria</i> (Buluga)	Fabaceae	34	10	0.294	2.4 $\pm$ 1.51	1.2	6.1	0.047
<i>Glochidion zeylanicum</i> (Anase)	Euphorbiaceae	19	7	0.368	2.0 $\pm$ 0.99	0.65	4.3	0.027
<i>Dalbergia latifolia</i> (Bite)	Fabaceae	15	2	0.133	4.5 $\pm$ 3.54	0.51	1.2	0.018
<i>Wendlandia thyrsoides</i> (Koli)	Rubiaceae	4	1	0.25	1.0	0.14	0.61	0.002

Table 2.2: *Influence of tree characteristics on probability and intensity of mistletoe infection in *P. emblica* and *P. indofischeri**

Estimates from mixed effect models (including back-transformed (bt) estimates of predicted probabilities of infection and mistletoe counts respectively), all terms significant to $p < 0.01$, * magnitude due to DBH in metres.

Host attributes	Probability of infection ($n = 167$)		Intensity of infection $n = 167$)	
	Estimate (bt)	Std. Error	Estimate (bt)	Std. Error
<i>P.indofischeri</i>	-6.06 (0.002)	1.79	-7.34 (0.0007)	1.23
<i>P.emblica</i>	5.32 (0.32)	1.93	2.96 (0.013)	1.28
Height - <i>P.indofischeri</i>	1.09 (0.75)	0.36	0.65 (1.9)	0.22
Height - <i>P.emblica</i>	-0.92 (0.54)	0.37	-0.58 (1.1)	0.22
DBH	-	-	46.53* (1.6e+20)	6.48
DBH^2	-	-	-115.33* (8.2e-51)	17.12

well as marginal (Figure 2.1). No difference in rates of mistletoes regrowth or branch recovery were found between the two *Phyllanthus* species (Table 2.3). Juvenile and small mistletoes were significantly more likely to re-grow following their removal than larger mistletoes (Table 2.3). Branch recovery (a notable improvement in branch health relative to original pre-removal condition), was more likely for branches from which larger mistletoes had been removed (Table 2.3). Mistletoe growth was characterised by new shoots sprouting 10 to 30 cm behind the original infection site, often from epicortical roots. Ninety-four percent of regrowing mistletoes did so within four months of initial removal (Figure 2.1). After 12 months, 14% of removed mistletoes had regrown to a size sufficient to resume fruit production and hence become sources of new infections (Figure 2.2).

2.4.3 Seed deposition and coppicing rates

Seed deposition was significantly greater on new coppice than on the branch sections proximal to cutting sites (Wilcoxon signed rank test, d.f. = 78, $p < 0.001$). In fact, very little seed deposition was observed on cut branches and then only on heavily infected trees. Seed deposition was greater on *P. indofischeri* than on *P. emblica*

Table 2.3: *Probability of mistletoe regrowth and Phyllanthus branch recovery following mistletoe removal*

Collapsed factor levels for mistletoe size classes: Large (medium and large) and Small (juvenile and small). Estimates from mixed effect models (including back-transformed (bt) estimates of predicted probabilities of mistletoe regrowth and branch recovery), all terms significant to $p < 0.01$.

	Mistletoe regrowth ($n = 98$)		Branch recovery ($n = 98$)	
	Estimate (bt)	Std. Error	Estimate (bt)	Std. Error
Size class - small	0.563 (0.64)	0.384	-2.991 (0.048)	0.723
Size class - large	-0.954 (0.4)	0.442	1.689 (0.21)	0.793

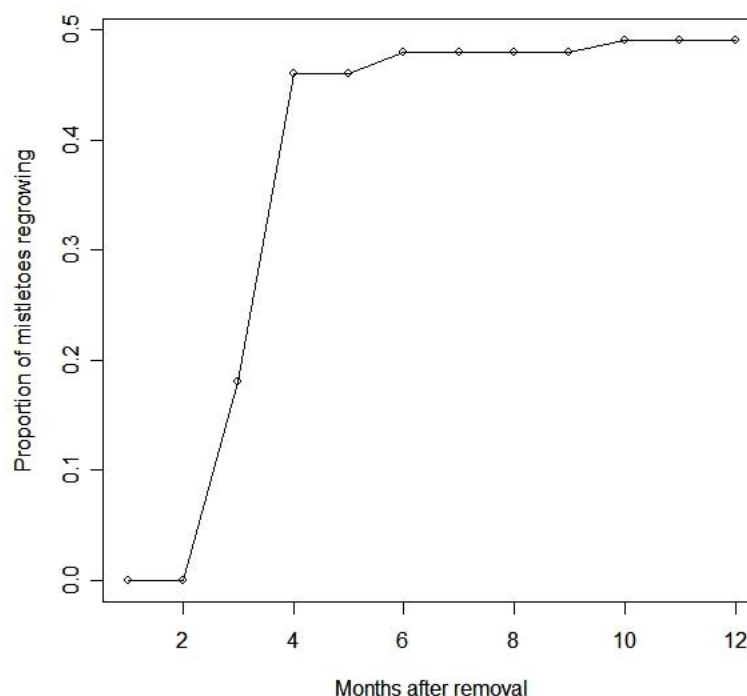


Figure 2.1: *Mistletoe regrowth following hand removal*
Proportion (cumulative) of mistletoes regrowing over a 12 months period following experimental hand removal, $n = 49$.

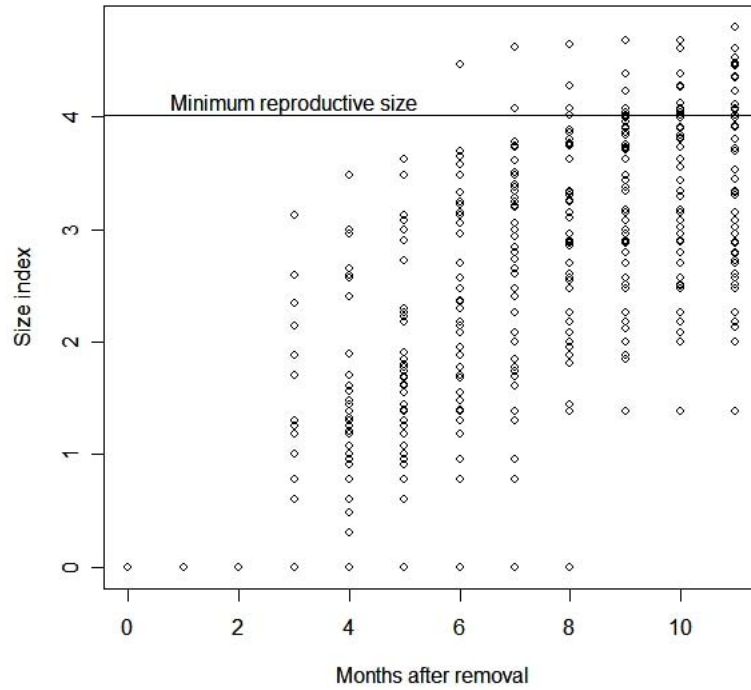


Figure 2.2: *Extent of regrowth in re-sprouting mistletoes*

Mistletoe size is presented as an index (mistletoe height x width), indices based on morphological traits have been used previously to assess age and size of mistletoes (e.g. Dawson et al. (1990); Norton & Ladley (1997)), the log of this index was used for presentation purposes. Regrowth over 12 months after initial removal, $n = 49$. Horizontal line indicates minimum observed size of reproductive mistletoes (based on phenological monitoring data from additional field studies (Chapter 3, Section ??)).

(d.f. = 77, $p < 0.001$). Coppicing rates were rapid, with branches cut during the recent harvesting season (i.e. about 4 months previously) having already attained a length of up to one metre (Figure 2.3, Appendix B).

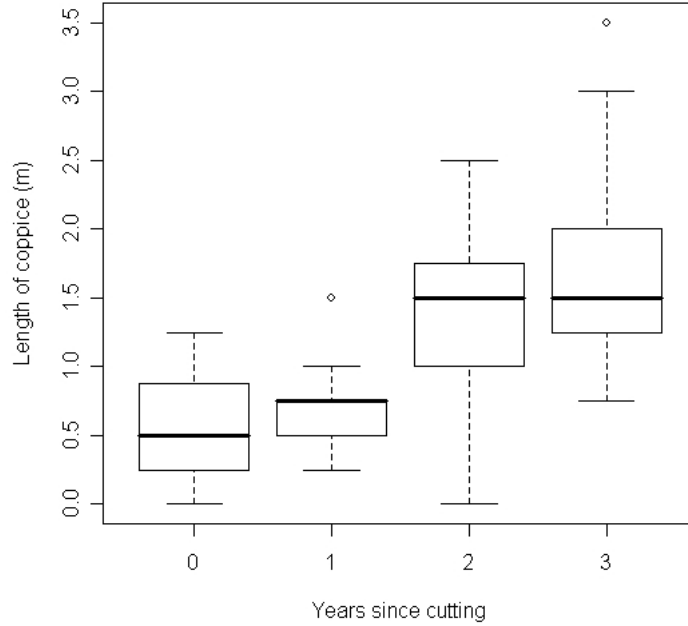


Figure 2.3: *Coppicing rates of P. emblica and P. indofischeri*

Rate as length of coppice in relation to years since cut. Zero years since cutting refers to branches cut in the most recent harvest, approximately four months prior to measurement.

2.5 Discussion

2.5.1 Mistletoe abundance and distribution

The results indicate that the mistletoe *T. tomentosus* is not host specific, but *P. emblica* and *P. indofischeri* appear to be the primary host species at BRT; 82% of the sampled mistletoe population being found on these two species. This pattern, a single common host (in this case two conspecifics), and a number of other less frequently

parasitised hosts, is common among mistletoes (Norton & Carpenter, 1998). The tendency of generalist mistletoe species to infect several closely related hosts (same genus or family) was not observed in this case (Hawksworth & Wiens, 1996). Levels of specificity are often considered to be lower in tropical parasite systems (Norton & Carpenter, 1998). Further study is required to establish the role of these additional host species in the maintenance and spread of infection in the Amla population, specifically for the planning of effective control strategies.

Prevalence of mistletoe in *P. emblica* was shown to be nearly twice that observed in all other species (including *P. indofischeri*). This finding supports previous studies (Setty (2004); Sinha (2000)) and harvesters also cite infection in scrub forest (*P. indofischeri*) as being both more recent and less intense (Chapter 4). Such patterns may be a consequence of three non-exclusive processes: (1) differential exposure (i.e. *P. emblica* receives more seeds as a consequence of the preferences of seed dispersing birds, which itself will have a number of different explanatory factors); (2) differential susceptibility (i.e. a greater proportion of mistletoe seeds establish successfully on *P. emblica*); or (3) differential survival of mistletoes due to host or environmental characteristics (i.e. post-establishment mortality of mistletoes is lower on *P. emblica*).

Observed infection patterns may reflect the perching preference of seed dispersing birds for trees of a specific structure or trees found in a particular habitat (Overton (1994); Martinez del Rio et al. (1996); Lei (1999); Aukema & Martinez del Rio (2002)). Vegetation characteristics suggest deciduous forest is less moisture-stressed than scrub forest and work on other host-mistletoe systems suggests mistletoe distribution can be influenced by water status (Norton & Stafford-Smith, 1999), many mistletoes having high transpiration rates (Lamont (1983); Ullmann et al. (1985); Davidson et al. (1989)). The ‘passive theory’ of mistletoes nutrient uptake suggests mistletoes are already profligate in their water use and may therefore exacerbate water stress of their hosts in dry environments (Goldstein et al., 1989). High mortality in desert *Acacias* had previously been linked to an increase in mistletoe infection, however, by investigating both host and mistletoe water potentials and nitrogen concentrations, Bowie & Ward (2004) instead provided evidence for the impact of host water status on mistletoe growth and flowering. Although *P. indofischeri* has greater exposure to

infection (seed deposition rates were greater on *P. indofischeri*), mistletoes may face higher mortality rates in drier scrub forest, resulting in the observed lower infection levels. However, counts of dead mistletoes as a proportion of all mistletoes sampled were comparable between scrub and deciduous forest.

Taken together, these findings suggest that variation in exposure, as well as in host suitability (susceptibility and acceptability) may be important in the infection patterns observed. A direct comparison of mistletoe mortality rates between the two *Phyllanthus* species (early mortality would lead to many mistletoes going unobserved), along with an investigation of the factors influencing exposure to infection via the deposition of mistletoe seeds, is required to provide support for this hypothesis. Harvesters suggest that mistletoes have spread more recently into scrub forest from deciduous forest, the more recent infection of *P. indofischeri* would account for the lower levels of infection observed in this species. At present, the validity of these alternative explanations for the differences in infection among the two *Phyllanthus* species cannot be determined, but distinguishing between these processes is likely to be useful in the development of strategies aimed at limiting the spread and intensification of mistletoe infection in the Amla population.

Mature trees were found both more likely to be infected and to have higher parasite loads. Several previous studies suggest dispersing birds prefer taller trees and hence disperse mistletoe seeds disproportionately into such trees (Martinez del Rio et al. (1996); Aukema (2001)). Additionally older trees may accumulate infections with age (Overton, 1994). Age-specific differences in host suitability or in the numbers of seeds dispersed to trees may also exacerbate this relationship (Roxburgh & Nicolson, 2008). Assessment of harvesting impacts on other plant NTFPs has already shown that survival of the largest size classes contribute most to population growth, while seed survival contributes very little (e.g. Peters (1992); Olmsted & Alvarez-Buylla (1995); Joyal (1996); Ratsirarson et al. (1996); Bernal (1998); Zuidema (2000); but see Peres et al. (2003)). Reducing adult mortality should therefore be a priority in the management of many NTFPs (Ticktin, 2004). The greater infection likelihood of mature *Phyllanthus* trees may have important consequences for population growth rates, and may possibly be an important cause of reported decline in *Phyllanthus*

populations in BRT (Ganesan & Setty, 2004).

Not only do mistletoes increase mortality of mature trees (Sinha (2000); Setty (2004)), but in the earlier stages of infection they reduce or eliminate fruit production. Sinha & Bawa (2002) found significant negative correlations between mistletoe load and fruit production in *P. emblica* and *P. indofischeri*. There is also evidence to suggest that mistletoe infection may impact fruit and seed weights (Setty, 2004), this may reduce the viability of seeds, therefore affecting reproductive rates still further. Previous studies suggest a mortality rate in infected trees of 14% per year (Setty, 2004). With half of the Amla population affected, and some harvesters indicating that as much as 50% of the Amla population has already been lost as a result of mistletoe-induced mortality, the implications for resource sustainability and livelihoods may be considerable. Regeneration problems have already been highlighted for *P. emblica* (Ganesan & Setty, 2004), and mistletoe infection may pose an additional threat to population persistence. In this NTFP system, fruit harvesting has previously been considered to threaten the local population viability of these two species (e.g. Sinha (2000)). Based on the extent of infection in the population combined with previously documented severity of impacts and reports from harvesters, mistletoe infection may pose a substantially more serious threat in terms of population stability and persistence than any reported demographic impacts from harvesting.

Although some mistletoes exist at high prevalence with no apparent impact on host populations (Roxburgh & Nicolson, 2008), many have negative impacts on their hosts. Such impacts include reductions in growth and fecundity, defoliation, die-back of branches (Hawksworth, 1983), along with stem and crown deformity (Robinson et al., 2002) and increased susceptibility to disease (Geils, 2002). Despite these, infection induced host mortality is generally rare (Aukema, 2003), occurring in instances where host trees are already stressed (e.g. Spurrier & Smith (2007)), or where certain constraints on the mistletoe population have been relaxed and they have become super-abundant (D.M Watson, personal communication). Increasing mistletoe populations have been documented in several locations throughout the world including Australia and North America (e.g. Reid (1977); Norton & Reid (1997)). In Australia and New Zealand, anthropogenic changes, including habitat fragmentation

and altered fire regimes, as well as a reduction in the density of natural herbivores have been implicated in these changes (Lavorel et al. (1999); Reid (1995)). A similar situation in BRT may have resulted in a larger mistletoe population, the increased intensity of infections leading to tree mortality. Alternatively, mortality may be a consequence of the interaction of mistletoe infection with additional, but unidentified environmental stressors, both possibilities require further consideration.

2.5.2 Effectiveness of management approaches

While clearance of mistletoes by hand generally involves removal of all stems and leaves, in many cases, due to a strong connection with the host tree, parts of the mistletoe haustorium remain within the host branch. Consequently, any benefits of hand removal are also likely to be temporary due to the high incidence of mistletoe regrowth. Results suggest that decline in host health as a consequence of mistletoe infection may be reversible following hand removal, but without recovery of previously infected branches the benefits for fruit production are less certain. Rates of regrowth following removal do not appear particularly rapid and given the time taken for mistletoes to reach reproductive maturity (personal observation), removal by hand could interrupt mistletoe fruit production for up to two years. Hand removal is, however, physically difficult and time consuming and, given the prevalence of infection in the Amla population, represents a considerable investment of time and labour which is likely to be prohibitive.

There are two management techniques favoured by harvesters; cutting of infected branches and controlled burning. Other authors have previously documented traditional knowledge and management strategies applied by the Soligas in their use of forest resources (e.g. Uma Shaanker et al. (2004)). Although probably motivated primarily by harvesting benefits (i.e. increased yield per unit effort), cutting of branches may actually have management benefits rather than being destructive as has been previously stated. Recovery of fruit production following cutting has been qualitatively demonstrated (Sinha (2000); Setty (2004)) and harvesters indicate that although one fruiting season may be lost, coppicing results in increased productivity in subsequent

years, several new fruiting branches arising from the cutting site.

Cutting of infected branches is an effective means of removing mistletoes. The results suggest that this behaviour may have additional benefits for mistletoe control, specifically by reducing mistletoe seed deposition by dispersing birds and therefore risk of infection. Coppicing rates are, however rapid, and mistletoe seeds were found on coppiced branches less than a year old. This suggests that the reduction in risk of infection due to chopping may be short lived, possibly providing a period of about one year before a tree becomes reinfected. Nevertheless, following dispersal, mistletoe seeds must successfully establish and grow to reproductive size, meaning an additional one or two years before they become sources of new infections themselves. This suggests a period of up to three years before possible reinfection, somewhat longer than the recovery period of hand-removed mistletoes. *Phyllanthus* displays a considerable interannual variability in fruit production and not all trees are harvested each year (Setty, 2004). If trees were cut during harvest in a rotation linked to these production patterns, such a cycle may have benefits for both mistletoe control and productivity, and could form a valuable component to management aimed at tackling the mistletoe problem. Further research is required in order to quantify the relative costs and benefits associated with this and other approaches.

The fire regime in BRT has undergone significant changes in recent years (Sinha & Brault, 2005). Although the exact nature of this change is still uncertain, evidence suggests a change from low intensity ground fires to more intense uncontrolled burning. In the past, the Soligas used carefully monitored, low intensity fires to control ground cover, facilitating collection of fruits from the ground during harvesting, and increasing accessibility and visibility. Fire was banned by the KFD when the area was declared a wildlife sanctuary, but it still occurs seasonally. Setty (2004) cites control of fire as a factor in the increased densities of *T. tomentosus*. Studies in other systems have implicated decreasing frequency of both wild and prescribed fire in the increase of mistletoes (Alexander & Hawksworth (1975); Wicker & Leaphart (1974)), and other research provides anecdotal evidence for the role of fire in the regulation of other mistletoe populations (C. Martinez de Rio, personal communication). *T. tomentosus* is certainly sensitive to fire, and burnt mistletoes can be seen on the

forest floor where they have become detached from the surviving host following fire (personal observation).

Although fire may have benefits in terms of regulating mistletoe population dynamics, Sinha & Brault (2005) suggested current burning patterns may have significant consequences for population persistence of *P. emblica* and *P. indofischeri*, including negative impacts on productivity, growth and survival. The impacts of fire on the Amla population, both directly as a limitation on regeneration and as a source of mortality, and indirectly by potentially regulating mistletoe populations, needs further consideration. The nature of fire, rather than fire itself, is likely to be the crucial factor. With the recent spread of dense stands of a highly flammable invasive species, *L. camara*, in the forest understorey, in addition to altered spatial and temporal burning patterns, fires may now burn more intensely, at greater temperature, and reach higher into the forest canopy than before (both as a consequence of *L. camara* abundance and fuel accumulation due to altered spatial and temporal burning patterns). This relatively recent change in community structure, and its possible effects on fire regimes, would explain some of the apparent contradictions between local management perspectives and scientific and institutional opinions on fire and management at BRT. The possibility that shared dispersal agents between *L. camara* and mistletoe may facilitate the spread of mistletoe infection also requires investigation (Ghazoul (2002); Ghazoul (2004)). An increase in *Plicosepalus acaciae* infection of *Acacia* having been linked to increased populations of its main disperser, the yellow-vented bulbul (Bowie & Ward, 2004).

2.5.3 Conclusion

These results suggest that given both the overall prevalence of infection, and the apparent greater susceptibility of the more significant resource species, mistletoe infection represents a serious threat to sustainable harvesting of Amla in BRT. The results also show that traditional practices previously considered to be detrimental in terms of Amla productivity and mortality may actually have management benefits including reduced risk of mistletoe infection and enhanced long term productivity.

Traditional harvesting and management techniques should therefore be objectively evaluated without preconceptions about their efficacy, and their benefits (as well as deficiencies) should be recognised and accepted within scientifically-driven management and policy frameworks.

CHAPTER 3

THE DISTRIBUTION OF MISTLETOE ON *PHYLLANTHUS EMBLICA* AND *PHYLLANTHUS INDOFISCHERI*: SEED DISPERSAL, HOST SUITABILITY AND FOREST FIRE

3.1 Outline

The spatial distribution of a tropical mistletoe, *Taxillus tomentosus*, and three contributing processes; seed deposition, host suitability and environmental heterogeneity, are examined. Mistletoe populations are plant metapopulations in which host trees are patches and the frequency of infection in each ‘patch’ is the result of a balance between mistletoe colonization and extinction processes. Seed deposition patterns act as templates for subsequent mistletoe recruitment, and later for multiple sources of mistletoe mortality. Seed dispersal by frugivorous birds generates mistletoe aggregation due to behavioural responses to the availability of mistletoe fruit, and previous studies have focused largely on exposure to seeds as the main determinant of mistletoe distributions. Heterogeneity among hosts, as well as multiple sources of environmental heterogeneity could also lead to such an aggregated or patchy distribution and the possible interactions between mechanisms influencing these subsequent processes, recruitment and mortality, including conflicting ecological forces affecting mistletoe aggregation can have significant influences. In the dry forests of the Western Ghats, India, *Taxillus tomentosus* parasitises two conspecific fruit trees *Phyllanthus emblica* and *Phyllanthus indofischeri*. The distribution of this mistletoe was characterised

at three scales; individual host trees, forest patches and forest landscape. The role of variation in exposure to mistletoe seed in the observed distribution was assessed, along with evidence for heterogeneity in host suitability (both biological and environmental). The possible influences of forest fire and infection induced host mortality on distribution patterns through associated mistletoe mortality were also considered. Geostatistical analysis suggested prevalence and intensity of mistletoe infection are correlated at scales of 3 and 4km, and investigation of small scale variation found aggregation of mistletoes at the tree and forest patch scales. Seed dispersal alone does not explain observed distribution patterns and may influence distributions only at very local scales. This study provides an example of a host-mistletoe system in which dispersal may not be the dominant influence on mistletoe distribution at the landscape scale; mistletoe mortality from forest fire and infection induced mortality of hosts may be central to explaining the observed patterns and require further investigation. Mistletoe distribution in this tropical forest appears to be shaped by opposing forces, and by those acting at different scales.

3.2 Introduction

The investigation of spatial pattern is important in understanding the factors that shape the distribution, dynamics, and interactions of species (Kareiva, 1994), providing valuable information for inference of process from pattern (Palma et al., 1999). However, pattern and process at one scale can be constrained by those at smaller, or larger scales (Wu & Louks, 1995), and processes can act in opposing directions to generate observed distributions (Borer et al., 2007).

Mistletoes, like other parasites, typically have distinct spatial patterns (Anderson & Gordon (1982); Overton (1994)). Many displaying aggregated distributions as a consequence of a mutualistic relationship with their avian dispersers (Aukema, 2003). Birds respond behaviourally to the abundance of fruit-producing mistletoes, varying visitation rates to individual plants and habitats (Sargent, 1990), with implications for seed transmission rates (Martinez del Rio et al., 1996). Dispersing birds have been shown to forage preferentially, and hence deposit seeds disproportionately, in infected

trees and in areas with a higher prevalence of infection (Reid (1989); Aukema & Martinez del Rio (2002)). This results in a positive correlation between infection level and seed transmission (Aukema, 2004), with heterogeneity in exposure generating a clumped distribution of mistletoes among their hosts. However, heterogeneity among hosts, as well as multiple sources of environmental heterogeneity could also lead to such an aggregated or patchy distribution. The occurrence of mistletoes may be influenced by variation in host suitability, both a consequence of the host itself (e.g. host resistance leading to reduced establishment success (Yan, 1993)), and its local environment (Hoffmann et al. (1986); Bickford et al. (2005)), as well as other possible sources of post-establishment mortality. In reality, plant distributions are likely influenced by both; spatial patterns of seed rain represent a template for recruitment (Clark et al., 2004) but subsequent genetic and environmental factors will interact with these to influence the spatial structure and dynamics of plant communities (Nathan & Muller-Landau, 2000).

Studies of mistletoe distribution to date have focused on linking patterns of seed dispersal directly to observed distributions, mainly in dry forest systems, with few simultaneously investigating the role of other influences (except see Aukema (2004)). The role of host suitability in infection patterns has been independently investigated (Roxburgh & Nicolson, 2008), and several studies document environmental limits to mistletoe distributions including temperature and rainfall (Abulfaith & Emara (1988); Aukema (2004)). Fire has not previously been demonstrated as a factor influencing distribution patterns outside of dwarf mistletoes (Hawksworth & Wiens (1996); Conklin & Armstrong (2001)). The extent, and way in which host and environmental heterogeneity may modify, and interact with patterns resulting from mistletoe seed dispersal has not been widely investigated, furthermore evidence for conflicting ecological forces influencing the spatial distribution of mistletoes has not been previously documented (Medel et al., 2004). There is a need to separate the importance of avian dispersal from other potential genetic, ecological and environmental factors in regulating mistletoe populations at different spatial scales and to investigate the net result of possible antagonistic mechanisms in observed aggregation patterns (Medel et al., 2004).

This chapter presents experimental and correlative information from field studies designed to investigate the distribution of mistletoe in a tropical forest and to evaluate the importance of different mechanisms in generating the observed patterns. Patterns in mistletoe distribution are considered at three nested scales, and four potential mechanisms that lead to differences in mistletoe prevalence and intensity at these scales investigated; variation in exposure to mistletoe seeds, host and environmental heterogeneity and mortality from forest fire. Finally distribution patterns are compared to those found in other systems and the reasons behind, and implications of, the identified differences are considered. The implications of infection induced host mortality in the context of conflicting ecological forces influencing mistletoe aggregation are also discussed.

3.3 Methodology

3.3.1 Mistletoe distribution

Sixty $500m^2$ plots were surveyed across scrub and deciduous forest using a staggered hierarchical sampling regime in a nested survey of mistletoe infection on *Phyllanthus* hosts ((Pettitt & McBratney, 1993); (Aukema, 2004)). Positive infection status was assigned to individuals bearing one mistletoe or more. The total number of mistletoes was also recorded. For each sample point infection prevalence and mean intensity of infection was calculated. ‘Prevalence’ refers to the fraction of hosts infected, ‘intensity’ to the mean number of mistletoes per tree (Aukema & Martinez del Rio, 2002). The mean number of mistletoes detected on all hosts at a given site was used to represent intensity of mistletoe infection at that site, hereafter ‘site intensity’. This value serves as an index of mistletoe abundance at a site, being more pertinent than mean mistletoe load on only infected trees (Kriger et al., 2007).

Geostatistics (Isaaks & Srivastava (1989); Rossi et al. (1992)) was used to examine mistletoe distribution at the landscape scale, specifically the spatial autocorrelation of infection, modelling variograms of two measures of infection; prevalence and site intensity. As the spatial structure of the mistletoe population is overlaid upon

the spatial distribution of hosts, plots with no host trees were removed from the analysis. The effect of host density on prevalence and intensity of infection was also removed by modelling the residual of the regression of prevalence (or intensity) on host density. Failure to take account of this can generate spurious results; even if mistletoes were distributed randomly across host trees a variogram of prevalence or intensity could indicate spatial patchiness due to patchiness in the distribution of the underlying host population (Real & McElhany, 1996). Variogram models were described with three parameters: the range of spatial correlation, the nugget and the sill. The range measures the distance at which spatial autocorrelation between data point pairs ceases, or becomes much more variable. The nugget represents non-spatial variation due to measurement error and unobserved microscale variability, and the sill the constant variance of the observations (comprising both the nugget variance and the signal variance). In models with a nugget effect, the partial sill therefore indicates that proportion of the variability which is spatially structured (Cressie, 1993). The maximum distance between sampling locations was approximately 35 kilometers. It is customary to limit the description of the spatial structure to at least half the maximum distance between sampling units, hence variograms of infection prevalence and site intensity of infection were constructed up to 15000 metres, using all pairwise comparisons of points for 57 plots. The sampling regime was designed to control for anisotropy and so this was not investigated in the models. Variogram modelling and cross-validation (Davis, 1987) was carried out using the *gstat* package (Pebesma, 2004) in R (R Development Core Team, 2007). From the cross validation results, bias (ME), the mean squared deviation ratio of the predictions vs. samples (MSDR) and the root mean squared error (RMSE) were calculated as diagnostic measures to evaluate the precision and quality of the models (Kitanidis, 1997).

The benefits of multiple analysis methods in characterising spatial patterns has been emphasised (Real & McElhany, 1996), therefore small scale patterns identified through the variogram modelling were investigated further. The methods described by Boulinier et al. (1996) were used to quantify the relative contributions of two local scales of aggregation to the overall observed distribution patterns. A scale-independent measure was used, J , representing the relative increase in crowding of

mistletoe individuals among forest patches caused by aggregation. J_j gives the increase in the expected number of other mistletoes found on a tree for a randomly chosen mistletoe. The global level of aggregation, J is made up of EJ_j , the weighted average of these within plot values i.e. an overall measure of within patch aggregation, and J_k , the residual aggregation generated by differences among patches, $J = EJ_j + J_k$. Therefore we estimated; a) the proportion of the total aggregation due to differences in mistletoe numbers among patches, and b) the remaining proportion of the total aggregation, reflecting the average aggregation within patches, i.e. on trees within a patch. This method examines the observed distribution of mistletoes among trees against that expected under the null hypothesis that mistletoes are distributed among trees in a random fashion at both the within patch and among patch scales. The relative importance of these two scales were compared using Chi-squared tests.

3.3.2 Exposure to infection

Seed deposition patterns were investigated in relation to infection at both the individual tree and forest patch levels. Thirty sites were selected, this being done non-randomly to include plots in areas along the full continuum of prevalence. Species, height, DBH, infection status and the number of mistletoes were recorded for a total of 988 surveyed trees. Each tree was climbed and all mistletoe seeds on all branches counted, thereby producing a total mistletoe seed count for each tree. Infection prevalence and site intensity of infection were calculated for each site as before. Generalized linear mixed effects models of the probability of receiving seeds, and the number of seeds received by individual trees were constructed using the lmer function from the lme4 package (Bates, 2007) in R (R Development Core Team, 2007). Five covariates were fitted as fixed effects; tree species (*P. emblica* or *P. indofischeri*), number of mistletoes, tree height and the prevalence of infection in the patch; plot was fitted as a random effect. Model simplification was conducted using analysis of deviance and Chi-squared tests (Crawley, 2007). Model checking was carried out on glms, with all terms fitted as fixed effects using the ‘binned plot’ function from the arm library

(Gelman et al., 2007) for binomial models, and standard checks of residual deviance for the count models (Crawley, 2007).

3.3.3 Host suitability and post-establishment processes

To investigate the possible post-dispersal role of host and environmental heterogeneity in the distribution of *Taxillus tomentosus* we looked at variation in mistletoe growth and mortality rates between individual hosts, between the two host species and between forest patches. Growth of 200 mistletoes (100 on 53 *P.indofischeri* trees and 100 on 25 *P.emblica* trees) were monitored over a 12 month period. Infected trees were selected across five sites. Within individual trees, mistletoes were chosen to give a good representation across size classes. Height, width and haustorium diameter of each mistletoe was recorded by climbing host trees (Norton & Ladley (1997); Dawson et al. (1990)). Mistletoes were tagged with identification numbers for monthly monitoring. Mortality was modelled as a function of host species and mistletoe size class using generalized linear mixed effects models as before. Models were fitted with tree species, mistletoe size class (1-4) and tree branch diameter (cm) as fixed effects. Tree and site were fitted as first- and second-level nested random effects respectively. Relative annual growth (RGR) (ratio of annual mistletoe growth to initial mistletoe size) for these three measures was compared between host species with linear mixed effects models using lme from the nlme library (Pinheiro et al., 2007) in R (R Development Core Team, 2007), fitting tree species and size class of the mistletoe as fixed effects and tree as a random effect. Model checking was carried out as before.

The role of forest fire as an additional source of mistletoe mortality had to be considered opportunistically. In one year of the study, an area of forest with a high abundance of mistletoes was surveyed immediately following a fire. 356 mistletoes were assessed over an area of approximately $1.5km^2$ and their health status recorded (i.e. green and healthy or damaged by fire). Trees with tagged mistletoes were revisited six months later to assess mistletoe survival and the occurrence of re-sprouting, classifying mistletoes as ‘alive’, ‘damaged but re-sprouting’ and ‘dead’. We tested the null hypothesis that fire does not cause mistletoe mortality using a binomial

proportions test on the total numbers of surviving and dead mistletoes.

3.4 Results

3.4.1 Mistletoe distribution

Variograms for both infection measures, prevalence and site intensity, showed maximum semi-variance at lags of about 3.5 and 4.3 km respectively, sampling locations separated by greater distances did not display autocorrelation in measures of infection (Table 3.1, Figure 3.1). Both models included a nugget effect suggesting small-scale variation in the data. The nugget represents all un-accounted for spatial variability at distances smaller than the smallest sampling distance (Rossi et al., 1992), in this case 50 metres, therefore suggesting considerable spatial variability below this minimum lag distance that cannot be modelled with the present sampling scheme.

This smaller scale variability in mistletoe distribution was resolved using the measure J to compare aggregation at two scales; the within and among patch levels. Aggregation could be produced by variability in mistletoe burdens among trees within a forest patch, variability in burdens among trees from different patches, or a combination of both. Chi-squared tests revealed significant aggregation of mistletoes both within ($X^2 = 83.59, p < 0.001$) and among ($X^2 = 487.4, p < 0.001$) patches (Table 3.2). Among-patch aggregation (J_k) was marginally stronger than within patch aggregation (EJ_i), explaining 56% of the total aggregation of mistletoes among trees (Table 3.2). The observed autocorrelation at short distances is likely a consequence of both mistletoe aggregation on individual trees, and at a very local scale within forest patches. The overall level of aggregation as indicated by the variance to mean ratio of mistletoe numbers/host ($s^2/\bar{x} = 5.96$), was towards the low end of values found in other published studies of host-mistletoe interactions (Table 3.3).

3.4.2 Exposure to infection

The probability of receiving seeds was significantly greater for infected trees, and within these trees, greater still for those with more mistletoe infections (Table 3.4).

Figure 3.1: *Sample variograms for infection prevalence and intensity.*
On the left the variogram for prevalence and on the right for intensity using bin width of 2000 h (lag distance) corresponds to a separation vector in distance. The variogram summarises the regression coefficient for the linear regression of pairs of all data values separated by a common distance h , and demonstrates the presence of spatial autocorrelation when plotted against the lagged value h

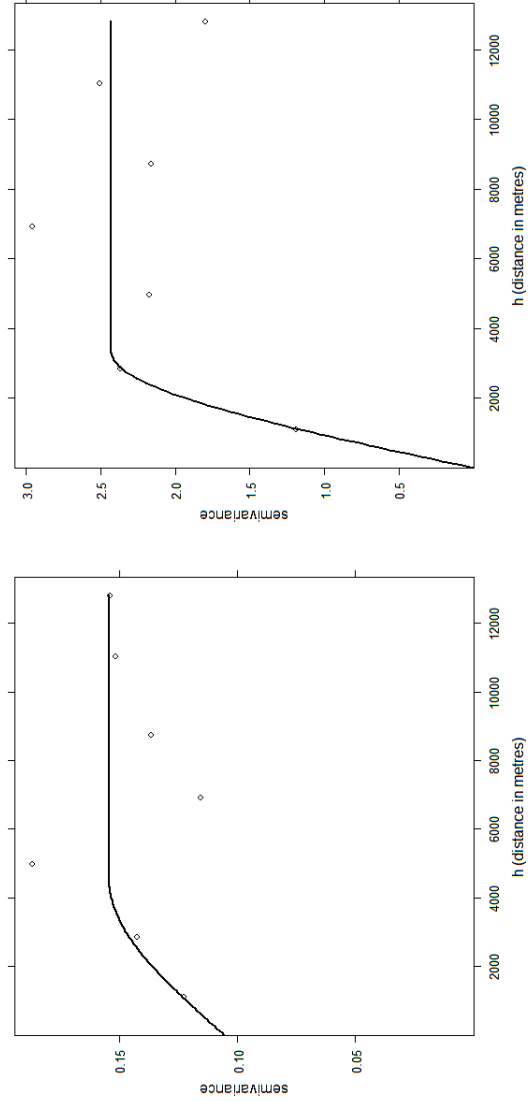


Table 3.1: *Models fitted with weighted least squares to experimental variograms for prevalence and site intensity of infection.*

The relative structural variability (RSV), the ratio of the partial sill to the sill (Robertson et al., 1993), indicates the proportion of variability that is spatially structured.

Variogram model	Nugget	Total sill	RSV (%)	Range (m)
Prevalence	0.01	0.14	33.6	4326
Intensity	0.21	2.44	91.5	3485

Similarly, seed counts were higher in infected trees and increased with the number of mistletoes on the tree (Table 3.4). We found no difference between species (after controlling for level of infection) in the probability of receiving seeds, or in the number of seeds received. Trees in areas with a higher prevalence of infection were more likely to receive seeds but prevalence had no influence on the actual number received. Height did not influence the probability of receiving seeds but infected trees of greater height received seeds in greater numbers (Table 3.4).

3.4.3 Host suitability and post-establishment mortality

Mistletoe growth and mortality rates were measured to investigate whether variation in suitability between individual host trees, between the two host species and in environmental characters between forest patches played a role in observed mistletoe distributions. Mortality rates were found to be higher on *P.indofischeri* ($p < 0.01$, Table 3.5). However, analysis of deviance components suggested that variability in mortality rates comes largely from tree to tree variation independent of either host species or site, particularly ‘good’ or ‘bad’ locations in terms of mistletoe survival not being apparent. No influence of host species on growth was identified, and analysis of deviance components suggested little variation attributable to either individual host tree, or to area.

Fire appears to be a significant cause of mistletoe mortality ($X^2 = 25.32$, $df = 1$, $p < 0.001$). Of 356 mistletoes, 63% were affected, additional mistletoes suffered damage but showed some extent of regrowth six months after the fire. To assess if mortality from forest fire could play a role in the identified landscape patterns,

Table 3.2: Aggregation of *Taxillus tomentosus* among *Phyllanthus* sp trees at the within- and among- plot scales. n = total number of trees; N = total number of plots; N_p is the number of plots with at least two trees and one parasite and n_p the number of trees in these plots. X_m = mean number of parasites/tree (global mean); P = the proportion of plots with a least one parasitised tree (i.e. prevalence). J and J_k measure, respectively, the total and among plot aggregation of parasites. $EJ_j = (J - J_k)$ measures the weighted average of the within-plot aggregation of the mistletoes among trees. Chi-squared tests of among-plot aggregation (X_{ap}^2), and within-plot aggregation (X_{wp}^2) of mistletoes; Chi-squares were computed with Yates correction. df_{ap} = degrees of freedom for chi-square among plots ($= N - 1$), $df_{wp} =$ degrees of freedom for chi-square within plots ($= n_p - N_p$). *** significantly aggregated ($p < 0.0001$)

Chi-square values									
Jmeasures									
n	N	n _p	N _p	X _m	P	J	J _k	EJ _j	J _k /J
227	57	191	26	1.85	0.754	5.21	2.94	2.27	0.563
								83.59***	
								df _{ap}	
								42	
								X _{wp} ²	
								487.4***	
								df _{wp}	
								167	

Table 3.3: *Magnitude of overdispersion in host-mistletoe systems.*

N = Number of hosts sampled, $\bar{x}$ = mean intensity of infection in sampled host population, $s^2/\bar{x}$ = variance:mean ratio, P = the percentage of total hosts surveyed infected (Prevalence), RHD= relative host density, i.e. hosts as a proportion of total stems. Lor. = Loranthaceae, Vis.=Viscaceae.

Species (Family)	N	$\bar{x}$	$s^2/\bar{x}$	P(%)	RHD	Habitat	Reference
<i>Phyllanthus</i> sp. (Lor.)	227	1.85	5.96	51	0.051	tropical deciduous forest	This study
<i>Tristerix aphyllus</i> (Lor.)	211	33.54	22.23	45.5	-	semi-arid mediterranean	Medel et al. (2004)
<i>Phrygilanthus sonora</i> (Lor.)	144	1.04	11.1	21	0.404	lowland desert	Overton (1994)
<i>Phrygilanthus sonora</i> (Lor.)	-	1.55	12.7	33	0.569	lowland desert	Overton (1994), Overton (1996)
<i>Tristerix aphyllus</i> (Lor.)	4711.2	-	18.8	15	-	semi-arid mediterranean	Martinez del Rio et al. (1996)
<i>Phoradendron californicum</i> (Vis.)	111	3.51	5.6	76	-	lowland desert	Overton (1996)
<i>Phoradendron californicum</i> (Vis.)	96	2.8	8.57	71	-	semidesert grassland	Aukema & Martinez del Rio (2002)
<i>Phoradendron californicum</i> (Vis.)	101	0.6	7.424	23	-	semidesert grassland	Aukema & Martinez del Rio (2002)
<i>Phoradendron californicum</i> (Vis.)	816	1.8	12.3	28	0.93	semidesert grassland	Aukema & Martinez del Rio (2002b)

mistletoe aggregation as indicated by variance:mean ratios before and after the fire were compared showing a decline from 10.93 to 7.20.

Table 3.4: *The influence of tree and area characteristics on the probability of receiving mistletoes seed and number of seeds received in P.indofischeri and P.emblica.*

The minimum adequate models are presented, all terms significant to $p < 0.01$, bt= back-transformed estimates from models with binomial and poisson error structures

	Probability of receiving		Seed count	
Coefficients:	Estimate (bt)	Std.Error	Estimate (bt)	Std.Error
Uninfected	-1.19 (0.23)	0.73	0.55 (1.7)	0.39
Infected	0.94 (0.72)	0.24	1.41(4.1)	0.23
No.of mistletoes	0.16 (0.54)	0.021	0.031 (1.0)	0.007
Prevalence	-2.64 (0.067)	1.07	-	-
Height	-	-	0.13 (1.1)	0.027

Table 3.5: *Mistletoe mortality in Phyllanthus emblica and Phyllanthus indofischeri.*

The minimum adequate models are presented, all terms significant to $p < 0.01$, bt= back-transformed estimates from models with binomial error structures

Model effects	Estimate(bt)/Dev.component	SE
Tree	0.82	-
Site	0.015	-
<i>P.indofischeri</i>	-1.70 (0.15)	0.44
<i>P.emblica</i>	-4.05 (0.017)	0.98

3.5 Discussion

3.5.1 Mistletoe distribution

The autocorrelation of infection measures up to large spatial scales combined with the identified small scale aggregation patterns indicate landscape ‘patchiness’ in mistletoe infection of Amla at three scales, tree, forest patch and landscape. Similar underlying mechanisms were identified for two of these scales as have been highlighted in other systems, specifically a positive correlation between infection at the tree and local

patch scale and seed transmission (e.g. Aukema (2004)). Results suggest that at small scales the spread of *T.tomentosus* may be driven by the response of dispersing birds to levels of resource abundance. *T.tomentosus* is dispersed predominantly by flowerpeckers (Davidar, 1978) which wipe the seed on to a branch after processing berries (Keeble, 1896), in such cases the vast majority of seeds are deposited on the same tree. Even if fruits are swallowed, gut passage times for these birds are extremely short, between 3-4 minutes (Ali, 1931). Flowerpeckers are also territorial around clumps of fruiting mistletoe, and restrict their feeding to these areas (Davidar, 1980). As such, dispersal distances are likely to be very short, the vast majority of seeds being deposited either on the same tree or on closely neighbouring trees.

3.5.2 Seed deposition

That mistletoe infections are transmitted by bird vectors in response to local resource levels has been well established (e.g. Aukema & Martinez del Rio (2002)), but the exact measure of local infection to which birds respond has not previously been demonstrated in mistletoes. The relative roles of tree fruit crop size (a function of tree mistletoe load) and the ‘fruit neighbourhood’ (Sargent, 1990) (infection characteristics of the area) may have significant consequences for distribution patterns (Garcia & Ortiz-Pulido, 2004), as well as genetic implications for population structure in relation to the relative contributions of intra-and inter-host dispersal. Saracco et al. (2005) found that neighbourhood variables had larger effects on visitation than focal tree fruit crop size.

Visitation was not directly observed but seed deposition can be used as a proxy. In contrast to previous studies these two processes were considered separately, the presence, or absence on a tree of mistletoe seeds, and subsequently, the total number of seeds present in the former case. That seed counts were explained largely by intensity of infection in the tree itself, suggests a more prominent influence of tree fruit crop size. Although infection prevalence in a forest patch did not influence the seed counts of trees within that patch, it did affect the probability of a tree receiving mistletoe seeds indicating that initial exposure (infection spread) may be a conse-

quence of infection characteristics in the local host population, whereas subsequent re-infection (intra-host seed dispersal) is dictated by characteristics of the tree itself (i.e a greater abundance of fruit or the availability of preferred perches). Further investigation is needed to clarify the relative roles of neighbourhood and tree infection characteristics, in particular, relating actual fruit resource availability at these levels directly to bird visitation. Quantifying the relative levels of intra and inter-host seed dispersal, understanding the processes by which uninfected trees become infected, and the dependence of infection rate on area based characteristics is particularly necessary for the development of management strategies aimed at reducing spread of mistletoe populations (Rist et al., 2008).

The fruit neighbourhood may be made up of other fruiting species in addition to mistletoes on neighbouring trees. Increased frugivore activity in areas where other species are providing additional fruit resources may have significant consequences for mistletoe dispersal rates. The widespread distribution of *Lantana camara* in BRT may be of particular significance as this plant produces large numbers of berries. Seed dispersal facilitation among neighbouring plants has been documented in other mistletoes (e.g. Carlo & Aukema (2005)), and dispersal of *T.tomentosus* seed may indeed be facilitated by the presence of fruiting *Lantana* neighbours, an influence that could be reflected in distribution patterns. Dispersal success has a dominating influence on mistletoe reproduction (Robertson et al., 1999), given the widespread distribution of *Lantana camara* (Barve et al., 2005) dispersal facilitation may have played a significant role in the recent and substantial increase in the mistletoe population (Rist et al., 2008), this possibility requires further investigation.

Few previous studies have investigated landscape level infection patterns in systems other than dwarf mistletoes. Aukema (2001) found spatial correlation in mistletoe prevalence at scales of 250m-1.5 km and at 4km. She attributed small-scale patterns to bird territory sizes but at the larger scale could not isolate the effects of elevation and tree density, the two being significantly negatively correlated with prevalence but uncorrelated with each other. Both factors are likely to influence infection prevalence (Abulfaith & Emara (1988); Overton (1994)). The landscape patterns identified in this system do not appear to be the result of similar influences; infection

measures were uncorrelated with elevation and scales of 3-4km are considerably larger than documented territory sizes for the known dispersers of *T.tomentosus* (Davidar, 1978). Combining the variogram models and aggregation analysis provided a greater insight into distribution patterns and extended the ability to make inferences about the underlying processes.

3.5.3 Post-dispersal mechanisms

The spatial distribution of plants is the result not only of dispersal. Seed deposition patterns act as a template upon which subsequent processes influencing establishment success and post establishment survival may act (Nathan & Muller-Landau, 2000), and in a spatially heterogeneous manner. Host suitability and environmental conditions such light and water availability in particular are known to influence mistletoe survival (Bach et al., 2005) and may vary spatially across a forest landscape as a consequence of variation among and within individuals of the two species, or heterogeneity in forest structure and biophysical variables. Mortality and growth rates provided no evidence of a significant influence of local scale environmental variation. Higher mortality of mistletoes on *P.indofischeri* suggests differential suitability between the two species, but to what extent the lower levels of infection in *P.indofischeri* are a consequence of this reduced mistletoe survival, and whether this is due to environmental or host specific factors (e.g. host immunocompetence or other host qualities such as bark thickness) is yet to be determined. As the two species occur in distinct forest types it was not possible to separate the influence of host species from environmental variables as dictated by forest type. Amla harvesters cite lower mistletoe survival on *P.indofischeri* resulting from reduced water availability in drier scrub forest and greater susceptibility as a consequence of smaller branch size (Rist pers comm. with Amla harvesters). Aggregation of mistletoes can occur as a consequence of seed dispersal as well as heterogeneity in host- or habitat-based susceptibility. Most studies designed to understand the factors involved in mistletoe aggregation have focused on just one of these mechanisms, this study highlights a potential role for all three.

Although aggregation is a common pattern in mistletoe systems, different factors

may also influence mistletoe aggregation in opposing directions. For example, in a study of a new world *Loranthaceae* (Medel et al., 2004) found aggregation to result from a balance between vector behaviour and host resistance traits, namely cactus spine length. Aggregation was lower than that identified in other host-mistletoe systems. Mistletoes rarely cause host mortality (Aukema, 2003), however in Amla, mortality among infected trees is high (Setty (2004); Rist pers. observation). In this location mistletoe-induced host mortality may be acting to counter forces producing aggregation (Anderson & Putz, 2002). Additionally, this study was conducted in a tropical forest of considerably higher diversity than other systems in which mistletoe distribution has previously been studied. Host community composition can fundamentally influence the establishment and prevalence of disease (Holt et al., 2003). Relative host density in this system influences mistletoe abundance (Rist, 2008) with possible implications for spread and consequently aggregation. Where hosts are more ‘diluted’ in a forest of high species diversity, development of aggregations may be restricted, distribution patterns at high levels of mistletoe abundance taking on quite different characteristics. Less intense aggregation may indicate that the mistletoe population is growing (Anderson, 1978). Information on temporal variability in aggregation is required to assess this hypothesis, specifically the temporal correlation between among-patch aggregation and mean tree mistletoe load. A decrease of among-patch aggregation would be expected in parallel to a local population increase if local spread of mistletoes explains a large part of the pattern of aggregation we observed.

The results also provide initial evidence that forest fire may be an additional opposing force to aggregation (Lavorel et al., 1999). Fire caused mistletoe mortality over a relatively large area, reducing levels of aggregation by a significant degree. Forest fire may therefore play a role in the landscape scale patterns we identified, maintaining lower mistletoe populations where burning occurs regularly at the same location, a pattern not uncommon in BRT. Although preliminary, these findings indicate a possible role of fire in the landscape level patterns observed, a possibility supported by other studies demonstrating the influence of fire on mistletoe populations dynamics (Conklin & Armstrong (2001)), including at the landscape scale (Kipfmüller & Baker, 1998).

3.5.4 Conclusion

The observed patterns in mistletoe infection are a consequence of several processes operating at different scales; at the level of individual trees, locally among neighbouring trees and also at a larger landscape level. In addition, these act in opposing directions, a balance of processes producing under- and overdispersion. Patterns of seed rain, a consequence of bird foraging and territorial behavior, represent a template for recruitment (Clark et al., 2004). Subsequently other processes, including differential establishment success and survival (including mortality from fire and infection induced host mortality), act on this template to determine observed distributions, including more emergent landscape distribution patterns. Management of the mistletoe infection problem has been informed by an increased understanding of both the spatial distribution patterns of mistletoe and the mechanism underlying these patterns, assisting in the identification of priority areas for management. Determining the relative roles of seed deposition and post deposition processes in observed mistletoe distributions, and the incorporation of a temporal component in future studies of host-mistletoe interactions in this system would be of significant additional value.

CHAPTER 4

ASSESSING THE CONTRIBUTION OF TRADITIONAL ECOLOGICAL KNOWLEDGE IN THE MANAGEMENT OF MISTLETOE INFECTION AND AMLA HARVESTING

4.1 Outline

Many forest communities possess considerable knowledge of the natural resources they exploit. Where management interventions are required to ensure the sustainability of harvested resources, traditional knowledge can provide vital baseline information. It can be used in monitoring, considerably less time and cost being required for the collation of such information from communities than from scientific studies designed to answer the same questions. In addition, traditional knowledge can fill information gaps that cannot otherwise be addressed, for example, historical trends in resource levels. However, traditional knowledge can also be anecdotal or biased, and its application to management must be critical and accompanied by validation studies.

Scientific data was contrasted with traditional ecological knowledge (TEK) in the context of the mistletoe threat to the sustainable harvesting of Amla, with the aim of informing both the development of specific management strategies and the further use of TEK in management at this site. The ability of harvesters to provide information more efficiently than scientific field studies, and to provide additional information and

insight for solving specific management problems was a target for investigation. Additionally the impact of mistletoe infection on the Amla harvest was assessed, including the discrepancy between sustainability as indicated by harvesting records, and local perspectives on resource status. TEK complemented previously gathered ecological data by providing concordant and additional information about: (1) mistletoe distribution among hosts, (2) mistletoe phenology, (3) mechanisms of mistletoe spread, and (4) the impacts of mistletoe infection on the Amla population. Reporting bias was found to be a significant problem in questions with a more direct link to management targets questions framed to elicit information on observations were found to be more accurate than those on opinion. Combining information on historical and current harvesting trends with official data from a government-established cooperative society suggested current assessments of sustainable harvesting may be inaccurate, potentially because an increase in harvesting effort may mask a declining NTFP resource.

The traditional knowledge, together with results from the previous two chapters should be used in the development of an experimental management program at BRT involving both resource users, scientists and forest managers (Chapter 5). The importance of a multidisciplinary approach to assessing sustainability is emphasised but areas where caution should be exercised in the use of TEK are highlighted. Additionally, it is demonstrated that threats other than unsustainable collection levels can undermine the potential of NTFP harvesting to support forest dependant livelihoods, and that monitoring of yields in isolation is insufficient to assess the sustainability of harvested resources.

4.2 Introduction

[Traditional ecological knowledge] is a cumulative body of knowledge, practice and belief evolving by adaptive processes and handed down through generations by cultural transmission, about the relationship of living beings (including humans) with one another and with their environment. (Berkes, 2004)

There is now much awareness of what is variously termed “Traditional”, “In-

igenous”, or “Local” ecological knowledge (TEK, IEK and LEK) and management (e.g. Traditional Resource Management (TRM)) in conservation science and natural resource management (Berkes et al. (2000); Pierotti & Wildcat (2000); Sheil & Lawrence (2004); Drew (2005)). These terms have often been used interchangeably, all essentially referring to sources of knowledge about species, ecosystems or practices held by people whose lives are closely linked to their natural environment. Traditional knowledge is the predominant usage among conservationists and is not restricted to indigenous peoples alone. The distinction between ‘Traditional’ or ‘Indigenous’, and ‘Local knowledge’ however may be more significant. The former both implying the development of such knowledge over a longer timescale (Gilchrist et al., 2005). However, some communities with a more recent association with an area or resource still possess a detailed acquired knowledge or understanding of the ecology and management of that area and the resources they utilise. Communities dependent on natural resources can rapidly develop insight into factors influencing their availability or quality. Such information can be shared among users and develop into a substantial body of knowledge (e.g. Hanna (1998); Acheson et al. (1998)). These current, or recent observations, can be as important as “traditional” information generated over a longer timescale (Akearok et al., 2003).

There are three arguments for the use of TEK in conservation planning and resource assessment, and for its development into participatory monitoring and incorporation into longer-term management: cost efficiency, additional information, and community engagement. TEK can be a more efficient method of acquiring information. Where TEK corresponds well to scientific data, rigorous social science methods can gather this information in considerably less time than formal ecological research which often involves time intensive and costly fieldwork. Data are also available on timescales rarely available from scientific studies (e.g. of sufficient length to cover several population “cycles”) and may be available at spatial scales that provide new understanding. In addition, programs that garner support of local people have a greater chance of successful outcomes and long-term sustainability (Bowen-Jones & Entwistle (2002); Borrini-Feyerabend & Buchan (1997); Schwartzman et al. (2000); Danielsen et al. (2005)). There are an increasing number of international mandates

for the inclusion of such knowledge in ecological restoration and conservation (Mauro & Hardison, 2000) (e.g CBD Article 8(j), (UNEP, 1992)), and the practical applications of TEK are growing. Many studies have already used such knowledge effectively to target conservation aims, for example the conservation of biodiversity (Fraser et al., 2006); of rare species (Colding, 1998); the management of protected areas (Johannes, 1998); sustainable resource use (Berkes, 1999); and climate change (Couzin, 2007).

Several studies have demonstrated the complementarity of traditional knowledge and scientific knowledge (Nabhan (2000); Berkes et al. (2000)), hence validating the use of this body of information in ecological research, including in harvesting assessment. However, there has been little discussion on how such information can be most effectively utilised. For example, as a source of baseline information, to fill specific information gaps, in monitoring, or for innovative ideas to feed into adaptive management programs; or indeed, where it may be misleading. Consideration of how this integration should occur in practice is also lacking, particularly in cases where data from TEK and scientific studies appear contradictory. Additionally, there have been few studies that have focused on the inadequacies of traditional knowledge, highlighting areas where more caution should be exercised in its use. A management system that is based on unreliable information, regardless of its source, jeopardizes the sustainability of resource populations through excessive harvest or, equally, may negatively impact livelihoods through unnecessary harvest restrictions (Freeman (1985); Freeman (1992)). Such restrictions can also cause unnecessary tension between communities and those regulating the the resources in question. Traditional knowledge can be considered one of a set of tools in natural resource management and requires the same validation as any other omponent of management or monitoring approach.

There has been a call to move beyond the process of comparing TEK and information from scientific studies to their more direct incorporation into resource management (Brook & McLachlan, 2005). However, only by making such comparisons is it possible to identify discrepancies in the results generated by the two methods, to investigate the reasons for these, including sources of bias, and therefore, to ensure that the basis for management is robust. Shifting baseline syndrome (Pauly, 1995) has been highlighted as an important restriction in studies using traditional knowl-

edge to assess historical trends in resource availability or environmental conditions. It refers to the inaccurate perception of changes in resource species abundance, or other environmental conditions, and leads to inappropriate reference points for evaluating economic losses resulting from overharvesting, or other factors affecting resource availability (e.g. Saenz-Arroyo et al. (2005)). Recall bias is another obstacle for TEK studies. Harvesters may seek to bias their reported activities, such bias operating in either direction depending on informant motivations. For example, under-reporting may occur to hide illegal hunting activities (Sheil & Wunder, 2002), or over-reporting to impress other community members. Resource users may also have preconceptions or conflicts of interest which influence their reporting (Danielsen et al., 2005), making consideration of the wider context and possible motivations of resource users crucial.

Previous studies have already demonstrated the Soligas' considerable knowledge of their forest environment (Uma Shaanker et al., 2004). Participatory resource monitoring carried out in BRT since 1994 (Lele et al., 1998), and in a more institutionalised manner since 1998, suggests that community based productivity estimations for NTFPs have a close correspondence to productivity estimates based on scientific surveys (Shanker et al., 2005). The potential for using such knowledge to target a specific management problem is assessed here. Data from ecological studies carried out over two years were compared with traditional ecological knowledge gathered from harvester interviews on mistletoe infection of Amla. The validity and additionality of TEK are investigated, and hence its potential for cost effectiveness in terms ongoing monitoring and management. By assessing the correspondence between the two sources of information, areas where TEK may be most useful, and where it may be misleading are identified. Current and historical harvesting trends as perceived by resource users are investigated and compared with official harvest records in order to quantify the impact of mistletoe infection on the Amla harvest, including investigating explanations for a discrepancy between sustainability as indicated by harvesting records and local perspectives on resource status. Additionally, harvester behaviour and opinions in relation to current and potential management approaches are assessed, outlining local techniques which could form the basis of a provisional management strategy for securing a sustainable Amla harvest.

4.3 Methodology

4.3.1 Interviews with harvesters

There were two main objectives in conducting interviews with harvesters. Firstly, to assess how different types of harvester knowledge regarding mistletoe infection corresponded to data gathered from ecological surveys, assessing TEKs role in management from a cost effectiveness perspective. Secondly, to assess its potential to provide novel and additional information on the mistletoe infection problem, specifically as a source of information on historical patterns in infection and the implications of mistletoe infection for harvesting incomes and potential directions for future management.

Between February and August 2006, 47 Amla harvesters from the BRT wildlife sanctuary were interviewed (Table 4.1). Respondents were selected from 16 podus (villages) spread across the sanctuary (Figure 4.1). By including individuals knowledgeable about all portions of the geographic area relevant to the resource under consideration the data represents a good picture of Soliga knowledge about Amla in BRT. Two to five respondents were interviewed from each podu. Interviewees were selected based on being identified as established Amla harvesters by one or more community peers. Harvesters can be confidently identified as those possessing traditional knowledge, as at this site only the indigenous communities have rights to harvest NTFPs. Other Soligas who do not participate in the Amla harvest, or indeed non-tribals, may also hold knowledge on this subject but experienced Soliga harvesters were expected to possess the greatest knowledge. The aim was to maximise the amount and quality of TEK obtained from respondents rather than to be representative. Participation in all surveys was voluntary, and respondents were interviewed by a local research assistant trained to conduct the surveys.

The questionnaire consisted of a mix of fifty specific and open-ended questions in two categories, with the opportunity for the interviewees to elaborate on questions as they saw fit rather than a strict question and answer format (Laird (2002); Martin (2004)). The interviewees typically indicated that they “did not know” if they could not answer a question. Questions were in two categories, the first category consisted

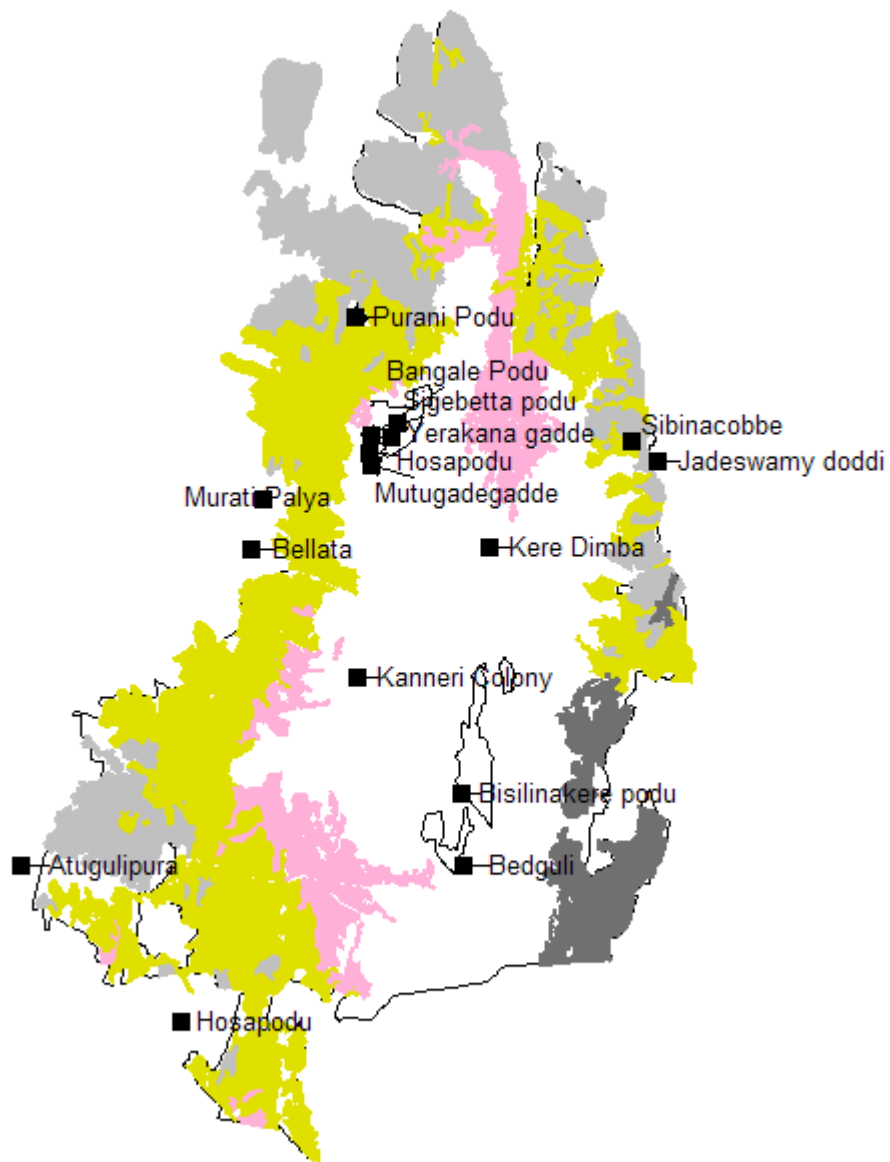


Figure 4.1: *Podus of interviewed TEK harvesters within BRT Wildlife Sanctuary*
The sanctuary map indicates dry deciduous (green, pink and dark grey) and scrub (light grey) forest.

Table 4.1: *Summary of interviewed harvester characteristics*

Character	
Podus (Villages)	16
Harvesters	47
Gender	Male 87%, Female 13%
Age	32 to 80 years ($\bar{x} = 50 \pm 13.1$)
Harvesting experience	10 to 30 years ($\bar{x} = 22 \pm 5.1$)
Main occupation	Agriculture (66%), NTFP collection (15%), daily wage labour (17%), housewife (2%)
Secondary occupation	NTFP collection (45%), no secondary occupation (43%), daily wage labour (9%), agriculture (4%)

of knowledge related to natural history observations and the second was specific to management. Questions in the natural history category targetted knowledge of host tree associations, mistletoe distribution, optimal growing conditions, timing of flowering and fruiting, pollination and dispersal agents, and the mechanism of infection spread. Questions associated with management pertained to more specific information on the impact of mistletoe infection on growth, productivity and mortality of Amla, the variation in suseptibility between the two Amla species, the comparative productivity levels of infected and uninfected trees and to the effects of fire on both Amla trees and mistletoes. The answers to each question were quantitatively categorized and compared to available ecological data. In this evaluation the term ‘accurate’ is used only in reference to whether the answer matched the available ecological data. Answers were used as the response variable in linear (lms) and generalised linear models (glms) with normal and binomial error structures respectively, testing harvesting experience, age, sex and main occupation as explanatory factors. All modelling was carried out using Statistical software R, version 2.5.1. (R Development Core Team, 2007). Model simplification for lms was conducted using anova and for glms anova with Chi-square deletion tests and model checking using the *plot* function from the *graphics* library (R Development Core Team, 2007).

Perceived changes in the density of Amla trees, both surrounding the podu and in the forest as a whole, were investigated. Harvesters are familiar with such distinctions

through their involvement in the participatory pre-harvest assessments of NTFP productivity conducted by a local NGO (ATREE Section 1.1.8); a one to two km radius around the podu was suggested as the basis for their assessment. Information on current and past (15 years previously) harvesting activities was collected, the average yield of Amla per day, number of days spent harvesting in a season and the standard rate earned per unit collected. The perceived total amount collected per season for each harvester was calculated based on the number of days spent harvesting multiplied by their individual per day collection amounts. Other studies support the validity of the assumption that harvesters accurately perceive harvest quantities (Jones et al. (2008))

Harvesters may react in various ways to the perceived opportunities and threats of being researched; for example seeking to bias their recorded forest uses upward so as to be better recognized, or downwards to hide illegal activities, or due to suspicion about the intended use of the information given (Sheil & Wunder, 2002). The interviewer was a resident of BRT and so was well known and trusted by local Soliga harvesters. Harvesters were open about their own management practices, even though some activities are prohibited by the Forest Department. Harvesters were considered to be describing basic ecological knowledge accurately and to the best of their abilities as well as not actively hiding information with regard to their own management practices. In addition, the accuracy of reporting of historical conditions in a changing system can be a source of error (Saenz-Arroyo et al., 2005). Shifting baseline syndrome (Pauly, 1995), (via the influence of age) was investigated in order to assess whether this affected results. Previous studies have demonstrated the high concordance between local Soliga names and scientific species classifications for plants as well as birds and animals (Ganeshiah & Uma Shaanker, 1998). In fact, although the Soligas recognise *Phyllanthus* as two species based upon vegetative characters (Ganesan, 2003), previously *Emblia officinalis* Gaertn., early studies did not distinguish between *Phyllanthus emblica* and *Phyllanthus indofischeri* (e.g. Murali et al. (1996); Uma Shaanker et al. (1998)).

4.3.2 Scientific knowledge and formal harvest records

This investigation was conducted in parallel with ecological studies, and with experimental fieldwork which assessed mistletoe distribution patterns and alternative management approaches; (Chapter 2; 3; Rist et al. (2008)). Additional published studies (Sinha & Bawa (2002); Ganesan & Setty (2004); Sinha & Brault (2005)) also provided data with which TEK could be compared.

Records of total Amla harvest in BRT over a fifteen year period (between 1990 and 2005) were obtained from LAMPS. Results on current and historical harvesting patterns from TEK interviews were compared with trends in these records and with official unit prices over this 15 year period.

4.4 Results

4.4.1 Ecological knowledge of Amla harvesters

All respondents were familiar with the mistletoe (*Taxillus tomentosus*), referred to locally as ‘*Bili Uppilu*’ or ‘*Antu Uppilu*’; in fact many respondents identified five or more mistletoe species present in BRT, their species descriptions matching results from mistletoe diversity surveys (unpublished data). Overall, harvesters are highly knowledgeable about mistletoe infection of *P.emblica* and *P.indofischeri* by *T. tomentosus* (Table 4.2), but some important discrepancies were also identified.

Harvesters identified 35 species as hosts in comparison to 14 species identified in forest surveys. *P. emblica* and *P. indofischeri* are the primary host species (Chapter 2; Rist et al. (2008)); all harvesters also considered this to be the case. Harvesters mentioned all but two of the secondary host species identified by forest surveys, and an additional 19 species not identified. The two species not cited as hosts by harvesters; *Stereospermum personatum* and *Wendlandia thyrsoides*, together accounted for only 0.004% of all observed mistletoe infections (Figure 4.2).

In considering the distribution of citations over the 19 additional species that harvesters identified as hosts, two species, *Kydia calycina* and *Mallotus philippensis*, were cited by over 40% of the harvesters. Rare host associations are likely to be

Table 4.2: Summary of harvester knowledge on mistletoe ecology, distribution and the impacts of mistletoe infection on *Amla*

	TEK	Ecological studies	Concordance
Primary hosts	<i>P. emblica</i> and <i>P. indofischeri</i>	82% sampled mistletoes observed on <i>P. emblica</i> and <i>P. indofischeri</i> (Rist et al., 2008)	Excellent
Secondary hosts	35 species	12 species (Chapter 2; Rist et al. (2008))	Some
Infection prevalence	0.5 to 0.8 ($\bar{x} = 0.57 \pm 0.11$)	0.51 (Chapter 2; Rist et al. (2008))	Excellent
<i>P. emblica</i> and <i>P. indofischeri</i>	<i>P. emblica</i> is more commonly infected, older, taller trees are more often infected and have more mistletoes	Prevalence 0.64 in <i>P. emblica</i> and 0.38 in <i>P. indofischeri</i> (Rist et al., 2008), greater probability and intensity of infection in taller trees (Rist et al., 2008)	Excellent
Forest type	Deciduous forest (Moist, dense forest and hill tops)	Deciduous forest (Rist et al., 2008)	Excellent
Phenology	Flowering during summer, fruiting at the end of summer prior to rains	Peak flowering April - July (Summer) Peak fruiting Aug - Nov (Rain in July/August) (Chapter 3)	Excellent
Flower visitors	<i>Sanna chitte hakki</i> (Flowerpecker/Sunbird), insects	Flowerpeckers (Davidar, 1983) Lorikeet, drongo (Shrestha, 2000)	Some
Dispersal	Birds, squirrels, bats, rats, monkeys and wind	Flowerpeckers (Davidar, 1978), Bulbuls (Ali & Ripley (1983); Shrestha (2000))	Poor
Favourable conditions for mistletoe growth	Moist, dense forest	High moisture (Reid & Lange, 1988) High light levels (Norton & Reid, 1997)	Good
Impact on growth	Reduces growth	Significant impact on growth (Setty, 2004)	Good
Impact on productivity	25-100% ($\bar{x} = 68\% \pm 20.7$) reduction in fruit production	ca. 44% decrease* (Setty, 2004)	Good
Impact on survival	All infected trees die, mortality occurs within between 2 and 10 years following infection	54% mortality rate of infected trees over four years (Setty, 2004)	Good
Differential susceptibility	Greater in <i>P. indofischeri</i>	Impact on growth only in <i>P. emblica</i> , Impact on productivity greater in <i>P. emblica</i> (Sinha, 2000)	Poor

*Calculated from mean fruit production figures presented in (Setty, 2004) for infected and un-infected trees (*Phyllanthus emblica* only). No information was given on the infection levels in these trees and the additional effects of temporal variability in fruit production cannot be isolated.

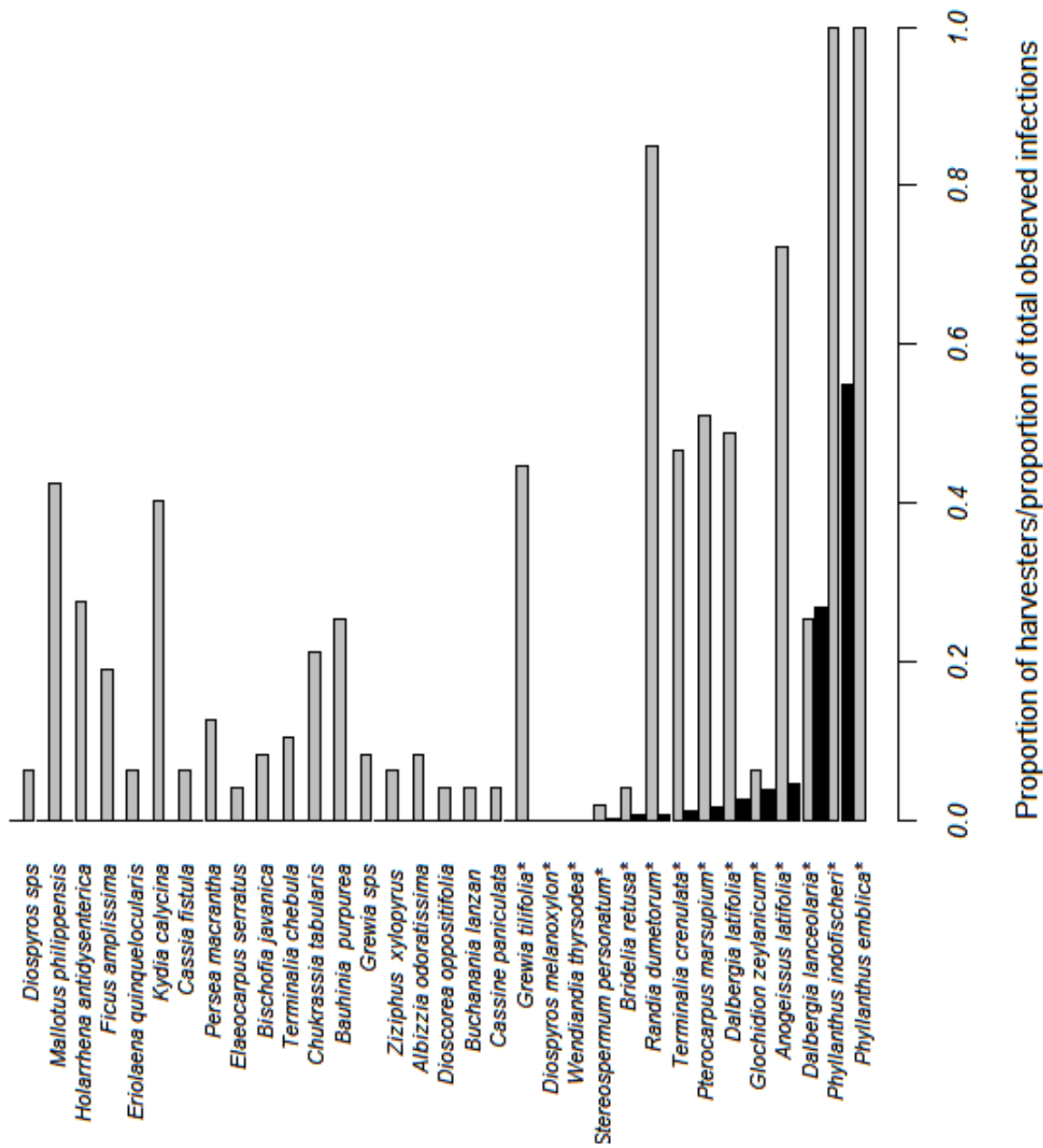


Figure 4.2: Host species of *T. tomentosus* identified by interviews and forest surveys. Grey bars indicate the proportion of harvesters (n = 47) citing each species; harvesters cited an average of 8.5 (± 2.7) species, black bars the proportion of the total number of observed mistletoe infections attributable to the particular host association (n = 512), *hosts identified in forest surveys.

particularly hard to detect when the relative abundance of that host species is low. Using the relative abundance of these two species and a global ‘per tree’ measure of probability of infection, the possibility that forest surveys could have missed an incidence of infection for one of these was considered (Table 4.3). The global measure was calculated by averaging the number of infected trees across all sampled species assuming equal probability of infection across species. Based on species abundance less than one infected tree was expected in the sample for both *Kydia calycina* and *Mallotus philippensis*. *Grewia tilifolia* was used for comparison, a species also cited by over 40% of the harvesters but which in contrast had been observed as a host species at a very low frequency. Using the same global measure of probability of infection, four infected trees were expected, a consequence of the relative abundance of this species.

In ecological assessments of infection prevalence, over half of all trees were found to be infected (64% of *P.emblica* and 38% of *P.indofischeri*) (Chapter 2; Rist et al. (2008)). The majority of harvesters made the same assessment, however some gave estimates of up to 80% of trees infected. Although all considered infection to be greater in *P. emblica*, in explaining landscape level patterns in infection, many harvesters chose to describe forest areas with specific microclimates e.g. “*hill tops*”, “*dense moist forest*”, rather than distinguishing on the basis of scrub and deciduous forest classifications. They suggested that mistletoe growth is better in moist, shaded conditions and that the higher prevalence of infection found in *P.emblica* is due to such conditions in deciduous forest; compared with *P. indofischeri* in drier scrub forest. “*They need a moist climate to grow well and are not able to tolerate a hot climate, so they grow more in dense forests and in hilly regions*” (Harvester 31).

All cited taller, older trees as bearing more mistletoes, and when asked about the distribution of mistletoe infections on an individual tree stated that mistletoes are mostly found on thin outer branches, and rarely on the trunk region, except in very heavily infected trees. Despite subsequently emphasising the preference of dispersing birds for these thin branches, few linked their observations to this fact, and several mentioned ease of establishment as the reason for this distribution pattern, mistletoe establishment being more difficult where tree bark is thicker.

Table 4.3: *Discrepancies in host species identification*

Host species of *T. tomentosus* as identified by forest surveys and TEK interviews. n = number of stems, Obs = number of observed infected individuals, Exp= number of expected infected individuals, Species in bold were cited by over 40% of harvesters but not detected in surveys (local names in brackets).

Species (local name)	Family	n	TEK?	Obs	Exp
All species		4889	-	163	-
<i>Phyllanthus emblica</i> (Nai nelli)	Euphorbiaceae	109	Yes	70	3.6
<i>Phyllanthus indofischeri</i> (Ittu nelli)	Euphorbiaceae	118	Yes	45	3.9
<i>Anogeissus latifolia</i> (Bejja)	Combretaceae	1233	Yes	14	41.1
<i>Dalbergia lanceolaria</i> (Buluga)	Fabaceae	34	Yes	10	1.1
<i>Glochidion zeylanicum</i> (Anase)	Euphorbiaceae	19	Yes	7	0.6
<i>Randia dumetorum</i> (Kare)	Rubiaceae	723	Yes	3	24.1
<i>Grewia tilifolia</i> (Daddasalu)	Tiliaceae	137	Yes	3	4.6
<i>Pterocarpus marsupium</i> (Honne)	Fabaceae	148	Yes	3	4.9
<i>Terminalia crenulata</i> (Matti)	Combretaceae	262	Yes	2	8.7
<i>Dalbergia latifolia</i> (Bite)	Fabaceae	15	Yes	2	0.5
<i>Bridelia retusa</i> (Sironne)	Euphorbiaceae	45	Yes	1	1.5
<i>Diospyros melanoxylon</i> (Tubare)	Ebenaceae	43	Yes	1	1.4
<i>Wendlandia thyrsoides</i> (Koli)	Rubiaceae	2	No	1	0.1
<i>Stereospermum personatum</i> (Padure)	Bignoniaceae	38	No	1	1.3
<i>Mallotus philippensis</i> (Kesilu)	Euphorbiaceae	9	Yes	0	0.3
<i>Kydia calycina</i> (Bende)	Malvaceae	28	Yes	0	0.9
<i>Cassine paniculata</i> (Kaneeru)	Celastraceae	57	Yes	0	1.9
<i>Buchanania lanzan</i> (Muruki)	Anacardiaceae	19	Yes	0	0.6
<i>Dioscorea oppositifolia</i> (Bellade)	Dioscoreaceae	4	Yes	0	0.13
<i>Albizia odoratissima</i> (Sele)	Fabaceae	4	Yes	0	0.1
<i>Ziziphus xylopyrus</i> (Gotti)	Rhamnaceae	1	Yes	0	0.03
<i>Grewia sps</i> (Udupe)	Tiliaceae	0	Yes	0	0
<i>Bauhinia purpurea</i> (Kanchuvala)	Caesalpinaceae	35	Yes	0	1.7
<i>Chukrassia tabularis</i> (KilANJI)	Meliaceae	0	Yes	0	0
<i>Terminalia chebula</i> (Arale)	Combretaceae	57	Yes	0	1.9
<i>Bischofia javanica</i> (Neelalu)	Euphorbiaceae	0	Yes	0	0
<i>Elaeocarpus serratus</i> (Kakkilu)	Elaeocarpaceae	5	Yes	0	0.2
<i>Persea macrantha</i> (Karavadi)	Lauraceae	35	Yes	0	1.7
<i>Chloroxylon swietenia</i> (Urigilu)	Rutaceae	139	Yes	0	4.6
<i>Diospyros sps</i> (Hasari)	Ebenaceae	52	Yes	0	1.7
<i>Holarrhena antidysenterica</i> (Ala)	Apocynaceae	25	Yes	0	0.8
<i>Ficus amplissima</i> (Itchi)	Boraceae	0	Yes	0	0
<i>Eriolaena quinquelocularis</i> (Kathale)	Sterculiaceae	8	Yes	0	0.3
<i>Cassia fistula</i> (Kakke)	Caesalpinaceae	38	Yes	0	1.3

Harvesters were asked what they had observed eating mistletoe fruits, and additionally how they thought mistletoes spread. Interestingly, although the two questions were aimed at establishing the same information, the responses differed. For example, although rats were cited as fruit predators they were not cited as dispersers (Figure 4.3). Responses also differed from the results of published research. Wind was cited as a dispersal mode for mistletoes by nine harvesters and one harvester considered wind the primary dispersal mechanism (Figure 4.3). Harvesters identified a much higher diversity of bird dispersers. Flowerpeckers have been cited in the literature as dispersers of *T. tomentosus* and Bulbuls also feed on fruit of the Loranthaceae (Ali & Ripley, 1983) but harvesters identified a total of nine bird species (Figure 4.3).

Harvesters had a detailed knowledge of the infection process and the mechanisms behind mistletoe spread; each mentioned all or several of the phases involved (Table 4.4). Harvesters identified growth of epicortical roots as a mechanism of infection accumulation within infected trees and considered this in their assessment of suitable management strategies. Epicortical roots are adventitious roots (arising from the mistletoe stem) that traverse the host bark, eventually establishing secondary haustorial connections (Calvin & Wilson (2006); Wilson & Calvin (2006)). The presence of epicortical roots was supported by observations during ecological studies (Chapter 2), but only the roles of intra- and inter-tree dispersal of mistletoe seed in the intensification of mistletoe infection on individual trees have previously been considered in the literature. All harvesters gave the same explanations when asked about the history of mistletoe infection in BRT. They stated that *T. tomentosus* is a native species but since the Forest Department prevented fire (approximately 20 years ago) it has become increasingly abundant in the forest. *“We have seen it right from our childhood but much less in numbers 20 years ago”* (Harvester 31). Harvesters also said that mistletoes were previously only seen in deciduous forest and not in scrub forest, or only extremely rarely, but that in the last 10-20 years they have spread into scrub forest, the population growing substantially in number. *“...25 years back there were mistletoes seen occasionally in the moist dense forest, from there it has spread widely to the open scrub forest”* (Harvester 34).

Harvesters were asked about the effects of mistletoe infection on Amla. All cited

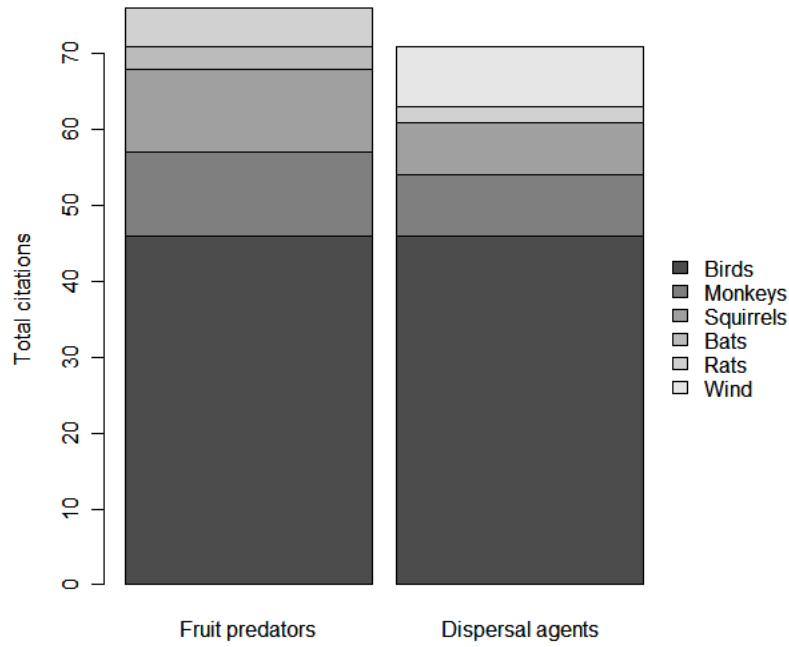


Figure 4.3: *Fruit eaters and dispersal agents as reported by harvesters*

Total citations of fruit predators ($n = 76$) and dispersal agents ($n = 71$) by harvesters; each harvester often mentioned more than one fruit feeder or dispersal agent. Specific birds cited by harvesters were: *Pycnonotus jocosus* (Red whiskered bulbul), *Dicaeum* sp./*Nectarinia* sp. (Flowerpecker/Sunbird), *Saxicoloides fulicata* (Indian robin), *Ducula badia* (Mountain Imperial-pigeon), *Streptopelia chinensis* (spotted dove), *Megalaima viridis* (white cheeked barbet), *Acridotheres tristis* (Common myna), ‘parrots’ and *Gallus* sp. (Junglefowl).

Table 4.4: *Harvester knowledge of infection initiation and spread processes*
Proportion of harvesters mentioning infection stages. Infection stages or processes not previously identified are given in italics.

	Stage of infection process	% harvesters
1.	Mistletoes are spread by birds, birds eat mistletoe fruits and defaecate the seeds onto tree branches	96%
2.	The seeds are sticky and adhere to tree branches where they germinate and the mistletoe grows	36%
3.	Following infection, mistletoe spread on a tree occurs via seed dispersal	15%
4.	<i>Following infection, mistletoe spread on a tree occurs via epicortical roots</i>	11%
5.	Infection spreads from the top to the bottom of the tree first outer twigs, then branches then finally the trunk	94%
6.	Infection results in progressive drying of the tree	83%
7.	Leaves dry and are shed and the bark of the tree splits	21%
8.	Flowering stops, productivity is reduced and eventually stops	91%
9.	Severity of effects increases as mistletoes grow and as their number increases	83%
10.	All infected trees eventually die	100%

reduced growth, reduced productivity and increased mortality rates (Table 4.2). Estimates of the reduction in fruit production caused by infection ranged from between 25 to 100% (Figure 4.4 b), with a mean higher than that indicated by previous experimental studies. All harvesters said that there is no recovery from infection, that all infected trees eventually die. Harvesters gave estimates of time to death following initial infection of between 2 and 10 years. Many mentioned that along with impacts on productivity, this mortality process is progressive and dependent on both the number and size of infections. “*Infection reduces the productivity of a nelli tree by 75% in the first 2-3 years, then later by 100%*” (Harvester 31). Opinion differed on the consequences of species (i.e. *P. emblica* or *P. indofischeri*) and tree age for infection impacts; a majority considered *P. indofischeri* more adversely affected by mistletoe infection, specifically due to the drier environment in which this species is found (Figure 4.5 a). Approximately equal numbers considered both older and younger trees to be more severely affected (Figure 4.5b). A previous study which investigated the

impact of infection on growth and productivity in both species using a 'before' and 'after' design, indicates a stronger influence of mistletoe infection on *P. indofischeri* than on *P. emblica* (Sinha & Bawa, 2002). Harvesters also mention effects of *T. tomentosus* on two other NTFPs, specifically 'Kanchuvala' (*Bauhinia acuminata*) and Lichen. Lichen is harvested from the bark of several tree species, including Amla. Harvesters reported that the drying of Amla and other host trees caused by mistletoe infection leads to lower lichen productivity on these trees. "*Lichen yield declines because infected trees lose lots of water due to the mistletoes, lichens are then unable to grow*" (Harvester 36). "*Lichen is one of the major NTFPs collected by tribes but infected trees also do not give good yield*" (Harvester 35).

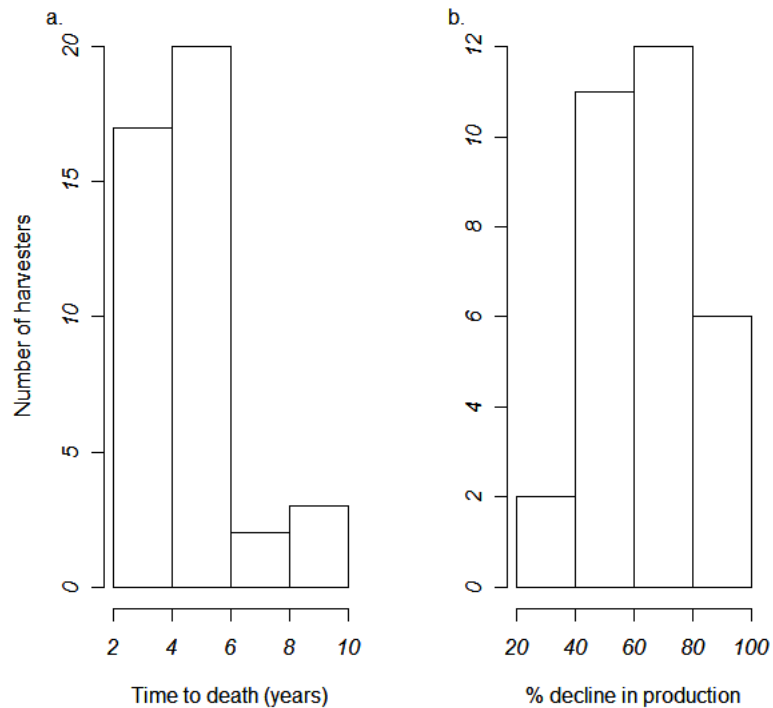


Figure 4.4: *Infection induced mortality and declines in fruit production*
Harvester estimates of *a.* the number of years following infection until tree mortality ($n = 43$), *b.* the percentage decline in tree fruit production as a consequence of infection ($n = 31$).

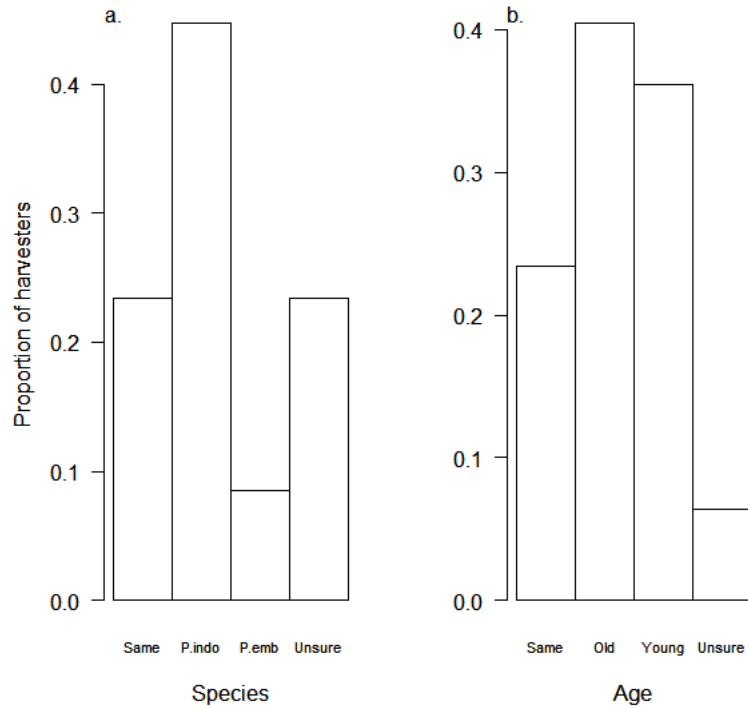


Figure 4.5: *Role of species and age on susceptibility to mistletoe infection*
Differential impacts of infection in relation to a. species (i.e. *P. emblica* and *P. indofischeri*) and b. age. Same= no difference in impact of infection between the two species, Unsure= respondent indicated they did not know if there was a difference ($n = 47$).

Harvesters were asked about the change in density of trees in the last 15 years, surrounding their podu, and also in the forest as a whole. They provided estimates of loss between 100 and 500 trees surrounding the podu and of 25-75% of the total Amla population (Figure 4.6). A few harvesters specified the greatest loss had been of older mature trees. “Now we are only seeing saplings and small nelli trees. 20 years ago we could see more than 500 trees within half a kilometre from our village, those trees now are all gone, ...” (Harvester 35). “We have lost all the high yielding trees, only the young and a few old fruiting trees are left, but these are also infected and only give a partial yield...” (Harvester 36). Harvesters described how their harvesting

behaviour and patterns had altered as a consequence of these changes, specifically an increase in distance travelled while harvesting. Previously they harvested close to the podu boundaries but now wander widely in the interior forest for collection, this being necessary to find an adequate number of trees to collect from. *“10 years ago we were harvesting sufficiently just 5-8km from the village, but now those trees are gone and we are going for harvest too great a distance from our village, ...”* (Harvester 31).

4.4.2 Variation in harvester knowledge

Overall knowledge held by harvesters was relatively homogenous, but possible explanatory factors were investigated where variation occurred. The differences in views found on infection prevalence, decline in productivity resulting from infection, time to death of infected trees, the influence of species and tree age on susceptibility, and the advocacy of fire as a potential tool in management were investigated. Years of harvesting experience had no effect on harvester estimates of prevalence, time to death following infection or lost productivity; however age partly explained harvester estimates of lost productivity. Gender was an important factor and suggested that men provided more conservative estimates for both prevalence of infection in the population and the time to death of infected trees. Gender however was not significant with respect to estimates of lost productivity (Table 4.5). Occupation influenced estimates of prevalence and lost productivity, the greatest differences being in estimates of infection prevalence (Table 4.5). Harvesters collecting NTFPs or practicing agriculture as main occupations gave estimates that were more accurate than those undertaking daily wage labour.

No effect of the explanatory variables investigated was found on harvester opinions regarding susceptibility (both tree, age and species), dispersal, or advocacy of fire versus chopping. Due to insufficient replication the effects of podu, and therefore location in the sanctuary on estimates, could not be investigated. Of particular interest was the possible influence of location on prevalence estimates. Chapter 3 demonstrated considerable spatial variation in infection patterns and consistent variation

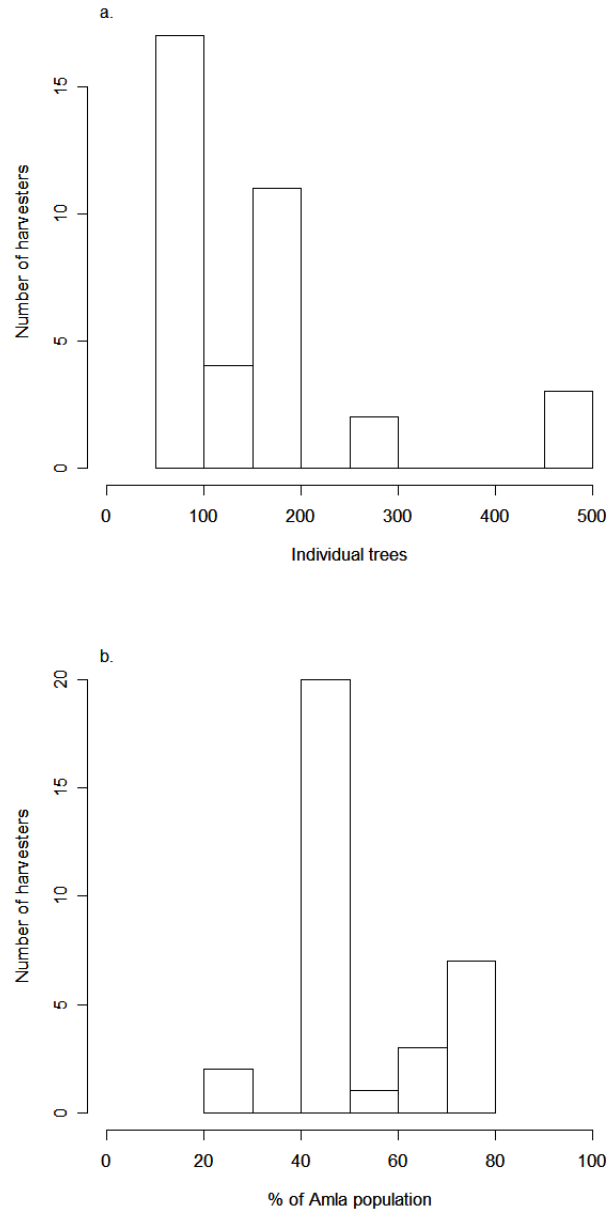


Figure 4.6: *Estimates of trees lost as a consequence of mistletoe infection*
Harvester estimates of a. the total number of trees lost surrounding the podu, and
b. the proportion of the total Amla population lost in the forest as a whole.

in harvester estimates in relation to spatial variation in infection levels was expected (Chapter 3). Harvesters in podus with a higher local prevalence of infection might be expected to provide consistently higher estimates of prevalence. However, the influence of the dominant forest type on the estimate of prevalence given was investigated by assigning podus either scrub or deciduous status based on the predominant forest type surrounding the podu, the hypothesis being that harvesters from podus close to or surrounded by mainly deciduous forest would provide higher estimates of prevalence than those in or close to scrub forest. However forest type had no influence on prevalence estimates.

Table 4.5: *Factors predicting harvester estimates of infection impacts*

Harvester estimates of infection prevalence (glm), declines in productivity (glm) and time to death of infected trees (lm). Harvester response was modelled as a function of gender, age, years of harvesting experience, main occupation and forest type. Only significant variables are presented, NS = non significant, Estimates from binomial glms were backtransformed from logits to mean proportions, all terms significant to $p < 0.01$ except *terms significant to $p < 0.05$.

Model term	Levels	Prevalence (d.f. = 42)	Productivity (d.f. = 26)	Mortality (d.f. = 40)
Sex:	Male	0.56	NS	5.2*
	Female	0.65	NS	3.2
Age		NS	0.5	NS
Years harvesting		NS	NS	NS
Main occupation:	Agriculture	0.54	0.79	NS
	Housewife	0.75	NS	NS
	Daily wage	0.62	0.59	NS
	NTFP collection	0.59	0.74*	NS

4.4.3 Effect on incomes

Harvesters reported significant declines in both average daily harvest and the duration of the harvesting season compared to 15 years previously (Figure 4.7). A mean decline of 88.5 kg in the amount collected per day and 15 days in the duration of harvesting represents a significant reduction in the total annual collection, a mean decline of

approximately 80% in individual annual harvests. Harvest records (from LAMPS) over a fifteen year period highlight the large temporal variability in fruit production (Setty (2004); Sinha (2000)) but do not appear to show evidence of this decline (Figure 4.8). Unit price changes over this period cited by harvesters match those in official harvest records (Figure 4.8).

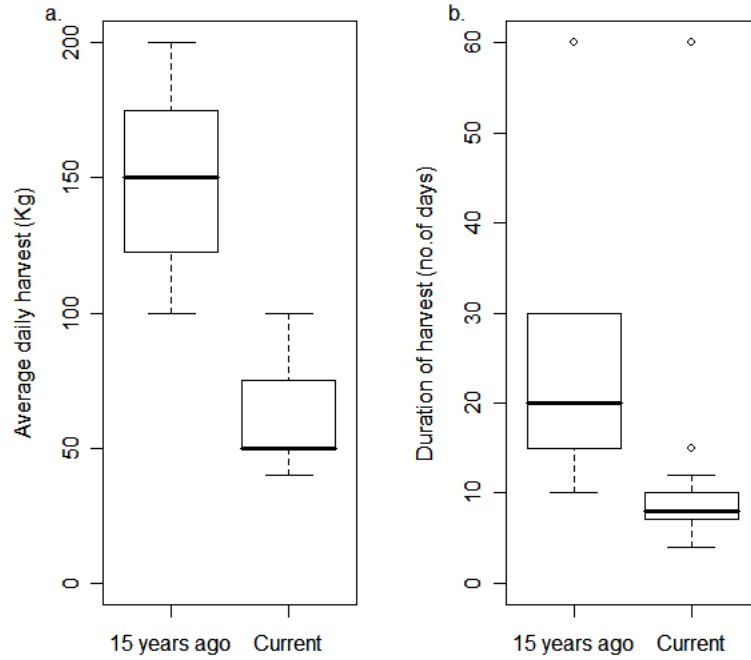


Figure 4.7: *Decline in average daily harvest and harvest duration* Current and historical (15 years previously), a. average daily Amla harvest (kg), and b. harvest duration (number of days).

4.4.4 Management behaviour and perspectives

Management suggestions made for the control of mistletoe included controlled burning, branch chopping and chemical control (Table 4.6). No significant influence of harvesting experience or other harvester characteristics on these views was identified.

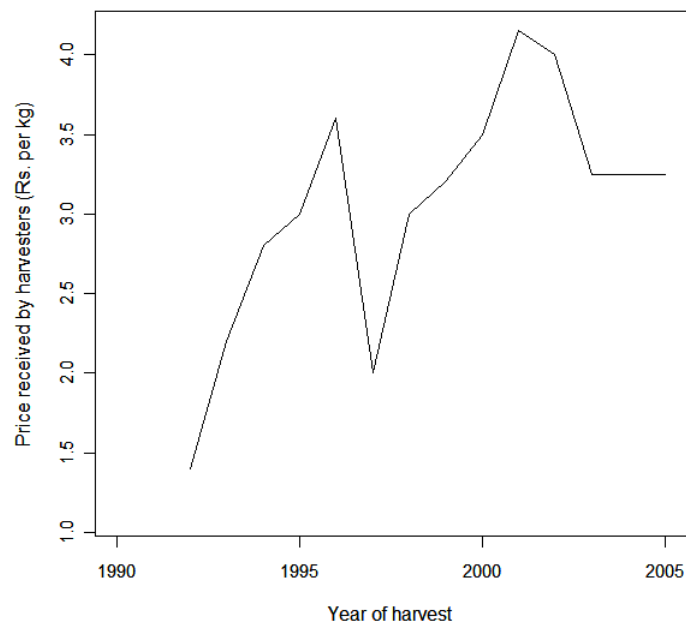
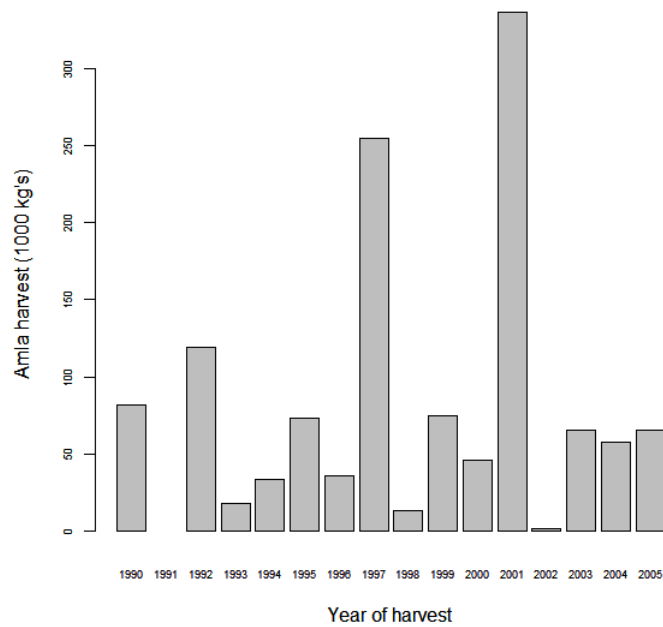


Figure 4.8: *Total Amla harvest and trading price between 1990 and 2005*
(Data provided for BRT 1990-2005 by LAMPs, Chamrajnagar)

Just under half suggested that branch chopping was the most suitable management response but many also pointed out the practical limitations, specifically the prevalence of mistletoe infection and the significant time required to do this for a large number of trees. Hand removal was mentioned as an option in theory but dismissed as not being a realistic consideration, this was due to practical limitations; being physically difficult and not permanent (mistletoe regrowth both from tissue remaining with the host branch (Chapter 2), and from epicortical roots (this chapter)) and for being ineffective in terms of regaining fruit production on branches from which mistletoes are removed. “...*infected branches are not growing again and therefore will not give a yield so removing the mistletoes does not help.*” (Harvester 15). “...*removing by hand is impossible because the mistletoe roots over tree branches are like a net, if a piece of root is left behind on a branch the mistletoe will grow again. Chopping is the best way to remove mistletoes. Infected branches of the tree are already partially dead so if we chop the branch it will help the tree to grow again with new leaves.*” (Harvester 33).

Harvesters all gave the same account with regards to the impact of fire on both mistletoes and on infected trees; fire kills mistletoes but does not adversely affect Amla trees. “*The fire acted as a remedy for the forest, pests and insects were destroyed and the ash fertilised the soil. 75% of mistletoes were killed with each seasonal fire.*” (Harvester 31). Harvesters were asked about current and past fire regimes. All said that the Forest department had banned fire. They said prior to the ban, fire occurred annually, burning only herbs and grasses at ground level and spreading 15 to 30 km over a period of up to two weeks. Harvesters said trees were not damaged and that when the rains came the forest grew back rapidly and healthily. In talking about fire history all harvesters also mentioned *Lantana camara*. They claimed that 20 years ago the forest was free from *Lantana*. They said that previously it was easy to see and move around in the forest and several suggested that the microclimate had also changed as a result, “... *with Lantana now we can't enter forest, is like a city in the village it so hot.*” (Harvester 1). However, only 21% suggested fire was the best approach for mistletoe management. Many harvesters that advocated chopping also mentioned fire, but said that the prevalence of *L. camara* in the forest prohibited its

use in mistletoe control. “*If we use fire to remove mistletoes the whole forest will burn due to Lantana.*” (Harvester 38). They believe that *Lantana* has replaced the grass understory, prevents the growth of tree saplings, and that forest fire is now more intense and sporadic, in some cases causing damage because it has not been properly controlled. “*In last few years there have been intense fires, causing lots of damage, this intense burning is because of Lantana.*” (Harvester 10).

Fire is suspected to be responsible for the high mortality rates of Amla seedlings identified in previous studies (Ganesan & Setty, 2004) but interestingly not one harvester mentioned the implications of fire for Amla regeneration. Several suggested fire could kill small Amla trees but there was no evidence to suggest they perceived this as a threat to the Amla population. Two harvesters suggested chemical means of control would be appropriate, one saying the government should spray herbicide because removal by hand or by chopping would be impossible due to the scale of infection. Additionally one harvester suggested the best approach would be to limit seed dispersal, and that by targeting the period when fruit are ripening the mistletoe population could be controlled. This is an interesting suggestion given recent publications investigating such strategies in the context of invasive species (e.g. Gosper & Vivian-Smith (2005)).

Table 4.6: *Management responses advocated by Amla harvesters*

Although harvesters provided a clear statement on the management approach they viewed as preferable, many mentioned additional approaches, therefore the percentages presented correspond to the total number of references to all approaches

Management approach	Proportion of harvesters
Branch chopping	45%
Fire	21%
Chemical control	4%
Don't know	30%

In terms of their own efforts while harvesting it is clear that despite discouragement from the Forest Department, removal by chopping is practiced commonly by harvesters. However, harvesters target only fruiting trees in their management. “*No-*

body removes mistletoes from non-yielding trees, they don't want to take risk. They concentrate only on the fruiting trees for their benefit so only a few yielding trees are chopped by harvesters each season..."(Harvester 15). *"This (chopping) does not give a complete result because again infection spreads from non-fruiting infected trees to these fruiting trees..."*(Harvester 17). Although this behaviour is common, a finding supported by previous observations, it is likely that its overall level of use as a management technique is lower as a consequence of Forest Department views on chopping (Chapter 2). *"Chopping has been prohibited by the Forest department and so no measures have been taken by the people to remove mistletoes..."* (Harvester 26). *"Removal is easy because every year we have contact with the trees while harvesting. Mistletoes should be removed, only then will the trees survive."* (Harvester 15). *"...there is no other way to prevent infection but the Forest Department prevent us entering the forest to collect NTFPs."* (Harvester 27). *"If the Forest Department provides wages to remove mistletoes people would take an interest and control it perfectly but the Department does not concentrate on the mistletoe problem and now they even prevent us from entering the forest, so no measures are taken to control it."* (Harvester 38). Two harvesters mentioned the use of fire in the neighbouring state, Tamil Nadu. There the Government formed village communities and through these committees, groups of young men were established that carry out seasonal burning in the forest during the summer. *"...they have been doing this for four or five years and now their forests are quite free from mistletoes, their forest grows better than our forest."* (Harvester 21).

4.5 Discussion

4.5.1 Correspondence between TEK and ecological studies

In general, ecological knowledge of harvesters closely matched the findings of ecological studies, in addition to those of previously published research. Harvesters provided accurate information on infection characteristics including primary host species, mistletoe distribution across forest types and within the Amla population, mistletoe

phenology and optimal growth conditions. However, there were also some discrepancies, most notably in secondary host species, dispersal mechanisms and the differential effects of mistletoe infection on *P. emblica* and *P. indofisheri*.

Although harvesters all identified *P. emblica* and *P. indofisheri* as the primary host species, a significantly larger number of secondary hosts were cited by harvesters than were identified in forest surveys. Harvesters could have a more detailed knowledge of mistletoe host associations than was detected by forest surveys. Some mistletoe-host associations may be particularly rare and surveys could have missed such rare pairings. However, sampling did in fact pick up several infrequent associations (e.g. a single infected tree in several species). Harvesters cited all species detected in forest surveys as host species except these very rare associations. In isolation suggesting that harvesters are in fact less able to provide information on these rare associations than the first finding in isolation might indicate.

Four of the species cited as hosts by harvesters were not observed in forest surveys. The additional 15 species cited may represent mis-identification by harvesters but this is unlikely. There are between five and seven mistletoe species found within BRT and another species may have been mistaken for *T. tomentosus*. However, harvesters had excellent knowledge of these other mistletoe species suggesting such mis-identification is unlikely. Additionally, two of these species *Kydia calycina* and *Mallotus philippensis* were widely cited, with over 40% of all harvesters mentioning them. More extensive sampling is required to conclusively establish the relative accuracy of TEK and field surveys in this respect, if shown to be reliable TEK would be considerably more efficient than field surveys in the identification of rare host-mistletoe associations.

Harvesters also identified significantly more bird species as fruit predators and/or dispersers than have been documented previously. This is not surprising as little work has focused on the ecological interactions of these mistletoes including their dispersal. However, animals were also identified as seed dispersers/consumers but dispersal of *T. tomentosus* by anything other than birds has also not previously been documented in the scientific literature. Mistletoes, including the Loranthaceae, are commonly dispersed by avian frugivores. However, *Romiciops australis*, an Argentinean marsupial,

has been documented as the exclusive disperser of the seeds of the Loranthaceous mistletoe *Tristerix corymbosus* (Amico & Aizen, 2000). Due to the lack of previous studies on dispersal of this particular mistletoe the possibility that mammals also play a role in the dispersal of this species cannot be discounted.

In gathering information on mistletoe dispersal, the question was framed in two different ways: harvesters were asked what they had seen eating fruit and also directly about the process of mistletoe spread. Fruits of *T. tomentosus*, as for other mistletoes, have an Endozoochorous dispersal mechanism (Calder (1983); Reid et al. (1995)); efficient dispersal requiring not only the ingestion and transport of the sticky seeds produced by these plants, but also their placement on the branches of an appropriate host. Although dispersal by mammals is feasible, wind dispersal, as cited by 20% of harvesters is not. This suggests that although harvesters might provide useful information when recounting their observations i.e. birds seen eating mistletoe fruits, care should be taken when gathering information that requires some kind of inference of an ecological process which may be more open to misinterpretation. Investigators should, where possible, frame questions to elicit information on specific observations rather than explanation of a process.

Opinion on the different susceptibilities of *P. emblica* and *P. indofischeri* was consistent among harvesters but conflicted with evidence from scientific studies. Failure to account for different mean intensities of infection between *P. emblica* and *P. indofischeri* in previous scientific investigations of impacts on growth and productivity (Sinha (2000); Setty (2004)) may account for this discrepancy and in fact harvesters may be correct. Information on differential susceptibility would be of value where management efforts seek to target the most vulnerable trees, particularly given the different importance of these two species in terms of their contribution to the harvested resource (Chapter 2). Therefore, further research is required to establish conclusively the nature and basis of differential susceptibility between *P. emblica* and *P. indofischeri*.

Harvesters were clearly aware of the consequence of mistletoe infection for tree growth, productivity and mortality and their estimates in terms of infection-induced mortality and productivity declines were accurate, their responses providing a good

match to study results. Harvesters were also accurate in estimating infection prevalence. However, some overestimated infection prevalence. Given the importance of the Amla harvest and the consideration of its decline as a major local problem by communities within BRT, some harvesters may have inflated their estimates of the number of trees infected in order to emphasize the severity of the problem. Although interviews were carried out by a local research assistant, some harvesters may have been aware of the involvement of a foreign researcher and biased their responses to questions they felt might elevate the recognition of mistletoe infection as a management issue.

Some evidence for the role of gender in the degree of reporting bias was found. Female harvesters provided higher estimates of infection prevalence in the Amla population and also provided lower estimates for time to mortality of infected trees, i.e. trees die faster following infection. Either women are more likely to bias their estimates upwards, or they are less knowledgeable in this context, viewing mistletoe infection to be a more severe problem than men. Harvesters who collect NTFPs or practice agriculture as their main occupations were found to give more accurate estimates with regard to infection prevalence and impacts of infection on productivity than those undertaking daily wage labour. These patterns may be because such harvesters (women and those with less forest-based primary occupations), may construct their estimates partly based on what they hear from others, and are less dependant on their own observations. These findings with respect to gender and occupation, indicate that even in a group of individuals selected as experts, those with more experience of the resource may provide more accurate information. However, although more experienced harvesters are generally expected to possess more knowledge, no influence of harvesting experience on responses was identified. Additionally, despite an age range of 48 years little influence of age on harvester responses was found. This also suggests that the potential for bias as a consequence of ‘shifting baselines’ (Pauly, 1995) may be minimal, but without ecological data on these trends it is hard to make a more solid inference.

The lack of an influence of location on harvester estimates (via either podu location or forest type) suggests harvesters are accurate in assessing mean conditions over

a large area and are not overly influenced by their own local conditions. Harvesters cover large areas of forest during their harvesting activities (aswell as when undertaking other activities at other times), therefore it may not be surprising that their assessments of prevalence at the population level are accurate. Indeed, other studies have shown that informant estimates tend towards the population mean, for example harvesters collecting below average amounts of a resource tend to overestimate their harvests while those collecting above average amounts underestimate (Jones et al., 2008). Thus, TEK appears to be particularly useful for assessing average conditions despite individual biases in reporting. In general, the results suggest that quantitative estimates relating directly to the impact of mistletoe infection on the Amla population can be reliable. At the level of the individual tree, harvester estimates showed a good concordance with scientific data and no evidence of bias was found.

Harvesters suggest that large numbers of Amla trees have been lost in the last fifteen years as a consequence of mistletoe infection, such estimates are of particular interest in establishing the full extent of the impact on harvester livelihoods. Information on the reliability of TEK data involving long recall periods is generally not promising (Jones et al., 2008). Recall could not be tested due to the lack of historical information against which to assess current tree densities, however the combination of various other pieces of information anecdotally supports harvester preceptions. Despite reports that *P. emblica* occurs in higher densities than *P. indofischeri* (Balachander, 2002), forest surveys failed to reveal a difference between the two species (Chapter 3). Where infection is causing tree mortality (Setty (2004); personal observation), this could reflect the higher prevalence of infection found in *P. emblica* (Chapter 2; Rist et al. (2008)); and harvester knowledge documented in this chapter). The extent to which any changes in host density have occurred as a consequence of mistletoe-induced mortality, and additionally the degree to which such changes could further influence the dynamics of Amla-mistletoe interactions, requires further attention. Commercialisation of the Amla resource occurred in the early 1980s. To what extent harvesting of this resource may also have played a role in the reported decline in tree density is yet to be established. Preliminary modelling studies (Sinha, 2000) suggest that current harvesting levels do not adversely affect population growth rates,

but information on past harvesting regimes would be necessary to eliminate the possibility that a previously more intensive harvesting regime played some role in the current population density and structure.

4.5.2 Management behaviour and perspectives

Although branch chopping has been demonstrated to have management potential (Chapter 2; Rist et al. (2008)), the Karnataka Forest Department (KFD) considers such chopping to be destructive. Results presented here suggest that the discouragement of this practice has reduced the prevalence of branch chopping among harvesters, despite it widely being considered effective. This indicates that although resource users may possess traditional resource management techniques, often with considerable potential for management, the practice of such techniques can be reduced where formal institutions have alternative, or conflicting management approaches and perspectives. Additionally such practices are also restricted by their opportunity cost. Even though branch chopping is considered an effective approach to controlling mistletoe, harvesters only do so for fruiting trees. This highlights the fact that cost is important and management-related knowledge held by resource users may not always be practiced, they may have the appropriate techniques but intervention is needed to provide the environment which makes this practical.

The reported reduction in tree population density also appears to have implications for sustainable harvesting practices. Previously it was common practice for harvesters to leave a proportion of fruit on the tree but they now collect fully from each fruiting tree. The implications of harvesting a greater proportion of fruit from a shrinking population may have consequences for regeneration and therefore sustainability.

Although the implications of the widespread growth of flammable *L. camara* for fire intensity were considered, and harvesters reported the damage caused by uncontrolled fire, they did not specifically acknowledge the potential impact of forest fire on Amla regeneration. This is important since the decision to promote a particular management strategy must be based on consideration of all potential effects. Fire

may reduce mortality of infected Amla adults through the removal of mistletoes, but due to mortality of seedlings and saplings it may have additional and negative consequences for population viability. Additionally, harvesters appear to over-report the effectiveness of fire in term of mistletoe control (Chapter 3).

Two other issues were of particular management significance. Harvesters consider the greater infection of *P. emblica* to be merely a consequence of a more suitable microclimate, rather than a greater susceptibility to infection. Given previous findings that mistletoe mortality is higher on *P. indofischeri* (Chapter 3), this offers important insights for further research and management, specifically that investigating a genetic basis for the different levels of infection in the two species could be a waste of resources.

As the impact of infection on fruit production and mortality rates are proportional to the number and size of mistletoes on an infected tree, information on the relative importance of epicortical roots in terms of infection intensification on individual trees in relation to seed dispersal is likely to be valuable for management. Specifically this mechanism could have important implications for attempts to model infection spread, and therefore the ability to target control measures most effectively. For example, a strategy that aimed at reducing dispersal by birds might reduce the risk of infection to uninfected trees but infection may intensify on previously infected trees as a consequence of clonal growth. The *Taxillus* genus has ‘basal’ epicortical roots (Calvin & Wilson, 2006), that initiate inflorescences (Barlow, 1997), subsequent fruit production and therefore intra- and inter-tree seed dispersal will occur. Epicortical roots can occur in response to wounding (Kuijt (1989); Calvin & Wilson (2006)), providing additional information to suggest hand removal may be ineffective as a management approach (Chapter 2; Rist et al. (2008)).

4.5.3 The effect of mistletoe infection on harvester income and harvesting effort as an indicator of sustainability

To date, it has been claimed that the harvesting of Amla in BRT is sustainable (Setty, 2004). Harvesting records do not suggest declining productivity, and in fact have been used as further evidence of this sustainability (Setty, 2004). Yet the number of har-

vesting days, and average Amla yield per day as reported by harvesters, has decreased substantially. Harvesters also report an increase in distance travelled for harvesting, an additional cost in terms of time foregone for other activities. Recall of historical harvest levels appears reliable as harvesters' reporting of the change in Amla price over this period matched official harvest records well, and there is no reason to suspect their collection quantities were any more open to errors in memory or other sources of bias. The considerable year to year variability in fruit production already documented (Sinha (2000); Setty (2004)) appears to be mirrored in official annual collection figures. However, in the absence of additional information on harvesting effort, harvested quantities cannot be taken as direct indicators of temporal trends in fruit availability. Combining harvester knowledge on current and historical collection patterns with official harvest records, suggests that an increase in the number of individuals participating in the Amla harvest may mask a declining resource base, with significant implications for harvester livelihoods. Monitoring on quantitative biological variables alone is insufficient (Stem et al., 2005). These findings emphasize the need to take account of other harvesting trends, for example, changing harvester behaviour (e.g. distance travelled by harvesters, number of trees harvested and method of harvesting) and the number of participating harvesters when assessing sustainability. Although it may also be a response to market changes, harvester behaviour may contain important information on resource status, e.g. an increase in travel distance might indicate a declining population. For example, harvester offtake per unit effort could be a useful gauge for monitoring the sustainable collection of NTFPs (?).

Although harvesters may switch to other activities, particularly as they may have previously been accustomed to year to year fluctuations in Amla availability, a decline in annual harvest amount of 80% represents a substantial lost contribution to cash income. Mistletoe infection also appears to impact other NTFP resources, most significantly lichen which is of significant livelihood importance in BRT (Hegde et al., 1996), providing a high return and being available for a longer time per year (more than 3 months) than any other NTFP. The nature of these effects, and the extent of their impacts, is yet to be determined but could further impinge on livelihoods dependent on NTFP collection in BRT.

4.5.4 Implications for the use of TEK in the management of harvested resources

This study illustrates the considerable body of traditional knowledge held by the tribal inhabitants of this area. It also demonstrates where such information may be most useful, and additionally where it may be misleading, and considers the sources of bias leading to such discrepancies. TEK can provide information more efficiently, and of the same resolution and accuracy, as conventional ecological studies. For example, TEK closely matched data from field research on mistletoe phenology. Phenological studies took place over a 12 month period requiring approximately 24 hours of fieldwork per month, involving two field workers, a total of 288 hours for the entire study. Social science methods to gather harvester information took considerably less time and resources, 47 interviews conducted by one individual taking a total of approximately 70.5 hours (1.5 hours per interview). TEK can also be a valuable source of information on rare events (i.e. rare host-mistletoe associations), that may require considerable fieldwork to identify, and a source of novel information, particularly relevant to the use of more experimental approaches to management. Such investigations can also help identify areas of concern for communities and resource users making conservation and management more locally relevant, and indeed the engagement of local people may be the most important reason for using TEK, particularly in areas where there is conflict over resource management.

Observational information from harvesters appeared to be more accurate than that based on understanding of processes or mechanisms, but this has not been the case in other systems where harvesters have elucidated complex biological, or ecological processes (e.g. Donovan & Puri (2004)). Essentially, the reliability of TEK is often highly variable (both within an individual body of knowledge and between TEK case studies), and also dependent on the perceived context in which it is gathered. There should be less focus on its degrees of “correctness” and more on what TEK can add to more standard scientific approaches. Tradeoffs are often required between accuracy, precision, and the resources available for conservation or resource assessment and management, in this context TEK may have considerable value. The limitations and

biases in both traditional knowledge and ecological studies should be recognised and conservation and resource management will benefit from their continued comparison.

4.5.5 Conclusion

Developing a sustainable production system for the Amla resource in BRT will require a clear understanding of how mistletoe infection affects Amla, the interactions between harvesting methods and mistletoe infection, fire and invasive species, and how these various elements function, both separately and synergistically. Traditional knowledge filled some information gaps and highlighted promising directions for management and further research. This knowledge but must be used in full recognition of its limitations and can be expanded upon through scientific methods (whose limitations must also be recognised), in addition to local experimentation based on traditional management practices. Such approaches, developing and testing our understanding with the participation of local people and resource users, will be both culturally appropriate and therefore more likely to be accepted and successful, and additionally, may be a more time efficient and cost effective approach to management and conservation.

CHAPTER 5

PARTICIPATORY ACTIVE ADAPTIVE MANAGEMENT, IS IT FEASIBLE? AMLA AS A CASE STUDY

5.1 Outline

The management of natural resources is often conducted under great uncertainty regarding future conditions, relationships among system components, responses to management, and particularly, the abundance of the resource itself. Adaptive management has been widely recommended as a way to deal with such uncertainty. However, to date, its implementation has been limited mainly to large complex systems in Europe and the United States.

Mistletoe infection represents a serious threat to the sustainable harvesting of Amla in the BRT wildlife sanctuary, but existing data, both from scientific studies and traditional knowledge, are insufficient to prescribe with certainty the best approach to mistletoe control. This chapter explores the theory and application of adaptive management, focusing on active adaptive management (AAM). Using a conceptual model of the Amla-mistletoe system to identify key management uncertainties, the viability of using this approach is considered. Managing mistletoe infection requires innovative methodologies that utilize all sources of existing information, and those that seek greater understanding of the impacts of infection, the mechanisms of infection spread, and the relative costs and benefits of alternative management approaches. Although an inherently scientific process, AAM offers a viable strategy for managing mistletoe infection in BRT, and its implementation, an opportunity to further investigate the merits of AAM in conservation and resource management.

5.2 Introduction

[Adaptive management] is an approach to natural resource policy that embodies a simple imperative: policies are experiments; learn from them... Adaptive management takes uncertainty seriously, treating human interventions in natural ecosystems as experimental probes. Its practitioners take special care with information. First, they are explicit about what they expect, so that they can design methods and apparatus to make measurements. Second, they collect and analyze information so that expectations can be compared with actuality. Finally, they transform comparison into learning, they correct errors, improve their imperfect understanding, and change action and plans. (Lee, 1993)

The conservation and management of natural resources must deal with considerable uncertainty; in the observations we make about the system we seek to manage, in the underlying behaviour of that system, and about the environment of which that system is a part (Hilborn & Mangel, 1997). Evidence suggests that we are still not able to deal adequately with these forms of uncertainty; this has been most clearly demonstrated in fisheries management (e.g. Ludwig et al. (1993); Walters & Maguire (1996)), but also in wildlife conservation (e.g. Doak (1995)), species re-introductions (e.g. Griffith et al. (1989)) and in biological control (e.g. McFadyen (1998)). Habitat destruction, invasive species, overharvesting, and the many other factors that threaten biodiversity are all urgent problems, reducing such uncertainties, even if theoretically possible, is limited by time and resource constraints (James et al., 2001). Consequently, there is a need for innovative approaches that allow us to utilise existing information in a manner that generates enhanced understanding of the system, while also engaged in its management.

Additionally, we endeavour to manage many natural resources in partnership with, and for the benefit of, local communities. Many resource users possess significant information on their environments and the resources they exploit (Berkes et al., 2000). They are also uniquely positioned to be, potentially, key components of management, particularly in implementation and monitoring. However, we lack practical

approaches for integrating local and traditional knowledge into management decisions, and the holders of this information into its implementation, as well as methods of assessing the relevance and appropriateness of specific local approaches to management (Berkes (2004); and the previous chapter). Such participatory and collaborative approaches to natural resource management can provide valuable biological insights but have their own uncertainties, and many still involve a rather superficial integration of resource users in the management process (Songorwa, 1999). Management approaches must take better account of system uncertainties in management decisions (Parma et al., 1998), using resource users themselves as part of this process, making management objectives more locally relevant, and hence management itself more sustainable (Berkes & Folke, 1998).

Adaptive approaches to management may offer one course for dealing more effectively with some of these challenges. Adaptive management explicitly recognises uncertainty and aims to reduce it through an experimental, hypothesis-based process (Walters, 1986), learning about the system becomes part of an iterative management cycle (Walters & Holling, 1990). It makes the best use of available information from all sources, and by prescribing flexible scenarios for the conservation and management of resources (Walters, 1986), makes management more robust to future challenges (Tompkins & Adger, 2004). It also provides a means to incorporate formal assessment of traditional or local management approaches alongside scientific ones, in a way that is transparent, and allows consensus building between all stakeholders; resource users, scientists and managers. Although such benefits have been recognised in existing applications, and many collaborative management approaches often incorporate ‘adaptive’ elements, the potential to implement management of an explicitly scientific and experimental form, in a participatory or collaborative manner has not been adequately explored.

Mistletoe infection of Amla in the BRT wildlife sanctuary represents an ideal case study for such an exploration. Although there is significant existing information upon which to base management actions, additionally considerable uncertainty remains. Institutional and local perspectives on the management of mistletoe infection differ, and the available scientific evidence does not provide enough information to

indicate definitively which approaches are best for mistletoe control. In particular, there is significant uncertainty regarding the complex ecological and human influences likely to underlie current mistletoe distribution (including the roles of fire, harvesting techniques, and interactions with invasive species), as well as the mechanisms of infection spread, and the full extent of the impacts for the Amla resource. The need for management intervention is urgent (Chapter 2) and generating the additional information required to fill these gaps will demand time and resources.

This chapter considers the potential for implementing an active adaptive, and participatory approach to the management of mistletoe infection of Amla in the BRT Wildlife Sanctuary. The theory of adaptive management is reviewed, and the adaptive management process outlined. The application of adaptive management to natural resource use and conservation to date, including identified barriers to implementation are assessed. The current uncertainties involved in managing mistletoe infection are outlined, using a conceptual model to bring the findings of the previous three chapters together with other information in the literature. This chapter concludes by considering if this participatory or collaborative version of active adaptive management is possible, and if not what the alternatives might be for managing Amla and other similar systems in tropical forests.

5.3 What is adaptive management?

“Adaptive management” (originally termed Adaptive environmental assessment and management (AEAM) Holling (1978)) first appeared in the natural resources management literature in the mid-1970s (Holling (1978); Walters & Hilborn (1976); Walters & Hilborn (1978)) in a response to a realisation of the extent of uncertainty involved in natural resource management and a frustration with attempts to use modelling to resolve these uncertainties (Walters, 2007). The value of active experimentation was first recognised in fisheries management. For example, implementing a deliberate policy of overexploitation in order to determine maximum sustainable yield; such manipulations being implemented as far back as the early 1950s (INPFC (1962); Skud (1976)).

More recently, adaptive management has received attention from both the conservation (McCarthy & Possingham (2006); Varley & Boyce (2006); Irwin & Freeman (2002)) and social sciences elements of resource management (e.g. Berkes (2004); Stringer et al. (2006); Olsson et al. (2004)). Associated with this wider influence, use of the term ‘adaptive’ in the context of natural resource management has broadened from the concept as first articulated by Holling (1978) and Walters (1986). In many cases, the claim of an adaptive approach in management has been justified simply by the idea that the results of initial policy choices will be monitored so as to identify a need for corrective action at a later stage. Many resource users and managers respond to changing circumstances in their use and management of resources, both fluctuations in resource availability (Ghimire et al., 2005) and more dramatic shocks (Berkes et al., 2000), for example in response to the “fast track” land reform programme in Zimbabwe (J Gambiza, personal communication). Consequently, failure to note the specific experimental element of this approach in presentations of its use has led some to question whether, in fact, adaptive management is anything other than how resource users normally behave, and even if it is a hollow concept.

It is important therefore, to outline the diversity of understandings, and to relate these to the terminology commonly encountered in the literature. Different forms of adaptive management have been highlighted (Walters & Holling, 1990). Most significant is the distinction between ‘active’ and ‘passive’ forms of the approach (e.g. Shea et al. (2002); Walters (2007)). Passive adaptive management (PAM) focuses on the implementation of an historically informed best practice or policy. New information gained from its implementation is incorporated into management plans, learning therefore occurring during the course of management. There are many examples of PAM in the ecological literature (e.g. Varley & Boyce (2006); Johnson (1999)), including examples of its successful participatory application (Aswani et al., 2007). However, in many respects, PAM can be considered analogous to the conventional approach to managing natural resources (Parma et al., 1998). Active adaptive management (AAM), in contrast, is explicitly experimental. AAM focuses on learning rather than implementation (Shea et al., 2002); with management actions being treated as deliberate, large-scale experiments (Walters, 2007).

This distinction is best illustrated with an example; a PAM approach to mistletoe management in BRT might involve supporting a historically informed policy of clearing mistletoe from 50% of infected Amla trees in a selected management area to reduce the spread of infection to other trees. In time, the benefits of this strategy may be confirmed. Alternatively, it might be learned that clearance of 50% is not sufficient and the clearance level would be increased. In AAM, a range of clearance levels would be implemented from the outset, some lower, and some greater than the current best practice. Designing and applying ‘treatments’ in this way allows more information to be gained in the same time frame. In this example, potentially learning that not only is 50% too little, but that there is no impact on the spread of mistletoe infection until, say, 75% of trees are cleared. Or maybe 25% would be found to be sufficient, in which case management resources can be diverted elsewhere. An active approach allows the current best practice to be changed more precisely (see Appendix C for an outline of the adaptive management cycle, and an outline of the AAM process).

Adaptive management was initially conceived as a technical-ecological model (e.g. Holling (1978); Walters (1986)). Understandings have since broadened. Within different interests and contexts, people focus on different aspects of the adaptive management approach, and a diversity of acronyms now describe management as an adaptive process. Adaptive Collaborative Management (ACM) is an approach rooted more in the social sciences and widely used in forest management (Pierce Colfer (2005a); Pierce Colfer (2005b)). ACM involves cooperative planning and management by a group of stakeholders. It is characterized by conscious efforts among such groups to collaborate and seek out opportunities to learn collectively about the impacts of their actions (Pierce Colfer et al., 2005). Adaptive co-management refers to a similar process, focusing more on the governance of natural resources (Olsson et al., 2004). Such methods concentrate on active reflection and evaluation, and mechanisms for incorporating learning and the sharing of rights and responsibilities into planning and management. They are not based on scientifically informed experimentation. Adaptive Collaborative Management and Adaptive co-management can therefore be considered participatory examples of passive adaptive management.

Adaptive management builds on methods from a range of disciplines, including

the social sciences, (for example recognising the importance of institutions and social structures to management and policy decisions), and it has benefits for attaining wider goals in management (e.g. conflict resolution (Stringer et al., 2006)). However, adaptive management is about optimal management not democratic resource governance, social learning or consensus building between stakeholders. Only AAM (also defined as the ‘scientific’ approach to adaptive management (McLain & Lee, 1996), captures the concept as originally defined by Holling (1978) and Walters & Hilborn (1976). It is the application of AAM, in a participatory manner, that is the subject of this chapters’ investigation.

5.4 Adaptive management in practice

A search of ISI Web of Knowledge for 1997-1998 found 67 papers dealing with resource management that used the words “adaptive management” in their title, abstract, or key words. A similar search for 2007-2008 found 166 papers. To date most applications of AAM plans have been to large, complex ecosystems in the US and Europe; mainly in riparian and coastal management (e.g. river management in the Kruger National Park (Rogers & Biggs, 1999)) or fisheries management (e.g. re-zoning and fisheries policies in the Great Barrier Reef (Mapstone et al., 1996)), along with some forestry examples, most famously the Northwest forest plan (Bormann et al., 2007). In such systems, complex proposals have been implemented by interdisciplinary, multi-institution teams. Stakeholders being made up of scientists, policy makers and system managers, significant financial resources being available for management and with well established institutions as a framework. Although there are recent examples of AAM being applied to ‘smaller’ problems (e.g. Armstrong et al. (2007); Varley & Boyce (2006); Dimond & Armstrong (2007)), no published examples were found of the use of AAM in a participatory manner, or applied to the less well resourced systems typical of natural resource management in the tropics.

The appeal of adaptive management has been extremely widespread. Legislative requirements for the use of the approach are not uncommon (for example, Californias Marine Life Protection Act (Department of Fish and Game, 1999), or the Northwest

Forest Plan (Bormann et al., 2007)). Many conservation organisations are now also looking at the potential of these methods (e.g. Salafsky et al. (2001); Margoluis & Salafsky (1998)). Partly a result of the intuitive value of the approach in their contexts, and additionally, perhaps more cynically, as they are under increasing pressure from donors to be able to demonstrate results; the iterative cycle of adaptive management offering a more obvious documentation of management progress. However, in such instances there has been no critical assessment of the feasibility of the approach for the smaller scale conservation and resource management issues which they seek to tackle. Although several limitations to the implementation of AAM have been widely cited (e.g. McLain & Lee (1996); Walters (1997); Lee (1999); Allan & Curtis (2005)), there has been little reference to this body of information. These barriers are considered here prior to investigating the potential for implementing adaptive management in its active form in the Amla case study.

Cost, risk and irreversibility

The most frequently cited barrier to adaptive management is cost. There are costs associated with developing system models and with the analysis of gathered information, in designing experimental management regimes and the management actions themselves may be more expensive than the status quo (Shea et al., 2002). In those cases where participatory methods have been a strong element, engagement with stakeholders can also be a costly process (Lynam et al., 2002). The major cost in adaptive management is often considered to come from monitoring (Walters, 1997). Costs depend on the variables being measured; the potential to capitalise on economies of scale by monitoring multiple variables, and the development of new methods of ecological field measurement offer opportunities to reduce these (Walters, 1997). As well as the costs associated with operationalising the management programme, there are those incurred as a consequence of its implementation, for example the opportunity costs of potential short term yield reductions. The timescale associated with the distribution of such costs and the benefits of management can have important consequences (Hauser & Possingham, 2008), as can their distribution across different stakeholders

(Balmford & Whitten, 2003).

Management may have two benefits, a short term pay off related to the specific management objective, and additionally learning about system responses to improve payoffs (or reduce damages) in the future. There is often a trade off to be made between these (the ‘dual control problem’), we might learn quicker by introducing larger disturbances and carrying out more monitoring but this will also involve a higher risk and more cost (Walters & Hilborn, 1978). Learning in management will always involve some level of ecological risk. In the Great Barrier Reef some conservation groups opposed experimental reopening of closed reefs, but in fact these actions provided evidence for the efficacy of no-take areas (Hughes et al., 2007). In some cases, the risk associated with the opportunity for learning is high (Parma & Deriso, 1990), and certainly irreversible management treatments should be avoided (Walters, 1997), but in most cases, implementation can be designed to reduce such risks substantially.

Self-interest and other institutional barriers

The “self interest in research and management organisations” described by Walters (1997) may be a significant obstacle. Stakeholders with different interpretations of system relationships, levels of power and abilities to communicate must frequently be brought together in the management process. Where a single lead management agency exists this may be straightforward but where institutional settings are more complex problems are common (Gunderson et al. (1995); McLain & Lee (1996)). Organizations and individuals involved in management may be reluctant to admit their mistakes or errors of judgement, or to alter previously advocated management approaches (Shea et al. (2002); McDougall (2007)). Walters (1997) outlined three factors which have impeded the implementation of adaptive management from this perspective; (1) a belief that a pretense of certainty is necessary to maintain management agency credibility, (2) a research process by scientists that attempts to answer ever more focused questions in the pursuit of ‘enough’ information with which to make management recommendations, (3) a tendency for decision makers to consider

inaction a rational choice, by delaying and avoiding substantial policy changes they can ensure the difficult decisions are made by someone else.

Ecological complexity

Ecological systems are extremely complex (Holling, 2001). A single management intervention may have many different impacts which are rarely easily separated. Cross-scale connections, indirect, synergistic effects, and nonlinear relationships are common (Antunes & Santos, 1999). Management actions can change the sensitivity of the system to natural environmental factors themselves with complex temporal patterns. Such ‘time-treatment interactions’, where treatment depends on the point at which the treatment was applied, mean comparisons between treatments may not provide a reliable estimate of their effects (Walters & Holling, 1990). Additionally, process uncertainty and transient responses, although issues for all management, can limit the ability to effectively assess the consequences of experimental interventions (Walters & Holling (1990); Parma et al. (1998)). Some have questioned whether ecological systems are even stable enough to permit learning (Johnson & Case, 2000).

The success of AM depends on the ability to learn quickly enough to make the knowledge gained useful. If the response time of the system is much longer than the frequency at which decisions are made, there will be no opportunity for learning (Shea et al., 2002). The value of information also declines with time, discounting favouring current ‘best bet’ strategies. While waiting for sufficient trends to emerge over the effectiveness of management, stakeholders may be incurring costs; those associated with management, monitoring, and also potentially lost production if the management is ultimately ineffective. Completing just one adaptive monitoring cycle took ten years in the case of the Northwest Forest Plan (Bormann et al., 2007). The information needed to improve management must be obtained on a timescale that allows it to be useful, and the benefits of such information must compensate for any forgone during the time required to get it.

In summary, the costs (perceived and real) of implementing management, including monitoring have been substantial and experimentation has been weak (partly due

to perceptions of risk). Self-interested institutions have also impeded progress and ecological complexities have limited the ability to reduce key uncertainties.

5.5 Amla: Present situation, management and uncertainty

Before considering the potential for implementing an AAM approach for Amla, a brief summary of the current uncertainties in the system are useful. Mistletoe infection represents a significant threat to sustainable harvesting of Amla in the BRT wildlife sanctuary; the prevalence of infection is high (Chapter 2; Rist et al. (2008)), and infection leads to tree mortality and reduced productivity (Sinha (2000); Setty (2004); Chapter 4). Management faces two crucial challenges; (1) despite evidence demonstrating the serious nature of this threat, current monitoring programmes have not highlighted a decline in the Amla population (Chapter 4), and (2) there are disagreements over current approaches to managing mistletoe infection and their potential for broad implementation remains uncertain (Chapters 2, 3 and 4).

Branch chopping currently emerges as the most promising method of mistletoe control (Chapters 2 and 4). Potentially it offers three management benefits; (1) effective removal of mistletoes, (2) enhanced productivity due to coppicing, and (3) reduced risk of infection (Chapter 2). But uncertainties over the interval of reduced infection risk, and the immediate impact on fruit production make its benefit over a longer management horizon uncertain. The temporal and spatial characteristics of infection spread and intensification are not well understood (Chapter 3), therefore the optimal approach for implementing a selected method of mistletoe control is also unclear, and additionally is likely to vary with specific management goals.

In addition to mistletoe infection, forest fire and the spread of *Lantana camara* are two key current concerns in BRT. Fire prevention has been implicated in the reported increase in mistletoe abundance, but its continued occurrence, also in Amla tree mortality and lack of regeneration. Historical fire prevention may have contributed to the current widespread abundance of *L. camara* (Chapter 4), but burning may now enhance its regeneration (Hiremath & Sundaram, 2005). Furthermore, *L. camara* may facilitate the spread of mistletoe infection (Chapter 3), and limit Amla regeneration

(Uma Shaanker et al., 2004), therefore the net impact of fire on Amla population dynamics is uncertain.

Infection patterns and impacts appear to differ between the two *Phyllanthus* species and as they do not contribute equally to the harvested resource, this also has implications for the implementation of management strategies (Chapters 2). A further complication is uncertainty over how impacts from mistletoe infection and harvesting might interact.

Drawing on the results of the previous three chapters, and additional studies in the literature, a conceptual model provides a representation of the processes influencing mistletoe infection in the Amla population (Figure 5.1). A conceptual model is a visual representation (and/or written description) of predicted relationships between ecological components of a system, and the influences to which they may be exposed. Presented as a box and arrow conceptualisation of stressors, system parameters and ecological attributes, this model highlights some of these uncertainties. The processes of infection spread and intensification, and the roles of fire, invasive species and harvesting techniques, as well as their interactions are included. The two species, *P. emblica* and *P. indofischeri*, are represented together in this model. Initial hypotheses about some of these model relationships are outlined in (Table 5.1).

5.6 Participatory active adaptive management for Amla?

Having identifying the main uncertainties with regards to the management of Amla (Section 5.5), and the major barriers to the use of AAM (Section 5.4), the feasibility of implementing the AAM process (Appendix C), in a participatory manner is now considered.

The management of Amla is an important issue for a diverse group of stakeholders including Amla harvesters, the Soliga of BRT more generally, the Karnataka Forest Department (KFD), local NGOs (Chapter 1; Section 1.1.6), as well as researchers from national and international institutions. Prior to starting this process, some form of ‘management body’ would have to be established encompassing members from across these different groups. The Soligas, being the largest group of stakeholders and po-

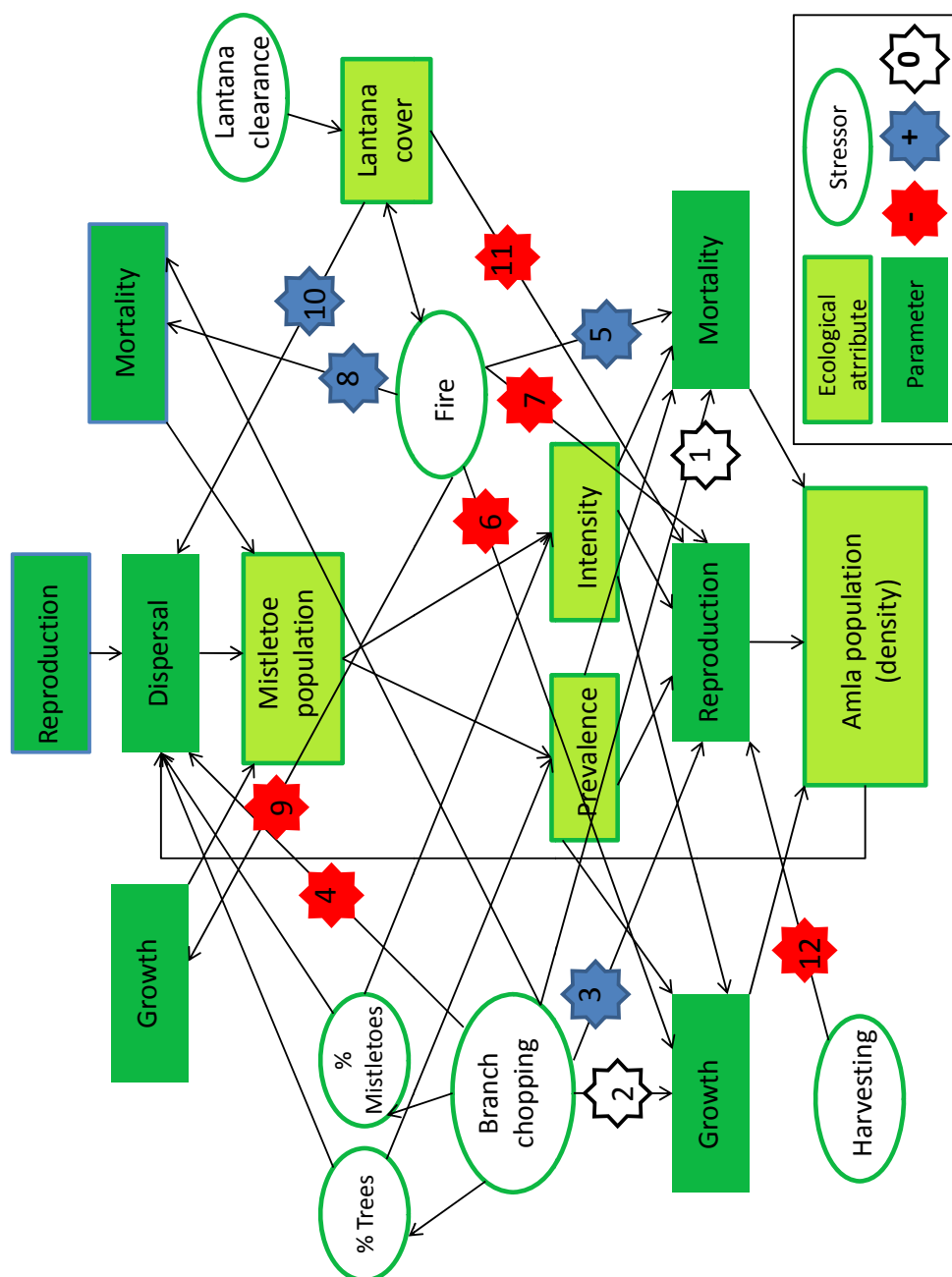


Figure 5.1: A conceptual model of mistletoe infection of Amla

The model identifies the major stressors, system parameters and ecological attributes that characterise the system. The direction of the effects are qualitative and correspond to initial hypothesis about the dynamics of the system.

Table 5.1: *Amla-mistletoe system hypotheses*

Initial hypotheses about the dynamics of the Amla-mistletoe system to be tested in an AAM programme. Model links refer to the conceptual model (Figure 5.1). The predictions given are qualitative, for example, link 1 specifies the initial hypothesis that branch cutting has no effect on Amla mortality.

Model link	Stressor	System parameter	Hypothesis
1	Branch cutting	Amla mortality	No effect
2	Branch cutting	Amla growth	No effect
3	Branch cutting	Amla Fruit production	Increase
4	Branch cutting	Risk of infection	Decrease
5	Fire	Amla mortality	Increase
6	Fire	Amla growth	Decrease
7	Fire	Amla fruit production	Decrease
8	Fire	Mistletoe mortality	Increase
9	Fire	Mistletoe growth	Decrease
10	Lantana	Risk of infection	Increase
11	Lantana	Amla regeneration	Decrease
12	Harvesting	Amla regeneration	No effect

tentially those with the most internally diverse perspectives, might initially form an ‘Amla Management Committee’. This committee would operate independently of the management body to compile, and later represent, community opinions. Existing Soligas institutions may be best placed to assume this function (Ostrom, 1990). Conflicts can occur in any stakeholder group despite good intentions and agreement about desired outcomes. At this initial stage, a protocol for conflict resolution within the management body should be established. A simple and common framework for resolving disagreements to which all parties involved agree at the outset. This protocol might specify where some issues can be resolved democratically in order to maintain progress, and for others where negotiation must continue until agreement is reached.

5.6.1 Define the management objective

The start of the process involves bringing all stakeholders together to discuss management of Amla. This assumes that all are willing to jointly identify a common management objective. There are positive indications that the groups can work together

in order to do this. Through participatory resource monitoring programmes, Amla harvesters, and the Soligas communities more generally, have worked with NGOs in NTFP monitoring and value-addition and processing programmes (Lele et al., 1998). Despite some historical tensions, as may be expected from organisations with differing mandates, local NGOs also have good working relations. This history of collaboration and partnership provides a solid framework for the implementation of an AAM program. The biggest uncertainty relates to the KFD. The KFD has formal relations with all other stakeholders, for example through LAMPS, but there have been recent tensions in these. Researchers have experienced difficulties obtaining research permits. There has also been conflict with the Soligas communities supported by VGKK with the KFD due to a recently implemented ban on NTFP collection, as well as incidents of forest fire (Kalpavriksh, 2007a). However, there is current discussion by the KFD regarding the establishment of a joint management committee at BRT which offers some promise.

Broad agreement must be achieved regarding the objective of Amla management. This objective may be articulated differently by the different parties; the KFD for example might focus on maintaining or increasing current densities of Amla, whereas harvesters might emphasise maximising the Amla harvest. These can be compatible, such an aim could be outlined as ‘sustaining or increasing Amla production such that the livelihood role of Amla can be maintained or enhanced without threat to the Amla population’. The Soligas, NGOs and researchers have already demonstrated similar management priorities. Key drivers of environmental change in BRT were identified in a series of meetings held from 1994 onwards (Lele et al., 1998). The Soligas identified fire, invasive species, and harvesting as potential drivers of system change, similar threats have also been outlined by the KFD in management documents and this shared understanding suggests agreement on a common aim should be obtainable. The ability to realise this however, depends on a shared vision between the stakeholders. There have been rumours of the KFD’s wish to relocate tribal villages from the sanctuary (as has been implemented for other protected areas in India (Rangarajan & Shahabuddin, 2006)). If the KFD’s future vision of BRT does not include its tribal inhabitants, and their use of forest resources, focusing instead, for

example on the potential for tourism, then the AAM process will not progress past this first stage.

5.6.2 Describe what is known about the system

As the initial step towards meeting the identified management objective, an assessment of all the available information on the system is required. This includes acknowledging uncertainties and specifying alternative hypotheses regarding relationships between components of the system (Figure 5.1). Together stakeholders would interpret the results from all previous studies at BRT, and combined with their own knowledge, develop a shared understanding of the system. All will have different interpretations of system relationships, levels of power and abilities to communicate making this stage a major challenge.

When advocating specific management approaches, the Soligas have demonstrated a clear understanding of the associated costs and benefits, having a pragmatic approach on many issues, for example, the challenges of re-introducing fire as a management tool. Branch chopping and fire in particular have already generated controversy, both among, and within stakeholder groups. These specific relationships are likely to major discussion points. If agreement cannot be reached, the management body can instead decide to focus on the resolution of these points in the first round of management with specific opinions being tested. Mistletoe management would follow once the influences of branch chopping and fire have been clarified. Indeed, some have emphasised that the adaptive management process is, itself, a way of overcoming disagreements and conflicts between stakeholders (Lynam et al., 2002). Methodologies for consensus building are available, including those specific to adaptive management (e.g. Lal et al. (2002); Walkerden (2005)) and professional facilitation (e.g. Foundations of Success (2000)) can be used, indeed experience suggests this is invaluable in this initial phase (P. Shanley, Personal communication).

Institutional barriers to developing this shared understanding are likely to be significant. For example, the KFD in particular may be unwilling to consider recent evidence questioning the utility of hand removal, an approach they have previously

supported, or indeed branch chopping, one they have discouraged. Researchers may have to admit to each other, and to other stakeholders, significant gaps in their knowledge. This may be particularly sensitive where the credibility of previously held positions are questioned. For example, advocating the lifting of the harvesting ban due to sustainable collection levels. Researchers will also perceive risks to their longer-term research activities at BRT.

The next challenge is how to represent a common understanding of the system so that it can be effectively used in the management process. Typically formulated mathematically, a system model is essential. For communication, for the identification of potential management actions and to identify, and therefore test, gaps in knowledge via predictions about the impacts of alternative policies (Walters, 1997). A dynamic model representation of the system (i.e. accounting for elements of time) has been identified by many practitioners as critical to the AM process (e.g. Walters (1986); Schreiber et al. (2004)). Although such models have been successfully developed in partnership with local communities (Lynam et al., 2002), mathematical modelling is an intensive activity, in terms of skill, time and cost. Attempts to implement the AAM process without a dynamic system model may well limit the resolution of learning about system relationships, and therefore result in less ‘precise’ management. However, this does not erode the value of an AAM approach. A conceptual model, as presented in Figure 5.1, can provide all that is necessary to convert the broad objective of management into specific, measurable indicators and to develop a set of clear hypotheses and policy options.

5.6.3 Identify indicators and define goals related to the management objective

Taking the objective outlined above, “sustaining or increasing Amla production such that the livelihood role of Amla can be maintained or enhanced without threat to the Amla population”. Most infected trees do not produce fruit, of significance for both harvesting and regeneration. Reducing the number of infected trees over a specified time period would therefore be an appropriate goal under this objective,

mistletoe prevalence (the proportion of the Amla population infected), being the relevant indicator. A decision structure is then specified, including a basis upon which to define the end of a management cycle. If monitoring of the indicator takes place annually this goal might then be articulated more precisely as ‘to reduce mistletoe prevalence in the Amla population by at least 25% within one year’. Time-bounded management goals are an element lacking in much conservation planning and resource management.

This decision structure can also include boundaries for responses of the system, these would act as safeguards in experimentation. For example, if after implementing a chopping regime, tree mortality appears to be correlated with the application of the chopping treatment then implementation can be halted immediately. Management would then be re-evaluated using an updated model. In this context, the Soligas can fulfill an important role due to their presence in the forest observing day to day changes. A more challenging aspect regarding the decision structure is defining a point (based on evaluation of the indicator), at which implemented strategies would be altered. For example, if after implementing a clearance strategy of 25% of infected trees, and upon reaching the first evaluation there appears to be no reduction in prevalence (i.e. infections removed are being compensated for by new infections), clearance could be increased immediately to e.g. 50% or another year of the same strategy implemented giving the system longer to respond to management. Such details would require detailed consideration by the management body.

5.6.4 Consider alternative management options and predict their outcomes

Several potential mistletoe control options have already been identified by harvesters including; branch chopping, fire, herbicides and manipulation of seed dispersal processes (Chapter 4). Other stakeholders may come up with additional possibilities. Researchers can investigate methods applied in systems elsewhere. For example, the growing literature on the control of invasives provides valuable information on the spatial aspects of implementing control measures. Alternative options identified by

researchers could be presented to the management body for consideration. After identification of a range of management options, predictions must be made regarding the outcome of their potential future implementation. These predictions are based on the system model and provide a basis on which to evaluate different management options. The final selection will be a compromise between the likely effectiveness of reaching the management goal and the costs of implementation.

5.6.5 Implement the management actions

Once an appropriate management option is selected, consideration must be given over how to implement it in selected management areas. Implementation should be designed to test specific hypotheses and it is at this stage that experimental design is critical (Walters, 1997). Continuing with the branch chopping example, and the goal of a 25% reduction in prevalence in one year, the following treatments might be selected: clearance of 100%, 50% and 25% of infected trees of all their mistletoes. The hypothesis to be tested being that infection spread to neighbouring trees occurs as a function of local infection prevalence. Another hypothesis may be that spread of infection occurs faster in areas of high host density (e.g. in an area of low Amla density, mistletoe clearance is more successful in attaining the prevalence goal after one year, than in a location with a higher density of hosts). By controlling for host density, this hypothesis could also be tested. It is not the aim here to provide a full experimental plan but merely to illustrate what would be required by presenting these two testable hypotheses. The challenge is to develop a nested experimental design that will permit the clear separation of the effects of as many of additional hypotheses as possible, for example, those relating to the impact of branch cutting itself. The design of diagnostic management experiments would be a key task for the researchers involved.

The practicalities of implementation must also be considered. Depending on the management actions selected, a significant amount of labour may be required for their implementation. This would be provided by Amla harvesters and other community members and coordinated by the management body. Such management activities

would provide much needed employment opportunities in BRT. The KFD has previously paid Soligas to clear fire breaks and carry out other manual work, they may therefore be able to make a similar small budget available for mistletoe removal. Additionally, LAMPs has previously funded participatory resource monitoring and may make a similar investment. Researchers and NGOs may also be able to use funds from existing budgets allocated to BRT. However, financial uncertainties could place additional strain on the management process. The management benefits in terms of enhanced Amla production may take some time to appear, and given that revenues are also subject to market influences, such a management programme is very unlikely to be able to pay for itself in the short or medium term. A more viable option would be to consider the entire management exercise as a research project and to seek external funding for its implementation. Investigating the theoretical potential of the AMM approach, in a practical rather than a theoretical manner, would be of interest to conservation practitioners and the research community, a well planned study would therefore stand a fair chance of attracting funding. Additionally, there might be opportunities for combining more ecologically and socially focused studies, e.g. mistletoe spread processes or the role of fire on mistletoe population dynamics are both topics of current interest in the ecology literature. There are any number of research programmes which could be linked to, and therefore financially support, the management of mistletoe infection in BRT.

5.6.6 Monitoring and evaluation of outcomes

Monitoring costs have been a major barrier to the implementation of AAM in other systems (Walters, 1997). However, there is growing evidence highlighting the cost-effectiveness of community-based monitoring (Mapstone et al. (1996); Danielsen et al. (2005); Jones et al. (2008)). Given that participatory resource monitoring is already in operation at BRT, and that the Soligas accuracy in such tasks has been well demonstrated, they are well placed to fill this role. Prevalence of mistletoe infection would not be a complicated ecological character to assess, although monitoring would have to be accurate (each tree in a treatment being climbed and searched carefully

for new infections).

Monitoring data would be gathered and stored centrally by the management body but made available to all. Adequate record keeping has been identified as a barrier in some community-based or participatory management systems (Bormann et al., 2007). In the NTFP enterprises at BRT, records of productivity and extraction have been successfully maintained by harvesters. Following data analysis by researchers, results would be presented at an evaluation meeting. Results must be presented in an accessible manner including visual descriptive statistics and careful explanation in order that all may be able to interpret for themselves. There is the potential that lack of trust may be an issue, this can never be wholly overcome, but with the raw data collected by Soligas and available to all this can be reduced somewhat. Monitoring results will be compared with initial predictions to assess both the accuracy of current system understanding, and the level of progress made in reaching the management goal, both being used to inform management in the next cycle. This evaluation is the final and crucial stage to the process, ‘closing the loop’ being a major gap in many of the large systems where AM has previously been implemented (Walters, 1997). The extent to which all stakeholders will be able to critically evaluate scientific predictions on their own terms and test the effectiveness of implemented management is uncertain. Where more complicated data analyses are required, especially given the replication and rigorous experimental design involved, there may be no way of maintaining transparency while also making the best use of the data collected.

5.7 Discussion

Active adaptive management presupposes a clear scientific framework and a system whereby information from experimental management interventions can be gathered and analysed, and then presented and interpreted in a manner that is amenable to the development of new management hypotheses. The potential for the application of this method in a participatory context with tropical forest systems has not previously been considered. In other settings several major barriers have been identified: (1) the costs of implementation, including monitoring have been substantial; (2) management

strategies have been considered too risky so experimentation has not occurred; (3) self-interested institutions have impeded progress; and (4) ecological complexities have limited the ability to reduce uncertainties. This feasibility assessment identifies several barriers to AAM of Amla with some similarities: (1) developing a common representation of the Amla-mistletoe system between stakeholders (institutional and ecological complexity); (2) the disparity between the time required to get answers and the urgency of the problem (ecological complexity); and (3) the vulnerability of the process to an uncertain social and political environment (institutional).

5.7.1 Alternative management scenarios

In sum, there many indications that success is possible, there are also substantial obstacles. If these limitations are not acceptable we should consider the alternatives. Responsibility could be devolved to Amla harvesters and the wider Soligas community allowing them to ‘manage’ as they choose. Where they regard the Amla resource to be of livelihood importance, and on experiencing some threshold of impact, local action may be taken to tackle mistletoe infection. However, with no formal tenure over the forest or its resources this is unlikely. Even if there were a strong economic incentive from market forces, it is probable that this would fuel the development of Amla plantations than the conservation and harvesting of wild populations. Additionally, as historically demonstrated, such opportunities are also often captured by other actors (Dove, 1993). In such a case, with no management intervention mistletoe infection would spread further. At a substantially lower Amla density an equilibrium between host and mistletoe populations may be reached preventing the local extinction of Amla (Anderson & May (1978); Anderson & Gordon (1982)).

Another option is a more technical, top-down approach. Drawing on the literature on mistletoe population dynamics and spread, a research team might develop a simulation model to investigate the likely impacts of alternative management interventions on infection spread. This would require substantial amounts of extra data, as well as time to build and validate the model. For example, Bogich & Shea (2008), used stochastic dynamic programming (SDP) in a metapopulation framework to de-

termine the most efficient management strategy for gypsy moth (*Lymantria dispar*) in North America. However, such an approach has no guarantee of obtaining the answers needed to inform management, or of being able to implement them once they are obtained.

Contrastingly, Johannes (1998) described ‘data less management’, management carried out in the absence of the data required for the parameterization and verification of models. This approach uses ‘information’ rather than data, gathering this from two sources; from previous studies on other systems, and traditional knowledge from resource users themselves. Management is not preceded by conventional research or followed by scientific monitoring. Johannes (1998) considered this approach most appropriate where no time or financial resources are available for data-gathering, but in a rapidly changing system such as this case study such an approach might offer little chance of long term success.

Centrally administered management by the KFD is the most likely alternative. Essentially meaning that the status quo of no intervention continues. There may be future opportunities for greater scientific input from researchers and NGOs, which might also lead to some positive outcomes, but with more limited opportunities for learning.

Despite its limitations, for Amla in BRT, an AAM framework has benefits over all of these alternatives; offering the opportunity to improve ecological status, livelihoods and stakeholder relations, while gaining critical information for guiding management. If AAM has potential in this case study, how generalisable is this to other NTFP harvesting systems in similar settings? Possibly not very widely. In several ways, BRT is quite a unique example. A substantial body of research has previously been conducted at the site, there is strong community cohesion, an existing familiarity with the activities involved in management as well as a demonstrated willingness to be involved. These elements serve as an important foundation and such existing capacity is likely to be a prerequisite for successful implementation of AAM elsewhere. Possibly a discouraging conclusion, this finding does offer some direction. Researchers should aim to establish longer term links with specific locations, and funding opportunities should support such efforts. Longer term projects provide better learning opportunities (suc-

cesses and failures) and real potential for capacity development and partnership with local communities. Many theoretical ecological research projects could more actively support community management activities by incorporating more locally relevant questions alongside pure ecological research.

5.7.2 Conclusion

Given increasing complexities in management, and the limited time and resources available, we must continue to look for more successful approaches and partnerships in our management of tropical ecosystems. Although we should be cautious in pursuing community involvement as a panacea, we must also be realistic regarding the dependency of many communities on areas prioritised for conservation, and should look harder for the opportunities to involve them in the management cycle. The application of community-based management is now widespread, but the integration of local knowledge and practices into the design, implementation, and monitoring of management programmes has been limited. AAM may offer an approach to correct this, while targeting specific management uncertainties and generating management relevant information in a shorter time frame than some conventional approaches. AAM is an inherently scientific process, and as such, will not be sustainable without the permanent involvement of a body with scientific expertise. However, as long as the objective is common, such partnerships between scientists, local communities and managers offer significant potential. In the study of many tropical ecosystems, we learn most by our experiments rather than from our observations, we should be taking the same approach to their management.

CHAPTER 6

DISCUSSION AND CONCLUSIONS

Since its emergence in late eighties and early nineties, the sustainable harvesting of non-timber forest products remains an issue of considerable interest in ecology, conservation science and rural development (Allegretti (1990); Nepstad & Schwartzman (1992); Belcher & Schreckenberg (2007)). NTFPs epitomise the integrated conservation and development paradigm; the search to find means of reconciling conservation and development goals, and even of achieving one via the other. With links to larger questions of tenure, indigenous rights, gender and poverty (Falconer (1990); Malhotra (1993); Sullivan (2000)), NTFP harvesting still maintains much interest in the academic literature and wider media (e.g. Purvis (2007); Phillips (2008)). As vital livelihood resources for many forest communities, the sustainable management of NTFPs will continue to be of significant importance.

Amla is an NTFP which makes a significant contribution to the subsistence incomes of Soligas households in the BRT Wildlife Sanctuary. However, the future viability of this role is uncertain. Amla harvesting illustrates many of the current challenges and opportunities in forest management, a complex biological system with an equally complex social and institutional management system attached to it. While this discussion highlights these challenges, it also provides a synthesis of research results in the form of specific management recommendations for Amla in BRT. Additionally the results of the previous chapters are placed in a wider context, providing insights for NTFP harvesting systems in other locations and highlighting important areas for further research.

6.1 A new management strategy for Amla in BRT

This thesis considers the threat posed by mistletoe infection to the viability of the Amla harvest and investigates how this threat might be tackled in management. Through joint use of ecological data and traditional ecological knowledge; mistletoe infection has been demonstrated as a serious threat to sustainable Amla harvesting, a conclusion based on: (1) the prevalence and distribution of mistletoe infection in the *P.emblica* and *P.indofischeri* populations, (2) the serious consequences of infection for tree productivity, growth and survival, (3) the reported historical decline in the Amla population, and (4) the inadequacies identified in current management and assessment.

Investigations into three approaches to management; mistletoe removal by hand, chopping of infected branches and forest fire, demonstrated some of their respective costs and benefits. Given the relative ineffectiveness of hand removal, its associated opportunity costs, and the lack of local support for this method, together with the potential negative impacts of fire and strong opposition to its use by the state government, branch chopping emerges as the best current mistletoe removal option. This control method is already practised in the sanctuary but should be formally supported and more systematically implemented. A program of rotational chopping linked to the annual Amla harvest would reduce the prevalence and intensity of mistletoe infection and enhance Amla yields in subsequent years.

The formulation of a robust management strategy requires not only a clear specification of management goals (i.e. enhanced viability of the Amla harvest or a healthy Amla population), and information on which actions are most effective (i.e. branch chopping), but also an assessment of how to implement these actions in the context of spatially structured and dynamic mistletoe and host populations and in recognition of specific resource constraints. The spatial characteristics of infection were found to be complex, with aggregative processes operating on interacting scales, involving mutualistic and facilitative interactions with several other species. Three levels of 'patchiness' in mistletoe infection were identified; highly infected trees, patches of high local prevalence, and areas of the sanctuary where infection is more prevalent.

Consideration of this spatial structuring, along with the processes of infection spread, can be expected to increase the effectiveness of implementation. Areas of particularly high infection prevalence, or Amla density should be targeted in the first years of implementation. Trees in areas of high prevalence are at the greatest risk of infection, clearing infection in these areas would reduce the spread of infection to neighbouring healthy trees. Targeting those trees with very intense infections would lower infection accumulation rates therefore reducing overall local mistletoe abundance, and by clearing mistletoes from taller such trees (within which this intensification process occurs more rapidly), benefits for fruit production and population growth may also be maximised. *Phyllanthus emblica* is the more significant of the two species in terms of its contribution to the Amla resource, and shows higher levels of infection, at both the individual tree and population levels suggesting this species should be a priority in initial management efforts. Evidence suggesting greater resistance to infection in *P.indofisheri* provides further justification for an initial focus on *Phyllanthus emblica*. Monitoring of local infection spread is a vital component to this management strategy particularly so where management implementation is used to consider remaining uncertainties over the dynamics and scale of mistletoe infection spread within an adaptive framework. The role of fire, and the potential role of lantana in facilitating spread of infection being two key uncertainties such a framework should aim to reduce.

A future management strategy for Amla must also consider the wider context of other forest resources, services and biodiversity, historical patterns of use and management, as well as the recent environmental history of the area which has seen an altered fire regime, the spread of several alien invasive species, human population growth and increased harvesting pressure driven by commercialisation

6.2 TEK and Science in Management

Amla harvesters provided the current ‘best option’ for mistletoe removal, a practice that in fact, has previously been considered by both scientists and Forest Department officials to be a destructive activity. More conventional ecological studies provided

information to inform how this technique might be best implemented in the context of a spatially structured mistletoe population. This study contributes to the growing literature that documents the relative benefits and deficiencies, of traditional knowledge and management practices, and additionally, emphasises the need to view these traditional practises without preconceptions over motivations or efficacy. Local management techniques must be assessed on a level footing with scientific ones, rather than post-hoc comparisons which test traditional knowledge and management against scientific techniques (Brook & McLachlan, 2005). A framework that will be more supportive of such assessments is presented in the previous chapter.

Although traditional knowledge provided extremely valuable information for managing mistletoe infection of Amla, traditional knowledge, or management techniques, will rarely be sufficient for the development of a management response. This example suggests maximum benefits will be achieved in ecosystem management where TEK is used side by side with information from scientific studies. Currently, such an integration, that feeds directly into management, is rarely achieved. The solution to this remaining challenge for resource management links directly with the broader issue of integrating communities more effectively into resource management and conservation.

6.3 Community-based management

“Community” has moved in and out of fashion in conservation and resource management; formerly seen as a barrier to conservation with policies to exclude local people, communities have now become a significant focus in conservationist thinking with international agencies directing enormous sums of money toward community-based conservation and management programs. Similar changes have occurred in the management of exploited resources, with a move away from command and control methods. Despite decentralised and participatory management being championed in this way, and its inclusion in many legislative frameworks, in reality, much participation has been token. Where scientists or managers work together with local people there is the tendency that they will maintain control and participation sim-

ply becomes a way of making environmental management appear more democratic. The tendency for participation rather than partnership has been identified as one of the main reasons for the limited success of collaborative management (Berkes, 2008). India, including the Biligiri Rangaswamy Temple wildlife sanctuary, is currently an interesting example in which to consider the potential for making progress in this area.

In India people have been repeatedly been alienated from forest management; first the colonial, and then the independent Indian state privatised and brought much forest under state control. Historically, occupied forest was declared state forest land, ending the previous traditional common property regimes of Adivasi (tribal) communities. Later many of these forests were designated as protected areas for the conservation of wildlife. Paradoxically, India has also been at the forefront of this trend towards participatory approaches. The National Forest Policy was revised in 1988 and a national forestry devolution program initiated. Popularly known as Joint Forest Management (JFM), the state (the Forest department) and local forest protection committees (FPCs), share rights and responsibilities over forest use and management (Ministry of Environment and Forests, 1990). This policy did not transfer ownership, but attempted to restructure the system of access, decision making and sharing of benefits. It had two objectives; the empowerment of forest-dependent communities, and the regeneration and improvement of degraded forests. However, after more than a decade of JFM implementation, achievements in terms of either of these objectives appear limited (Bharracharya & Basnyat (2003); Ghate & Nagendra (2005); Ministry of Environment and Forests (2006)). There are strong calls for further and drastic reforms e.g. (Tewari, 2008). Significantly, as JFM was originally conceived as a way of encouraging the rehabilitation of degraded 'forest' the scheme is not applicable to India's six hundred or so protected areas. The communities living in these forests continue to have little involvement in management. In fact, many thousand's of forest dwellers have been forcibly relocated and resettled. Such schemes have generally been poorly implemented with no prior consultation, leading to serious social and economic impacts for the affected communities and with uncertain environmental benefit (Rangarajan & Shahabuddin (2006); Shahabuddin et al. (2005)).

Forest management in India is currently at an important junction. The Scheduled Tribes and Other Forest Dwellers (Forest Rights) Act aims to ‘compensate’ forest communities for the denial of their traditional rights to forest lands and resources. Passed in 2006, rules are currently being framed to make this act operational. While several provisions of the act will strengthen the rights of traditional forest communities, other provisions are inadequate and are likely to cause significant damage to forests and wildlife, ultimately being of little benefit to either community or conservation (Kalpavriksh, 2007b). Joint management of protected areas has been advocated (Kothari et al. (1997); Apte & Kothari (2000)). Although, the act as it is currently formulated, is a missed opportunity in this regard, careful reform could make this conservation model a possibility for implementation across India bringing with it many new opportunities for co-management.

The application of community-based natural resource management is widespread, yet the full integration of local knowledge and practices into the design, implementation, and monitoring of management programmes has been limited. While we need new approaches for combining traditional knowledge and scientific data in assessing resource status, we also need new institutions that foster management partnerships with local communities; hybrid institutions that combine customary management with western models of resource management and conservation.

6.4 A future role for Amla

This study has not specifically looked at Amla harvesting, and is therefore limited in its ability to make an assessment of the likely sustainability of current harvesting practises, or indeed of future harvesting scenarios. However, given the threat mistletoe infection poses to the Amla population, the likely potential of Amla to contribute to local livelihoods should be questioned. Much discussion has centred on barriers to the successful commercialisation of NTFPs, but were Amla to represent such a livelihood opportunity, efforts to raise incomes through harvesting may fail, not because of overharvesting or a lack of economic potential, but because additional forest disturbances have undermined the resource base. In the context of NTFPs

as an means to improve local livelihoods, this study suggests that Amla could have been be a missed opportunity. The sustainability of current Amla collection patterns should be re-assessed in the light of mistletoe infection impacts, and future NTFP commercialisation attempts must consider these wider threats to sustainability.

The consideration of a potential future role in livelihoods for Amla also leads to larger questions over the desirability of focusing on an NTFPs a development path for these, and indeed other communities. Amla was one of four species included in an ‘enterprise based conservation project’ in BRT funded by WWF during 1994-1997 (Lele et al., 1998). Efforts in this direction continue through local NGOs and LAMP societies, but there have been no formal assessments of this project’s outcomes in terms livelihood improvements. A study over ten years ago assessed the contribution of NTFP extraction more broadly to income in BRT (Hegde et al., 1996), at that time NTFPs contributing half of total annual income. This assessment should be repeated, few studies having investigated the changing dependance on NTFPs over such a time period. An investigation of the impacts of the recent harvesting ban, assessing both livelihood, and ecological consequences of such legislation, would also be valuable and provide further evidence in support of management reform.

The future viability of the Amla resource is uncertain but even if it were possible to increase or sustain Amla harvests, we should reflect on how desirable such a goal may be from the perspective of the Soliga community. Putting further resources into Amla, a species well studied and ‘relatively’ well managed, may not be justifiable on developmental grounds. Other NTFPs may be a better investment for management revenue, for example both honey and Lichen in BRT are of higher value. In fact, NTFP harvesting may not offer the best route to improving local livelihoods, and these communities may desire a different future altogether. The importance of this aspect is rarely considered in many integrated conservation and development interventions. The extent to which sustainable NTFP harvesting plays a part in the future ‘vision’ of these communities remains to be established. They may, in fact, anticipate minimal economic benefits from the forest in the longer term, having more of an agricultural vision for the future. Alternative land uses, including clearing more land for subsistence agriculture (although unlikely without relocation), may indeed

provide the best means for raising living standards. Alternatively, they may imagine an ongoing dependence on the forest, and consequently, attribute greater significance to the changing availability of forest resources. The distinction between these has implications for the allocation of management resources, and most significantly, for longer-term management objectives. Although a challenging task, such an investigation in BRT would have considerable merit, particularly in the context of the new Forest Rights Act and the opportunities and threats that this new legislation may bring.

6.5 New threats to non-timber forest products

The impacts of over-harvesting on non-timber forest product (NTFP) sustainability have received considerable attention (Boot & Gullison (1995); Bernal (1998); Peres & Lake (2003); Ticktin (2004); Endress et al. (2006)). However, as this thesis demonstrates, other forest disturbances may have more immediate, and substantial consequences. Mistletoe infection represents an “emerging threat” to the in BRT forest (Laurance, 2006), and may successful link with several other such threats (Rodriguez, 2006). Threats to NTFP sustainability come not only from harvesting, they can be multiple and interacting. Although some NTFP species may respond favourably to particular disturbances (Laird (1995); Davies et al. (2001)), in many instances, emerging threats may impact negatively on the collection of forest resources. Environmental synergisms such as interactions between invasive species, disease, fire and climate change, are both very damaging and extremely common (Laurance & Williamson (2001); Peres (2001); Crowl et al. (2008); Dukes & Mooney (1999); DAntonio & Vitousek (1992)). In situations where management must tackle many of these threats, separating cause and effect can be challenging, and as demonstrated in the Amla example, ascribing infection patterns to the incorrect processes can lead to poor management, in terms of both social and environmental outcomes. NTFP assessments based solely on harvesting pressure will lead to incorrect conclusions over population status. Similar ecological complexities will influence most NTFP management systems and it may become increasingly difficult to separate the effects of

utilisation from other environmental pressures. Management of NTFPs must look beyond harvesting to consider a plethora of interacting processes, operating at different scales, and with many feedbacks and must anticipate potential future threats to sustainable harvesting from the livelihood or ecological perspective.

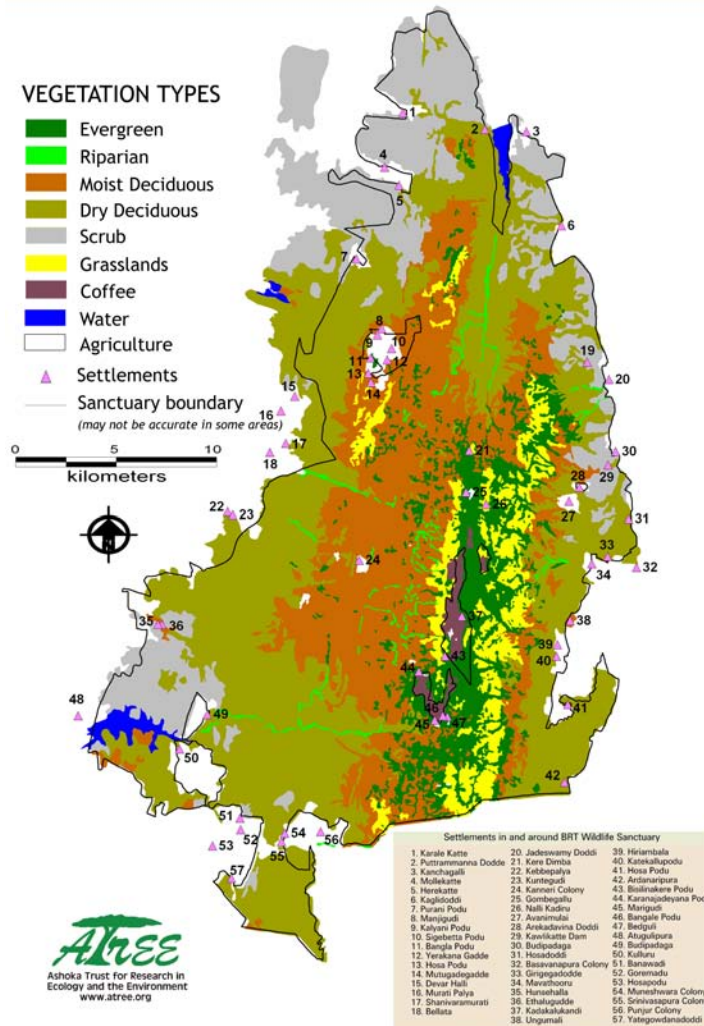
The harvesting of NTFPs has received a significant amount of attention regarding the potential for enhancing the incomes of forest communities. In many respects, this focus on the commercialisation of NTFPs has been misplaced. Although some commercialisation schemes have been successful, including some aspects of efforts in BRT itself, for the vast majority of forest communities NTFPs will continue to be about fulfilling day to day needs rather than livelihood enhancement opportunities. Access to these resources has already been restricted and risks being further reduced as forests are subject to increasing pressures from climate change, invasive species and other disturbances. In this context a focus away from commercialisation towards NTFP security would be timely.

APPENDIX A

THE BRT WILDLIFE SANCTUARY

Map of the Biligiri Rangaswamy Temple (BRT) Wildlife sanctuary illustrating vegetation types and settlement locations (Courtesy of GIS/RS Laboratory at ATREE).

BILIGIRI RANGASWAMY TEMPLE WILDLIFE SANCTUARY



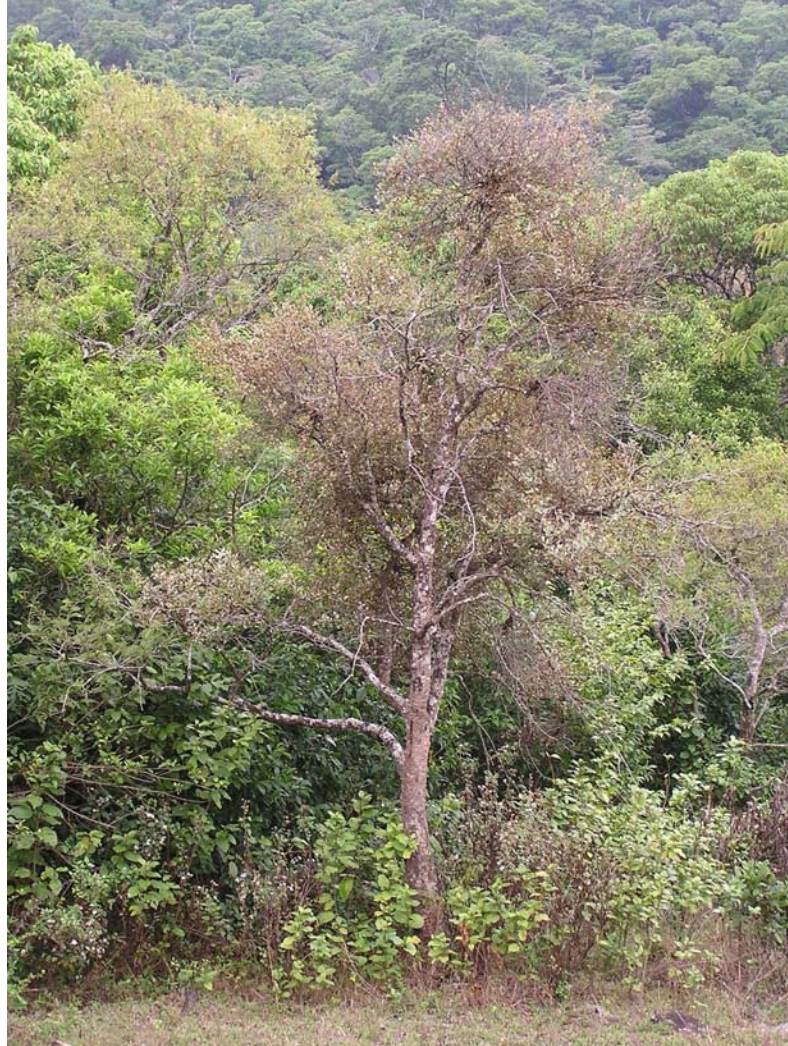
APPENDIX B

INFECTED TREES & COPPICING BRANCHES

A healthy, uninfected *Phyllanthus emblica* tree



A *Phyllanthus emblica* tree infected by *Taxillus tomentosus*.



Regrowth following branch chopping during harvest. Coppicing sites are circled and fruits (and mistletoe) visible on top outer right branches.



APPENDIX C

THE ADAPTIVE MANAGEMENT CYCLE

There is significant variation in descriptions of an adaptive management process, both the number, and nature, of the stages involved. Most differences reflect a greater level of detail (i.e. more stages), others reflect emphasis on specific elements of adaptive management, for example, a focus on modelling (e.g. Johnson & Case (2000); Bearlin et al. (2002)), or monitoring (e.g. Parma et al. (1998)). The stages outlined here draw on other cycles in the literature (e.g. Walters (1986); Parma et al. (1998); Shea et al. (2002); Bearlin et al. (2002)), and present the cycle in its basic unbiased form (Figure C.1).

1. *Define the management objective*: the first stage is to identify exactly what management intervention is trying to achieve. This objective is frequently refined following subsequent stages of the AM process.
2. *Model existing knowledge*: an assessment of all the available information on the system is required. This includes acknowledging uncertainties and specifying alternative hypotheses regarding relationships between components of the system. Understanding of the system is sometimes formulated into mathematical models. Although this dynamic model representation of the system has been identified by many as critical to the AM process (Walters, 1986), the system model may also be a conceptual description of relationships. The fundamental objective is that it allows predictions to be made about the impacts of alternative policies, and therefore testing of current knowledge at the end of the cycle.

3. *Identify indicators and define goals related to the management objective:* an indicator of achievement of the management objective must be identified. It will be chosen based on the specified management objective, and as a function of what is known about the system. The indicator is a measureable characteristic of the system, used to evaluate the outcome of management actions. It must be cost effective to monitor, and predictable in the context of making forecasts about the impacts of management interventions. A time-bounded goal in relation to reaching the management objective should also be identified.
4. *Consider alternative management options and predict their outcomes:* a set of possible management strategies are identified. Some may be simply what is currently implemented or may correspond to current best practice (having been implemented successfully elsewhere), others may be more novel strategies. Using the system model (conceptual or mathematical), the likely performance of this set of options are evaluated (predictions will later be compared with management outcomes). Based on this evaluation, a choice is made over what to implement. Some strategies may be preferred based on their predicted chances of success, others due to their potential for learning about the system. Choices must be made on how to apportion the available resources between the alternative strategies, balancing learning and management (McCarthy & Possingham (2006); Hauser & Possingham (2008)).
5. *Implement the management actions:* the selected management strategies must be implemented with a rigorous experimental design to allow the simultaneous testing of multiple hypotheses and the separation of site specific differences.
6. *Monitoring and evaluation of outcomes:* indicators will be monitored to assess progress towards the management objective. By comparing the observed with the predicted results of management, we learn to what degree our understanding of the system is correct, the management that we apply in the next iteration will reflect this improved knowledge. Monitoring itself can be adaptive; the monitoring protocols can be updated if monitoring is not providing the correct

information (Ringold et al., 1999). Evaluation of management also includes the process of evaluating, and where necessary, re-defining the management objective.

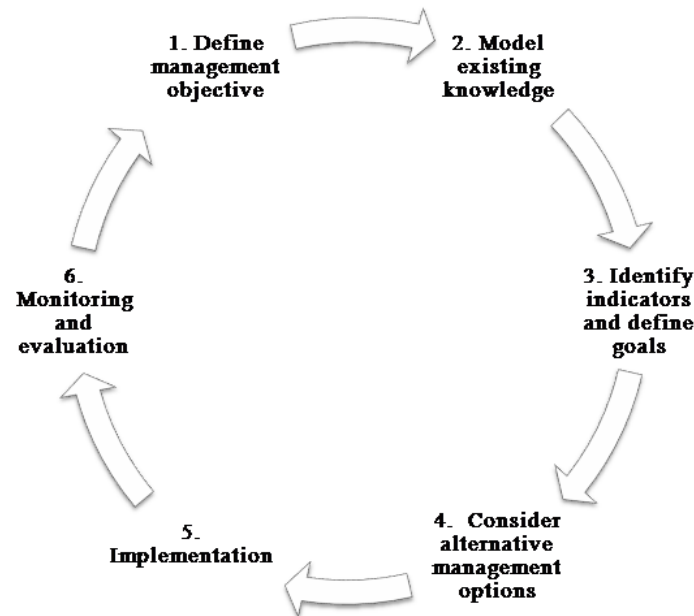


Figure C.1: *The adaptive management cycle*

The adaptive management cycle. The cycle takes place over a defined management period. Sometimes managers will bypass stages one. In every cycle managers pass through stages two to six; from the evaluation of hypotheses regarding the dynamics of the system and their relevance to management objectives, through to inference from the management treatments applied. The six stages are repeated in an iterative management process.

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