

Forest Cover, Condition, and Ecology in Human-Impacted Forests, South-Eastern Madagascar

Jane C. Ingram and Terence P. Dawson

Abstract: *The littoral forests of south-eastern Madagascar are a national conservation priority due to high degrees of biodiversity, but face pressures from regular human use and a future mining operation. A landscape scale assessment of deforestation patterns, forest condition and composition of remaining forest stands is important for understanding the nature and distribution of human pressures and could act to inform land use management and identify conservation priorities throughout the area. In light of these issues, the aims of this study were threefold: to document patterns of littoral forest loss at multiple spatial and temporal scales; to map littoral forest structure across the landscape; and to assess the abundance and diversity of littoral forest tree species valuable to both humans and conservationists. The methods applied include satellite remote sensing applications and ground-based ecological surveys. We demonstrate three results: first, patterns of forest loss are spatially and temporally variable; second, forest basal area, an indicator of forest condition, can be estimated using multi-spectral satellite data, artificial neural networks and ground survey data; and third, littoral forests possess high abundance and diversity of tree populations of importance for both conservation and human livelihoods, despite regular use by local people. A landscape perspective, combined with an understanding of the local human and environmental context, is crucial for understanding the nature and impact of human pressures on forest resources, and, thus, determining optimal management possibilities.*

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INTRODUCTION

Conservationists have classified Madagascar as the world's highest conservation priority or biodiversity 'hotspot' based on two criteria: exceptional concentrations of endemic plant species and high degrees of threat facing the persistence of those species (Mittermeier et al. 1998; Myers et al. 2000). It has been estimated that the country retains only 9.9% of its primary vegetation (Myers et al. 2000) and that deforestation continues 'unchecked' due to population growth and poverty (Hannah et al. 1998). Despite the importance of land-cover change estimates for determining Madagascar's global conservation status, considerable discrepancies exist among studies on the extent of the island's remaining forest cover (Silander 2000).

Discrepancies concerning deforestation estimates have resulted due to the application of different remote sensing methodologies, the use of satellite data with incompatible spectral, temporal and spatial resolutions and the use of variant definitions of forest (as discussed in Agrawal et al. 2002; Dufils 2003). Furthermore, temporal patterns of deforestation are rarely addressed due to the expense of acquiring satellite data and the difficulty of obtaining cloud-free imagery for the humid tropics. For these reasons, many deforestation assessments are cross-sectional analyses, which represent deforestation as an average rate of forest loss and, thus, fail to incorporate important underlying temporal trends in land cover change (Koop and Tole 2001). Typical 'snapshots' of forest cover from a few select years of satellite imagery may result in misleading interpretations of deforestation and natural resource use by human populations (Leach and Mearns 1996). Similarly, deforestation estimates are often extrapolated across spatial scales. Such generalisations may mask important local trends in land-cover change across Madagascar, where the causes of deforestation vary greatly throughout the country (Jolly and Jolly 1984) and in many regions these causes are poorly understood (Lowry et al. 1999). Areas that are very close to one another geographically can have different temporal patterns of forest loss and different drivers behind deforestation (Sussman et al. 1994). Thus, national and regional estimates of tropical forest loss may misrepresent deforestation processes at local scales, where conservation and land-use policies have their greatest impact.

The quantification of the discrete process of tropical deforestation has often overshadowed the measurement of tropical forests that have been impoverished or reduced by human activity, such as logging, but not eliminated (Nepstad et al. 1999). Such a process is often referred to as 'degradation' which has been defined as the temporary or permanent deterioration in the density or structure of vegetation cover or its species composition (Grainger 1996). Al-

though conservation of old-growth forests is of extreme importance, management of 'degraded' forests can also be a crucial component of conservation programmes in developing countries where local people utilise forests on a daily basis. Degraded forests often provide valuable resources to human communities, retain significant amounts of biodiversity and may relieve pressure on old growth forests. To date, little attention is focused on the assessment or conservation of Madagascar's human-impacted forests, yet these stands could represent valuable opportunities for protecting biodiversity, providing important resources and ecosystem services to local communities and, thereby, relieving human pressure on old growth forests.

Deforestation and forest impoverishment have undoubtedly occurred on the island, but, as a growing body of literature suggests, the process and the role of humans in contributing to land-cover change is complex (Jarosz 1993; Kull 2000; McConnell et al. 2004) and has not been explained adequately by rate-based accounts of deforestation. In various African countries, over-simplified descriptions of environmental change have influenced the development of environmental programmes that, in many cases, have been ineffective at protecting the environment and have resulted in the exclusion of people from the natural resources upon which they depend for their livelihoods (Hoben 1995; Barraclough and Ghimire 1996; Fairhead and Leach 1998; Sheil and Wunder 2002). Understanding the patterns, processes and impacts of land-cover change is of importance because the way in which environmental problems, such as deforestation, are defined, delimited, and discussed shapes the possibilities for solving them (Jarosz 1993).

South-Eastern Madagascar

This study investigates such issues in south-eastern Madagascar proximate to Tolagnaro, also known as Fort Dauphin. The region is home to a subtype of rainforest, known as littoral forest on sand, which has been identified as a national conservation priority (Ganzhorn et al. 2001) due to its limited extent and due to high concentrations of national and local endemic plant species (Du Puy and Moat 1998; Dumetz 1999), a diverse tree flora (Dumetz 1999) and high diversity of faunal taxa (Ganzhorn 1998; Ramanamanjato et al. 2002; Watson et al. 2005). The future of these forests is uncertain due to pressures from both local human use and a future multi-national mining project sponsored by the company, QIT Madagascar Minerals (a subsidiary of Rio Tinto), which will extract deposits of ilmenite from the area. The mining would progressively remove the majority of the current littoral forest cover but would also bring much needed economic development to the region. The forest is also of great value to the inhabitants of 44 villages who live within or proximate to the exploration zone, an area of approximately 325 square kilometres, and who utilise forest resources for their daily needs.

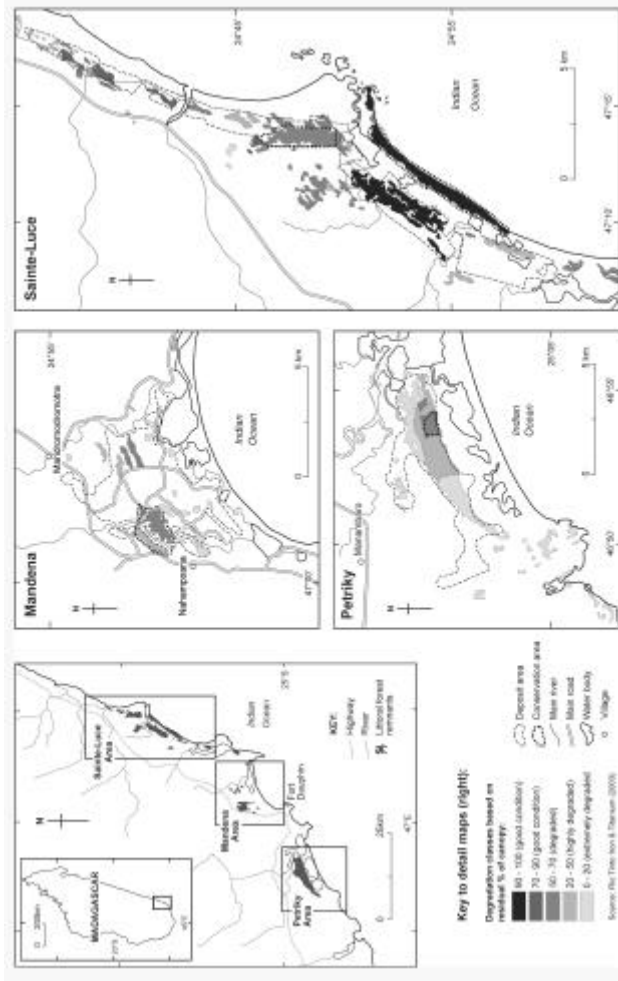
Within the Social and Environmental Impact Assessment (SEIA), the mining company and hired consultants have predicted that even in the absence of mining, most of the littoral forest cover will be removed within the next four decades if deforestation due to human use continues at current rates (QMM 2001, MIR Télédétection Inc. 1998). Based on their assessments of deforestation, QMM has predicted that if the average annual rate of deforestation is maintained over the next several years (86 ha/yr) the littoral zone's forest cover could be completely removed by approximately 2040 (QMM 2001: 3–102). Similarly, the SEIA reports that much of the current littoral forest cover is already degraded (QMM 2001). The company's conservation plans will involve extensive replanting of trees following mining and the establishment of 650 ha of conservation zones for littoral forest, where mining will not occur (QMM 2001). Nevertheless some conservationists and non-governmental organisations are sceptical about the rate and nature of forest change presented by the SEIA (WWF 2001; WRM 2003).

This study aimed to understand the nature of human impact on littoral forests in south-eastern Madagascar, where the processes of deforestation and forest degradation have had a strong influence on land use policy and management. The specific objectives are to 1) assess spatial and temporal patterns of forest change in south-eastern Madagascar to supplement previous studies of littoral forest loss; 2) apply a quantitative assessment of littoral forest structure for assessing patterns of forest condition at a landscape scale; and 3) assess distribution and abundance of trees useful to people and trees endemic to Madagascar, that are of interest for conservation purposes. Although the latter two aims have been developed more fully in Ingram et al. (2005a,b) the consideration of these issues in concert provides a more holistic understanding of the interactions between humans and the forest environment in this area.

Description of Study Site

The littoral forests are a mosaic of forest stands distributed across three sites, *viz.* Ste. Luce (also known as Manafiafy), Mandena and Petriky (Figure 1). The forests are located at elevations less than 50 m above sea-level and comprise a relatively narrow band of coastal plain and adjacent foothills that is approximately 7 km at the widest point and extend for approximately 56 km from 24°35'S to 25°08'S latitude (Lewis Environmental Consultants 1992). A distinct climatic gradient exists across the study area, with conditions becoming drier and hotter to the south (Goodman et al. 1997). The wet season occurs from November to May with annual average rainfall amounting to 2400 mm in Mandena and Ste. Luce and 1200 mm in Petriky (QMM 2001). The progressive decrease in rainfall from north to south has discernible impacts on Petriky, the southernmost of the forest sites. Petriky is floristically distinct from Ste. Luce and Mandena, which have a higher number of tree species.

Figure 1
 QMM's map of forest condition at the three study sites: Ste. Luce, Mandena and Petriky.



Notes: The shades of black and grey represent different classes of forest condition: black represents forest in very good condition, dark grey represents forest in good condition, grey represents forest of moderate degradation, light grey represents forest of strong degradation and very light represents forest of extreme degradation. Map provided by Rio Tinto Iron and Titanium (2003).

Petriky is a drier, more nutrient-poor zone, similar to a scrub woodland, and for this reason has been classified as a distinct subtype from Ste. Luce and Mandena (Dumetz 1999). However, collectively, the three littoral forests are floristically and structurally distinct from other regional lowland rainforests due to their location on sandy soils and their low height and low diameter at breast height (dbh) values (Dumetz 1999).

Local people depend on the forests for subsistence purposes, which include fuelwood, construction materials, food and medicine. Excessive dependence has resulted in degradation of the forest (QMM 2001), yet, few studies have sought to quantify patterns in littoral forest structure and diversity in relation to each other and to human impact.

METHODS

Assessment of Regional and Local Deforestation Patterns

Previous Deforestation Assessments

The previous deforestation study conducted for the mining zone of Petriky, Mandena and Ste. Luce was completed by consultants contracted by the mining company. Deforestation estimates were derived from panchromatic aerial photographs from the year 1950 (1:50000), a colour aerial photograph from 1989 (1:200000) and a SPOT panchromatic satellite image from 1995. The study concluded that 76 ha of littoral forest per year were lost collectively across the three sites on the basis of two images collected in 1950 and 1995. This rate did not incorporate the forest loss that occurred between the intermediary time intervals between 1950 and 1989 or between 1989 and 1995. If these figures had been included, a decline in the rate of deforestation from 76 ha of forest loss per year from 1950–1989 to 72 ha of forest loss per year between 1989 and 1995 would have been observed.

The company updated the rates of forest loss for the littoral forest sites of Petriky, Mandena and Ste. Luce (MIR Télédétection Inc. 1998) by using aerial photography acquired in the year 2000 (QMM 2001). Based on these revisions, the previous rate of 76 ha of forest loss per year was updated to 86 ha of forest loss per year (0.86 km²/year) occurring between the years 1950 and 2000.

Methods Used Within this Study

We used data from the Landsat Thematic Mapper (TM) from November 1984, November 1992, November 1999 and the Landsat Enhanced Thematic Mapper (ETM) from January 2002. For the study, only the red and near-infrared bands of the sensors were used, in which spatial resolution of approximately 30 m is identical, and were integrated into the Normalised Difference Vegetation Index (NDVI) (Tucker 1979). All NDVI values were rescaled to byte format (0–256).

Image Pre-Processing

The detection of land-cover change using multi-date satellite images requires pre-processing procedures that minimise differences between images resulting from seasonal, geometric or atmospheric changes rather than real land cover change. We controlled for seasonal change by using Landsat TM and ETM images collected in the wet season, from November to May (Goodman et al. 1997; QMM 2001). However, the wet season may not begin at exactly the same time every year. Thus, slight phenological differences may exist between the images due to the semi-deciduous nature of some tree species located in the forest sites. However, the change detection method used here reduced errors associated with subtle phenological differences between images. All the images were geo-rectified upon acquisition of the data to the Universal Transverse Mercator (UTM) projection system. Known geographical positioning system (GPS) points were imported as a vector layer and validated across each image to ensure consistent geo-location across all of the images. To isolate forest from other land-cover types, a vegetation mask was created from the 1984 image by retaining only pixels with an NDVI value ≥ 184 . Historical littoral forest cover maps (QMM 2001) and GPS ground-truthed land classifications verified that this threshold was optimal for isolating littoral forest and other forest types from non-forest vegetation. To correct for atmospheric and illumination differences between the images, we applied a relative method of radiometric calibration, due to the simplicity and reliability of this method (Collins and Woodcock 1996). The image differencing procedure further eliminates pixels that have experienced systematic change due to atmospheric differences between years.

Change Detection Procedures

Image-differencing, the subtraction of the pixel digital values of an image recorded at one date from the corresponding pixel values of an image acquired on a different date (Lillesand and Kiefer 1994), is a highly favoured change detection method on account of its accuracy, simplicity in computation and ease in interpretation (Hayes and Sader 2001). This method has proved useful for land-cover change assessments in various tropical environments (Miller et al. 1978; Singh 1986; Hayes and Sader 2001). Vegetation indices, such as the NDVI, generally have a stronger relationship to the phenomena of interest within a scene than any single spectral band (Nelson 1983; Collins and Woodcock 1996) and, thus, vegetation index differencing is a reliable, fast and convenient method for identifying changes in vegetation cover over large areas (Lyon et al. 1998).

All of the calibrated NDVI images were differenced in chronological order (the 1992 image was subtracted from the 1984 image; the 1999 image was subtracted from the 1992 image; the 2002 image was subtracted from the

1999 image). Each differenced image was overlaid with the forest vegetation mask and the mean difference and standard deviation were calculated within this boundary for each time interval. In a histogram of pixels comprising the differenced image, the pixels clustered around zero represented pixels that had experienced zero to little change in reflectance between image dates while pixels at the tails of the distribution represented pixels that had changed substantially in reflectance between the two dates (Jensen 1996). The images were reclassified so that only pixels with a value of \geq two standard deviations (SDs) below the mean difference (negatively changing pixels) were retained. The selection of threshold values that discriminate real land cover change from systematic change is a primary difficulty in image differencing procedures (Hayes and Sader 2001). However, ground surveys in the study area confirmed that the chosen threshold of two SDs below the mean difference of change captured littoral forest loss known to have occurred in Mandena during late December of 1999.

The number of significantly, negatively changing pixels between successive image dates were calculated for the individual forest sites of Petriky, Mandena and Ste. Luce. To achieve consistency between studies, we used the boundaries of Ste. Luce, Mandena and Petriky as defined by QMM. Non-forest vegetation was omitted from the analysis. From the number of significantly, negatively changing pixels calculated at each spatial scale, we calculated i) the area of forest loss between successive images and ii) forest cover for each year. Forest cover was calculated by removing all forested pixels that had significantly negatively changed between two years from the forest boundary in the earlier image. Pixels that had significantly changed twice (the maximum number of times any single pixel changed throughout the time period) were removed from this analysis so that only single deforestation events were assessed.

Assessment of Forest Condition

The littoral forests have been mapped and pre-classified by degradation level as determined from a semi-quantitative ground assessment of canopy closure conducted from 1999 to 2001 (QMM 2001, Figure 1). This assessment related degree of canopy openness observed from visual observation to degree of human-induced degradation. Natural forest stands were classified using the five qualitative categories of forest condition designated to the sites by QMM: very good condition, good condition, moderate degradation, strong degradation and extreme degradation. Classification was based on per cent canopy cover as determined from visual assessments made by at least two observers walking transect(s) throughout forest parcels. Scientists from the Missouri Botanical Gardens, Royal Botanic Gardens Kew and the Centre National de Recherche Appliquée au Développement Rural (FOFIA) in Madagascar reviewed this classification system as valid for the assessment and mapping of canopy degradation in littoral forests (Lowry et al. 1999). Yet, the research

teams suggested a need for a quantitative study of forest condition and information on patterns of forest degradation, such as the spatial correlation of tree cutting to villages, roads and paths (Lowry et al. 1999). Forest resource use may intensify differentially across the landscape when the mining starts as progressively less forest is available to local people. Thus, because such pressure may directly impact biodiversity and resource needs of local communities, it is important to identify and map patterns in forest structure.

A more detailed description of the methods used and results of this research component can be found in Ingram et al. (2005a). Twenty-one belt transects measuring 4×100 m were surveyed across the three study sites in November 2001. Samples were taken in six forest stands belonging to four of the five different classes of forest condition (two strongly degraded, two moderately degraded, one in good condition, one in very good condition). No samples were collected in the extreme degradation class because much of the forest had been removed at these sites and, thus, measuring was not possible. Geographic coordinates were recorded at each site with a GPS allowing georeferencing to the satellite image. All tree stems with a diameter at breast height (dbh) (dbh taken at 1.3 m from the forest floor) > 5 cm within each transect were identified and the dbh recorded. Mean basal area and stem density per transect were calculated from these measurements.

Landsat ETM Data

Each transect was located on the Landsat ETM scene from January 2002. Radiance values of the red (band 3), near-infrared (NIR) (band 4) and mid-infrared (MIR) bands (bands 5 and 7) were extracted at each transect site and averaged across a 3×3 pixel window. The window was centred on the GPS position at each transect, which minimised any potential geo-location errors (similar to Foody et al. 2001; Asner et al. 2002). Bands 3, 5 and 7, of the Landsat ETM were used for this study because of the distinct spectral responses of these bands with vegetation, soil and litter. These interactions have successfully been exploited in similar studies to assess forest structural features such as age and basal area (Jensen et al. 1999; Steininger 2000).

Statistical Analyses

Spearman's correlations and artificial neural networks (ANNs) were used to assess the relationship of mean stem density and basal area values per transect to reflectance values at each transect site. Based on these relationships, ANNs utilised the spectral data from bands 3, 4, 5 and 7 in the image to predict structural attributes of pixels comprising littoral forest where no ground data had been collected. To do this, ANNs were first trained to learn the relationship between the structural feature of interest and spectral reflectance in each wave band at each transect site and then tested for their ability to predict the

desired structural feature from spectral data alone. We tested the predictive ability of the ANN using two different data sets. First, we used only the ground data collected in 2001. In this approach, structural values for all but one of the twenty-one transects collected during 2001 were used to train the network and the structural measure from the one excluded transect was used as testing data. This process was repeated twenty-one times, which produced a predicted value of each structural feature at each of the twenty-one transects. Predicted and actual structural measures were then assessed in relation to each other. We then used a supplementary data set of stem density and basal area measures collected using a similar transect method in the year 2000 to increase the sample size and, thus, permit an additional assessment of the predictive power of the ANN. The structural measures for each of the twenty-one transects collected in November of 2001 were used as the training data and the structural measures recorded at fifteen transects collected in 2000 were used as the testing data. The time lag between the image acquisition date of January 2002 and the field data collected in 2000 is not ideal for this analysis since removal of trees occurs in this forest due to regular use. This selective process may not result in widespread forest clearance, such as deforestation but may reduce forest biomass in localised areas. For this reason, this data was used to supplement the results obtained using only the data collected in 2001 and the predictions were considered in the context of this temporal gap.

Predictive Map of Basal Area

Due to the strong relationships between basal area and spectral reflectance within bands 3, 4, 5 and 7, we used the ANNs to predict basal area for all unsurveyed littoral forest pixels from spectral information alone. Stem density was omitted from this part of the analysis due to the weak correlations between stem density and spectral reflectance. The training set consisted of basal area measurements from each transect assessed during the 2001 field survey and spectral reflectance values for each transect site in bands 3, 4, 5 and 7.

Tree-Diversity Inventories

Tree diversity was recorded for each of the transects surveyed in November 2001 plus one additional transect¹, for a total of twenty-two transects. All trees which had been cut were also recorded and identified, if the tree was sufficiently intact to permit identification. All trees were identified to Malagasy vernacular name by a local botanist, Ramesy Edmonds. Botanists from Kew Gardens and local Malagasy taxonomists have matched the vernacular names of tree species with the Latin names (Ingram et al 2000b, also listed in Appendix 1) for the trees in the study sites using voucher specimens collected in the littoral forests of Mandena, Petriky and Ste. Luce.²

From the total species list, species were categorised into utilitarian and/or endemic species groups. Utilitarian species lists were provided by Dr. Clement Sambo, social anthropologist at the University of Tulear, who conducted research in the study area to determine the littoral forest tree species local people use and for which purposes (Sambo 2001, unpublished data). The uses of tree species included firewood, timber, medicine/spiritual, animal food, human food, fibres and oil. We obtained a list of tree species used for charcoal production, a use practised by itinerant groups, from QMM (QMM 2001). We have reorganised these broad categories into the human use categories of timber, firewood, charcoal, medicinal value, human food and other uses (an amalgamation of oil and fibrous material within a single category). Oil and fibrous materials were combined into one category due to the few species recorded for these purposes. The list of endemic plant species was provided by Missouri Botanical Gardens (MBG) within QMM's EIA (QMM 2001; Lowry 2001). Endemic species were categorised into one of four different conservation priority groups: *Priority 1* refers to species presumed to be endemic to the mining sector; *Priority 2* refers to species presumed to be endemic to the littoral forest; *Priority 3* refers to species presumed to be endemic to the south-eastern region; *Priority 4* refers to species, often widespread and found elsewhere in Madagascar, but endemic to the island. Further details of the tree diversity inventory research and species classifications can be found in Ingram et al. (2005b).

Analytical Methods

Diversity indices were calculated for i) the total group of tree species; ii) the utilitarian species; and iii) the endemic species recorded at each transect. We utilised three indices that provide metrics of species richness, evenness and dominance: the Shannon Weiner (SW) Index, the Simpson's Index and the Berger Parker Index (Henderson 2003). The relationship of diversity indices to mean basal area and stem density at each transect was calculated using Pearson's correlation coefficients. The total basal area in each transect was calculated for each human use. Although density is a useful comparative measure, the basal area of stems above some marketable diameter is a better measure of site occupancy by timber species (Chazdon and Coe 1999) and of biomass available for human use across the landscape. Frequency, rather than basal area, is likely to be the more important measure for medicinal species since species substitutability is likely to be lower for medicinal purposes when compared to other forest uses (Gordon et al. 2003). For this reason, the number of individuals of medicinal species per transect was also calculated.

RESULTS

Deforestation Assessments

Table 1 shows that forest cover and rates of deforestation have varied both spatially and temporally throughout the study period of 1984–1992, 1992–1999 and 1999–2002. The highest amounts of forest loss occurred in Petriky between the years 1984 and 1992 (74.00 ha), a rate of 9.25 ha/year. After 1992, the amounts of forest loss was considerably lower: 17 ha of forest were lost between 1992 and 1999 (2.42 ha/year) and 18 ha between 1999 and 2002 (6.00 ha/year). In contrast, Ste. Luce and Mandena showed the highest amounts of forest loss towards the end of the study period. In Ste. Luce, 16 ha of forest were lost during both the time intervals 1984–1992 (2 ha/year) and 1992–1999 (2.14 ha/year). A sharp increase of forest loss occurred in Ste. Luce between 1999 and 2002, during which 109 ha of forest were lost at the rate of 36.33 ha/year. In Mandena, only 1 ha of forest loss was recorded between 1992 and 1999 (0.13 ha/year), 11 ha were lost between 1999 and 1992 (1.57 ha/year), and 134 ha between 1999 and 2002 (44.67 ha/year). Figure 2 shows the rate of forest loss that occurred during each time period. It shows nonlinear variability in the rates of deforestation throughout the 18-year period.

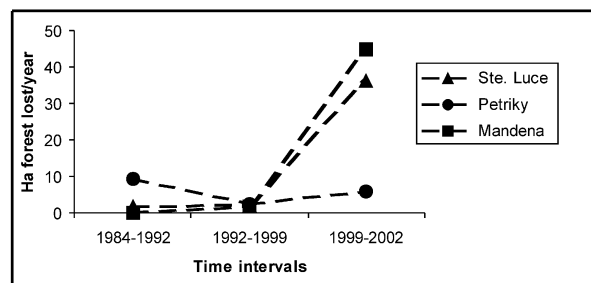
Table 1

Estimates of forest cover (ha) for Ste. Luce, Petriky and Mandena from 1984 to 2002

Year	Ste. Luce	Petriky	Mandena
1984	1,404	932	760
1992	1,388	858	759
1999	1,373	841	748
2002	1,264	823	614

Figure 2

Forest loss from 1984 to 2002



Notes: Forest loss during time intervals spanning 1984 to 1992, 1992 to 1999 and 1999 to 2002. The deforestation rates for Ste. Luce, Mandena and Petriky forest cover values are graphed on the 'y' axis.

Assessment of Forest Condition

Relationship of Forest Structure to Spectral Reflectance Values

The relationship of forest structural measures to spectral reflectance values using correlation analyses varied with stem density and basal area and individual wavebands (Table 2). Weak and insignificant relationships were observed between stem density and spectral reflectance in all the wavebands, while basal area was strongly and significantly correlated to spectral reflectance in all the wavebands.

The ANNs gave promising results in their ability to predict mean basal area per transect, but did not have a strong predictive capacity for estimating mean stem density. Using only the field data collected in 2001, the relationship between actual and predicted values of stem density was weak and insignificant ($r = 0.02$, $n = 21$, $p > 0.05$), while the relationship between actual and predicted values of basal area was strong and significant ($r^2 = 0.62$, $n = 21$, $p < 0.01$) (Figure 3a). The regression line for this method was closely aligned with the 1:1 line. When using the field data collected in 2001 as training data and the field data collected in 2000 as testing data, the ANN produced similar results: a weak and insignificant relationship between actual and predicted measures of stem density ($r = 0.04$, $n = 15$, $p > 0.05$) and a strong and significant relationship between actual and predicted values of basal area ($r^2 = 0.67$, $n = 15$, $p < 0.01$) (Figure 3b). Although, the relationship is strong between predicted and actual values of basal area when using the combined data sets from 2001 and 2000, the relationship substantially deviates from an exact 1:1 relationship. The tendency of the model to under-estimate basal area values at sites with medium to high observed basal area values and to over-estimate basal area values at sites with relatively low observed basal area values is a systematic error that can be explained by potential growth in basal area (at

Table 2

Spearman's rank correlation coefficients for bands 3,4,5 and 7 and NDVI from January 2002 Landsat ETM + image with ground measurements of mean stem density and basal area per transect collected in November 2001 (n = 21)

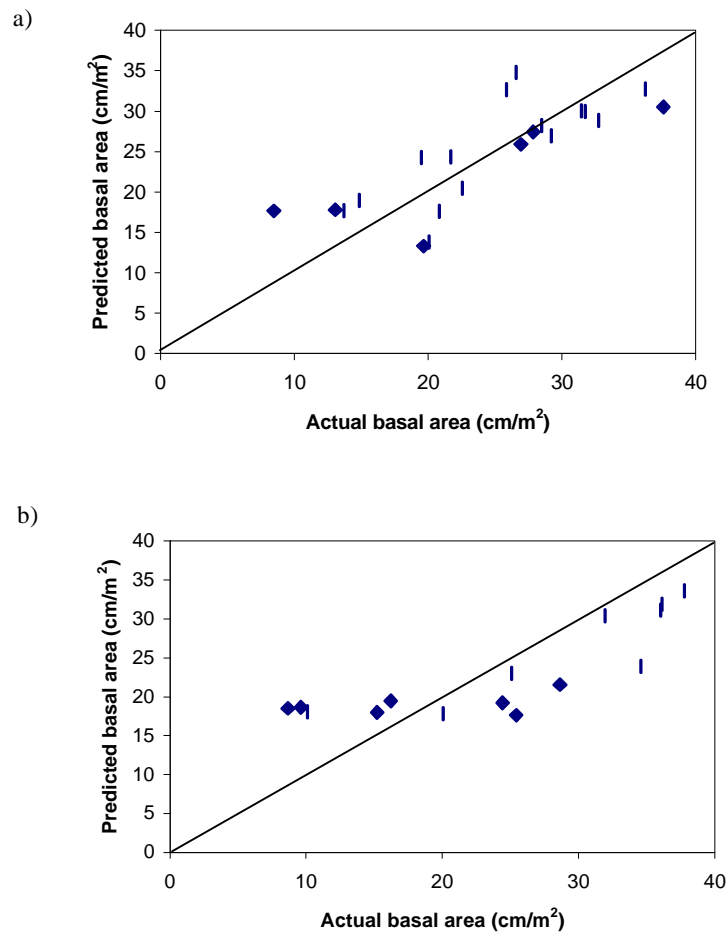
<i>Band</i>	<i>Stem density</i>	<i>Basal area</i>
Red (band 3)	-0.21	-0.61**
NIR (band 4)	-0.27	-0.66**
MIR(band 5)	-0.33	-0.77**
MIR (band 7)	-0.37	-0.76**
NDVI	-0.09	0.14

Note: **Correlation is significant at the 0.01 level.

Source: Table adapted from Ingram et al. 2005.

Figure 3

Relationship of artificial neural network predicted values of basal area plotted against actual measurements of basal area.



Notes: a. The relationship between actual and predicted measures of basal area derived from the ANN using the field data collected in 2001 ($r^2 = 0.62$, $p < 0.01$, $n = 21$). b. The relationship between predicted and actual values of basal area derived from the ANN using data collected in 2001 as the training data and data collected in 2000 as the testing data ($r^2 = 0.67$, $p < 0.01$, $n = 15$). Adapted from Ingram et al. 2005a.

Source: Adapted from Ingram et al. 2005a.

sites recorded as having low to moderate values of basal area in 2000) or loss in basal area (at sites recorded as having a high basal area in 2000) since the data was collected. These changes could alter the relationship between basal area measurements obtained in 2000 and the spectral reflectance values at these sites taken from the 2002 satellite image.

The Predictive Map of Basal Area

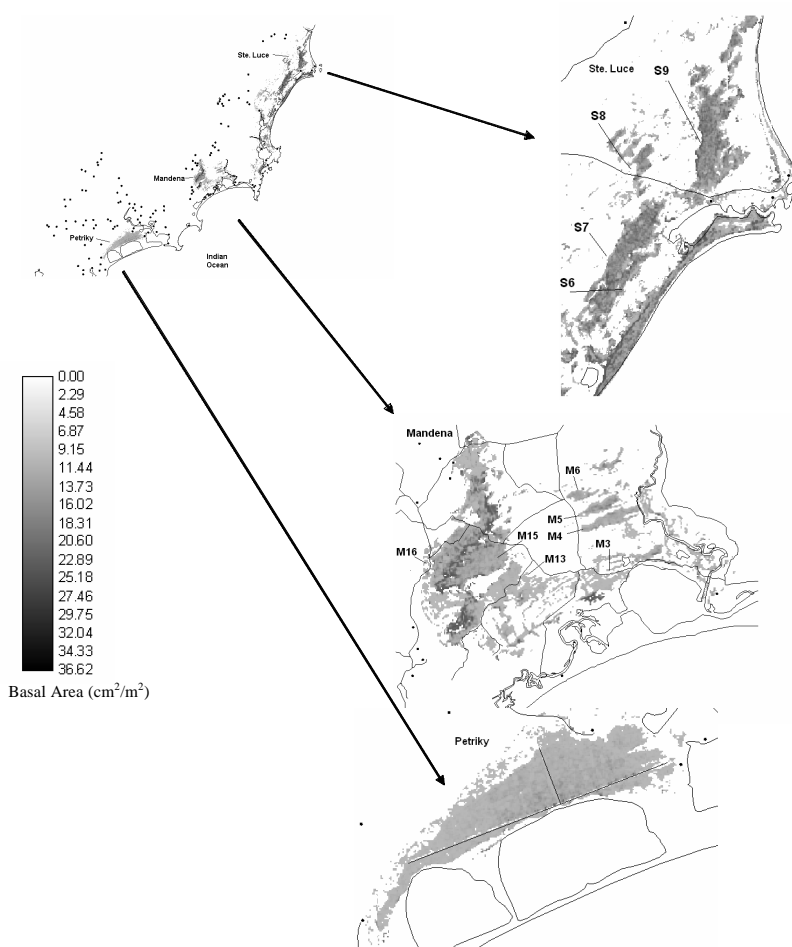
The predictive map of basal area (Figure 4) portrays environmental influences on forest structure and trends in basal area that can be associated with human access to forest fragments and distance from the forest edge. The model predicts Petriky to have an overall lower basal area when compared to Ste. Luce, which can be associated with known climatic gradients across the study sites. Within each forest site, where climate and other physical conditions are generally uniform, the map reveals site-specific trends in basal area related to human accessibility. Basal area values were more strongly related to distance from villages ($r^2 = 0.28$, $p > 0.05$) than from roads ($r^2 = 0.01$, $p > 0.05$). However, when these two factors are considered together in relation to basal area within a multiple regression, a higher degree of variability in basal area is accounted for by these measures ($r^2 = 0.44$, $p < 0.01$). Thus, the two factors of distance to roads and villages must be considered in concert with respect to their influence on basal area. This can be seen in Petriky, where pixels predicted to have the highest basal area occur within the middle of the forest fragment. Although these high predictions of basal area are within pixels proximate to the road, they are at a maximal distance from the three villages located on the periphery of the forest.

The basal area predictions across the Mandena site were highly heterogeneous. The forest stands of M15 and M16 have been designated by the mining company as conservation zones and were predicted to have relatively high basal area values when compared to the other forest stands in the site. The remaining forests in the stands that comprise areas classified as M13 and M3 are not protected as conservation zones, are relatively close to villages and roads and were predicted to have comparatively low values of basal area. These predictions have been confirmed by ground observations and are sites of previous, extensive charcoal-making activities. In comparison, fragments M4, M5, M6 and M7, which are set slightly back from the road and are bordered by a river on one side, were predicted to have more inter-stand heterogeneity in basal area values and a higher abundance of high basal area pixels when compared to other unprotected fragments within Mandena. Generally, the pixels predicted to have the lowest basal area within the stands of M4, M5, M6 and M7 were located predominantly along the edge of these fragments. Multiple pixels across the Mandena site were predicted to possess very high basal area values ($>20 \text{ cm}^2/\text{m}^2$). However, ground observations have confirmed that these areas are swampy habitats dominated by trees, with a large

leaf area index and, thus, are likely to have a spectral signal similar to dense vegetation.

Figure 4

Map of forest basal area as estimated from the ANN using spectral values from Landsat ETM imagery in bands 3, 4, 5 and 7 and ground collected basal area measurements from the year 2001 as training data



Notes: Points represent villages and lines represent roads, rivers, lakes and coastal boundaries. The primary access for humans to the forest stands is by traveling on foot along the roads. Legend presents basal area in cm²/m² values.

Source: Map from Ingram et al. 2005a.

In the Ste. Luce area, there is one main road that passes through the three surveyed forest parcels (S9, S8 and S7). Three villages are located along this road, two of which are close to the coast and one is more proximate to the forest, particularly the forest stand S9. The forest pixels closest to the village were predicted to have a relatively lower basal area than pixels farther from the village and the road. The core areas of the forest parcels were generally comprised of pixels with higher predicted values of basal area.

Tree Diversity Inventories

A total 2155 individuals of 135 tree species were recorded. The maximum number of individuals and species recorded within any one transect were one hundred and fifty-eight and forty-five, respectively. Thirty-two (24%) species were represented by only one individual tree. Seventy-nine of the recorded species (58%) were utilitarian species and fifty-six (42%) species were non-utilitarian. Twenty-three endemic species (17% of the total species) were recorded and seventeen (74%) of these were also utilitarian species. Out of the 2155 individuals recorded 1827 were classified as utilitarian (84%).

The correlations between mean basal area values per transect and measures of diversity per transect were stronger than with mean stem density values (Table 3). The relationships between basal area and diversity were strongest for the endemic species and weakest for the utilitarian species. For 'all' species and utilitarian species, only the SW index and species richness showed a significant relationship to mean basal area. The correlations with stem density were considerably weaker. Only the correlations between stem density and species richness of the total species recorded and utilitarian species were significant.

The occurrence of singletons, species recorded only once, in the utilitarian species group (13% singletons) was much lower than in the non-utilitarian species group (40% singletons). *Zoralahy* (*Dicoryphe stipulacea*) was the most abundant non-utilitarian species and was represented by fifty-one indi-

Table 3

Pearson's correlation coefficients among mean basal area (bold face) and stem density (italics) values per transect with diversity measures at each transect

Type of species	SW index	Simpson's index	Species richness
All tree species	0.64** , 0.41	0.35 , 0.02	0.74** , 0.57**
Utilitarian	0.58** , 0.37	0.30 , 0.02	0.71** , 0.58**
Endemic	0.85** , 0.57	0.74** , 0.28	0.74** , 0.57**

Notes: Results are based on untransformed data for 22 transects. **denotes significance at $P < 0.05$. The correlation between basal area and stem density: 0.60**.

Source: Table adapted from Ingram et al. 2005b.

viduals. In contrast, eleven utilitarian species were represented by > sixty individuals, the most abundant of which were *kalavelo* (*Suregada baronii*) (one hundred and fifteen individuals), *harandrato* (*Intsia bijuga*) (one hundred and eight individuals), and *rotry* (*Syzygium emirnesis*) (ninety-four individuals). Although non-endemic species outnumbered endemic species by far, several endemic species were fairly abundant with only three species were recorded as singletons. The two most abundant endemics were also utilitarian species, *rotry* (priority four species) and *fanolamena* (*Asteropeia micraster*) (seventy-nine individuals, priority two species). The third most abundant endemic species was *zoralahy* (priority two species).

The basal area of utilitarian species as a group comprised the majority of the total basal area across the landscape (84%) and the majority of the basal area in each transect. The species that constituted the highest amount of basal area to the landscape were utilitarian species: *harandrato* (7.54% of total basal area), *rotry*, (6.68%), *hazomainty* (*Diospyros mampingo*) (4.77%), *fanolamena* (4.44%) and *meramaintso* (*Sarcolaena multiflora*) (3.77%).

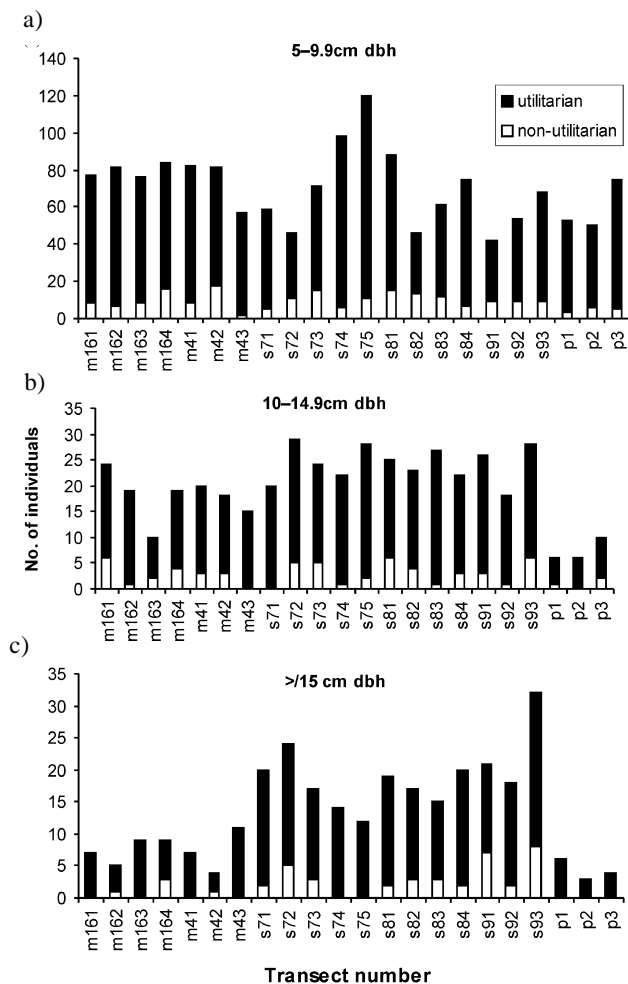
Utilitarian species also contribute the highest proportion of individuals for each of the three size classes in every transect (Figure 5). In the ≥ 15 cm dbh class, eight of the twenty-three transects contained only utilitarian species and in the 10 cm–14.9 cm dbh class, three transects were comprised of only utilitarian species. *Rotry* (twenty-five individuals), *hazomainty* (twenty-two), *harandrato* (twenty-one), *fanolamena*, (sixteen) and *vahabahatra* (*Cinnamosma madagascariensis*) (fifteen) were the most frequently recorded species across the landscape in the ≥ 15 cm dbh size class. The most abundant species in the smallest size class of 5–9.9 cm dbh, which were also abundant within the largest size class, were *harandrato* (sixty-nine individuals), *rotry* (sixty-one), *hazomainty* (fifty-seven) and *fanolamena* (forty-two).

The amount of basal area available at a landscape scale varied across the different utilitarian purposes (Figure 6). The field inventories recorded eight species used for charcoal (QMM 2001), sixty-three species for firewood, twenty-six species for construction, twenty species for medicine, two species for human food and eight species for fibrous material or oil. Basal area across the landscape was highest in the firewood category, followed by construction, medicine, charcoal, food and other non-timber uses. Basal areas values for fibrous material, human food and charcoal were lowest within M4. The basal area available for fibrous material usage and human food was also very low in other transects outside of M4 while the basal area available for charcoal was comparatively higher in all transects located in forest stands other than M4. The number of species available for medicine (Figure 6g) was between five and fourteen for all the transects, with the lowest found in Petriky. However, this could be a consequence of the fact that fewer samples were collected in Petriky.

A total of 323 trees were recorded as cut; however, only 232 were identifiable to a species level, which resulted in fifty-nine species recorded as cut.

Fifteen of the fifty-nine identified cut species were not considered utilitarian and thirteen of these fifteen were cut only once, one species was cut twice and

Figure 5
Abundance of utilitarian and non-utilitarian species divided into three size classes

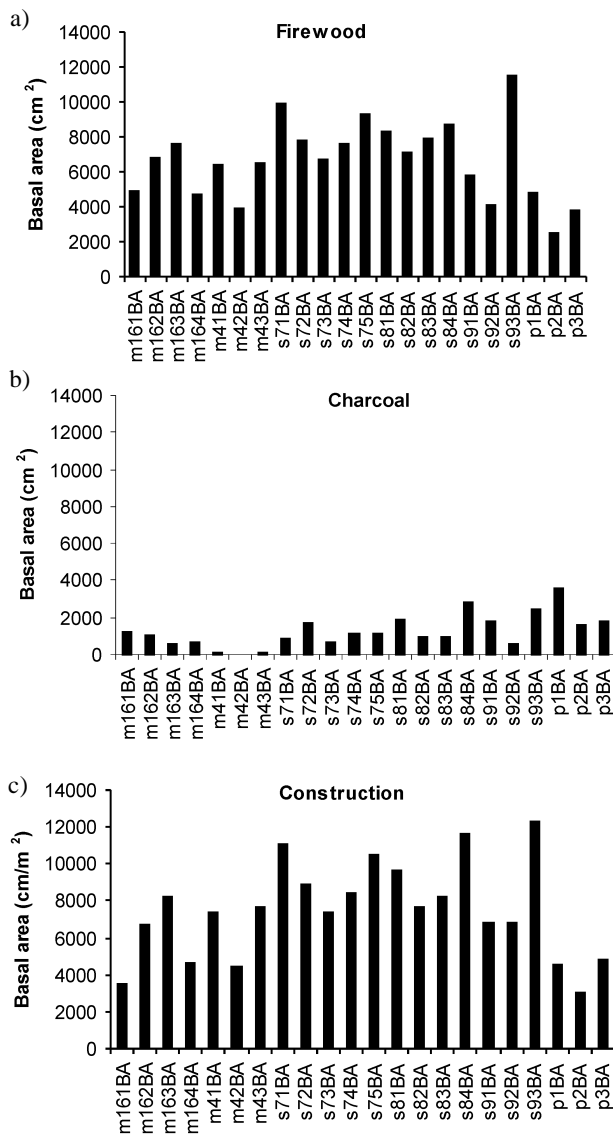


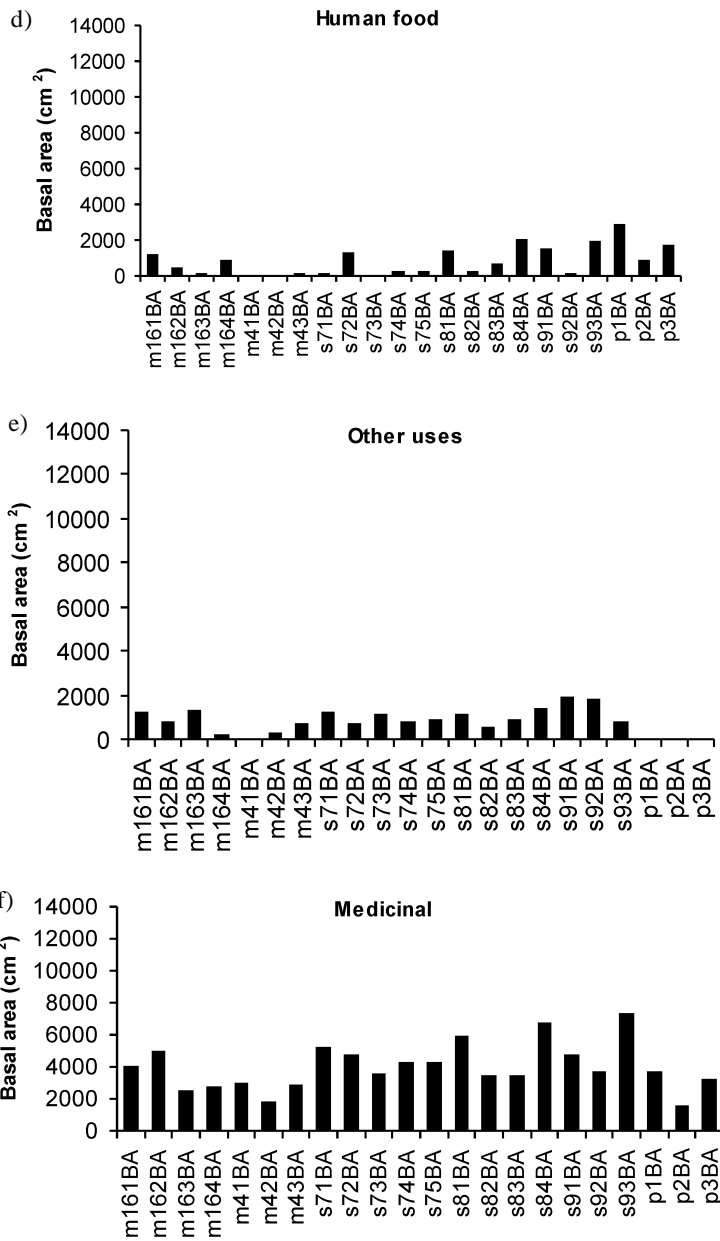
Notes: Number of individuals is plotted on the 'y' axis and the transect number is on the 'x' axis. a. Number of individuals recorded within the 5-9.9 cm dbh size class. b. Number of individuals recorded within the 10-14.9 cm dbh size class. c. Number of individuals recorded within the ≥ 15 cm dbh size class.

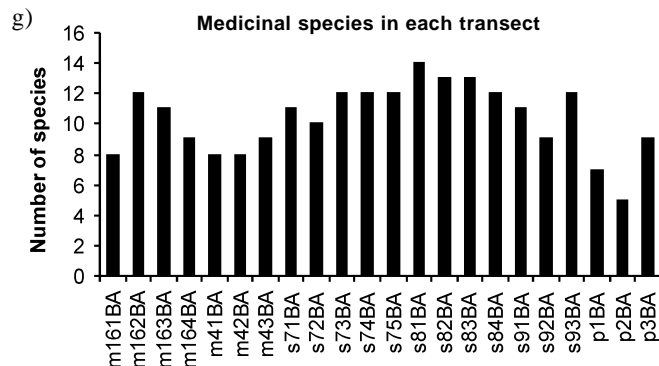
Source: Figure adapted from Ingram et al. 2005b.

Figure 6

Total basal area present within each transect for the different use categories a) firewood, b) charcoal, c) construction, d) human food, e) other uses, f) medicinal, g) total no. of medicinal species







Source: Figure from Ingram et al. 2005b.

one species cut four times. Thirty-four of the forty-four utilitarian species were cut more than once. The most commonly cut species was *harandrato* (fifty-seven cuts), which is used for timber and medicine. *Forofky* (also known as korofky) (*Diospyros littoralis*) was cut twelve times and is used for firewood, timber and medicine. *Rotry* was cut eleven times and is used for energy, timber, medicine and human food.

The species with the highest number of coppicing individuals (which may have coppiced from natural or anthropogenic disturbance) were the utilitarian trees *ambora* (ten), *meramaintso* (ten), *harandrato* (nine) and *rotry* (nine). Four coppiced non-utilitarian species were recorded once (*beronono*, *Trilepisium madagascariense*; *disaky*, *Garcinia aphanophlebia*; *hazofotsy*, *Homalium planiflorum*; *tombobitsy*, *Psorospermum* sp.), one coppiced non-utilitarian species, *katrafay* (*Terminalia fatrae*) was encountered twice and six coppiced individuals were recorded for the non-utilitarian tree, *zoralahy*.

DISCUSSION

Deforestation Studies

The estimates of deforestation obtained in this study varied from the previous estimates of forest cover obtained during the studies commissioned by QMM for inclusion in their SEIA (QMM 2001; Vincelette et al. 2003). These discrepancies are likely due to the analysis of different time intervals between the studies, different classifications of forest and deforestation, and the application of different methods. The delineation of littoral forest and non-forest is difficult due to the very patchy nature of the smaller forest stands. For this

reason, the visual interpretation of aerial photographs employed in the SEIA to assess forest cover in Mandena, Petriky and Ste. Luce involves a considerable degree of user subjectivity (MIR Télédétection Inc.1998). The method presented in this study, however, applies a systematic, repeatable technique, which can be applied by future researchers for comparative purposes and monitoring.

The spatial and temporal scale of a study can significantly influence estimates of deforestation (Geist and Lambin 2001) as has been demonstrated here. Across the Mandena, Ste. Luce and Petriky sites, forest loss amounts and rates varied spatially and temporally. Of the three sites, Petriky had the highest amounts and rates of forest loss during 1984–1992, while a different pattern of forest loss was observed for Ste. Luce and Mandena, in which forest losses were highest during the shortest time between 1999 and 2002. The present results indicate that perceptions of forest change are highly dependent on the spatial and temporal scales of the analysis. Particularly, the selection of image dates chosen for an analysis can greatly influence interpretations of how deforestation is proceeding in the region.

The difference in rates of deforestation at the three sites can be associated with different human-induced changes occurring at each site. The high rate of deforestation in Petriky between the years 1984 and 1992 coincides with the clearing of forest to construct a major access road that bisects the forest parcel (Figure 4). The road was constructed by the mining company during this period. In Mandena, the high degree of forest loss occurring later in the study period was primarily due to a single, widespread deforestation event caused by itinerant charcoal makers within a large block of intact littoral forest, previously known as M3 (Henderson 1999). Such widespread deforestation in Mandena has been directly attributed to the recent incursion and practices of itinerant charcoal makers (QMM 2001). The recent influx of charcoal makers into the area could be related to questions over traditional land tenure. In rural Madagascar, deforestation associated with external pressures, such as influx of migrants, is often exaggerated along territorial boundaries, between communities or on land where ownership by government, clans, lineages, or communities is relatively uncertain (Durbin et al. 2003). The government or a select few private individuals own most of the land in the Fort Dauphin area, although, traditionally, the populations that used the land gained customary rights to it (Lewis Environmental Consultants 1992; QMM 2001). Land rights in the area, particularly in Mandena, now seem uncertain, although it has traditionally been used and managed by local populations. In the mid to late 1990s before the government granted QMM a permit to mine Mandena, the company established field stations for researchers working in Ste. Luce and Mandena as well as a botanical garden and a visitor centre in Mandena. Although mining has not yet begun, QMM's established presence in Mandena

may have served as an invitation to migrants, such as charcoal makers because now it may not be clear which group, if any possesses access rights to the land. This is not an unusual situation in rural Madagascar, where migrants are often responsible for deforestation in their non-native communities and, thereby, may disrupt the local social structure and cohesion that is based on a respect for traditional values and production systems (Durbin et al. 2003). More generally, the incursion of outside pressures of any sort such as changing values, livelihoods and markets are common factors known to alter the relationships between local people and their natural environment causing intensified resource use (Colchester 1981; Adams and McShane 1992). Thus, the presence of a multi-national mining company operating in an area that has traditionally been managed by local people could influence social and environmental change in ways that extend beyond the direct impacts of the mining operation.

Assessment of Forest Condition

The ANN-produced landscape scale predictions of basal area comparable to the semi-quantitative map of forest condition produced by QMM, captured between-site differences in basal area associated with environmental variability, and portrayed trends in basal area associated with human impact and accessibility to forest fragments. Soil nutrients and climate are known to vary among the three study sites (QMM 2001) and, in the absence of humans, spatial patterns of forest basal area will be determined by environmental variables (Salvador 2000) such as soil type, soil nutrients, climate, disturbance regime and topography (see Clark and Clark 2000 for a review). The ANN predictive map captured the climate and soil nutrient-related differences between Petriky and Ste. Luce by predicting low basal area values throughout Petriky and higher basal area values throughout Ste. Luce. Although Mandena has been classified as floristically more similar to Ste. Luce than to Petriky, structural differences were not as evident between Mandena and Petriky. Mandena lies between the two climatic extremes of Ste. Luce and Petriky, is the closest forest site to the largest town in the area, Fort Dauphin, and is surrounded by a higher density of villages than the other two forest sites. Thus, basal area in Mandena appears to have been shaped more significantly by the comparatively dense human populations living close to the forests rather than environmental factors alone. Furthermore, Mandena's proximity to Fort Dauphin has resulted in more pressure on the forest resources that are sold in the local market. In each of the three forest sites where environmental variables, such as a precipitation and soil nutrients, are relatively constant across the site, predicted patterns of basal area are correlated with distance from villages, roads and relative ease of accessibility to the forests. Changes in forest structure and decreases in

basal area have been observed with distance from human settlements or roads and related to human pressure on forest resources (Medley 1993; Vermeulen 1996; Chittibabu and Parthasarathy 2000). In this region, specifically, degree of stand isolation and human accessibility has also been shown to influence community composition of trees and shrubs (Cadotte et al. 2002). The predictive map of basal area showed basal area values increasing as distance from forest edges, villages and roads increased. However, these trends must be considered contextually in each forest site. For example, in the forest of Petriky, the basal area is highest in the middle of the forest parcel, at a maximal distance from the three villages, but, proximate to the road built by QMM. Although the construction of this road coincided with an initial increase in deforestation in the area (as discussed earlier), the predictive map shows that most of the forest use in Petriky, as indicated by comparatively lower values of basal area, occurs at distances closest to human populations, regardless of the presence of a road. However, in Mandena where village density is higher and the road network is more diverse and sprawling, basal area appeared to be affected by a combination of factors associated with accessibility, such as distance to roads, villages and water bodies. An exception to this pattern was seen in the forest stands M15 and M16 in Mandena, which are located in close proximity to both roads and villages, but were predicted to have the highest values of basal area of any forest stands located in Mandena. This pattern is attributed to the presence of researchers and QMM employees working in/around these forest patches and their status as conservation zones, where forest exploitation is restricted. In both Ste. Luce and Mandena, unprotected stands that were bisected by roads and located close to villages were predicted to have lower basal area values when compared to fragments that were slightly more removed from the road, villages and/or protected by river or water boundaries. Such findings underline the fact that local context is crucial when considering trends in forest cover and land use (Kull 2000). The ANN also predicted well-known edge effects by estimating lower values of basal area for pixels located along stand edges. This was especially evident around the larger stands, such as S8 in Ste. Luce. The ability of the ANN to portray trends of basal area in relation to environmental and anthropogenic factors known to influence forest systems from other studies combined with predictions similar to broad trends in littoral forest condition mapped in previous studies (QMM 2001) demonstrates the utility of this method for providing quantitative assessments of forest structural patterns at a landscape level. The advantage of this quantitative method is that it diminishes the likelihood of making subjective assumptions of forest condition in relation to human impact, particularly, when climatic and other physical factors may also influence forest structure. This method also permits a landscape scale understanding of

the nature of human impact on forest resources and, thus, may be useful for conservation and land-use planning.

Relationship between Structure and Biodiversity

Structural measures, such as basal area, that can be assessed and mapped with remote sensing imagery can greatly aid conservation planning if those measures can also be related to measures biodiversity (Steininger 2000; Ingram et al. 2005a). Basal area is not only a structural feature that we were able to estimate and map accurately with spectral data and ANNs, but was also the structural feature most strongly correlated to species diversity measures for all of the tree species groups considered, utilitarian, endemic and total number of species, but particularly endemic species. Thus, in this region, maps of forest structure may be useful for identifying areas of high basal area and, thus, sites of potentially high species richness. Such tools may be useful for prioritising areas on the landscape for conservation, but should be supplemented by ground assessments.

Species Abundance, Basal Area and Human Impact across the Landscape

These forests have important conservation value not only for biodiversity but also for human livelihoods according to our results. The high utilitarian species richness and abundance recorded in all the transects were similar to the findings of Kremen et al. (1999), who found that 90% of the individual plants recorded during forest inventories in Madagascar's Masaola peninsula were utilitarian species. The comparatively large number of singletons in the non-utilitarian category indicates that this group comprises a large proportion of relatively rare species and, thus, people are using the most common species across the landscape. The results demonstrate that human use of a species does not unequivocally result in low abundance, low regeneration potential or basal area of that species across a landscape. However, because this assessment represents a snap-shot in time, it is not possible to determine whether degradation is progressing and if root stocks could eventually deteriorate if coppicing occurs too frequently. Thus, if current disturbances increase beyond a sustainable threshold, it is possible that highly used tree species could decline. However, if degradation by local people had already reached an unsustainable point and the forests had been completely over-exploited, we would expect very low abundances and low basal area values for the species that are used regularly by local people and higher abundances and basal area measurements for the non-utilitarian species. Our results did not support this expectation and, in contrast, showed that these forests continue to harbour a high diversity and abundance of useful and endemic species. Frequent use did not

seem to impact species abundance or basal area of the most regularly used trees. *Harandrato* and *rotry* were the two most frequently cut species across the landscape and, also, two of the most abundant species and contributors of basal area. *Harandrato* was the second most abundant species across the landscape, constituted the highest basal area overall, and recorded as the most frequently cut and coppiced tree encountered in the surveys. This species was also highly abundant in both the smallest and largest size classes, which demonstrates that the species has adequate reproductive capability. *Rotry*, both a utilitarian and endemic species, was abundant in the largest size classes, constituted the second highest amount of basal area, was widely used across the landscape as the second most frequently cut tree, and was also recorded as having a relatively high number of coppiced individuals. Tree species that coppice may possess an advantage for recovery from disturbance (McLaren and McDonald 2003), which may be a reason why *Harandrato* and *Rotry* are so abundant despite frequently cut and used. *Rotry*, which can reach relatively high diameters at breast height for this forest system, may also be protected from over-exploitation due to the cultural belief that spirits are thought to live in very large trees (QMM 2001). Therefore, *Rotry* may not be cut once it exceeds a certain diameter despite its utility for a variety of purposes. Spiritual uses were not considered categorically in this study, but may have a considerable influence on tree community composition and conservation. For example, several forest patches across the landscape in Mandena, known as 'taboo forests', are not cut by local people because they are considered to be sacred (McConnell et al. 2004). Such observations highlight the importance of understanding the cultural context that influences land-cover change and composition.

The amount of basal area available for certain purposes varied across transects and use categories. The firewood category possessed the highest amount of basal area over any other use category. Approximately 90% of the energy needs in rural areas are met from firewood, while charcoal is the primary energy source for Fort Dauphin (QMM 2001). Many local people living proximate to the littoral forests collect dead wood rather than cutting live trees for energy usage (QMM 2001), which has a lesser impact on forest structure and diversity than cutting. For this reason, deforestation in the littoral forests due to firewood use may be minimal in comparison to exploitation due to charcoal production, which is a much more destructive practice. Furthermore, the high frequency of coppicing amongst firewood species at the study sites suggests that many of these tree species can recover from a certain degree of disturbance.

Charcoal production has recently increased in the Mandena region due to the influx of outside groups, such as Androy migrants from the south, who

have come to the area to harvest trees for charcoal to sell in Fort Dauphin (QMM 2001). As mentioned previously, this activity has been a primary source of recent, large-scale deforestation in the region and is detectable from satellite imagery and aerial photographs. The impact of this practice on species diversity was apparent in the transects located in the unprotected fragment of M4 in Mandena; the only forest stand that was completely lacking tree species used for charcoal. The paucity of charcoal species in M4 could be due to the fact that these species do not naturally occur in this region or because these species have been over-harvested from the Mandena region. The former scenario is unlikely because the species used for charcoal, *fanola*, was recorded as cut in one of the M4 transects, although there were no intact individuals of this species recorded in any of the transects. The fragment of M16, also in Mandena, has a higher abundance of tree species used for charcoal than that found in M4. This may be due to the fact that destructive charcoal making is prohibited in M16 because it is part of a conservation zone recently established and managed by QMM under an agreement made with local communities. As shown in the predictive map of basal area, M4 possessed comparatively higher values of basal area than other unprotected stands in the Mandena site, such as M3 and M13, both of which have been severely degraded because of charcoal exploitation. Thus, the stand of M4, which harboured both endemic and utilitarian species, may be threatened by pressures of charcoal production.

Medicinal plants are also very important across the landscape since villagers consider the forest to be their pharmacy and, thus, rarely use western medicine (QMM 2001). Compared to the other use categories, very few of the medicinal plant species actually occurred within the >15 cm dbh class. This trend could be due to the fact that these species do not naturally obtain large girths, because they have been over-harvested or because people are harvesting leaves, berries or flowers rather than the entire tree for medicinal uses. Due to the importance of these species for human health and the future loss of tree biodiversity that will result from mining, more research is needed to determine if medicinal plant species are decreasing in abundance and species richness and what percentage of the local medical needs are met by forest tree and plant species.

CONCLUSIONS

This research has highlighted the complexity of littoral forest change, condition, and composition in the cultural landscape of south-eastern Madagascar. The results, have shown that a landscape scale perspective is critical for identifying patterns of human impact on nature and for understanding the domi-

nant factors, which influence these patterns. The different cultural, physical and geographic contexts of these sites are shown to influence patterns of land-cover change, condition, and composition. Assessment of forest cover change conducted at each site individually at a high temporal resolution illustrated spatially and temporally dynamic patterns of forest loss and, thus, discourage simplistic, linear portrayals of littoral deforestation. This disaggregated assessment of forest cover change permitted exploration of a variety of factors threatening forest cover at different sites, which is important for curbing deforestation and managing forest resources. This more nuanced depiction of deforestation was supplemented with a quantitative assessment of the structure of the remaining forest stands of the littoral landscape. This assessment showed that a combination of environmental factors, at the broadest scale, and the degree of isolation and human accessibility at a site-scale, influence forest basal area, which can be related to forest condition. An understanding of how different human and natural factors interact across the landscape and where anthropogenic pressures are the greatest can support management of resource use and biodiversity conservation.

In order to understand how forest condition may influence community composition, we conducted inventories of tree species diversity and abundance. These surveys revealed a strong relationship between basal area and diversity measures, suggesting that human impact may influence not only forest structure but also species composition. However, despite the impact inferred by this relationship, species richness and diversity of tree species communities remained high across the landscape. Human impact on species diversity varied across user groups: forest use practices by local people seem to be more sustainable with respect to maintaining diversity and abundance of utilitarian trees than the forest use practice of migrant groups. The high overlap between endemic and utilitarian species lists presents opportunities for conservationists and local people to work together to maximise conservation goals and local human needs across this landscape, which is facing many different use pressures.

This study has demonstrated that though humans have had a discernable impact on the littoral forest landscape in south-eastern Madagascar, this impact is variable throughout time and space and is a function of the cultural and environmental factors that vary across each site. Although, often described as severely degraded, these forests remain repositories of biodiversity and resources important for human well being. For these reasons, special attention should be given to these areas as the social, environmental, and economic atmosphere of the region continues to change, influencing forest cover change, condition, and the use of forest resources.

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Notes

1. It was not possible to acquire a GPS coordinate for one of the transects and, for this reason, it was omitted from the satellite imagery analysis.
2. Voucher specimens were collected by Dr. Aaron Davis, H el ene Ralimanana, Franck Rakotonasolo, and Rolland Ranaivojaona. The scientists undertook vegetation surveys in the proposed Fort Dauphin mining area, between 31st January and 29th February 2001. From this work, construction of vernacular names database was made for the proposed mining area in Fort Dauphin. All of the herbarium specimens from this work are permanently stored at RBG, Kew and Antananarivo (PBZT).

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Appendix 1

Latin and common names for trees recorded during ground surveys in October–December 2000.

Latin names were not available for some species

Latin name	Common name
<i>Ophiocolea delphinense</i>	Akondronola
<i>Plagiosciphus</i>	Ambiropiso
<i>Tambourissa</i> sp.	Ambora
<i>Dyopsis scotiana</i>	Amboza
<i>Vepris eliottii</i>	Ampoly
-----	Arotsy
<i>Noronhia</i> sp.	Belavenoke
<i>Crataeva obavta</i>	Belataka
<i>Bembicia uniflora</i>	Bemalemy
-----	Benalessy
-----	Beondue
<i>Dombeya mandenensis</i>	Berehoka
<i>Trilepisium madagascariense</i>	Beronono
<i>Betarepokala</i>	Betarepokala
<i>Dyopsis stlucei</i>	Boaka
<i>Dyopsis</i> sp.	Boakabe
<i>Garcinia</i> sp.	Disaky
-----	Dona
<i>Dracaena reflexa</i>	Falinandro
<i>Dracaena fontanesiana</i>	Falinandrobe
<i>Euphorbia laro</i>	Famanta
<i>Vernonia pectoralis</i>	Famoty
<i>Polycardia phyllanthoides</i>	Fandrianakanga
<i>Asteropeia</i> sp.	Fanola
<i>Asteropeia multiflora</i>	Fanolafotsy
-----	Fanolalohy
<i>Asteropeia micraster</i>	Fanolamena
<i>Physena madagascariensis</i>	Farisaty
<i>Canthium</i> sp.	Fantsikahitra
<i>Clerodendrum</i> sp.	Fantiskoho
<i>Vepris fitoravina</i>	Fitoravy
<i>Leptolaena pauciflora</i>	Fonto
<i>Diospyros littoralis</i>	Forofoka
<i>Leptolaena delphinensis</i>	Fotombavy
<i>Schizolaena elongata</i>	Fotondahy
<i>Ambavia gerrardii</i>	Fotsivavo
<i>Grewia delphinense</i>	Hafipotsy
-----	Hainiha
<i>Podocarpus madagascariensis</i>	Harambilo
<i>Elaeodendron pauciflorum</i>	Haramboanzo
<i>Intsia bijuga</i>	Harandrato
-----	Harompanitsy
<i>Dais glaucescens</i>	Havohoa
-----	Hazimina
<i>Homalium planiflorum</i>	Hazofotsy
-----	Hazohiha
<i>Diospyros mampingo</i>	Hazomainty

<i>Anisophyllea fallax</i>	<i>Hazomamy</i>
<i>Homalium</i> sp.	<i>Hazombato</i>
<i>Mammea parviflora</i>	<i>Hazomiteraka</i>
<i>Tricalysia cryptocalyx</i>	<i>Hazongalala</i>
<i>Oncostemum</i> sp.	<i>Iona</i>
<i>Mammea sessiliflora</i>	<i>Jambo</i>
<i>Cerbera manghas</i>	<i>Kabokala</i>
<i>Suregada baronii</i>	<i>Kalavelo</i>
<i>Drypetes madagascariensis</i>	<i>Kambatsikambatsy</i>
<i>Rhus thouarsii</i>	<i>Kangy</i>
<i>Terminalia fatrae</i>	<i>Katrafay</i>
<i>Tricalysia cryptocalyx</i>	<i>Kotofotsy</i>
<i>Noronhia</i> sp.	<i>Lahinampoly</i>
<i>Scolopia erythrocarpa</i>	<i>Lapivahatra</i>
-----	<i>Lengohazo</i>
<i>Monoporus bipinnatus</i>	<i>Lona</i>
<i>Brochoneura acuminata</i>	<i>Mafotra</i>
<i>Macaranga obovata</i>	<i>Makarangana</i>
-----	<i>Malotra</i>
<i>Cynometra cloiselii</i>	<i>Mampay</i>
-----	<i>Manary</i>
-----	<i>Mangaroo</i>
<i>Enterospermum</i>	<i>Mangavao</i>
-----	<i>Manobary</i>
<i>Enterospermum</i> aff. <i>berieranum</i>	<i>Maranitratoraka</i>
<i>Homalium nudiflorum</i>	<i>Marankoditra</i>
<i>Blotia hildebrandtii</i>	<i>Maroando</i>
<i>Aspidostemon elliotii</i>	<i>Menahihy</i>
<i>Sarcolaena multiflora</i>	<i>Meramaintso</i>
-----	<i>Merana</i>
<i>Zanthoxylum tsihanihimposa</i>	<i>Mohongo</i>
<i>Faucherea hexandra</i>	<i>Nato</i>
<i>Mimusops coriacea</i>	<i>Natobonaka</i>
-----	<i>Natohetike</i>
<i>Faucherea hexandra</i>	<i>Natotendrokazo</i>
<i>Vitex bracteata</i>	<i>Nofotrakoho</i>
<i>Ficus reflexa</i>	<i>Noroka</i>
<i>Pandanus concretus</i>	<i>Pandanus</i>
<i>Rakotofotsy</i>	<i>Rakotofotsy</i>
<i>Homalium albiflorum</i>	<i>Ramirisa</i>
<i>Canarium madagascariensis</i>	<i>Ramy</i>
<i>Beilschmiedia madagascariensis</i>	<i>Resonjo</i>
<i>Eugenia cloiselii</i>	<i>Ropasy</i>
<i>Syzygium emirnesis</i>	<i>Rotry</i>
<i>Astrotrichilia elliotii</i>	<i>Sagnira</i>
-----	<i>Sagnirana</i>
<i>Macphersonia radlkoferi</i>	<i>Sagnirambaza</i>
-----	<i>Sahinipotsira</i>
<i>Turraea lanceolata</i>	<i>Sakaimboalavo</i>
<i>Elaeocarpus alnifolius</i>	<i>Sana</i>
-----	<i>Sanikalal</i>
<i>Astrotrichilia elliotii</i>	<i>Sanirana</i>

<i>Malleastrum mandanese</i>	<i>Sarigoavy</i>
<i>Rhus taratana</i>	<i>Sarinato</i>
<i>Sinoronoro</i>	<i>Sinoronoro</i>
<i>Poupartia chapelieri</i>	<i>Sisikandrongo</i>
-----	<i>Sivory</i>
-----	<i>Soazanahary</i>
<i>Rhodocolea racemosa</i>	<i>Somotsoy</i>
<i>Phylloxylon xylophyloides</i>	<i>Sotro</i>
<i>Gaertenera arenaria</i>	<i>Tagnatagnanala</i>
<i>Tahambasiky</i>	<i>Tahambasiky</i>
<i>Hyperacanthus mandenensis</i>	<i>Taholanga</i>
<i>Erythrina</i> sp.	<i>Talanosy</i>
<i>Petchia madagascariensis</i>	<i>Tandrokasy</i>
<i>Taolonana</i>	<i>Taolonana</i>
<i>Rothmania mandenensis</i>	<i>Tavlana</i>
<i>Cryptocarya oblonga</i>	<i>Tavolohazo</i>
<i>Ocotea laevis</i>	<i>Tefimoa</i>
<i>Psorospermum</i> sp.	<i>Tombobitsy</i>
<i>Erythroxyllum corymbosum</i>	<i>Tomizo</i>
<i>Homalium involucreatum</i>	<i>Tsanhiposa</i>
<i>Rhopalocarpus coriaceus</i>	<i>Tsilavimbato</i>
<i>Cinnamosma madagascariensis</i>	<i>Vahabahatra</i>
<i>Agelaea pentagyna</i>	<i>Vahimainty</i>
<i>Clematis</i> sp.	<i>Vahivoraka</i>
<i>Pandanus concretus</i>	<i>Vakoa</i>
-----	<i>Vaksa</i>
-----	<i>Vantrilana</i>
<i>Protorhus ditimena</i>	<i>Varongy</i>
<i>Brexia madagascariensis</i>	<i>Voakarepokala</i>
<i>Homalium albiflorum</i>	<i>Voankazoala</i>
<i>Uapaca louvelii</i>	<i>Voapaky</i>
-----	<i>Voavoa</i>
<i>Schefflera vantsilana</i>	<i>Voatsilana</i>
<i>Phyllarthron ilicifolium</i>	<i>Zahambe</i>
<i>Mammea perrieri</i>	<i>Zambo</i>
<i>Homalium viguirei</i>	<i>Zora</i>
<i>Dicoryphe stipulacea</i>	<i>Zoralahy</i>
