

**HABITAT FRAGMENTATION, TREE SPECIES DIVERSITY AND
LAND COVER DYNAMICS IN A RESETTLEMENT AREA IN
CHIMANIMANI DISTRICT OF ZIMBABWE: A SPATIO-TEMPORAL
APPROACH**

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requirements for the Master of Philosophy degree in Geography and
Environmental Science.

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Dedication

To a father who knows the value of a good education, who sees it as the one inheritance they can't take away. A man whose eyes sparkle at the achievements of his offspring. The one whose talented hands make it all possible

To a mother so full of amazing love. A woman so simple, soft spoken and gentle yet always that pillar of immense strength

To a sister so good a friend. One always quick to listen and with a solution to it all. Who knows how to light up your world, laughing life's miseries away. The best friend ever

To a young brother so full of potential and destined for greatness. My most treasured friend

To a niece so tiny and adorable, may you be inspired

Declaration 1: Originality

I hereby declare that this thesis submitted for the Master of Philosophy degree at the University of Zimbabwe is my original work and has not been previously submitted to any other institution of higher education. I further declare that all sources cited or quoted are indicated by means of a comprehensive list of references.

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Declaration Statement: Publication in Preparation

Details that form part and/or include research presented in this thesis include publications submitted

Publication 1

Charity Nyelele¹, Amon Murwira², Munyaradzi D. Shekede³ and Prisca H. Mugabe⁴ (Tropical Ecology 3rd issue of Vol. 55, 2014. REF MSE-3228) *Woodland fragmentation explains tree species diversity in an agricultural landscape of Southern Africa.*

The work was conducted by the first author under the guidance and supervision of the second, third and fourth authors.

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Publication 2:

Charity Nyelele¹, Amon Murwira², Munyaradzi D. Shekede³ and Prisca H. Mugabe⁴ (Submitted to South African Geographical Journal-Manuscript ID RSAG-2012-0025) *Modelling past, present and future spatial dynamics in land cover in a resettlement area in Zimbabwe.*

The work was conducted by the first author under the guidance and supervision of the second, third and fourth authors.

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Abstract

Agricultural expansion into forests leads to habitat fragmentation and the creation of habitat patches that can only support a limited amount of biodiversity. Adopting theoretical frameworks which seek to understand biodiversity variations in agricultural landscapes currently undergoing rapid land cover changes mainly due to agricultural expansion into forests is important for promoting biodiversity-agriculture coexistence. The main objective of this thesis was to test whether the area-diversity prediction of the island biogeography theory can successfully be used to explain differences in tree species diversity among different woodland patch sizes in Nyabamba resettlement area of south-eastern Zimbabwe. We also tested whether cropland expansion drives land cover change in resettled landscapes of Zimbabwe. The area-diversity prediction of the island biogeography theory was used to explore whether woodland patches of different sizes had significant differences in tree species diversity. We also used remotely sensed data in a GIS-Markov chain modelling framework to determine historic as well as future predictions of land cover dynamics in the study area. Our results show that larger woodland patches had significantly higher species diversity than smaller woodland patches, indicating that the island biogeography theory can be used to explain tree species diversity differences in agriculturally fragmented woodlands. Results of historic land cover modelling and futuristic spatial predictions showed increases in cropland and wooded grassland accompanied by decreases in plantation and woodland. We also found that soil types and distance from rivers significantly influence land cover conversions. Results of this thesis imply that habitat fragmentation has a significant effect on tree species diversity and that cropland expansion is a major driver of land cover conversions in newly resettled agricultural landscapes.

List of Acronyms

ANOVA	Analysis of Variance
ASTER	Advanced Spaceborne Thermal Emission and Reflection Radiometer
DEM	Digital Elevation Model
EDA	Exploratory Data Analysis
FTLRP	Fast Track Land Resettlement Programme
GIS	Geographic Information System
GPS	Global Positioning System
Ha	Hectare
ILWIS	Integrated Land and Water Information System
Landsat TM	Landsat Thematic Mapper
LSD	Least Significant Difference
MLP	Multi Layer Perception
SPSS	Statistical Package for the Social Sciences

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Chapter 1: Introduction

1.1 General background

Habitat fragmentation results in the division of large, continuous habitats into smaller, more isolated remnants (Didham, 2001). Agricultural land expansion into woodland areas is a major driver of biodiversity changes the world over (IUCN, 2002) mainly through the process of habitat fragmentation. Habitats are progressively subdivided into smaller, geometrically more complex and isolated patches (Harris and Weiner, 2003) which can only support a limited amount of biodiversity due to habitat fragmentation. The amount, pattern and spatial scale of habitat fragmentation can be quantified from remotely sensed data taken at different times. Understanding how habitats are occupied through space and time allows predictions to be made for biodiversity conservation as well as promoting agriculture and biodiversity co-existence.

According to the Convention of Biological Diversity (2013), the proportion of total land area covered by forest in Zimbabwe has been falling significantly, at a rate of 330,000 hectares per year. Woodland cover declined from 53% in 1992 to 42.34% in 2008, while cropland expanded from 27.48% in 1992 to 41.24% in 2008. The Ministry of Environment and Tourism (2006) highlights that the main drivers of biodiversity change in Zimbabwe include population pressure and poorly defined tenure systems, excessive harvesting for both domestic and commercial uses, as well as conversion of forest areas to agricultural land. The Fast Track Land Reform Programme (FTLRP) initiated in 2000 by the government has

resulted in an increase in the number of small-scale farmers who upon settlement convert previously wooded areas into cropland.

Several theories have been formulated to understand species dynamics in space including MacArthur and Wilson's island biogeography model. The theory emphasizes the role of area and geographical isolation as the basic determinants of species diversity through their effects on colonization and extinction rates. According to Kadmon and Allouche (2007), species diversity in a local community reflects a dynamic balance between colonization (arrival of new species) and extinction of species already present in the community. Based on the assumption that colonization rates are determined by the degree of geographical isolation and extinction rates are determined by the size (area) of the island, the theory predicts that species richness should be positively correlated with island size and negatively correlated with the degree of isolation (MacArthur and Wilson 1967). It has further been suggested that the area of an island may influence the rate of colonization because large areas receive more colonizers than small areas (Connor and McCoy 1979).

The theory although formulated for migratory oceanic species may still apply to terrestrial species and might be critical for balancing conservation and agricultural production objectives. However terrestrial species have received limited attention within the context of this theory. This study will attempt to apply one of the predictions of the island biogeography theory which hypothesizes that biodiversity is a function of island area to the study of tree species in an agricultural landscapes. Based on this prediction larger islands are predicted to support more species than smaller islands (Whitcomb *et al.*, 1981). Adopting such theoretical

frameworks may offer a way to analyse area-diversity relationships in agricultural landscapes.

In Zimbabwean agricultural landscapes that are currently undergoing land cover conversions there is also need to investigate the nature, extent and pattern of these conversions so as to assess resultant forest cover changes and their consequences. Although such studies may improve our understanding of the nature of land cover conversions as well as the impact of human activities, few studies (Elliot *et al.*, 2006; Petit *et al.*, 2001) have tried to understand land cover conversions in areas experiencing fast resettlement processes and resultant deforestation. In this study deforestation is defined as the direct conversion of woodland to non-woodland land. Several models that can be used in modeling land cover change exist and include stochastic models such as Markov models. However, most studies that have used Markov chain models to understand land cover conversions have done so outside of agricultural landscapes. Markov modelling has been used extensively in the study of urban environments and forest systems but rarely in modelling land cover conversions in agricultural landscapes such as newly resettled agricultural areas. The futuristic perspective of Markov chains helps in building proactive environmental management programmes.

1.2 Thesis objectives

In this study we had two main objectives. The first objective was to test whether the area-diversity prediction of the island biogeography theory can be successfully used to explain differences in tree species diversity among different woodland patches in newly resettled areas. The second objective was to test in what ways does cropland expansion drive land

cover change in resettled landscapes now and in the future so as to understand the nature, pattern and extent of land cover conversions which lead to habitat fragmentation.

1.3 Outline of the thesis

This thesis consists of five chapters testing the area-diversity relationship in woodland patches as well as understanding how cropland expansion drives land cover change in resettled landscapes now and in the future.

Chapter 1 illustrates the importance of understanding area-diversity dynamics to promote agriculture and biodiversity co-existence in agricultural landscapes as well as monitoring historic and futuristic land cover dynamics in such landscapes. This chapter also describes the objectives of the thesis.

Chapter 2 outlines the methods used to collect and analyse the data. The chapter explains how we used the area-diversity predictions of the island biogeography theory to predict the effects of habitat fragmentation on tree species diversity. The chapter also outlines how we used remotely sensed data coupled with Markov chain modelling in a GIS to model historic as well as futuristic land cover dynamics.

Chapter 3 presents the results and discussion on the effects of habitat fragmentation on tree species diversity. It explores testing the area-diversity predictions of the island biogeography theory in an agricultural landscape.

Chapter 4 presents results on historic and projected land cover dynamics in the resettlement area based on remotely sensed data coupled with Markov chain modelling in a GIS.

Chapter 5 synthesizes the main findings of the thesis. In particular the chapter discusses the scientific contribution of the thesis in the context of biodiversity conservation in agricultural landscapes. We conclude this chapter by discussing how future studies can improve this thesis.

Chapter 2: Methods

2.1 Study area

The study was conducted in Nyabamba A1 resettlement area of Chimanimani district in the south-eastern part of Zimbabwe, located between 19°59' and 20°00' South and 32°49' and 32°50' East on an area covering 747,7ha. Figure 2.1 indicates the study area. In Zimbabwe, the A1 resettlement model is a communally-owned type of settlement meant to extend and improve the base for productive agriculture in the small-holder farming sector as well as eliminate squatting and other disorderly settlements in both urban and rural landscapes (Ministry of Lands, 2007).

Zimbabwe is classified into five agro-ecological regions based on capacity to support agriculture (Vincent and Thomas, 1962). The study area is located in agro-ecological region one with an average mean annual rainfall of 1000 mm (Vincent and Thomas, 1962). The climate is seasonal being characterized by a dry season spanning from May to September and a wet season from October to April. The average mean annual temperature of the area is between 17 and 21°C (Jones and Harris, 2008).

The elevation of the area ranges from 1600-1800 m above mean sea level. Due to the steep terrain of the area forestry used to be the dominant land use activity in Nyabamba. At present, wet miombo, characterized by small fragments of *Parinari curatellifolia*, *Erythrina abyssinica* and *Ficus sycomorus*, is now confined to a few remnant patches surrounded by an agricultural landscape. Prior to 2000 the area was *Acacia dealbata* plantation managed by the

Wattle Company of Zimbabwe as well as commercial *Pinus spp* and *Eucalyptus spp* plantations. This explains the existence of remnant plantation patches in the area. Small patches of exotic tree species such as *Acacia dealbata*, *Eucalyptus spp* and *Pinus spp* also exist in the area. Some tree species of societal importance (traditionally sacred species such as *Bridelia micrantha*, *Parinari curatellifolia*, *Pterocarpus angolensis* and *Syzigium cordatum*) are left standing when other tree species are cleared.

The soils of the area are predominantly orthoferallitic soils particularly Rhodic and Haplic ferralsols (FAO-UNESCO, 1988). These soils are brightly coloured due to high oxide content. Owing to their relatively poor nutrient status, together with the slopes characteristic of the area these soils are not used for normal cultivation and are largely taken up by forestry and the growing of tree crops especially tea and coffee (Nyamapfene, 1991). In addition, these ferralsols are poor soils, with a low ion exchange capacity and nutrient reserves that are easily disrupted by agricultural practices (FAO, 1993).

Most of the human settlement in Nyabamba dates back to 2000 when the most recent phase of land reform in Zimbabwe, the Fast Track Land Reform Programme (FTLRP) was initiated. Rapid land cover changes are taking place with most natural woodlands and wooded grasslands being converted into cropland and bare land around most parts of the country creating a new pathway for deforestation “hot spots” (Matsa and Murunganiza, 2011). At present, clearance for agricultural purposes and logging for fuel wood are some of the underlying causes of land cover conversions.

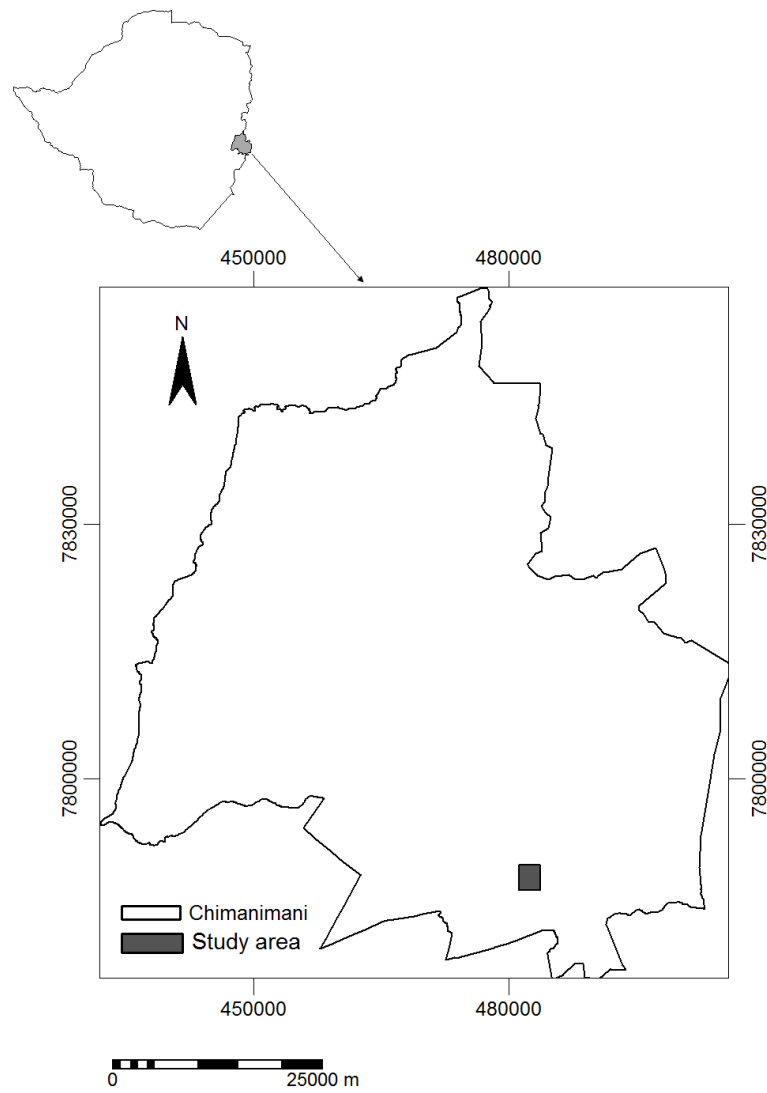


Figure 2.1: Location of the study area in Chimanimani district, Zimbabwe. Map coordinates are in WGS84 UTM Zone 36 South

2.2 Determining area-diversity dynamics

2.2.1 Sampling and tree species data collection

Prior to tree species diversity assessment in the field, we classified an IKONOS satellite image (www.geoeye.com) for the study area using the maximum likelihood classifier. We used the maximum likelihood classifier because of its robustness and the availability of sufficient training data which improves the classification result. Furthermore, the fine spatial, spectral and radiometric resolution of IKONOS image used in this study greatly reduces the mixed pixel problem, providing a greater potential to extract much more detailed information on land cover. The image had a spatial resolution of 4m and was acquired on the 6th of March 2009. Four dominant land cover types resulted from the classification namely cropland (predominantly maize (*Zea mays*)), plantation (dominated by *Acacia dealbata*, *Eucalyptus spp* and *Pinus spp*), woodland (dominated by *Parinari curatellifolia*, *Erythrina abessinica* and *Ficus sycomorus*) and wooded grassland (Figure 2.2).

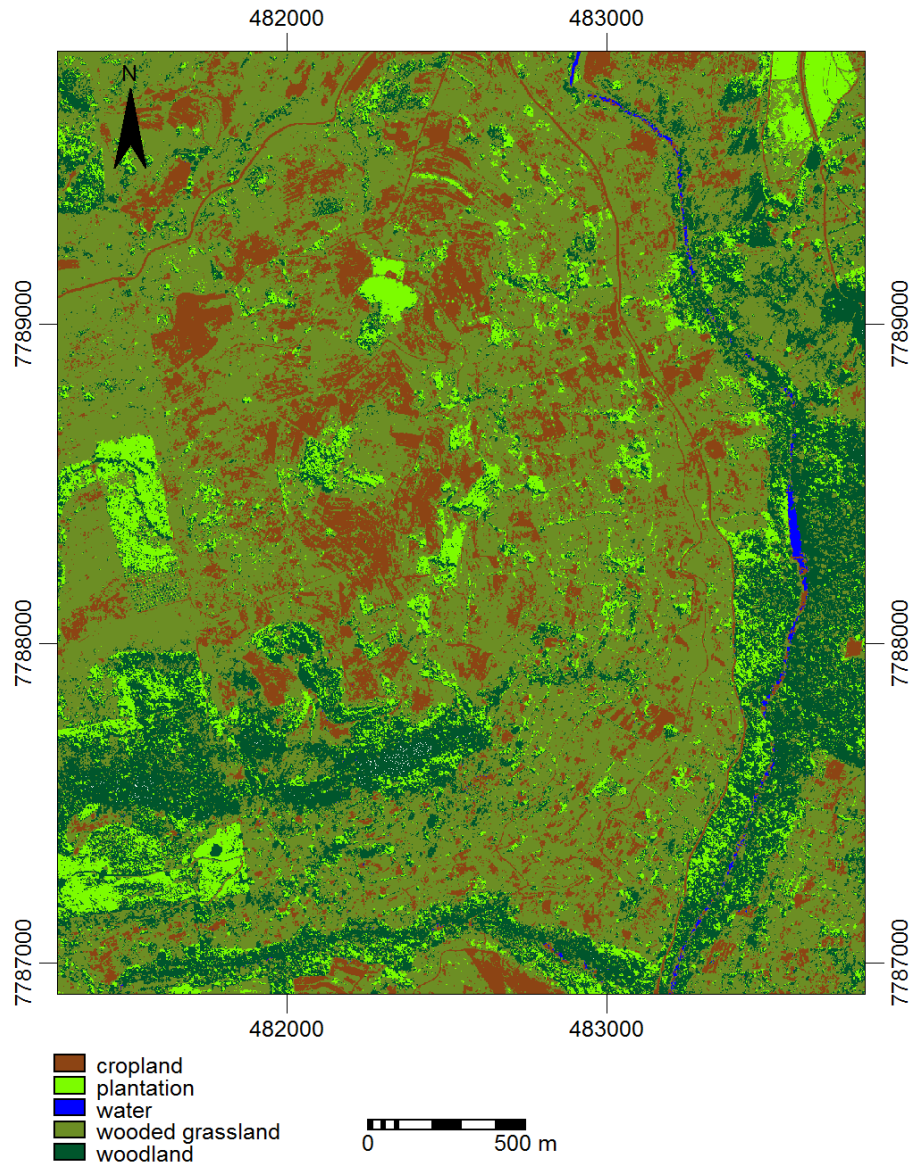


Figure 2.2: The different land cover classes in the study area

Post classification accuracy was carried out using ten independent ground control points per land cover class and yielded 84 % accuracy based on the Kappa statistic. We used the commonly used Kappa statistic because it provides a quantitative measure of the magnitude of agreement between observations. Class spectral separability was analysed using the Jeffries-Matusita separability measure where values that are less than 1.9 are considered not separable and separable otherwise. The Jeffries-Matusita is used to assess the potential of

band pairs to discriminate between two different region classes. All comparisons yielded values greater than 1.9 indicating that the land cover types have good separability. We then extracted woodland patches using a logical operator in ILWIS GIS. Based on the resultant woodland map, we used stratified random sampling to generate thirty random sampling points distributed equally across the study area for purposes of detailed field based measurement of tree species diversity (Figure 3). We classified the thirty woodland patches using the area numbering algorithm and came up with size ranges. The patches assumed three distinct size categories i.e. below 4 hectares, between 4-25 hectares as well as above 25 hectares. These categories were then used to classify individual patches into three patch sizes for subsequent data analysis. This clustering of tree species diversity in different sizes was done so as to avoid arbitrary classification of size classes. Under this sampling strategy ten (10) plots were selected in each of the patch size categories and were located at the edge of the patch as well as in the interior.

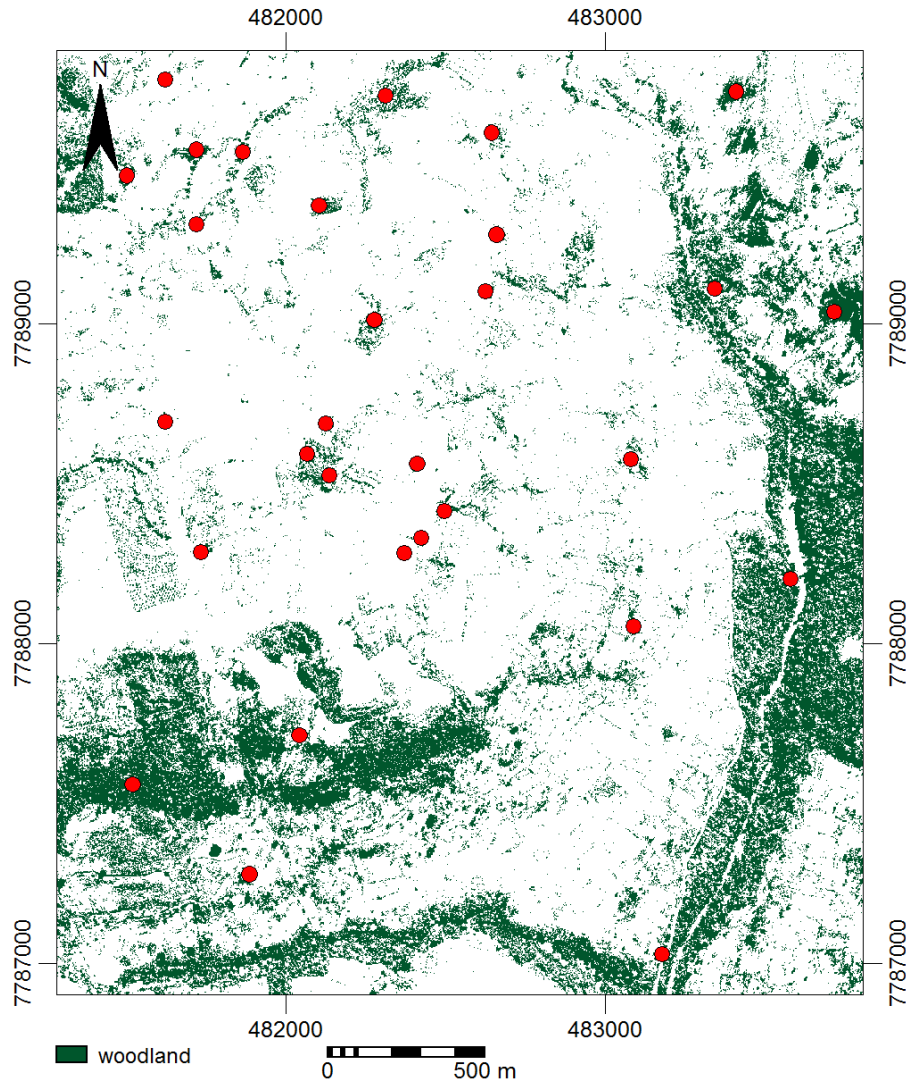


Figure 2.3: Woodland patches and sampling points (red circles)

We entered the thirty sampling points into a handheld Global Positioning System (GPS) unit as waypoints for fieldwork carried out in June 2010. The GPS was used to navigate to each sampling point. In each case the generated sample point became the centre of a 15 by 15 metre sampling plot. Within the sampling plots data on tree species composition and frequency of occurrence were measured in the sampling plot. The size of the plot (15m by 15m) was derived from the minimum species area curve. The 15m plot size was adapted from previous tree diversity surveys that used plot sizes ranging from 25 to 400 m² and at times , as

large as 25 000 m² for trees in large woodlands and forests (Mutowo and Murwira, 2012; Sutherland 1996). All tree species with a height $\geq 1.3\text{m}$ located within the plot were sampled and data recorded. The threshold of greater or equal to 1.3m on tree sampling was used in order to eliminate tree saplings and is also in line with FAO guidelines of measuring tree diameter (dbh) at 1.3m breast height above the ground since dbh was also collected in the field for later use. Identification of tree species that we failed to identify in the field was carried out by a qualified botanist at the National herbarium and botanical gardens in Harare using voucher samples collected during fieldwork.

2.2.2 Tree species diversity assessment

Diversity indices provide important information about rarity and commonness of species in a community. Species diversity was quantified using the Shannon Weiner, commonly referred to as the Shannon Weaver's index of diversity (H). The index relates diversity to the order of one and is the only diversity which can be consistently decomposed into meaningful independent alpha and beta components. It also has the advantage of favoring neither rare nor common species disproportionately; it counts all species according to their frequency. It is therefore the "fairest" index, weighting each species exactly by its frequency in the sample. The Shannon Weaver Index (H) is a measure of the average 'uncertainty' of predicting to what species an individual chosen at random from a collection of species belong. This average uncertainty increases as the number of species increases, and as the distribution of individuals among the species become more even. The Shannon Weaver index, $H = 0$ if and only if there is one species, and reaches a maximum when all species present are represented by the same number of individuals (perfect even distribution). It is calculated using the formula:

$$H = -\sum P_i (\ln P_i) \text{ (Equation 1)}$$

where P_i is the proportion of each species in the sample.

Diversity indices such as the Shannon Weaver index are not themselves diversities, they are just indices of diversity. Converting indices to true diversities (effective numbers of species) gives them a set of common behaviours and properties. After conversion, diversity is always measured in units of number of species, no matter what index and this facilitates interpretation of results. We converted the indexes to true diversity using the formula:

$$D = \text{Exp} (H) \text{ (Equation 2)}$$

where H is the Shannon Weaver index.

We then tested the tree species diversity data for normality using the Komolgorov Sminorv test. The Kolmogorov-Smirnov test is used to decide if a sample comes from a population with a specific distribution. An attractive feature of this test is that the distribution of the Komolgorov Sminorv test statistic itself does not depend on the underlying cumulative distribution function being tested. Another advantage is that it is an exact test (the chi-square goodness-of-fit test depends on an adequate sample size for the approximations to be valid). Results of the normality test indicated that the data conform to a normal distribution curve ($P > 0.05$) thus parametric tests were used for subsequent data analysis. Explanatory Data Analysis (EDA) on area-diversity relationships was then carried out. To test the area-diversity hypothesis descriptive analysis was carried out on the three patch sizes to derive the mean tree species diversity of each patch size category. One Way Analysis of Variance (ANOVA) and Least Significant Difference (LSD) Post Hoc multiple comparisons were done

to test for any significant differences in tree species diversity within as well as between the patch sizes.

2.3 Modelling historic and futuristic land cover dynamics

2.3.1 Land cover classification

In a Geographic Information Systems (GIS) environment, we classified two Landsat TM satellite images obtained from United States Geological Survey (<http://glovis.usgs.gov/>) for the study area acquired on the 25th of April 2000 and 26th of April 2009 respectively. Four dominant land cover types i.e. cropland, plantation, wooded grassland and woodland were classified during this process using the maximum likelihood algorithm. Post classification accuracy was carried out using 1:50000 topographical maps and Google Earth as reference data for the 2000 image and an IKONOS image obtained from GEOEYE (www.geoeye.com) for the 6th of March 2009 as a reference image for the 2009 classification. In both instances ten (10) reference points per land cover class were used for accuracy assessment. Accuracy assessment for the 2000 classification was 78% whilst the 2009 image yielded 73% accuracy based on the Kappa statistic. We then separated all the land cover types and came up with individual land cover parcels for all the land cover types for subsequent analysis.

2.3.2 Environmental variables

Land cover change can be explained by several environmental factors. In this study we considered and used GIS to map distance from homesteads, distance from rivers, distance from roads, soil type as well as slope as possible drivers of land cover change. Specifically, homesteads, rivers and roads were digitised in Google Earth and co-registered with the land

cover data in ILWIS GIS. Distance from rivers, distance from roads and distance from homesteads were derived in ILWIS GIS using the distance calculation functionality. The distance calculation function assigns to each pixel the smallest distance in meters towards user-specified source pixels, for example in our case distance to homesteads, roads as well as rivers. The slope data was calculated from a 30m ASTER Digital Elevation Model (DEM) of the area obtained from www.earthexplorer.usgs.gov. The soil map of the area was derived from the (FAO-UNESCO, 1988) classification.

Homesteads affect changes in land cover in the sense that most changes are made to satisfy the basic demands of the households for example clearance of trees to open up land for agricultural fields, cutting of trees for wood fuel and construction purposes. Roads tend to open up corridors for human movement facilitating conversion of woodlands into grassland and other forms of land cover. Slope steepness dictates the possibility of resource exploitation. Steeper slopes are less likely to be cultivated or cleared for wood fuel and other woody products than gentler accessible slopes. Land cover change varies according to soil type; clearance for cultivation purposes is more likely to be undertaken on fertile soils than on poor soils. Water courses such as rivers regulate deforestation and cropland practices, based on institutional regulations at play in most areas, deforestation and cropland are likely to be carried out with distance away from the river channel.

We also used logistic regression to map cropland expansion and deforestation. In order for the cropland and woodland maps to be used in logistic multiple regression to determine cropland expansion and deforestation, land cover change maps were reclassified into binary variable maps using the map calculator function in ILWIS GIS. Areas where change occurred were given 1 while those without change were given 0. The soil map was categorized from a raw soil map into binary data for the two soil types Haplic ferrasols and Rhodic ferrasols.

This was done to create presence and absence maps ideal for logistic regression modelling that requires reclassification of each individual condition of interest. This means that for each soil type zero values in the input map are regarded as 0 (false), and all other defined values as 1 (true).

2.3.3 Modelling environmental links resulting in deforestation and cropland expansion

All the environmental variables determined above were analyzed using logistic regression in SPSS based on the Wald forward algorithm to determine which variables significantly ($P < 0.05$) explain land cover dynamics in the study area particularly deforestation and cropland expansion. Our null hypothesis being there is no significant relationship between deforestation or cropland expansion with the environmental factors i.e. distance from homesteads, distance from rivers, distance from roads, slope as well as soil type. The binary variable maps were treated as dependent variables. The environmental factors were used as independent variables. In a GIS, logistic regression of environmental factors was carried out. Our spatial predictions are futuristic and based on the assumption that land cover change dynamics are a function of various environmental factors. Based on logistic regression, we came up with two models: one for predicting deforestation and the other for predicting the likelihood of cropland expansion. Using the image calculator in IDRISI Andes we performed a logical query by land cover attributes to create maps of change (1) and no change (0) for woodland and cropland between 2000 and 2009. The change and no change maps were in binary format because logistic regression is one modelling technique that requires such a presence and absence format. Each binary variable map was treated as a dependent variable whilst the environmental factors were treated as independent variables during logistic regression. Only the independent environmental factors that had shown a significant relationship with either deforestation or field expansion were included in each of the models.

2.3.4 Non spatial predictions

The process of land cover change was modelled by a Markov chain to predict land cover distributions in the future. Numerous studies have demonstrated the utility of the Markov

chain model in land cover change modelling (Baltzer, 2000; Coppedge *et al.*, 2007; Fuller and Hardiono, 2011; Guan *et al.*, 2008; Kamusoko *et al.*, 2009; Khoi and Murayama, 2011). Markov chain models have a relatively simple and intuitive logic that makes them attractive alternatives to more complex formulations of stochastic land use models. Markov chains are able to describe the complex and long-term process of land use conversion in terms of simple transition probabilities, making them a potentially useful planning tool. In this study, the Markov model was used to compliment the spatial predictions done based on logistic regression. Markov models occupy an intermediary position between stochastic and deterministic models as they are developed in the structure of probabilities and the chain sequence is deterministic in nature. The probabilistic nature of Markov chains helps in simulating the complex nature of land cover dynamics. Markov models have substantial scientific appeal as they are mathematically compact, easily developed from observed data and serve as an effective tool for simulation exercises. Markov models are relatively easy to derive from successional data. The Markov model does not require deep insight into the mechanisms of dynamic change, but it can help to indicate areas where such insight would be valuable and hence act as both a guide and stimulant to further research.

2.3.5 Land cover change determination for 2018 and 2027

For the scope of this work, the Markov process is considered to have discrete states in the form of four land cover classes. Woodland, wooded grassland, plantation and cropland were the adopted finite states that transition at discrete nine year intervals. Futuristic predictions of land cover up to 2027 were done in IDRISI Andes. First we performed a cross tabulation of the 2000 and 2009 images to derive cross classification and tabulation of the areas that had changed as well as the direction of change between 2000 and 2009. This also yielded our

initial state vector, the state of our system in 2009. Using the Markovian transition estimator function we created transition probability matrices for 2018 as well as conditional probability maps of transitioning into a particular land cover type in 2018 based on the 2000 and 2009 images. Transition probability matrices are the basic framework of a Markov model and describe the probability (\mathbf{P}) of movement from one state to any other state during a specified or discrete time interval. We can obtain \mathbf{P}^1 (the state of the system in 2018) by multiplying the initial state vector $\mathbf{P}^{(0)}$ by the probability transition matrix \mathbf{P} (derived by crossing the 2000 image with the 2009 image). If we assume that the conditions under which land cover change occurs remain the same then we can determine the distribution of land cover for \mathbf{P}^2 (the state of land cover in 2027) by multiplying \mathbf{P} by \mathbf{P}^1 .

Next we used the Land Change Modeler for Ecological Sustainability for our land cover simulations. Using the change analysis sub functionality we derived statistics on gains and losses per land cover category as well as change maps showing persistence, gains and losses per land cover class between 2000 and 2009. Land cover transitions and exchanges between the land cover classes as well as the spatial trend of change were also mapped in the process. In the transition potentials utility we factored in the driver variables i.e. distance from homesteads, distance from rivers, distance from roads, slope and soil types to explore the potential power of explanatory variables. Futuristic transition potentials from one land cover type were then determined using the Multi Layer Perception (MLP) neural network classifier which allows for modelling multiple transitions. Under the change prediction sub functionality Markov chain modelling was used to generate future predictions for land cover i.e. 2018 and 2027 based on the soft prediction model as it yields a map of vulnerability to change for the selected set of transitions and provides a comprehensive assessment of change potential.

To establish if our land cover change process was of 1st order and had the Markovity property we used the maximum likelihood ratio criterion ($-2 \ln \lambda$ statistic) test. The test determines whether our knowledge of the state of the system in 2009 and the probability of moving from one land cover class to any other was sufficient to estimate the state of the system in each successive nine year interval. The test statistic ($-2 \ln \lambda$) is obtained through the following formula:

$$-2 \ln \lambda = 2 \sum_{i,j}^n f_{ij} \ln \frac{p_{ij}}{p_j} \quad (\text{Equation 3})$$

where p_j = column total as proportion of the total number of pixels

p_{ij} = probabilities of transition

f_{ij} = the number of observations in each cell

Table 2.1 shows the results of the Markovity test.

Table 2.1: Significance tests for first order Markov property (Markovity) ($\alpha = 0.05$)

Change likelihood period	$-2 \ln \lambda$	χ^2	Degrees of freedom (n-1)2
2009-2018	436.916	16.92	9
2009-2027	1086	16.92	9

Our calculated chi-square distribution was compared to the tabled chi square at 0.05 significance level and 9 degrees of freedom. Our test indicates that the calculated chi square is greater than the tabled chi square for the period up to 2027. Therefore we assume that the land cover data in all the transition periods satisfies the Markov chain assumption of dependency and future land cover change can be simulated using Markov chain modelling.

RESULTS AND DISCUSSION

Chapter 3: Woodland fragmentation explains tree species diversity in an agricultural landscape of Southern Africa

3.1 Results

3.1.1 Land cover types in the study area

The proportion covered by each of the four land cover types in the study area is illustrated in Figure 2.2. It can be observed that wooded grassland is the major land cover type found in Nyabamba resettlement area. Wooded grassland covers 55.9% (417.8 hectares), woodland 19.6% (146.5 hectares) cropland 14.6% (108.8 hectares) while plantation is 9.7% (72.7 hectares) of the total area. Water covers 0.2% of the area (1.8 hectares).

3.1.2 Tree species composition

Several indigenous and exotic tree species were observed in the study area with the indigenous species being the most dominant. Specifically, there were twenty-two indigenous tree species and seven exotic tree species (Table 3.1). The frequent tree species found in the area include *Acacia dealbata* (925), *Albizia adianthifolia* (76), *Annona senegalensis* (54), *Erythrina abyssinica* (88), *Parinari curatellifolia* (59), *Psidium guajava* (60) and *Trema orientalis* (59).

Table 3.1: A list of the tree species per patch size category. Numbers in brackets indicate the number of trees found.

Patches below 4ha	Patches between 4-25ha	Patches above 25ha
<i>Acacia dealbata</i> (627)	<i>Acacia dealbata</i> (73)	<i>Acacia dealbata</i> (225)
<i>Erythrina abessinica</i> (63)	<i>Parinari curatellifolia</i> (35)	<i>Albizia adianthifolia</i> (64)
<i>Trema orientalis</i> (17)	<i>Trema orientalis</i> (31)	<i>Syzigium cordatum</i> (52)
<i>Annona senegalensis</i> (13)	<i>Psidium guajava</i> (22)	<i>Rhus chirindensis</i> (40)
<i>Bridelia micrantha</i> (6)	<i>Erythrina abessinica</i> (21)	<i>Psidium guajava</i> (37)
<i>Persea americana</i> (5)	<i>Brachystegia spiciformis</i> (20)	<i>Annona senegalensis</i> (30)
<i>Cussonia kirkii</i> (5)	<i>Rhus chirindensis</i> (15)	<i>Bridelia micrantha</i> (27)
<i>Combretum apiculatum</i> (1)	<i>Bridelia micrantha</i> (14)	<i>Parinari curatellifolia</i> (24)
	<i>Ficus sycomorus</i> (13)	<i>Pterocarpus angolensis</i> (13)
	<i>Combretum apiculatum</i> (13)	<i>Trema orientalis</i> (11)
	<i>Macaranga capensis</i> (12)	<i>Cussonia kirkii</i> (11)
	<i>Uapaca kirkiana</i> (11)	<i>Vangueria infausta</i> (10)
	<i>Annona senegalensis</i> (11)	<i>Macaranga capensis</i> (9)
	<i>Mangifera indica</i> (9)	<i>Burkea Africana</i> (8)
	<i>Persea americana</i> (7)	<i>Mangifera indica</i> (7)
	<i>Cussonia kirkii</i> (7)	<i>Julbernardia globiflora</i> (7)
	<i>Albizia adianthifolia</i> (7)	<i>Ficus sycomorus</i> (6)
	<i>Vangueria infausta</i> (5)	<i>Combretum apiculatum</i> (5)
	<i>Heteropyxis dehniae</i> (3)	<i>Heteropyxis dehniae</i> (5)
	<i>Anthocleista grandiflora</i> (2)	<i>Eucalyptus spp</i> (4)
	<i>Jacaranda mimosifolia</i> (1)	<i>Erythrina abessinica</i> (4)
	<i>Catha edulis</i> (1)	<i>Anthocleista grandiflora</i> (3)
	<i>Pinus spp</i> (1)	<i>Brachystegia spiciformis</i> (1)
	<i>Strychnos cocculoides</i> (1)	

3.1.3 Tree species diversity as a function of patch size

Tree species diversity in the woodland patches found in the study area ranges from 2.74 to 5.28 (true diversity based on the Shannon Weaver index) as illustrated in Figure 3.1. Patches with sizes between 4-25ha have relatively higher mean tree species diversity (5.28) compared with patches greater than 25ha (4.65) and patches less than 4ha (2.74).

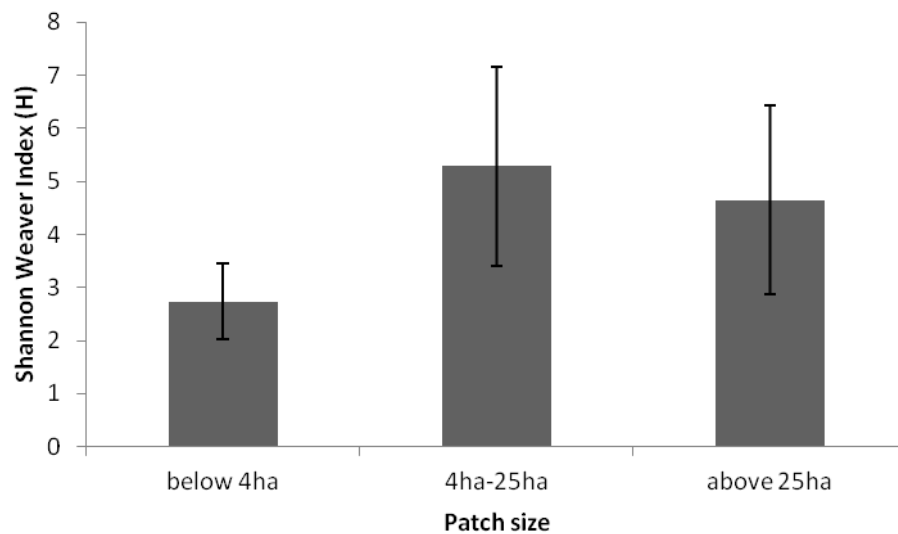


Figure 3.1: Mean tree species diversity distribution. Whiskers show 95 % confidence interval

It can be observed that tree species diversity differs significantly between the different patch size classes (ANOVA $F = 4.41$, $P = 0.022$). Specifically, there is a significant differences in mean tree species diversity between patches less than 4ha and patches between 4-25ha (Mean difference = -1.976, $P = 0.017$). Furthermore, we also observe that tree species diversity does not differ significantly between patches less than 4ha and patches greater than 25ha (Mean difference = -1.35, $P = 0.08$) as well as between patches whose size is between 4-25ha and patches greater than 25ha (Mean difference = 0.62, $P = 0.42$).

3.2 Discussion

Based on the island biogeography theory, patch area and secondarily isolation have traditionally been considered the major variables indicating the species diversity of a patch. The basic island area effect, though is mainly due to habitat diversity, in most cases, larger islands simply have more habitats which therefore support more species. Results of this study indicate that tree species diversity varies significantly between small and large woodland patches thereby confirming the area-diversity predictions of the island biogeography theory that high species diversity is associated with large patch sizes and low species diversity with small patches (MacArthur and Wilson, 1967). Similar studies (Echeverria *et al.*, 2007; Hill and Curran, 2003; Lindborg *et al.*, 2012; Raghubanshi and Tripathi, 2009) found significant effects of patch size on tree species diversity. Larger woodland patches are able to support a greater number of individual tree species compared with smaller patches, and thus are also likely to have species with larger population sizes (Kisel *et al.*, 2011). Conversely, species diversity in small patches is relatively low as small patches contain less habitats resulting in small tree species populations that are prone to density-dependent stochastic extinction processes (Honnay *et al.*, 1999). Thus, we deduce that results for this particular study are consistent with the area-diversity predictions of the island biogeography theory.

The result in this study that woodland patches larger than 4ha had significantly higher tree species diversity than woodland patches of less than 4ha suggests that leaving remnant woodland patches of greater than 4ha in the landscape could improve chances of promoting agricultural production and biodiversity conservation in resettled landscapes. These results are consistent with the observations of Blann (2006) who found that remnant habitat patches are responsible for maintaining biodiversity currently present in agricultural landscapes. Thus, we make a claim that the maintenance of woodland patches at sizes greater than 4ha

could promote tree species diversity in agriculturally fragmented landscapes although we should caution that this might be true for areas more or less the same size as our study area (747,7ha).

Results of accuracy assessment performed on the classified images indicate a relatively high level of classification accuracy as shown by a Kappa statistic of 76 %. In addition, the Jeffries-Matusita separability measure yielded values greater than 1.9 indicating that the land cover types have good separability. Both measures of accuracy and spectral separability used in this study were high and thus considered appropriate for testing island biogeography theory on patches of different sizes. Although, research has indicated that non-parametric classifiers such as neural network and decision tree classifier may provide better classification results than parametric classifiers (Lu and Weng, 2007), we used the maximum likelihood classifier because of its robustness and the availability of sufficient training data which improves the classification result. Furthermore, the fine spatial, spectral and radiometric resolution of the IKONOS image used in this study greatly reduces the mixed pixel problem, providing a greater potential to extract much more detailed information on land cover.

Where our study differs from previous studies (Lampila *et al.*, 2005; Ruiz-Gutierrez *et al.*, 2010; Vergara and Armesto, 2009; Collinge and Forman, 1998; Ockinger *et al.*, 2009; Braschler *et al.*, 2009; Ducheyne *et al.*, 2009; Browsers and Newton, 2009) is in testing whether woodland fragments in an agriculturally fragmented landscape conform to the predictions of the island biogeography theory specifically borrowing on the area diversity prediction and not the entire framework. While our findings may not be surprising, it is significant to note that studies that have successfully used the island biogeography theory to explain the impact of habitat fragmentation on biodiversity have mainly focused on migratory

species and not terrestrial ones such as trees. However, we have to caution that the results of this study are based on a study area of a smaller spatial extent (747,7ha) and specific geographic location. Further studies may need to be conducted to find out if the island biogeography area-diversity prediction is applicable at large spatial extent and in different locations.

Chapter 4: Modelling past, present and future spatial dynamics in land cover in a resettlement area in Zimbabwe

4.1 Results

4.1.1 Land cover dynamics 2000-2009

Figure 4.1 illustrates the major land cover types found in Nyabamba for the year 2000 and 2009 respectively. For both time periods wooded grassland formed the dominant land cover type in the area covering 294.12ha in 2000 and 317.79ha in 2009. In the year 2000 cropland constituted 183.78ha and increased to 276.66ha in 2009. Woodlands made up 124.74ha in 2000 but decreased to 84.78ha in 2009. Plantations amounted to 105.84ha in 2000 but were reduced to 29.25ha in 2009.

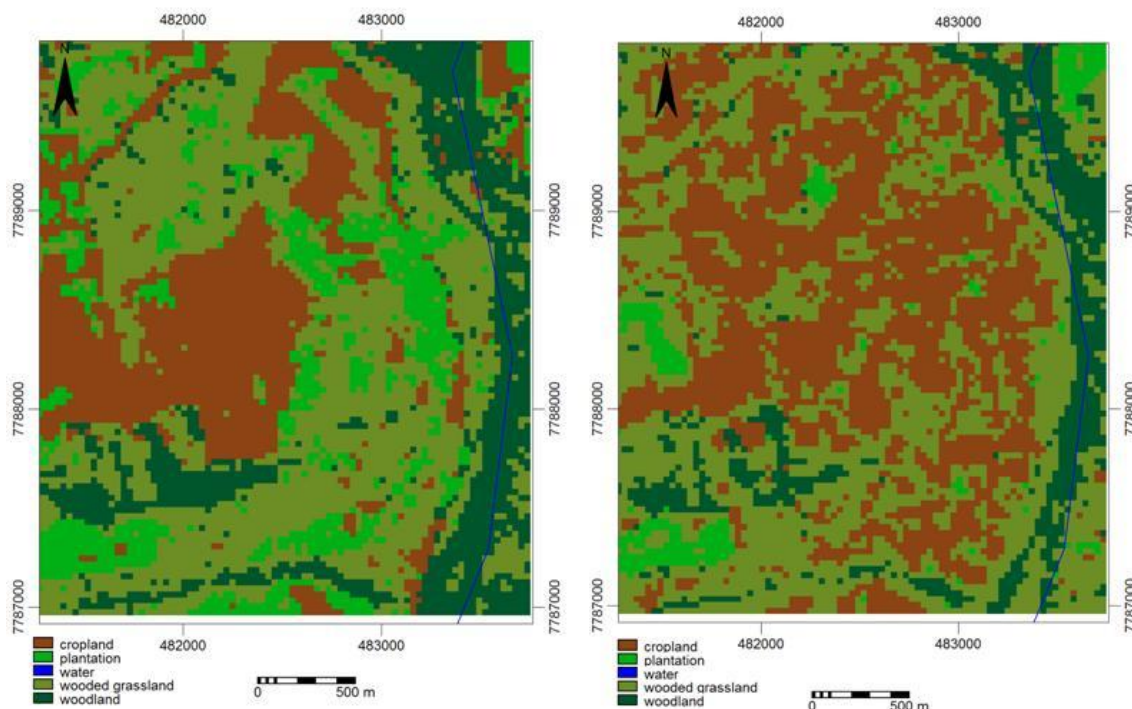


Figure 4.1: Nyabamba land cover classes for 2000 (left) and 2009 (right).

Both positive and negative land cover changes were experienced between 2000 and 2009. Table 4.1 illustrates the gains (positive change); losses (negative change) as well as the persistence per land cover type. Percentage change from the total study area is also shown in the table.

Table 4.1: Gains and losses per land cover class (2000-2009)

Land cover class	Gains	Losses	Persistence
Woodland	12.26ha (1.7%)	52.56ha (7.4%)	72.18ha (10.2%)
Plantation	16.56ha (2.3%)	93.15ha (13.1%)	12.69ha (1.8%)
Cropland	179.01ha (25.3%)	86.13ha (12.2%)	97.65ha (13.8%)
Wooded grassland	156.87ha (22.1%)	133.2ha (18.8%)	160.92ha (22.7%)

Cropland gained the most between 2000 and 2009, 179.01ha in total gained mainly from wooded grassland (120.6ha). Woodland experienced the least gains (12.26ha) during this period with the greater of these coming from wooded grassland (7.2ha). Wooded grassland experienced the largest losses for the period, 133.2ha in total whilst woodlands had the smallest loss of 52.56ha. In terms of persistence wooded grassland had the highest amount of area that did not change (160.92ha) whilst plantation had the smallest area (12.69ha).

4.1.2 Relationship between environmental factors and land cover change

Tables 4.2 and 4.3 show the results of the logistic regression relating spots that were deforested between the year 2000 and 2009 to environmental factors i.e. distance from homesteads, distance from rivers, distance from roads, slope as well as soil type. Table 4.2 shows environmental factors that showed a statistically significant relationship with deforestation.

Table 4.2: Environmental factors that showed statistically significant relationships with deforestation 2000-2009 ($\alpha = 0.05$)

Environmental factor and constant	Slope(β_1)	Odds ratio (eβ)	<i>p-value</i>
Distance from rivers	.001	1.001	.007
Constant (β_0)	-1.248	.287	.025

Our results show that there is a significant positive ($P < 0.05$) relationship between deforestation and distance from rivers.

Table 4.3 shows factors that had no significant ($P < 0.05$) relationship with deforestation based on logistic regression of environmental factors.

Table 4.3: Environmental factors that had no significant relationship with deforestation ($\alpha = 0.05$)

Environmental factor	Slope(β_1)	<i>p-value</i>
Distance from homesteads	1.038	.308
Distance from roads	.58	.810
Slope	.619	.431
Soil type	1.528	.216

Our results show that there is no significant ($P>0.05$) relationship between deforestation and distance from homesteads, distance from roads, slope as well as soil type.

Tables 4.4 and 4.5 show the results of logistic regression between cultivated areas (between 2000 and 2009) and environmental factors. Table 4.4 shows factors that have a significant ($P<0.05$) relationship with cropland expansion.

Table 4.4: Environmental factors that showed significant relationships with cropland expansion ($\alpha = 0.05$)

Environmental factor and constant	Slope(β_1)	Odds ratio (e^{β})	<i>p-value</i>
<i>Rhodic ferralsols</i>	-1.797	.166	.009
<i>Rhodic ferralsols</i> Constant (β_0)	2.159	8.667	.000
<i>Haplic ferralsols</i>	1.797	6.029	.009
<i>Haplic ferralsols</i> Constant	.363	1.438.	.265

There is a significant ($P<0.05$) relationship between cropland expansion and soil types as shown in Table 4.4. We observe that there is a significant ($P<0.05$) positive relationship between Haplic ferralsols and cropland expansion as well as a significant ($P<0.05$) negative relationship between Rhodic ferralsols and cropland expansion. 207.63 hectares of cropland areas are located on Haplic ferralsols and 69.03 hectares are on Rhodic ferralsols.

Table 4.5 illustrates the environmental factors that had no significant ($P < 0.05$) relationship with cropland expansion.

Table 4.5: Environmental factors that had no significant relationship with cropland expansion ($\alpha = 0.05$)

Environmental factor	Slope(β 1)	<i>p-value</i>
Distance from homesteads	1.502	.220
Distance from roads	.241	.624
Distance from rivers	.275	.600
Slope	1.444	.229

Our results illustrate that there is no significant ($P > 0.05$) relationship between cropland expansion and distance from homesteads as well as distance from roads, distance from rivers and slope.

4.1.3 Spatial predictions for deforestation and cropland expansion using 2000-2009 land cover data

Figure 4.2 shows the spatial predictions for probability of deforestation and cropland expansion based on logistic regression using 2000 to 2009 land cover data. Our results show that more areas are likely to be expanded into cropland than will be deforested. Deforestation is marked by a north eastern-south eastern trend whereas cropland expansion has a central-south western trend.

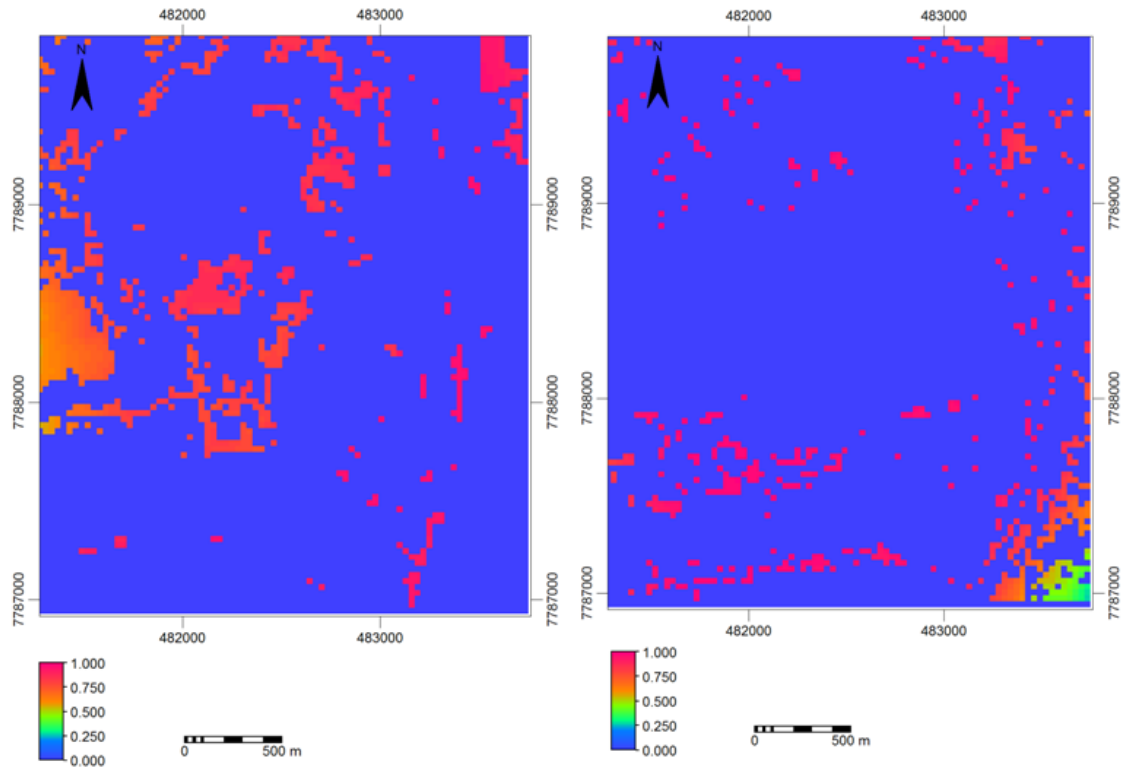


Figure 4.2: Spatial prediction of cropland expansion and deforestation

4.1.4 Future dynamics of land cover (2018-2027)

Our predictions of future land cover illustrate that wooded grassland will continue to be the dominant land cover class in the area covering 324.36ha in 2018 and 325.98ha in 2027. Our 2018 land cover data shows that woodland and plantation will continue decreasing whilst wooded grassland and cropland increase. Results project that in 2018 woodlands would have decreased by 18.27ha from the area they covered in 2009 whilst plantations will decrease by 3.69ha. Results also show that wooded grasslands will increase their 2009 area by 6.57ha in 2018 whilst cropland is expected to increase by 15.39ha.

Figure 4.3 illustrates land cover gains between 2009 and 2018.

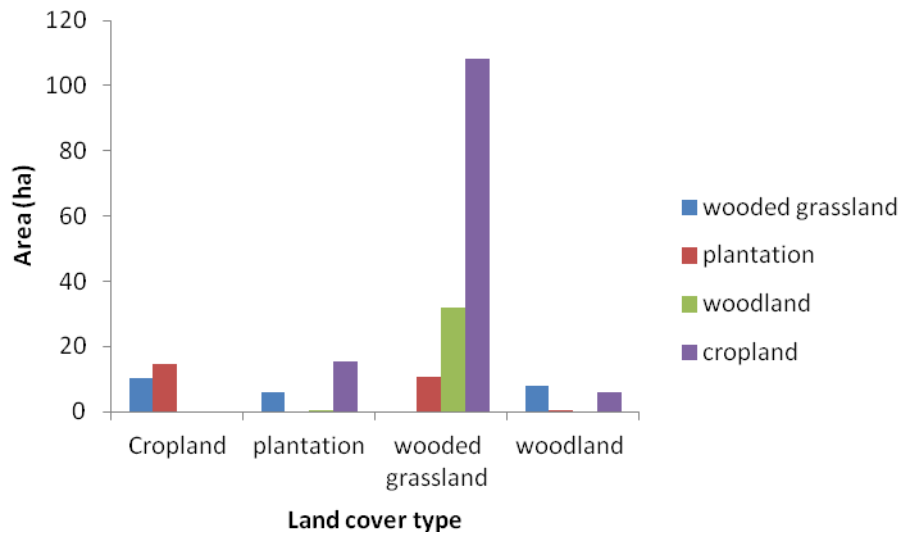


Figure 4.3: Land cover gains per land cover type between 2009 and 2018

Results show that wooded grassland will gain the most especially from cropland (108.27ha). Woodland on the other hand will gain the least, with the greater part of these gains coming from wooded grassland (7.83ha).

Projecting further to 2027 results illustrate that woodlands will decrease by 15.39ha from their 2018 figure of 66.51ha to cover 51.12ha. Plantations will increase by 0.45ha to amount to 26.01ha. 13.32ha of land will be converted to cropland in 2027 so that the area under cropland increases to 305.37ha. Wooded grasslands will increase by 1.62ha between 2018 and 2027. Figure 4.4 highlights the potential land cover transitions between the different land cover types for the period 2018 to 2027.

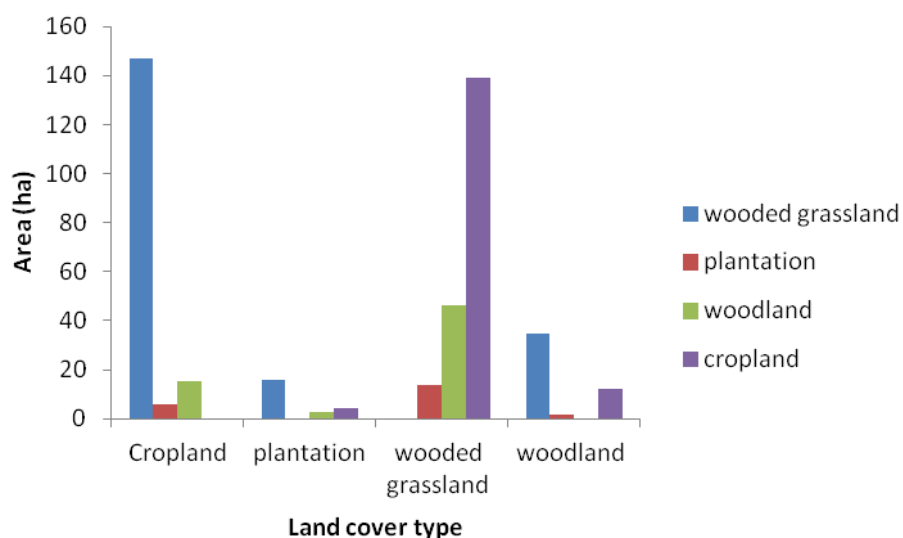


Figure 4.4: Land cover gains per land cover type between 2018 and 2027

Results show that wooded grassland will gain the most from the land cover transitions, 199.08ha in total with most of it coming from cropland (139.14ha). Plantation on the other will gain the least (22.86ha) with most transition coming from wooded grassland (15.93ha).

Figure 4.5 shows the area covered by each of the four land cover types from 2000 to 2027. Our results show that woodland and plantation decrease whilst wooded grassland and cropland increase in terms of total area covered.

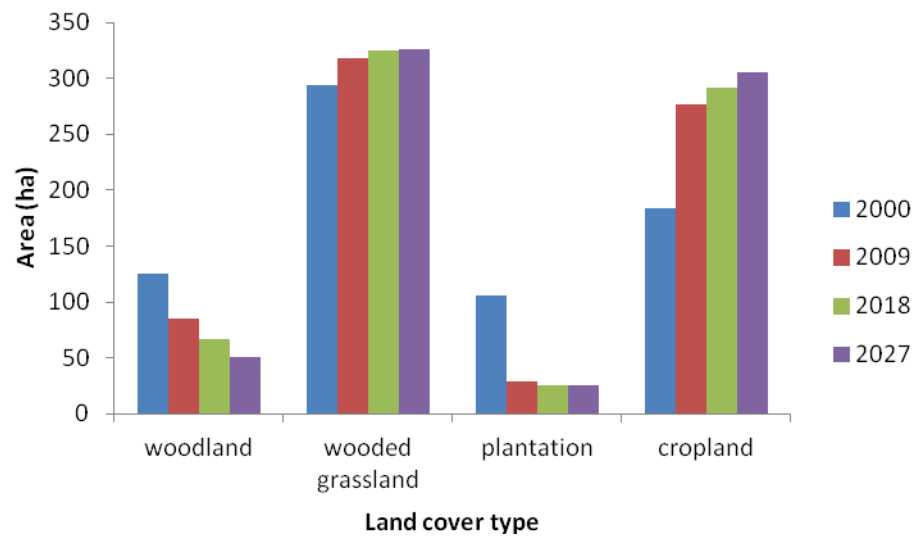


Figure 4.5: Land cover changes (2000-2027)

4.2 Discussion

Our analysis of historic land cover indicated an increase in cropland and wooded grassland accompanied by a simultaneous decrease in plantation and woodland between 2000 and 2009. Similar trends have been documented in several other studies. For example, a study by Chigumira (2010) observed significant losses in woodland and bushland as well as an increase in cultivation and grassland between 2002 and 2008 in three resettlement farms in Kadoma, Zimbabwe. Also, Elliot *et al* (2006) document similar trends of increases in cropland and decreases in woodland in Mupfurudzi, Sengezi and Tokwe resettlement areas. Thus, we claim that agriculture is a major driver of changes in woodland in the study area.

An analysis of the probability of occurrence of deforestation and cropland expansion shows that a greater part of Nyabamba is likely to be converted into cropland in future. These results are similar to those of Petit *et al* (2001) who projected an increase in cultivated land and a rapid decline in forests and other natural vegetation cover types in Lusitu, Zambia. The direction and magnitude of future land cover conversions is not surprising as it conforms to the fact that when people were resettled in the area it was mainly for agricultural purposes. In this regard, most of the land will be opened up into cropland. Agriculture competes for land with forestry, it then follows that deforestation will take place in areas of increased agricultural activity (Benhin, 2006).

Our results also indicate that there is a significant positive relationship between deforestation and distance from rivers suggesting less exploitation of woodlands close to rivers compared with those further away from rivers. In order to explain this relationship it may be important to take into account institutional controls on land use and natural resources management (Lambin *et al.*, 2003). Although this was not the main focus of this study, our result may indicate the effects of the Environmental Management Act of 2002 that prohibits any form of

cropping at a distance of 30m from a river channel. In addition, local traditional and cultural norms in place in Nyabamba may have partially led to the observed pattern in deforestation. For example, traditional rituals and ceremonies are performed at sacred places along these rivers and as such stiff fines and penalties are imposed on offenders reportedly curbing deforestation and agricultural practices. Traditionally protected areas such as shrines along rivers contain traditionally sacred trees such as *Bridelia micrantha*, *Parinari curatellifolia*, *Pterocarpus angolensis* and *Syzigium cordatum*. Thus areas close to rivers tend to be protected and remain relatively well forested.

We found a significant relationship between cropland expansion and soil types. Clearance for cropland purposes tends to be undertaken on fertile soils than on poor soils. In Nyabamba we found that cropland activities are mostly located on Haplic ferralsols (which are fertile) with a less fields found on Rhodic ferralsols (which have poor fertility). Ferralsols in general are deep, strongly weathered soils that are chemically poor, but physically stable subsoils which are suited to tree crops such as coffee and are poor for agricultural practices such as maize production (FAO/IIASA/ISRIC/ISS-CAS/JRC, 2008). The less preferred Rhodic ferralsols contain rhodium as well as iron and aluminium which are not conducive to crop growth. According to Foth and Schafer (1980), these soils could be productive after proper management techniques that involve extensive use of fertilizers. Heavy rates of lime are often required to reduce aluminum toxicity. We claim that farmers in the area go for Haplic ferralsols that are easier and less expensive to work on.

Furthermore, projections of future land cover based on Markov chain analysis indicated an increasing trend of wooded grassland and cropland increase in terms of total area covered and

a decrease in woodland and plantation in Nyabamba up to 2027. We are confident in the use of a single change prediction based on the test for Markovity that we carried out that shows that our land cover change process was of 1st order and had the Markovity property. The test also determines that our knowledge of the state of the system in 2009 and the probability of moving from one land cover class to any other was sufficient to estimate the state of the system in each successive nine year interval.

Where our study differs from previous studies is in the application of a GIS coupled-Markov chain modelling framework in a resettled agricultural landscape to understand the nature, extent and pattern of these conversions. Most studies that have applied stochastic models such as Markov models to simulate future land cover conversions have done so outside of agricultural landscapes. Markov modelling has been used extensively in the study of urban environments (Attua and Fisher, 2011; Guan *et al.*, 2008; Falahatkar *et al.*, 2011; Rimal, 2005) and forest systems (Fuller and Hardiono, 2011; Kamusoko *et al.*, 2011; Khoi and Murayama, 2011) but rarely (Petit *et al.*, 2001; Kamusoko *et al.*, 2009) in modelling land cover conversions in agricultural landscapes such as newly resettled agricultural areas. In addition, the Markov chain model, as applied in this study, does not account for neighbor effects. That is, land use in a particular location may be influenced not only by its previous land use, but also by the land uses of its neighbors. This principle has been incorporated into other types of cellular models of land use, such as cellular automata, which model land use as a function of the states of cells in a defined neighborhood. This study has shown the utility of combining remotely sensed data and stochastic models in monitoring land cover conversions and the influence of underlying environmental factors to explain land cover conversions. The futuristic perspective helps in building proactive environmental management programmes.

SYNTHESIS

Chapter 5: Land cover and tree species diversity changes in a newly resettled agricultural landscape: A synthesis

5.1 Introduction

Agricultural land expansion into woodland areas is a major driver of biodiversity changes (IUCN, 2002) mainly through the process of habitat fragmentation. Species habitats are progressively subdivided into smaller, geometrically more complex and isolated patches (Harris and Weiner, 2003), which can only support a limited amount of biodiversity. Implicit in the discussion of habitat fragmentation are the inter-related concepts of amount, pattern and spatial scale which can be quantified from remotely sensed data taken at different times. Adopting theoretical frameworks which seek to understand biodiversity variations in agricultural landscapes is important for the promotion of agriculture and biodiversity co-existence as well as determining the optimal habitat patch size to be considered for diversity conservation (Pearson *et al.*, 1999). The island biogeography theory offers a mechanism to explain biodiversity as a function of patch area. However the application of this theory has been limited to migratory species which it was formulated for. Although this might be the case, we test the hypothesis that the area-diversity prediction of the island biogeography theory can successfully be used to explain diversity in terrestrial species such as tree species.

In newly resettled areas, it is also necessary to understand the magnitude, patterns and direction of land cover conversions to guide sustainable biodiversity conservation measures in agricultural landscapes (Petit *et al.*, 2001). Research on how cropland expansion drives land cover changes in agricultural landscapes is pivotal especially considering the agriculture and forestry co-existence paradigm that we are moving towards. However, few studies (Elliot

et al., 2006; Petit *et al.*, 2001) have tried to understand land cover conversions in resettlement areas, where agricultural practices are exacerbating deforestation and impacting negatively on the biodiversity there. Remotely sensed data combined with stochastic models aid in the analysis of change trajectories and predictions that may improve our understanding of the nature of land cover conversions as well as the impact of human activities. The futuristic perspective of stochastic models such as Markov chains helps in building proactive environmental management programmes.

In this thesis we had two main objectives. The first objective was to statistically test whether the area-diversity relationship prediction of the island biogeography theory can be successfully used to explain differences in tree species diversity among different woodland patches in newly resettled areas. The second objective was to test whether cropland expansion drives land cover change in resettled landscapes now and in the future.

5.2 The predictions of the island biogeography apply to agriculturally fragmented woodlands.

In this thesis, we found that tree species diversity differs significantly between small and large woodland patches. In fact, we found that high tree species diversity is associated with woodland patches that are larger than 4ha whilst patches less than 4ha exhibit lower tree species diversity in the resettlement area. These findings suggest that 4ha woodland patch size could be the threshold patch area below which tree species diversity becomes compromised and above which tree species diversity is maintained in an agricultural landscape although we should caution that this might be true for areas more or less the same size as our study area (747,7ha).

5.3 Remotely sensed data and stochastic models help develop insights into historic and futuristic land cover dynamics

Understanding the role of land cover conversions on biodiversity conservation requires historical reconstruction of past land cover conversions and projections of likely future changes. In this thesis we used remotely sensed data and spatial Markov chain modelling in a GIS to determine past, present and future land cover dynamics in the resettlement area. We also explored the potential of environmental factors (soil type, slope, distance from homesteads, distance from roads and distance from rivers) to explain deforestation and cropland expansion. Our analysis of historic land cover shows an increase in cropland and wooded grassland as well as a simultaneous decrease in plantation and woodland between 2000 and 2009. Furthermore futuristic predictions of land cover conversions up to 2027 based on Markov chain modelling indicated an increasing trend of cropland and deforestation expansion. Results indicate that agriculture competes for land with forestry; as such deforestation will take place in areas of increased agricultural activity. Thus, in this thesis, we claim that agriculture is a major driver of land cover changes especially woodland change in newly resettled agricultural landscapes such as the Nyabamba area.

5.4 A summary of findings

In this thesis we drew a number of conclusions. Firstly, we concluded that woodland patch size explains tree species diversity in agriculturally fragmented woodlands. Next, we concluded that human driven forest fragmentation has a significant effect on tree species diversity. Thirdly, based on spatial Markov modelling results, we concluded that cropland and wooded grassland are likely to increase whilst woodland and plantations decrease. We

also concluded that distance from rivers and soil types are the environmental factors that significantly relate to land cover conversion.

5.5 The future

In this thesis we showed that although the island biogeography theory was developed for true oceanic islands, its predictions can be applied to agriculturally fragmented woodlands. However, it would be useful if more studies were done to test whether the island biogeography area-diversity prediction is applicable at large spatial extent and in different locations. Further research on where and how natural resources change over time should also be considered because information of this nature allows for informed environmental planning.

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