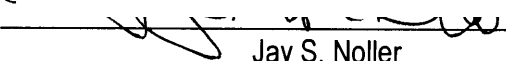


## AN ABSTRACT OF THE DISSERTATION OF

Abel Lufafa for the degree of Doctor of Philosophy in Soil Science presented on May 23, 2005.

Title: Spatial Analysis and Modeling of Carbon Storage in Native Shrubs of Senegal's Peanut Basin.

Signature redacted for privacy.

Abstract approved:  \_\_\_\_\_  
Jay S. Noller

Farmlands, grasslands, and savannas in semi-arid sub-Saharan Africa have the potential to store carbon (C) in soil. There is a great need to manage ecosystems to increase C storage in semi-permanent soil pools. This will improve soil quality, impede desertification and by reducing atmospheric CO<sub>2</sub> levels, will slow climate change. Several previous studies highlight the role of trees as a source of C to the soils of Senegal's Peanut Basin. This study is the first to recognize the role of *Guiera senegalensis* and *Piliostigma reticulatum* (two native shrub species widely found in natural and agro-ecosystems in sub-Saharan Africa), as key vegetative C sink in this landscape. This dissertation reports on three components of study. First, investigation of factors mediating distribution and total abundance of these shrubs shows that shrub distribution is controlled by geological substrate, mean annual temperature, mean annual rainfall and terrain profile convexity. Relative abundance of shrubs within their communities is influenced by mean annual rainfall, maximum annual temperature and elevation (for *G. senegalensis*) and mean annual rainfall, mean annual temperature, elevation and profile convexity (for *P. reticulatum*). The second study reports on the biomass C stocks of the shrubs and the relative influence of the shrubs and trees on spatial dependence of soil C levels. Estimates of peak-season biomass C ranged from 0.9 to 1.4 Mg C ha<sup>-1</sup> with an overall mean of 1.12 Mg C ha<sup>-1</sup> (SEM = ±0.079) where *G. senegalensis* dominates and from 1.3 to 2.0 Mg C ha<sup>-1</sup> (mean = 1.57 Mg C ha<sup>-1</sup>; SEM = ±0.18) where *P. reticulatum* dominates. Geostatistical analyses showed that shrubs had greater influence on landscape-level soil C levels than trees. The third study presents the regional C stocks and development of a method that couples the CENTURY model and GIS to model C dynamics for *G. senegalensis* in a portion of the basin. There is approximately 247000 Mg of biomass C at peak-season for the studied section and CENTURY model projections indicate that C sequestration in the area is hinged on long-term effectiveness of fire suppression (through residue burning) with actual rates depending strongly on scenarios of management and soil type.

©Copyright by Abel Lufafa  
May 23, 2005  
All Rights Reserved

SPATIAL ANALYSIS AND MODELING OF CARBON STORAGE IN NATIVE SHRUBS OF  
SENEGAL'S PEANUT BASIN.

by  
Abel Lufafa

A DISSERTATION

submitted to  
Oregon State University

in partial fulfillment of  
the requirements for the  
degree of

Doctor of Philosophy

Presented May 23, 2005  
Commencement June 2006

Doctor of Philosophy dissertation of Abel Lufafa presented on May 23, 2005

APPROVED:      Signature redacted for privacy.

Major Professor, representing Soil Science

Signature redacted for privacy.

Head the Department of Crop and Soil Science

Signature redacted for privacy.

Dean of the Graduate School

I understand that my dissertation will become part of the permanent collection of Oregon State University libraries. My signature below authorizes release of my dissertation to any reader upon request.

Signature redacted for privacy.

Abel Lufafa, Author

## ACKNOWLEDGMENTS

Philippians 4:13; "I can do all things through Christ who strengthens me". Ephesians 5:20; "Give thanks always for all things unto God and the Father in the name of Christ Jesus". Am very grateful to the almighty God for His generous provision of wisdom, knowledge and guidance that are invaluable in such an undertaking. Thank you God for all You have done and continue doing in my life. May Glory and Honor be exclusively Yours now and forever.

There is one man in my life that urged me on by way of his untiring support and seemingly unlimited belief in me, to that man, all else fades. Thanks Jay for ably guiding me through this study and consoling my frustration whenever we hit a rock. Am grateful to Richard Dick for his unwavering support and keeping me on track throughout the study. I am indebted to John Bolte whose thought provoking comments prompted me to ask the "tough" questions and Dawn Wright for the suggestions and leads that were invaluable especially in the GIS-CENTURY modeling.

Moses Tenywa remains my inspiration and my friend. I can only hope to repay my indebtedness to him by helping someone else with the same sacrificial attention he has given me. His devotion to me and ardent interest in all my academic works is a living testimony of unfettered love. There are special mentors that I must acknowledge due to their importance in my work. It is not practical to list all of those that have contributed because then I would be reciting names of many that I even never met, but whose published work inspired me. Suffice it to say that to all those from Paul Woomer the great to Fred Kizito, thank you!

There are many that from behind the scenes have encouraged and supported my studies, and I wish to thank them. Thank you Uncle David, SamB, Mukisah@mtn, Mubiru, Nicholas and a special thanks is due my friend Lina and Big Julio for cheering my progress. Last but not least, my thanks are due to the Senegalese group of the NSF project for all their contributions to this work and to Joan Sandeno for making this work read more and more of English and less and less of my native Lusoga!!

## CONTRIBUTION OF AUTHORS

Seven additional authors made significant contributions to the individual manuscripts presented in this dissertation. Richard Dick, Fred Kizito and Ibrahima Diedhiou made considerable contributions to all the manuscripts, Samba Ndiaye contributed to manuscript one and two, Modou Sene and Mamadou Khouma to manuscript two and three. Dawn Wright and John Bolte provided guidance and insight on manuscript three.

## TABLE OF CONTENTS

### Page

CHAPTER ONE. Introduction to the Dissertation.....	1
1.0 Background.....	1
1.1. Soil degradation in the Peanut Basin.....	2
1.2. Other ramifications of soil degradation .....	3
1.3. Role of carbon in soil productivity: soil quality and fertility .....	4
1.4. Carbon sequestration in degraded Parkland systems of the Peanut Basin .....	6
CHAPTER TWO. Determinants of distribution and abundance of two shrub species; <i>Guiera senegalensis</i> and <i>Piliostigma reticulatum</i> in Senegal's Peanut Basin.....	9
Abstract.....	10
1.0 Introduction .....	11
2.0 Materials and Methods.....	13
2.1. Study area characteristics .....	13
2.2. Vegetation data .....	15
2.3. Environmental data .....	15
2.4. Data analysis .....	17
3.0 Results and Discussion.....	24
3.1. Determinants of species distribution.....	24
3.2. Determinants of species abundance .....	28
3.3. Influence of management intensity on shrub abundance .....	30
4.0 Conclusion .....	32
5.0 Acknowledgements.....	32
CHAPTER THREE. Carbon stocks and patterns in native shrub communities of Senegal's Peanut Basin.....	33
Abstract.....	34
1.0 Introduction .....	35
2.0 Materials and Methods.....	36
2.1. Study area characteristics .....	36
2.2. Carbon stock estimation .....	37
2.2. Soil C spatial structure.....	39

## TABLE OF CONTENTS (Continued)

	<u>Page</u>
3.0 Results and Discussion.....	41
3.1. C stock estimation .....	41
3.2. Soil C spatial structure.....	47
4.0 Perspectives .....	54
5.0 Acknowledgments.....	56
CHAPTER FOUR. Regional carbon stocks and dynamics in native woody shrub communities of Senegal's Peanut Basin. ....	57
Abstract.....	58
1.0 Introduction .....	59
2.0 Materials and Methods.....	60
2.1. Study area description.....	60
2.2. Regional biomass C estimation .....	61
2.3. Spatial modeling of SOC dynamics .....	65
3.0 Results and Discussion.....	69
3.1. Regional biomass C stocks .....	69
3.2. Spatial SOC dynamics.....	72
3.3. Impact of climate change on SOC sequestration.....	80
4.0 Acknowledgments.....	80
CHAPTER FIVE. Conclusions to the Dissertation.....	81
Bibliography .....	83
Appendices .....	103



## LIST OF FIGURES

<u>Figure</u>	<u>Page</u>
1.1. Above: Millet and Below; Peanut production trends in the Diourbel region of the Peanut Basin. (Source: Division de la Statistique). ....	3
1.2. Relationship between organic carbon and cation exchange capacity of selected experimental soils. (Source: FAO, World Soil Resources Reports 96).....	5
1.3. Soil management practices (weighted points) in 14 villages of the Bambey region, Senegal. (Source: Petra Tschakert, Presentation at "Landscape Carbon Sampling and Biogeochemical Modeling Workshop" 25 <sup>th</sup> February to 9 <sup>th</sup> March, 2001, Dakar, Senegal). ....	6
2.1. Location and regions of the Senegalese Peanut Basin. ....	13
2.2. Monthly long-term average rainfall (mm) and temperature (°C) for the study area.....	14
2.3. Shrub spatial patterns for (a) <i>G. senegalensis</i> and (b) <i>P. reticulatum</i> . ....	21
2.4. Semivariograms of model residuals for <i>G. senegalensis</i> and <i>P. reticulatum</i> . ....	24
2.5. Determinants of presence of <i>G. senegalensis</i> and <i>P. reticulatum</i> in the Peanut Basin. ....	26
2.6. Potential environmental envelopes for the shrub species: a). <i>G. senegalensis</i> ; b). <i>P. reticulatum</i> . ....	27
2.7. Predicted shrub abundance based on the logistic regression model and krigged residuals. ....	29
3.1. Unity plots of observed and predicted biomass. Top left to right: aboveground, belowground and total <i>G. senegalensis</i> biomass. Bottom left to right: aboveground, belowground and total <i>P. reticulatum</i> biomass. ....	43
3.2. Semivariograms of soil C based on 339 samples from the 480 m <sup>2</sup> plot ....	49
3.3. Separate semivariograms for soil C with the small-grid data around the shrubs removed. ....	51
3.4. Separate semivariograms for soil C with the small-grid data around the trees removed. ....	52
3.5. Potential distribution maps for <i>G. senegalensis</i> (left) and <i>P. reticulatum</i> (right) in the Peanut Basin of Senegal. ....	55

## LIST OF FIGURES (Continued)

<u>Figure</u>	<u>Page</u>
4.1. Study area location.....	61
4.2. Conceptual framework for the loose coupling between GIS and the CENTURY model. ....	67
4.3. Relationship between biomass C stock and shrub abundance from spectral unmixing.....	71
4.4. Predicted aboveground biomass C stocks in the study area. Shown values have been resampled to a 90 m resolution. ....	71
4.5. The CENTURY-simulated SOC under the four management scenarios. ....	73
4.6. Spatial distribution of predicted SOC under the "prunings burned" regime in 2005, 2010, 2050 and 2100. ....	76
4.7. Spatial distribution of predicted SOC under the "prunings returned" regime in 2005, 2010, 2050 and 2100. ....	77
4.8. Spatial distribution of predicted SOC under the "prunings returned-low fertilization" regime in 2005, 2010, 2050 and 2100.....	78

## LIST OF TABLES

<u>Table</u>	<u>Page</u>
2.1. Confusion matrix for classification on MTMF image using ground validation pixels. ....	16
2.2. List of potential predictors used to model distribution and abundance of the two shrub species. ....	18
2.3. Edaphic characteristics of the sites used to study influence of management on shrub abundance.....	22
2.4. Error matrix and accuracy measures for the shrub predictions. ....	28
2.5. Factors associated with <i>G. senegalensis</i> and <i>P. reticulatum</i> abundance in the Peanut Basin. ....	29
2.6. Comparison of cluster parameters for the shrubs in the different sampling locations.....	31
3.1. Location, elevation, number of plot replicates and average number of shrubs at the biomass inventory sites. ....	39
3.2. Selected model coefficients of determination ( $R^2$ ), MSEPs and Pearson correlation coefficients ( $r$ ) for observed and predicted biomass. ....	42
3.3. Peak-season biomass, soil and system C stocks in native shrubs in Senegal's Peanut Basin. Ranges within each site are in parentheses.....	45
3.4. Regression relationships between shrub numbers and <i>P. reticulatum</i> biomass C stocks as $Mg\ ha^{-1}$ .....	46
3.5. Peak-season and regrowth biomass for <i>G. senegalensis</i> and <i>P. reticulatum</i> at selected sites in Senegal's Peanut Basin. ....	46
3.6. Descriptive statistics for the study grids of C structure at the two sites. Shown are means ( $\mu$ ), standard deviations ( $\sigma$ ), range and sample size ( $n$ ) for the soil C measurements. ....	48
3.7. Summary of model parameters fitted through each of the semivariograms in Figure 3.2 through 3.4 (based on different grid combinations). ....	50
4.1. Vegetation and elevation variation across the biomass C scaling-up training plots.....	63
4.2. Carbon stocks and satellite image derived shrub abundances at the training plots. ....	70
4.3. Simulated average SOC in the different soils of the study area in 2010, 2050 and 2100.. ....	79

## LIST OF APPENDIX FIGURES

<u>Figure</u>	<u>Page</u>
A.1.1. Point pattern representation of the shrubs (a) <i>G. senegalensis</i> , b) <i>P. reticulatum</i> . ....	107
A.1.2. Shrub spatial patterns (a) <i>G. senegalensis</i> and (b) <i>P. reticulatum</i> . Ripley's L(d) plotted against d(distance of successive neighboring shrub from "focal shrub"). (---) is the approximate 95% confidence limits (simulation envelopes; see text) about the expected value (.....) for complete spatial randomness, which = 0. ....	108
A.1.3. Ripley's L(d) for the shrubs (a) <i>G. senegalensis</i> and (b) <i>P. reticulatum</i> with simulation envelopes (---) based on 100 replications of a Neyman-Scott cluster process.....	109
A.2.1. Biomass allocation expressed as shoot:root ratio for the sampled shrubs.....	120
A.2.2. Unity plots of observed and predicted biomass. Top left to right: aboveground, belowground and total <i>G. senegalensis</i> . Bottom left to right aboveground, belowground and total <i>P. reticulatum</i> biomass. ....	124
A.4.1. Conceptual framework of the loose coupling between GIS and the CENTURY model .....	130
A.4.2. i_Century configuration dialog box.....	136
A.5.1. Symbols used in the flow diagrams showing the database routines used .....	138
A.5.2. Routine used to create the Site Data table. ....	139
A.5.3. Routine for creating the Monthly Weather table. ....	140
A.5.4. Routine for creating the Control Records table and relating the Control Records table to the Monthly Weather and Soils tables. ....	141
A.5.5. Routine for creating the Event Blocks table. ....	142
A.5.6. Routine for creating the Events table. ....	143

## LIST OF APPENDIX TABLES

<u>Table</u>	<u>Page</u>
A.1.2. Comparison of cluster parameters for the shrubs in the different sampling locations.....	112
A.2.1. Number of samples, their location, average number of stems and average diameter for the sampled shrubs.....	114
A.2.2. Biomass predictive equations and their adjusted $R^2$ values for <i>G. senegalensis</i> and <i>P. reticulatum</i> in 2003.....	117
A.2.3. Biomass predictive equations and their adjusted $R^2$ values for <i>G. senegalensis</i> and <i>P. reticulatum</i> in 2004.....	118
A.2.4. Biomass predictive equations and their adjusted $R^2$ values for the shrubs over the two study years. ....	121
A.2.5. General allometric models for estimating shrub biomass.....	122
A.2.6. Selected model MSEPs and their Pearson correlation coefficients for observed and predicted biomass.....	122
A.2.9. Regression statistics for predicting biomass fractions of the two shrubs with individual species combined over the two years. Parameters are from the relationship $\log Y = \beta_0 + \beta_1^*(X_1) + \dots + \beta_n^*(X_n)$ , where $Y$ = dry weight of biomass in grams, $X_1 \dots X_n$ are the respective explanatory variables in each model. ....	127
A.2.10. Regression statistics for the general biomass predictive models. Parameters are from the relationship $\log Y = \beta_0 + \beta_1^*(X_1) + \dots + \beta_n^*(X_n)$ , where $Y$ = dry weight of biomass in grams, $X_1 \dots X_n$ are the respective explanatory variables in each model....	128
A.4.1. Output information from GIS. ....	132
A.4.2. Overview of i_Century tables .....	133

## DEDICATION

For this piece of work, may the name of the almighty God be lifted up. It is your will God that this course comes to pass. My strength and success come from your love and kindness and it is in this vein that I whole-heartedly dedicate this piece of work to your glory.

I also dedicate this work to my loving mother Mrs. Lufafa Merb (RIP) who lost her protracted struggle for life barely a week into this course. Life is almost meaningless without you but I am soothed by the knowledge that I will see you one beautiful day when the Lord beckons us for supper after this great tribulation. Yes, there will be an abundance of flowers and singing and joy! The deserts will become as green as the mountains of Lebanon, as lovely as Mount Carmel's pastures and the Plain of Sharon. There the Lord will display His glory, the splendor of our God and we will be reunited forever. Amen.

## **CHAPTER ONE**

### **Introduction to the Dissertation**

#### **1.0 Background**

Land degradation persists as the most important environmental problem facing semiarid sub-Saharan Africa (Harrison, 1987; Blaikie, 1989; Lal, 1990; World Bank, 1996). Although defined variously (Blaikie and Brookfield, 1987; Barrow, 1991; Oldeman, 1994; Johnson and Lewis, 1995; Conacher, 2001; Stocking and Mumaghan, 2001), the differences in definition are mere semantic conjectures (often with distinct disciplinary-oriented meaning) given the fact that there is unanimity regarding the toll it exacts on the natural resource base. Land degradation due to natural causes is believed to occur at a rate which is in balance with that of natural rehabilitation but human-related factors are said to be responsible for the accelerated forms of degradation (Larson and Frisvold, 1996).

Land degradation has negatively impacted the quality of life for a significant number of the predominantly agrarian inhabitants of semiarid sub-Saharan Africa (SSA). It has exacerbated global environment change by fostering land conversion in light of a swelling population and adoption of poor farming methods by farmers struggling to balance and buffer their food security. Land degradation in SSA Africa has been equated to desertification, food crises and overall impoverishment (Lal, 1999; Barbier, 2000). United Nations reports (UNECA, 2001; 2002) show the high economic cost of land degradation to the SSA countries exclusively attributed to lost agricultural productivity. The thesis, that the fall of civilizations is linked to the degradation of their land is quite difficult to refute under the conditions pertaining in semiarid SSA.

Whereas there might be varying processes of degradation (Lal and Stewart, 1994) at least three stages are recognized in the drylands (Zewge, 2003). The first stage referred to as "woodland degradation" relates to natural woodlands and is occasioned by increased conversion to cropland or increased demand for tree products. The second stage relates to agroforestry parklands which are established when farmers convert natural woodlands to croplands, by cutting most of the trees and leaving behind that they value most to fulfill their basic needs. The third stage known as "cropland degradation," occurs as dewooded lands

degrade as a result of intensive cultivation. Land degradation takes a number of forms (including soil degradation, vegetation degradation, agrochemical pollution), but the single most important element in the semiarid sub-Sahel (due to its influence on productivity) is soil degradation (Reardon et al., 1999) which is attributed to mismanagement, deforestation, over-grazing, increasing cultivation of fragile ecosystems, and increasing rural populations (Drechsel et al., 2001).

### **1.1. Soil degradation in the Peanut Basin**

The Peanut Basin located in Senegal is one of the most intensive Sahelian agricultural zones (Tschakert et al., 2004). Situated in the center of the country, the basin straddles approximately 46,367 km<sup>2</sup>, holds traditional Serer and Wolof agricultural lands with intensively cultivated village territories and is important in the food supply equation of the country. Traditional Serer and Wolof (the two major ethnic groups in the Peanut Basin) agricultural practice is characterized by a “triangular” management approach including crop rotation, fallowing and application of animal manure (Pelissier, 1953; Lericollais, 1972; Gastellu, 1981; Pontie and Lericollais, 1995). Long ago, these indigenous agricultural practices were on the whole, sufficient and adequately met the food, fuel and fiber needs of the existing population. However, with mounting demographic pressure, diminishing land resources (leading to continuous cultivation) buttressed by decreases in amount and reliability of rainfall, these lands have progressively declined in productivity.

The reduction in soil productivity in this basin can be traced to the late 1800s following introduction of Peanut cultivation under the French colonial rule. The European industries desired the high oil content of Senegalese peanuts, which resulted into extensive tracts of land being opened-up to expand production. This trend of extensification with attendant soil degradation peaked during World War II when intensive peanut production was designated to support the allies (Kelly et al., 1996). According to anecdotal information and unpublished reports, the period from independence in 1960 and the years of African socialism saw a return to good agricultural practices under “Programme Agricole,” the state’s major agricultural policy designed to provide large-scale support for smallholder farmers. Unfortunately, the gains attained during these days were eroded when overemphasis on peanuts reinforced extensive land clearing and deforestation and prevented early agricultural diversification.



After the transition period from 1980-1983, during which the state incrementally retired from its multiple functions, a new era of agricultural policies began in 1984 with the “Nouvelle Politique Agricole”. At the backdrop of Senegal’s growing population, increasing national debt, inefficient resource allocation through the public sector, and most of all, pressure from World Bank and IMF through structural adjustment policies, this new agricultural policy brought fundamental changes for farmers. The state retiring from its main duties, no longer provided for agricultural subsidies and national use of agricultural inputs declined by more than 60% compared to the 1970s. The State disengagement, juxtaposed to the two serious droughts of 1982-1984 and 1996-1997, gave way to increased soil degradation (Tschakert, 2001), decline in productivity and corresponding negative increases in per capita yield of crops (Figure 1.1).

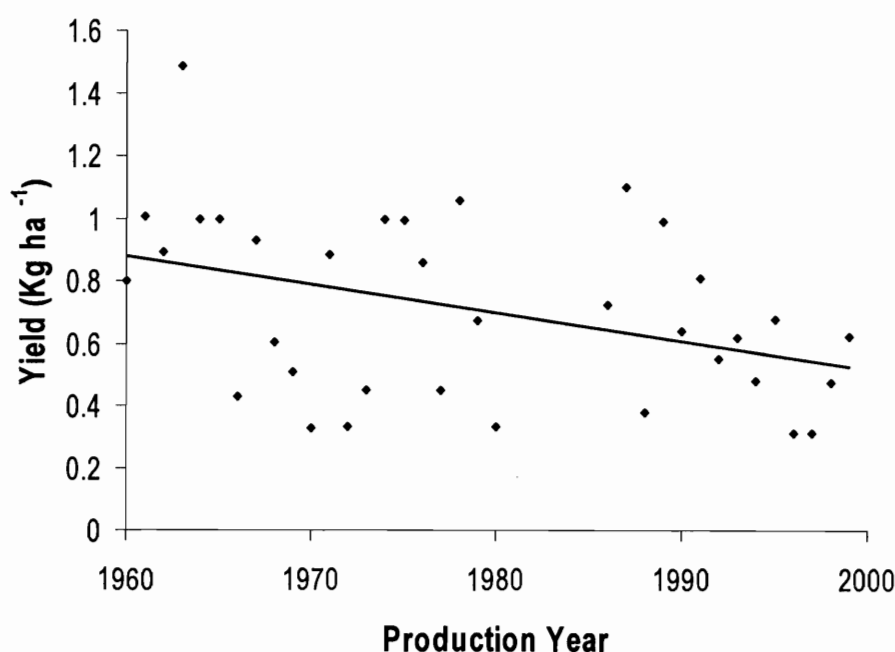


Figure 1.1. Per capita Peanut production trends in the Diourbel region of the Peanut Basin. (Source: *Division de la Statistique*).

## 1.2. Other ramifications of soil degradation

Besides the local concerns of lost productivity, degradation in these fragile ecological landscapes is also implicated in the continued release of greenhouse (especially CO<sub>2</sub>) gases to the atmosphere; an issue that is of concern to global society seeking to mitigate climate change. Currently, the international community enjoined by several multilateral environmental agreements

(MEAs), is preoccupied with the quest for mechanisms to curtail degradation and to check the increasing atmospheric load of carbon dioxide and other greenhouse gases. Under the twin conventions: UN Framework Convention on Climate Change (UNFCCC) and the UN Convention to Combat Desertification (UNCCD), a range of mechanisms to promote sustainable generation of environmental goods and services together with economic development have been proposed and in some cases implemented. The Kyoto protocol (Grubb et al., 1999) as espoused in the UNFCCC provides an excellent opportunity for synergistically addressing local soil productivity concerns and ameliorating global climate change. The protocol recognizes that net emissions may be reduced either by decreasing the rate at which greenhouse gases are emitted to the atmosphere or by increasing the rate at which they are removed from the atmosphere through sinks and considers the two means as complementary. Under the protocol, promotion of C sequestration through a variety of mechanisms is recognized as a means of offsetting emissions. Furthermore, implementation of the protocol will provide incentives for shifts in land management to systems that store carbon in forms and pools that are relatively inert. All management interventions aimed at enhanced carbon uptake and accumulation in croplands will also restore degraded soils and increase crop productivity (Batjes, 2001; Olsson and Ardo, 2002).

### **1.3. Role of carbon in soil productivity: soil quality and fertility**

Through photosynthesis, agricultural practices sequester and fix C; a key component of organic matter into soil and plants. Organic matter (OM) has essential biological, physical and chemical functions in soils and its content is generally considered one of the primary indicators of soil quality, both for agriculture and for environmental functions (Lal, 2000). An important property of soils for crop and forest productivity is the ability to store and provide nutrients for plant uptake. This ability otherwise known as cation exchange capacity (CEC) is a function of organic matter content (Figure 1.2). Organic matter therefore, becomes of an even more special value in semiarid sub-Saharan soils which are inherently sandy, characterized by low activity clays (FAO, 1998) and hence of low CEC. Through formation of chelates and other bonds, organic matter is known to mediate and improve the bioavailability of some important elements such as phosphorus while inhibiting the toxicity of others, e.g. aluminum (Robert, 1996). Sustainability of low input agriculture such as is practiced in semiarid sub-Saharan, is contingent upon recycling of

nutrients (N, P, K, Ca) by gradual decomposition of plant and crop residues (Sanchez and Salinas, 1982; Poss, 1991) and an increase in organic matter is also accompanied by a corresponding increase in the operational biodiversity and more effective soil biological functioning, which is normally very low in most agricultural soils.

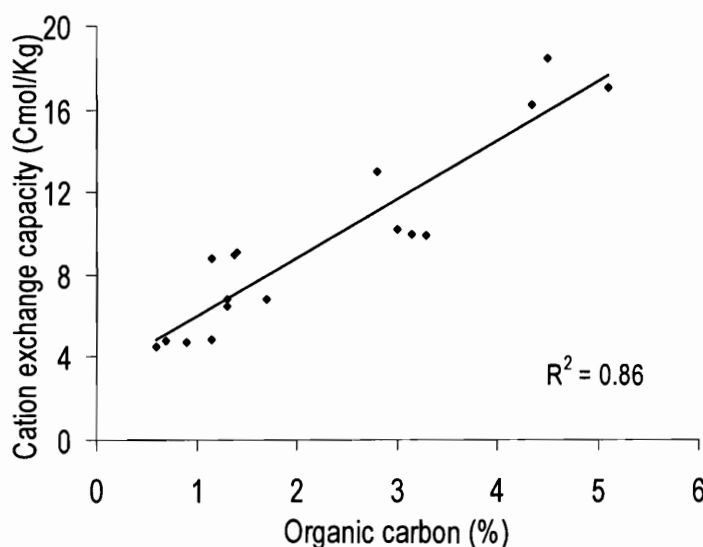


Figure 1.2. Relationship between organic carbon and cation exchange capacity of selected experimental soils. (Source: FAO, World Soil Resources Reports 96).

With regard to physical properties, organic matter and associated living organisms play the main role in soil aggregation (Balesdent et al., 2000) at different scales of soil organization (Tisdall and Oades, 1982; Robert and Chenu, 1991). Many soil properties (e.g. water retention and release to plants, infiltration rate and resilience to erosion and other physical degradation processes) depend on the aggregation and stability of the soil fabric. In the case of erosion, a correlation has been established between the historical decrease in soil organic matter and erosion development (Lal, 1990). Accumulation of organic matter leads to structural stability (resistance to both wind and water erosion) and favors soil cover, thus preventing erosion and increasing water conservation (Lal, 1997; Gregorich et al., 1998).

#### 1.4. Carbon sequestration in degraded Parkland systems of the Peanut Basin

Batjes (1996, 2001) discusses C sequestration potential relative to the magnitude and severity of land degradation. He distinguishes slight and moderate degradation, which can be restored by improved land management in comparison to strong and extreme degradation, where restoration strategies will likely require conversion to a new land use. The magnitude and severity of land degradation in the Peanut Basin although not quantified, is recognized as being highly heterogeneous; varying spatially and temporally (R.P. Dick, Pers. comm). Tschakert et al. (2004) underscore the biophysical potential for soil carbon sequestration in these degraded agricultural systems of the Peanut Basin and through modeling with CENTURY soil organic matter model, demonstrate carbon gains for some agricultural practices. Figure 1.3 from a survey of 14 villages in the Bambey region of the Peanut Basin, shows various soil management practices tailored towards stalling degradation over the last four decades.

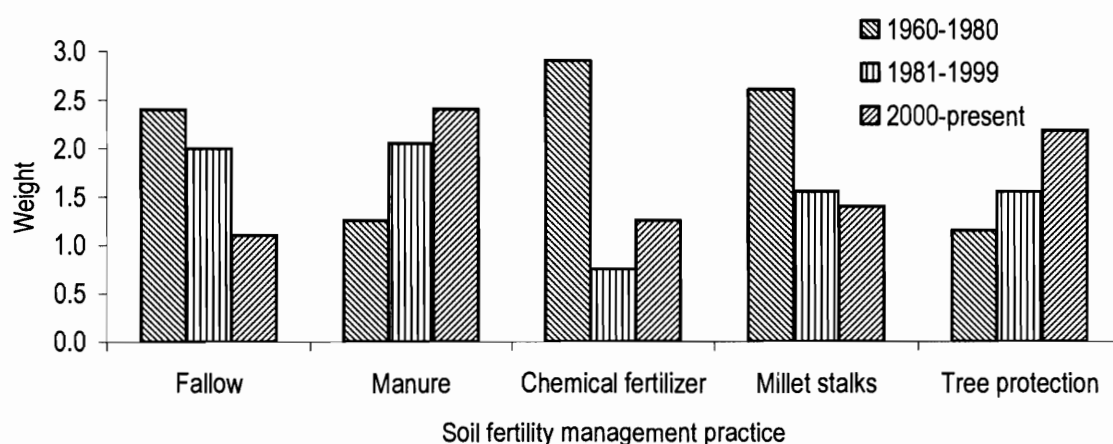


Figure 1.3. Soil management practices (weighted points) in 14 villages of the Bambey region, Senegal. (Source: Petra Tschakert, Presentation at "Landscape Carbon Sampling and Biogeochemical Modeling Workshop" 25<sup>th</sup> February to 9<sup>th</sup> March, 2001, Dakar, Senegal).

The figure reveals a conspicuous and increased recognition over the years, of the protection and retention of trees in farmers' fields (agroforestry) and use of organic manure as important indigenous soil management strategies. It also shows modest reversion to use of chemical fertilizer. However, the low amounts of animal manure produced and the persistent unwillingness (motivated by fear of crop and market failure) of farmers to use chemical fertilizer (Tschakert et al., 2004)

render these two interventions unsustainable in the long-term. Association of trees with crops therefore stands out as representing the most viable and sustainable alternative for averting degradation in these systems. The ability of agroforestry systems to improve soil quality and increase crop yields has been widely shown (Fassbender et al., 1991; Schroeder, 1994; Dixon, 1995; Nyberg and Hogberg, 1995; Palm, 1995; Rhoades, 1997; Buresh and Tian, 1998) while its huge potential to sequester carbon is also well documented (Winterbottom and Hazlwood, 1987; Schroeder, 1994; Sanchez, 1995; Karjalainen, 1996; Sanchez et al., 1999; Pandey, 2002; Montagnini and Nair, 2004).

The tree component of the Parkland agroforestry systems of the Peanut Basin is dominated by *Acacia tortilis* to the north, *Faidherbia albida* in the central and *Cordyla pinnata* to the south; probably reflecting the north-south rainfall gradient. Interspersed between the trees, is a woody vegetative component that principally owes its existence to a regrowth of annually pruned shrubs. Two native species: *Guiera senegalensis* J.F. Gmel and *Piliostigma reticulatum* (DC) Hocht dominate this vegetative layer and depending on soil type, topography and rainfall amounts; they cover the entire landscape and form the only contiguous vegetation block in large portions of the Basin (A. Lufafa pers.comm). Several studies (Woomer, 1993; Bationo et al., 1998; Manlay, 2000; Batjes, 2001; Manley et al., 2002; Liu et al., 2004; Tschakert et al., 2004; Woomer et al., 2004a; Woomer et al., 2004b) have investigated or alluded to the role carbon sequestration in these Parkland systems could play in improving productivity and mitigating climate change and have demonstrated positive impacts. Studies of Parkland systems have largely been on trees with virtually no detailed investigations of these shrubs and their role in mitigating soil degradation. Thus, the goal of this study was to assess the influence of the two native shrub species on landscape-level C cycling.

The specific objectives of the study were:

- to determine factors influencing the distribution and abundance of the two shrub species across the landscape;
- to quantify C stocks in the shrubs and underlying soils and evaluate the relative influence of the shrubs and trees on soil C spatial structure;

- to provide projections of landscape-level soil C stocks, under a series of landuse and management options over decadal times through biogeochemical modeling with the CENTURY model.

Each of the above objectives constitutes a manuscript in this dissertation.

## CHAPTER TWO

### **Determinants of distribution and abundance of two shrub species; *Guiera senegalensis* and *Piliostigma reticulatum* in Senegal's Peanut Basin.**

A. Lufafa, I. Diedhiou, N.A.S. Ndiaye, F. Kizito, R.P. Dick and J.S. Noller

Acta Oecologica (in review)  
Ecologie des populations et communautés  
Institut national agronomique Paris-Grignon  
16, rue Claude-Bernard  
75231 Paris cedex 05  
France

## Abstract

The ability to predict and manage the course of landscape-level ecological change and its longer-term consequences on ecosystem functions (e.g. carbon stabilization and soil degradation mitigation) depends on the ability to understand how a particular ecosystem functions and the mechanisms that control the distribution, configuration and abundance of key species. *Guiera senegalensis* and *Piliostigma reticulatum* are two native shrub species that are widely found in sub-Saharan Africa but unrecognized in their potential role in regulating hydrological and carbon cycles in both natural and agro-ecosystems. Our objective was to conduct a study on the determinants of landscape-level distribution and abundance of these shrub species as a basis for ecological modeling and management of this fragile semiarid environment. Formal Inference Recursive Modeling was used to adduce determinants of species presence, while logistic regression and geostatistical approaches were used to estimate shrub abundance within their communities. Results show that the distribution of the shrubs is strongly controlled by four factors: geological substrate, mean annual temperature, mean annual rainfall and terrain profile convexity. Relative abundance within the shrub communities is under the influence of mean annual rainfall, maximum annual temperature and elevation (for *G. senegalensis*) and mean annual rainfall, mean annual temperature, elevation and profile convexity (for *P. reticulatum*). Predictive models for shrub distribution and abundance were generally poor, probably highlighting the weakness of statistical models in analysis and quantification of the spatial structure of ecosystems.



## 1.0 Introduction

Sequestering carbon (C) in the terrestrial biosphere is one of the options proposed to compensate greenhouse gas emissions (DOE, 1999). Agricultural lands, if judiciously managed, are believed to be a major potential sink that could uptake and store between 42 and 90 Pg of C from the atmosphere over the next 50-100 years (Albrecht and Kandji, 2002). In addition to ameliorating global warming, sequestration of C in soil organic matter has the potential to enhance local land productivity (Woomer et al., 1994; Ringius, 2002), which is particularly important for restoring degraded lands and improving the livelihood of resource-poor farmers in semiarid agro-ecosystems (FAO, 2001).

There is scientific consensus that agroforestry, which is the deliberate retention or introduction of trees or other woody perennials in agricultural lands, is a key intervention to synergistically achieve agricultural sustainability (Schroeder, 1994), reduce land degradation (Schroeder, 1993) and mitigate climate change (Dixon, 1995; Paustian et al., 1997; Paustian et al., 2000). Tschakert et al. (2004) using the CENTURY model show there is a C sequestration potential of up to 0.12 Mg C ha<sup>-1</sup> yr<sup>-1</sup> in the dryland systems of the Senegalese Peanut Basin for agroforestry associations involving *Leucaena leucocephala* (a non-native species) and proffer that improving agricultural practices is key to enhancing food production and mitigating climate change in this area.

Attempts to introduce novel agroforestry systems in dry-land systems such as those of Senegal's Peanut Basin have been largely unsuccessful. The most intractable problem is how to retain the positive effects of tree litter inputs and roots on soil physical and chemical properties while reducing the negative effects of belowground competition between trees and crops. In the Senegal's Peanut Basin, there are two naturally occurring woody shrubs: *Guiera senegalensis* J.F. Gmel and *Piliostigma reticulatum* (DC.) Hochst. These shrubs are typically left to grow in farmers' fields where they are coppiced at the end of the dry season (March to April) and subsequently are pruned at every weeding during the growing season of row crops. Following the growing season, the shrubs are allowed to regrow reaching > 2 m diameters and > 1 m heights with average total dry weight biomasses of 2.4 and 3.7 t ha<sup>-1</sup>, respectively, for *Guiera*

*senegalensis* and *Piliostigma reticulatum* (A. Lufafa, pers. comm., 2005). A casual conservative analysis estimates landscape-level C stocks residing in these shrubs at over  $22.5 \times 10^5$  Mg and that through improved management, these shrubs could be a repository of up to  $30 \times 10^5$  Mg of C over the next 15 years. Furthermore, preliminary studies indicate these shrubs can provide nutrients (Gerard and Buerkert, 1999; Wezel, 2000; Iyamuremye et al., 2001) to improve crop productivity.

The unique niche where these shrubs coexist with human agricultural activities in semi-arid West Africa is of interest for understanding how these ecosystems function and for improving agricultural productivity. Our qualitative observations showed wide variations in density and distribution at both field- and landscape-level scales. However, little is known about what factors mediate the distribution and abundance across this region. Many interacting factors determine the range and abundance of a plant species at a given locality. Species distributions show clear associations with climate, typically on regional scales of hundreds of kilometers and the physiology of these associations and links is reviewed in detail by Woodward (1987). Webb (1959, 1968) described variation in species distribution in relation to a wide range of annual rainfall and inferred soil nutrients in Australia. Species distributions also show association with soil type, which is more often recognized on relatively small scales because extensive soil data are rarely available. At larger scales, Baillie et al. (1987), Newbery et al. (1988) and ter Steege et al. (1993) found correlations between tropical tree species abundance and soil type. A combination of climate and other environmental factors have long been used to explain the main vegetation patterns and types around the world (Salisbury, 1926; Cain, 1944; Good, 1953; McArthur, 1972; Stott, 1981; Walter, 1985; Ellenberg, 1988).

The principal objective of this study was to determine factors that control the distribution and abundance of *Guiera senegalensis* and *Piliostigma reticulatum* in the Peanut Basin. Improved knowledge of the shrub environmental envelopes would be useful to guide C sequestration to mitigate elevated atmospheric CO<sub>2</sub> levels, improve soil quality, help predict shrub species shifts in distribution and stability under an altered climate, and enable more refined hypotheses about the functioning of these shrub systems to be conceived.

## 2.0 Materials and Methods

### 2.1. Study area characteristics

The general study area is the Senegalese Peanut Basin (Figure 2.1) in Senegal, West Africa which is located east of Dakar, centered at about at  $16^{\circ}\text{W}$ ,  $14.7^{\circ}\text{N}$  with a spatial coverage of ~44000 sqkm. The climate is semiarid, with on average more than 85% of precipitation falling between August and October. Mean annual precipitation (MAP) is approximately 540 mm, skewed towards the south and with high variability from year to year (Dacosta, 1989).

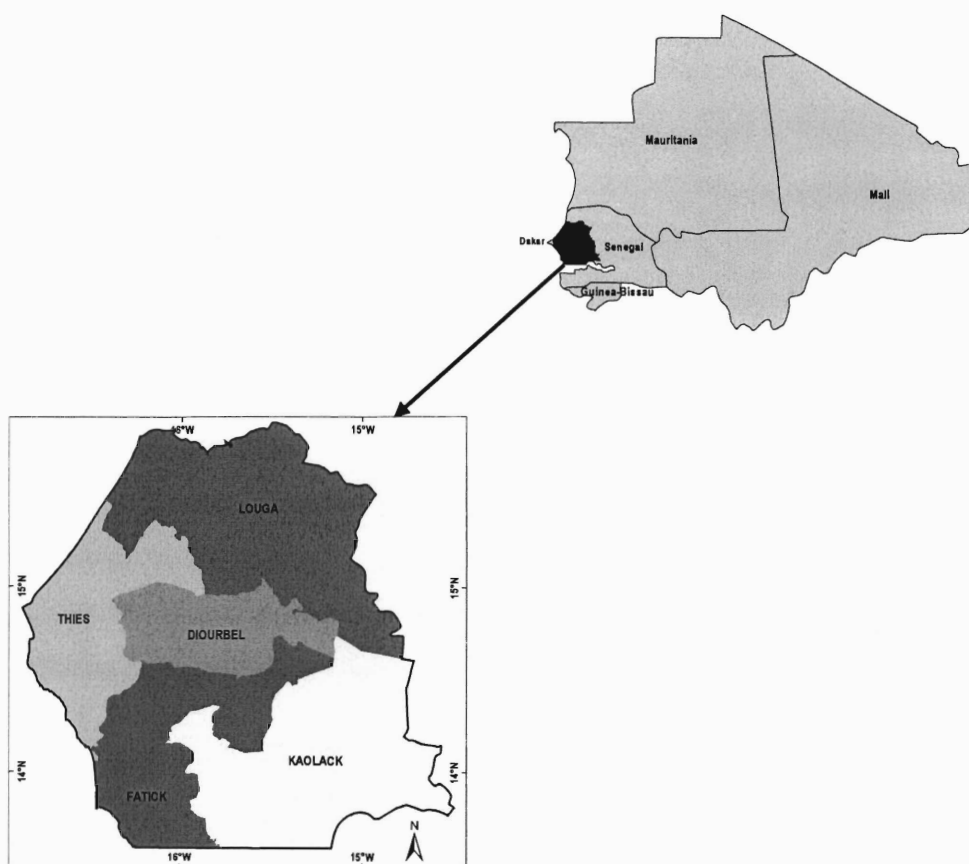


Figure 2.1. Location and regions of the Senegalese Peanut Basin.

The minimum and maximum mean annual temperatures are  $20^{\circ}\text{C}$  and  $34^{\circ}\text{C}$  respectively, with a marked seasonal variation (Figure 2.2). Elevation ranges from 8 to 249 m a.s.l. Geological substrates in the area include mainly aeolian deposits of Harmattan wind sand (Herrmann, 1996) of Quaternary age over sedimentary rocks of Cretaceous to Miocene (Monciardini, 1966), and highly eroded colluvial-alluvial ferruginous sediments derived from paleosols (Neogene) and

Precambrian bedrock (Renaud, 1961; Michel, 1973). Fluvial deposits consist largely of fine gravelly sand derived from weathered bed rock (Leprun, 1967; Pieri, 1969; ORSTOM, 1984).

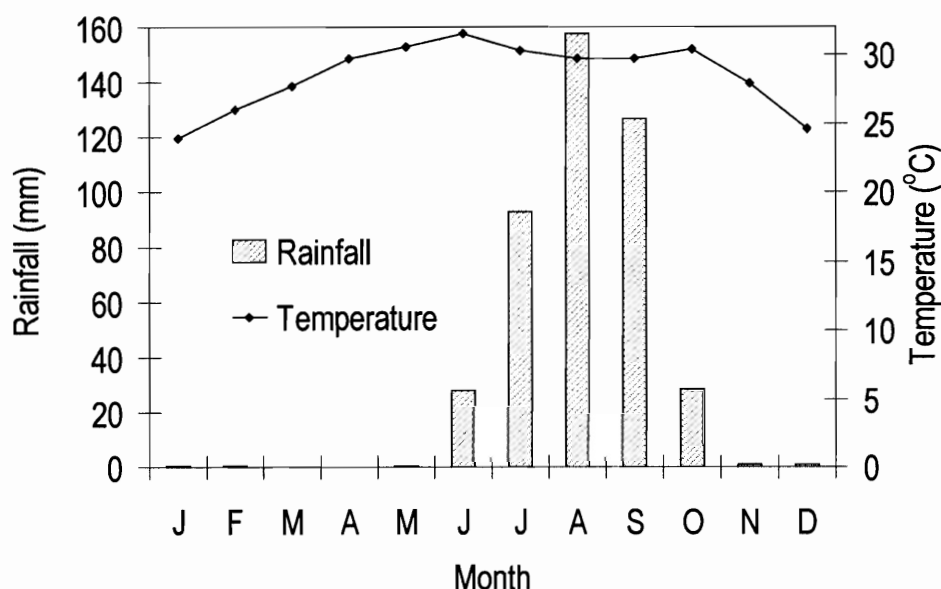


Figure 2.2. Monthly long-term average rainfall (mm) and temperature (°C) for the study area.

Basin soils are sandy, classifying as *Psamments* and *Calcids* according to Soil Taxonomy (Soil Survey Staff, 2003) and fall broadly into two indigenous types, i.e. Dior and Deck (Badiane et al., 2000). Dior soils are characterized by a sandy texture (over 95% sand), low organic matter (0.2%) and low N content (0.1%). In comparison, Deck soils are hydromorphic, have up to 8% clay, slightly higher in both organic matter (0.5-0.8%) and N content (0.2-0.4%), and lower in P. Basin vegetation is shrubland with scattered trees (Diouf and Lambin, 2001). The shrub layer is dominated by *G. senegalensis* in the central and northern areas, whereas by *P. reticulatum*, *Combretum glutinosum* and the evergreen *Icinia senegalensis* co-dominate in the south. The tree component is dominated by *Acacia tortilis* to the north, *Faidherbia albida* in the central and *Cordyla pinnata* to the south; probably reflecting the north-south rainfall gradient. Basin vegetation is intermittently grazed and browsed by cattle, sheep and goats. Growing seasons for the shrublands typically last from the first major rainfall after the dry winter season until December of the same year.

## 2.2. Vegetation data

We used a Landsat ETM<sup>+</sup>-based vegetation map to derive geographical presence/absence and abundance data for the two shrub species. The map is based on spectral analysis of a March, 2004 Landsat image using the Mixture Tuned Matched Filtering (MTMF) algorithm in ENVI<sup>TM</sup> (ENVI, 2004). The MTMF algorithm is designed and optimized to detect extremely weak signals that are essentially in the noise (Funk et al., 2001). Given the sparse nature of arid-land vegetation, enhanced detection significantly improves their inventory and mapping (Huete et al., 1985; Huete and Jackson, 1987; Smith et al., 1990). In addition to detecting presence, it provides a relative abundance of the target vegetation in a pixel and outputs an infeasibility image that is used to reduce the number of false positives in the original scene (ENVI, 2004). Reflectance spectra used in the analysis were obtained in the field using an Analytical Spectral Devices VNIR spectroradiometer (FieldSpec, 1997) at a time approximately coincident with the date of acquisition of the Landsat image. At this time of year, the grass and most of the tree canopies were fully senescent while the shrub canopies were fully green. The confusion matrix of the resulting classification, which describes the predicted and actual classifications, is shown in Table 2.1.

A random set of 8000 (30 m x 30 m) pixels; 6000 for *G. senegalensis* and 2000 for *P. reticulatum* was chosen as the basic modeling data representing shrub presence/absence. Abundance data was generated by overlaying a 0.9 km x 0.9 km grid over the MTMF image and the individual grids considered sampling units. The number of Landsat pixels in the grid identified as containing the target shrubs were enumerated using GIS procedures and divided by the total number of pixels (900) in the grid to obtain counted proportions (Ramsey and Schafer, 2002), which is the standard for recording species abundance (Guisan and Zimmermann, 2000). These counts of detections per unit area were used as an index of landscape-level shrub abundance.

## 2.3. Environmental data

We selected a series of biophysical factors based on perceived and/or documented importance to shrub growth. Existing GIS data layers (including soils, geology and hydrogeology maps) at a scale of 1:250,000 were acquired in Dakar from Centre de Suivi Ecologique (CSE) and climatic

Table 2.1. Confusion matrix for classification on MTMF image using ground validation pixels.

Class	<i>G. senegalensis</i>	<i>P. reticulatum</i>	<i>C. glutinosum</i>	<i>I. senegalensis</i>	Total
<i>G. senegalensis</i>	412	34	3	0	449
<i>P. reticulatum</i>	117	213	16	22	368
<i>C. glutinosum</i>	10	40	38	21	109
<i>I. senegalensis</i>	3	76	8	35	122
Total	542	363	65	78	1048

Class	Producer's accuracy	User's accuracy
<i>G. senegalensis</i>	76.01	91.76
<i>P. reticulatum</i>	58.68	57.88
<i>C. glutinosum</i>	58.46	34.86
<i>I. senegalensis</i>	44.87	28.69

Overall accuracy = 66.6%; kappa coefficient = 0.48.

data was obtained from the Direction de la Météorologie Nationale (DMN). Four variables were used to characterize the climatic regime: MAP, MAT, maximum annual temperature and minimum annual temperature. A Shuttle Radar Topography Mission (SRTM) digital elevation model (DEM) with a 90 m resolution and an elevation accuracy of 2 m was used for deriving terrain variables. Six terrain variables-slope gradient, aspect, profile convexity, plan convexity, minimum and maximum curvature were obtained directly from the DEM at 90 m grid resolution with 3 by 3 pixel filters in ENVI™ which implements the Wood (1996) formulas. Table 2.2 summarizes the selected environmental variables.

## **2.4. Data analysis**

### **2.4.1. Determinants of shrub distributions**

Formal-based Inference Recursive Modeling (FIRM), a recursive partitioning technique (Hawkins, 1995), was used to describe the relationships between environmental variables and the distributions of the shrub species. Recursive modeling (tree-regression) is a useful exploratory technique for uncovering structure in data with variables that may be hierarchical, nonlinear or categorical (Breiman et al., 1984). It is adept at capturing non-additive behavior, where the relationships between the response and predictor variables depend on values of other predictors (Chambers and Hastie, 1992) and is invariant to monotone re-expressions of predictor variables. The technique does not rely on assumptions of linearity, can handle continuous and categorical data and has in several cases been shown to yield more accurate estimates than standard linear-models (Reichard and Hamilton, 1997; Iverson and Prasad, 1998; Rouget et al., 2001). In FIRM, a binary partitioning algorithm recursively splits the data into smaller subsets (nodes) based on the values of the predictor variables until either the node is maximally homogeneous or the node contains too few observations. The grouping of classes is assessed by chi-squared tests between each pair, and the overall significance of the split is assessed by a Bonferroni procedure. Regression trees were generated in S-Plus 6.1 (Insightful, 2001) for general patterns of distribution of the shrubs using species occurrence as the dependent variable and the environmental variables. The regression trees were parameterized with stringent *p*-values to help reduce the number of variables and produce parsimonious models with relevant predictors.

Table 2.2. List of potential predictors used to model distribution and abundance of the two shrub species.

Variable	Code
<i>Geology</i>	
Terminal Continental on marine Oligo-Miocene	1
Holocene Fluvial deposits	2
Miocene-Eocene sedimentary rock	3
Holocene marine deposits	4
Quaternary lacustrine deposits	5
Pleistocene fluvial deposits	6
Lower Eocene (Lutetian) sedimentary rock	7
Upper Eocene (Lutetian) sedimentary rock	8
Holocene fluvial deposits in Pleistocene valleys	9
Lower Eocene (Ypresian) sedimentary rock	10
Miocene sedimentary rock	11
Miocene-Eocene sedimentary rock	12
Undifferentiated Eocene sedimentary rock	13
Quaternary lacustrine deposit on undifferentiated Eocene sedimentary rock	14
Paleocene limestone and sandstones	15
Cretaceous (Maestrichtian) sandstone	16
Pleistocene dunes on Eocene (upper Lutetian) sedimentary rock	17
Pleistocene dunes on lower Eocene (Ypresian) sedimentary rock	18
Pleistocene dunes on Eocene (lower Lutetian) sedimentary rock	19
Quaternary lacustrine deposits on Eocene	20
Pleistocene dunes on Oligocene sedimentary rock	21
Holocene dunes	22
Pleistocene dunes on undifferentiated Eocene sedimentary rock	23
<i>Terrain morphometry</i>	
Slope	Slop
Elevation	Elev
Aspect	Asp
Profile Convexity	ProfC
Plan Convexity	PlanC
Maximum Curvature	MaxC
Minimum Curvature	MinC
<i>Climate</i>	
Mean annual precipitation	MAP
Mean annual temperature	MAT
Average maximum temperature	Tmax
Average minimum temperature	Tmin
<i>Soils<sup>†</sup></i>	

<sup>†</sup> Twelve soils series are mapped in the study area and are not shown here because soil is not a significant factor in our models.



#### 2.4.2. Determinants of abundance

The determinants of abundance within the shrub envelopes were assessed using ordinary logistic regression techniques (Hosmer and Lemshow, 1989). Data were first screened for logically related variables and then for highly correlated pairs of independent variables. Relationships between species abundance and each individual potential explanatory variable (same as those used in the recursive modeling) were investigated by inspection of scatter-plots and by single-variable regression analysis. Each of the explanatory variables was adjusted for all of the others by performing multiple regression. Non-linearity in the relationship between shrub abundance and a predictor variable was explored by adding polynomial terms and then grouping the values of continuous variables into categorical ones. Variable selection for the multiple logistic regression models was carried out by a combination of automatic (stepwise) procedures and goodness-of-fit criteria. An additional criterion for selection of the final model was the degree of spatial correlation of the model residuals.

#### 2.4.3. Influence of management intensity on abundance

Based on distance from a village settlement, three village production rings (compound, bush and savanna rings) with varying management intensities (decline in management intensity with increasing distance from a village) can be identified in the Peanut Basin (Ruthenberg, 1971; Badiane et al., 2000; Manlay et al., 2002). Second-order neighborhood analysis using Ripley's K function (Ripley, 1977; Ripley, 1981; Diggle, 1983) indicated these shrubs exhibit a clustered pattern (summary in Figure 2.3). We therefore hypothesized that shrub clusters in the different production rings should exhibit differences in values of their parameters (e.g. cluster radius, distance between clusters, distance between individual shrubs in the clusters and number of shrubs per cluster) reflecting differences in management intensity.

Six sites, three each for *G. senegalensis* and *P. reticulatum*, were selected for studying shrub cluster parameters along the north-south rainfall gradient. Table 2.3 is a summary of some edaphic factors for the six sites. Once in a selected site, a table of random degrees (45° to 360°

in 45° increments) was used to identify a direction in which a transect, cutting across the three village production rings, was located starting 50 meters away from the homesteads. 20

A 100 m x 100 m quadrat was located well inside each of the three rings (to avoid edge effects) and in cases where the quadrat would not fit, it was oriented perpendicular to the transect. The position of individual shrubs in the quadrat was mapped along an X-Y plane and where shrubs were very close that suckering was suspected, they were considered as one individual. Results from Ripley analysis of the quadrat data were used to derive cluster parameters as follows: the point of maximum deviation from the Poisson expectation in each curve provided an estimate of radius (r) of the cluster (Getis and Franklin, 1987); the point at which the k-function begins to rise gives an estimate of distance to the nearest neighbor (Boots and Getis, 1978); the proportion of the population within h units of one another (number of points per cluster) was estimated using the relationship:  $p(d) = \frac{\pi L(d)^2}{A}$ , where, p is the number of points per cluster, L is Ripley function at a given distance d and A is the area of the study quadrat; dividing the total number of points in the population by the number of points per cluster provided an estimate of the number of clusters in the population.

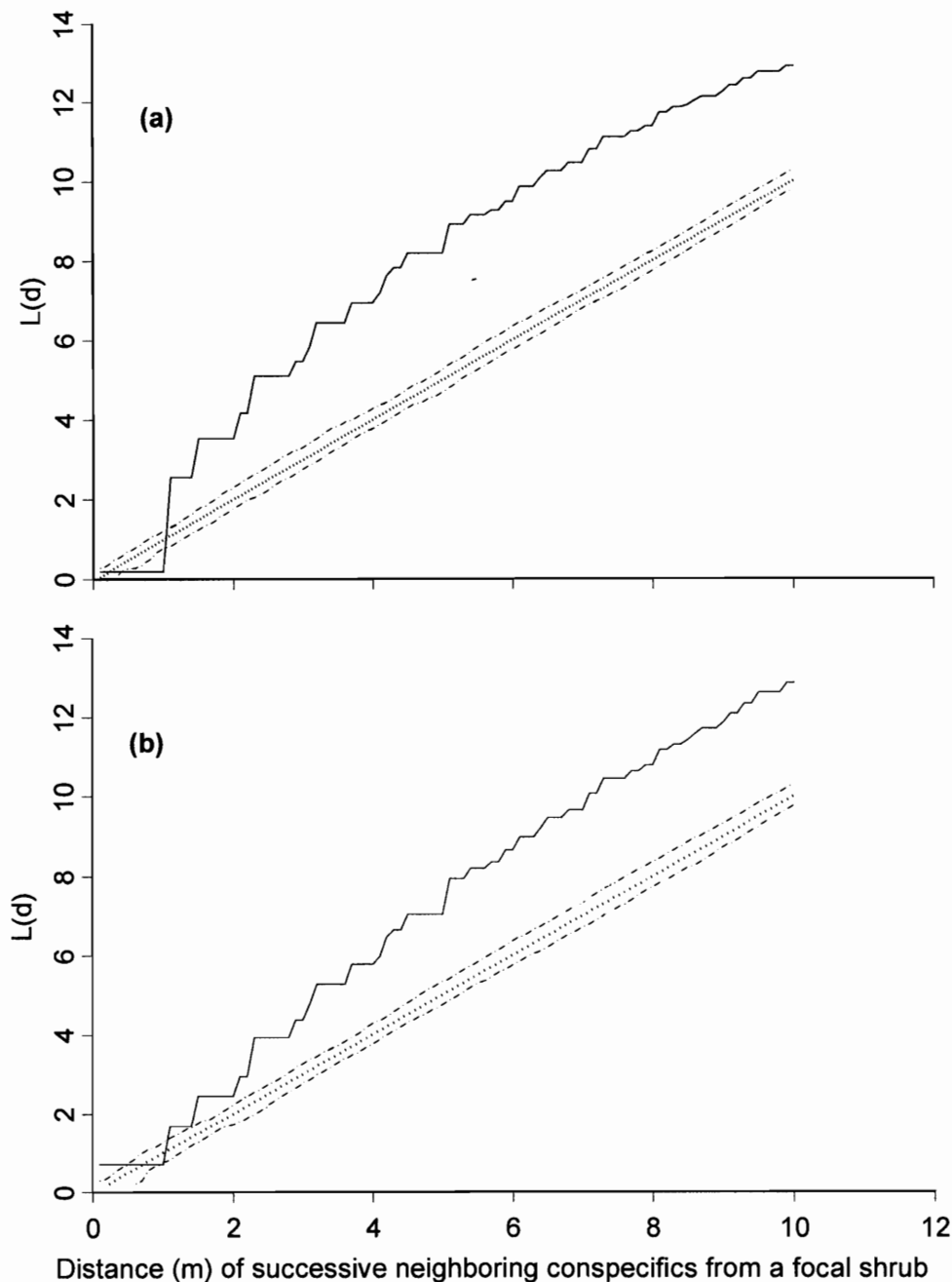


Figure 2.3. Shrub spatial patterns for (a) *G. senegalensis* and (b) *P. reticulatum*. Ripley's  $L(d)$  plotted against  $d$  (distance of successive neighboring shrub from "focal shrub"). (---) is the approximate 95% confidence limits about the expected value (.....) for complete spatial randomness, which = 0. The test of the hypothesis is: If,  $L(d) = d$ , randomness (hence a plot of  $L(d)$  against distance is linear); if  $L(d) > d$ , clustering and if  $L(d) < d$ , regular (Boots and Getis, 1978; Cressie, 1993). Observed values occurring above the upper 95% confidence interval indicate aggregation (i.e. shrubs closer to each other than expected) and values occurring below the lower 95% limit indicate a regular pattern (i.e. shrubs farther from each other than expected).

Table 2.3. Edaphic characteristics of the sites used to study influence of management on shrub abundance.

Shrub species	Village	Elevation (m)	Aspect	Slope (%)	Soil texture (%; 0-40 cm)			Soil classification (USDA)
					Sand	Silt	Clay	
<i>G. senegalensis</i>	Thilla Ounte N14.79, W16.68	27.3	WSW	4 - 9	87.4	9.6	3	<i>Typic Torriorthents</i>
	Tabakali N14.37, W16.06	27.8	N	3 - 6	91	4.0	5	na
	Keur Matar Aram N14.77, W16.86	50.5	NW	1 - 4	70	22	8	<i>Typic Torripsaments</i>
<i>P. reticulatum</i>	Boubname N14.09, W15.62	21.0	NNW	1 - 2	65	20	15	<i>Typic Petrocalcids</i>
	Babanene N14.21, W15.38	26.7	SE	2 - 6	69	19	12	<i>Typic Haplocalcids</i>
	Nioro N13.97, W15.47	27.3	WNW	1 - 5	68.6	14	17.4	na

#### 2.4.4. Prediction and model performance

##### 2.4.4.1. *Presence/absence model*

For each shrub species, decision rules derived from the recursive models were applied to digital maps of the environmental variables in order to assign a 'probability' of species presence (estimated from the proportion of observations in which the species was present in a terminal node of the regression tree) to each cell in the study area. Thresholds, derived from the model, were applied to these probability maps to convert them to binary presence/absence predictions for each species. Accuracy of prediction was assessed using a series of measures derived from an error matrix (Franklin, 2002).

##### 2.4.4.2. *Shrub abundance model*

Derived predictive models were used to produce maps of shrub abundance based on predictor variables available as raster images in ArcGIS. Predicted abundances for known grids were extracted and residuals calculated on the logit scale. Based on variograms (Gotway and Hergert, 1997; Goovaerts, 1999) of these residuals (Figure 2.4), a kriged map of residuals was calculated and added to the predicted values on the logit scale before transforming the result back to abundances. Addition of kriged residuals helps take into account local spatial dependence and allows local deviation from the prediction of the logistic model, if the deviation is supported by observed values in the neighborhood (Isaaks and Srivastava, 1989). This improves the final map in that it does not deviate too severely from the observations, which is particularly important if the model does not adequately explain the observed variation in shrub abundance. Comparisons of model performance were achieved by calculating predictions on a random sample of 50 grids (32 for *G. senegalensis* and 18 for *P. reticulatum*) from the MTMF image, excluding those locations used for model building.

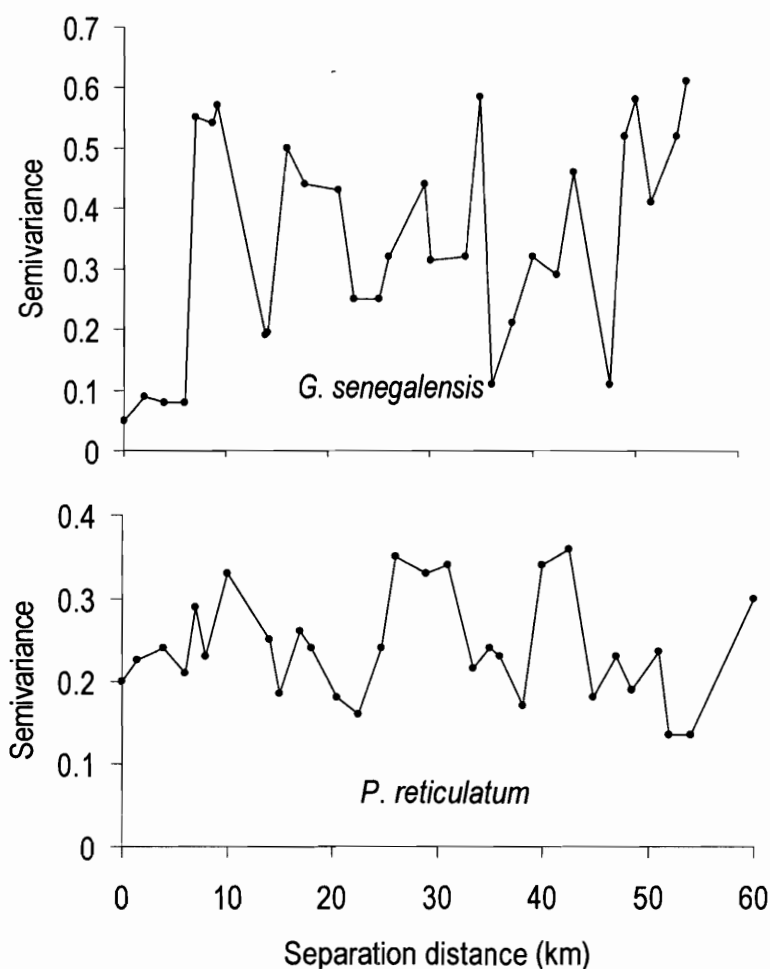


Figure 2.4. Semivariograms of model residuals for *G. senegalensis* and *P. reticulatum*.

### 3.0 Results and Discussion

#### 3.1. Determinants of species distribution

Factors most influencing the distribution of the two shrub species are revealed by the dendrograms of the tree-regression analysis (Figure 2.5). Among the suite of potential predictors, only very few variables significantly influenced shrub distribution. Three factors: geology, MAT and MAP correctly classified more than 75 % of the species distribution. Of the derived terrain morphometric variables, only profile convexity exhibited a discernible effect on shrub presence although this was limited to *P. reticulatum*. Surficial geology was the most important predictor for

both shrub species range, whereas MAT and MAP were secondary and tertiary predictors, respectively. Both shrub species are candidates for slightly different geological substrates although their distribution within the substrates is refined by variations in MAT and MAP (Figure 2.5).

An expectantly important predictor- soil type does not contribute significantly to either shrub distribution model even though there is a difference in soil order between the reference study sites (Table 2.3). This apparent lack of significance of soil in determining shrub distribution is most likely a reflection of the inadequacies of the coarse scale nature of the map, as soil mosaics are intrinsically less than the DEM resolution. Studies show that associations between species distribution and soil type are often evident on large scales (Baillie et al., 1987; Newbery et al., 1988; ter Steege et al., 1993). We suspect that there is co-dependence within the spatial data, as soil order in this area strongly reflects surficial geology age. For example, with respect to the latter, *P. reticulatum* prefers older soils landscapes with carbon-cemented fluvial terraces and Pleistocene dunes.

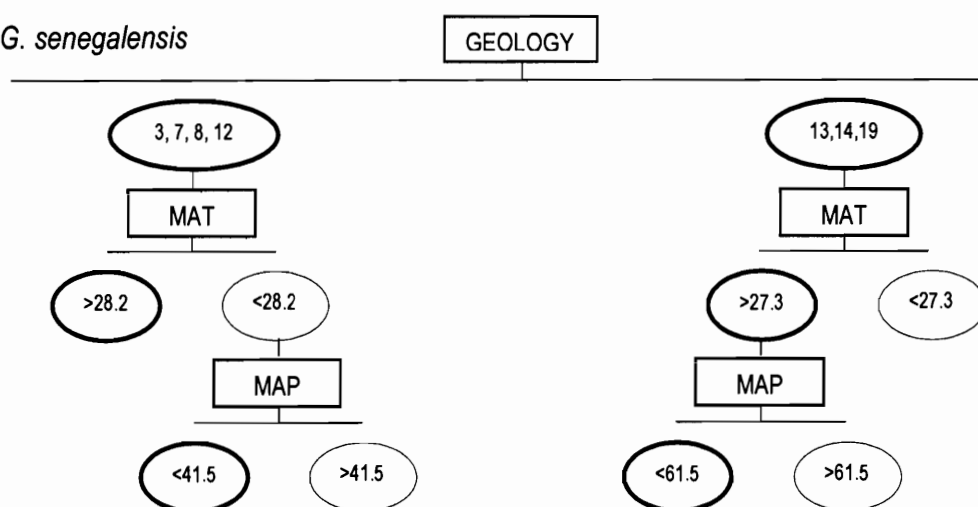
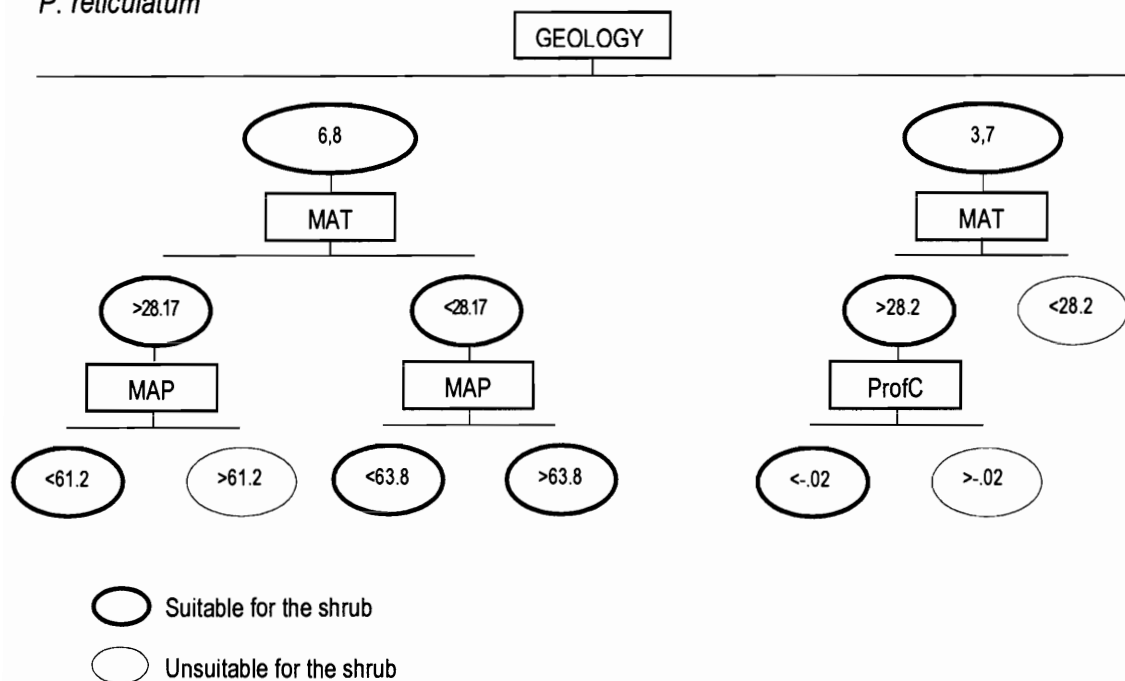
*G. senegalensis**P. reticulatum*

Figure 2.5. Determinants of presence of *G. senegalensis* and *P. reticulatum* in the Peanut Basin. Boxes indicate factors that significantly influence the species distribution while threshold values that distinguish suitable from non-suitable areas are indicated in ellipsoids. The relative importance of the factors decreases with progression down the dendrogram.



The potential environmental envelopes for the two shrub species as derived from the regression tree probabilities are shown in Figure 2.6 and the attendant error matrix for the predictions is provided in Table 2.4.

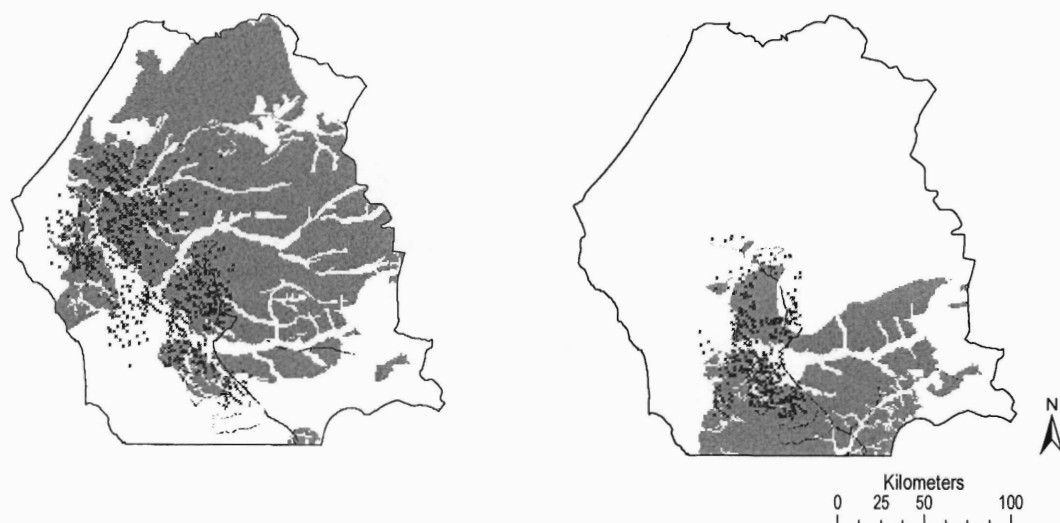


Figure 2.6. Potential environmental envelopes for the shrub species: a). *G. senegalensis*; b). *P. reticulatum*. Shaded areas are suitable for shrub occurrence (probability > 0.5). x denotes areas classified as having the species in the MTMF derived vegetation map whereas black dots are field obtained locations with the shrubs.

*G. senegalensis* exhibits a fairly more ubiquitous distribution as compared to *P. reticulatum* whose range is restricted to the wetter southern parts of the basin (MAP > 580 mm). These predicted ranges of distribution are generally consistent with the known distributions of the subject species. Correct classification rates for predicting shrub presence were relatively low for both shrub species. The high rate of *P. reticulatum* false positives suggests that the model overestimates its presence, whereas the high rate of false negatives for *G. senegalensis* indicates that its extents are underestimated by the model. Sensitivity was higher than specificity (Table 2.4) for both shrub species indicating that shrub presence was predicted more reliably than absence. The low prediction accuracy could be an artifact of several factors. Many studies (e.g. Gordon and White, 1994) show that the environmental conditions where a species is found are very broad and this inherently limits the ability of static/probabilistic models of site conditions to predict species distributions (Decoursey, 1992; Korzukhin et al., 1996; Lischke et al., 1998; Franklin, 2002).

The modest prediction accuracy may also reflect misclassification errors in the Landsat-derived vegetation map, spatial uncertainties because of interpolation errors and/or effects of combining explanatory variables of differing scale sensitivities.

Table 2.4. Error matrix and accuracy measures for the shrub predictions.

		Actual	
		Present	Absent
Predicted			
<i>G. senegalensis</i>			
	Present	132	34
	Absent	52	41
N = 259; Sensitivity = 71%; Specificity = 55%; Positive Predictive Power = 79%; Correct Classification Rate = 67%			
<i>P. reticulatum</i>			
	Present	83	41
	Absent	15	26
N = 165; Sensitivity = 84%; Specificity = 39%; Positive Predictive Power = 67%; Correct Classification Rate = 66%			

### 3.2. Determinants of species abundance

A summary of the factors influencing shrub abundance across the landscape is given in Table 2.5. The selected logistic models contain three (MAP, maximum annual temperature and elevation) and four (MAP, MAT, elevation and profile convexity) significant explanatory variables respectively for *G. senegalensis* and *P. reticulatum*. The models reveal a complex relationship between shrub abundance and the environmental/edaphic factors. The difference in significant variables in the two shrub models indicates differences in factors responsible for variation in either shrub abundances. The proportion of explained deviance is 51% and 38% respectively for *G. senegalensis* and *P. reticulatum* models implying that considerable amount of variation in shrub abundance is unaccounted for. The resulting maps of shrub abundance are shown in Figure 2.7.

Table 2.5. Factors associated with *G. senegalensis* and *P. reticulatum* abundance in the Peanut Basin.

Variable	Odds ratio <sup>†</sup>	Confidence interval
<i>G. senegalensis</i>		
MeanR	2.980	1.610-5.510
Tmax	0.227	0.081-0.634
Elev	1.250	1.230-1.273
<i>P. reticulatum</i>		
MeanR	1.012	1.005-1.020
MeanT	1.032	1.013-1.052
Elev	0.973	0.959-0.987
ProfC	0.180	0.070-0.470

<sup>†</sup>Odds ratios not adjusted for spatial dependency. The ratios were calculated from the models:  $\log \text{Abundance} = 1.092 * \text{MeanR} - 0.0181 * \text{Tmax} + 0.0000095 * \text{Elev}$  for *G. senegalensis* and  $\log \text{Abundance} = 0.012 * \text{MeanR} + 0.031 * \text{MeanT} - 0.028 * \text{Elev} + 0.467 * \text{ProfC}$  for *P. reticulatum*.

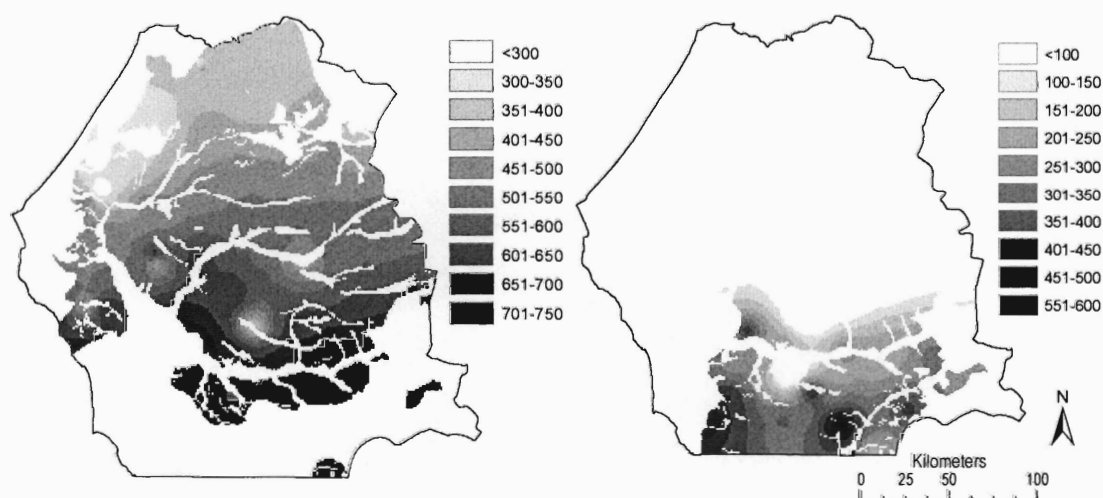


Figure 2.7. Predicted shrub abundance based on the logistic regression model and krigged residuals. The predictions were restricted to modeled environmental envelopes. Left). *G. senegalensis*; Right). *P. reticulatum*.

Of the 32 grids of *G. senegalensis*, 22 were within  $\pm 50$  counts of their known abundance, 12 out of the 18 *P. reticulatum* grids were within this count range of their known abundance. A significant proportion of this variation is likely to be noise due to errors propagated from the original vegetation map that yielded shrub abundances. The plausibility of the models is constrained by confounding between explanatory variables (which are often highly correlated with each other) leading our interpretation to be putative to some extent. In addition, these models are not hypothesis-driven (not derived from explicit biological mechanisms) but rather are data-driven. This could have led to chance associations which have no biological or climatic explanation. We curtailed this possibility by using a stringent  $p$ -value ( $p = 0.01$  instead of the typical 0.05) making it more difficult for variables to enter the regression model. Therefore, even if the interpretation is not very obvious, the relationships documented here are strong associations, which provides evidence for some mechanisms that are responsible for variation in either shrub abundance. No doubt, recourse to the realm of the respective shrub physiology is the leeway to unravel these associations and this can best be done by controlled experiment. Within the study area, finer-scale variables related to soil properties and site history might be expected to improve model performance.

### 3.3. Influence of management intensity on shrub abundance

Table 2.6 is a summary of cluster parameters for the entire set of quadrats studied. We expect the overall distribution of cluster radius and distance to nearest neighbor to be biased towards smaller clusters and greater distances to nearest neighbors in areas proximal to the village center. This difference was expected on the basis of intensified continuous cultivation in the compound rings (Manlay et al., 2002) as opposed to longer periods of fallow as one moves from the bush ring to the savanna ring (Manlay, 2000). However, there was no consistent pattern that differentiated between the cluster radius and distances to nearest neighbor distributions of shrubs in the different village production rings. Even though the cluster density estimates for the shrubs differed between village production rings, patch radii within each species was not significantly different ( $p$ -value from one-way ANOVA = 0.335 and 0.817 for *G. senegalensis* and *P. reticulatum*, respectively). Thus the estimates of patch parameters appear to demystify our hypothesis that management intensity has an influence on the abundance of the shrubs at the landscape level.

Table 2.6. Comparison of cluster parameters for the shrubs in the different sampling locations.

Shrub species	Site name	Distance from village (m)	No. of shrubs	Cluster density m <sup>-2</sup>	Cluster radius(m)	No. of shrubs/ cluster	Nearest neighbor distance (m)
<i>G. senegalensis</i>	Thilla Ounte	50	129	0.000806	5.8	16	1.05
		464	234	0.000585	6.6	40	1.21
		655	206	0.000665	3.3	31	1.10
	Tabakali	50	386	0.001608	5.9	24	0.90
		281	561	0.001145	8.4	49	1.00
		543	566	0.001204	7.5	47	1.16
	Keur Matar Aram	50	164	0.000586	5.6	28	1.30
		212	479	0.001261	6.1	38	1.07
		487	487	0.001218	4.8	40	1.22
<i>P. reticulatum</i>	Boubname	50	102	0.000785	6.2	13	1.15
		433	189	0.000591	4.7	32	1.00
		802	167	0.000642	7.4	26	1.00
	Babanene	50	78	0.00078	3.8	10	0.98
		398	201	0.000558	6.4	36	1.11
		676	237	0.000608	4.2	39	1.00
	Nioro	50	124	0.000729	7.3	17	1.20
		378	344	0.000905	4.8	38	1.05
		708	314	0.000766	6.7	41	1.15

In fact, it seems very likely that the manner of management (the annual pruning notwithstanding) in these Parkland systems is such that it preserves the shrubs in the landscapes due to the importance the farmers attach to them in replenishing the fertility of soils (A.Sene pers. comm., 2004). This notion is buoyed by the lack of a significant difference ( $p$ -value = 0.324 and 0.065 respectively for *G. senegalensis* and *P. reticulatum* from a one-way ANOVA) in number of shrubs in the different village production rings.

#### 4.0 Conclusion

Although the statistically based models used in this study were not derived from explicit biological mechanisms, they offered a tractable theoretical framework for analyzing and quantifying the spatial structure of two shrub communities, *G. senegalensis* and *P. reticulatum* in the Peanut Basin. Our results are consistent with those of Woodward (1987), Prentice et al. (1992), Reed et al. (1993), and Bridge and Johnson (2000) in confirming the importance of edaphic factors in mediating regional patterns of species distribution and abundance. Results argue for a strong role of water supply and temperature in mediating both distribution and abundance of shrubs. This information is of utility in assessing the shrub dynamics and implicitly ecosystem functioning (e.g. carbon stabilization and soil degradation mitigation) of these communities under different climate scenarios. Addition of measurements of energy (i.e. temperature variables), water availability, and climate seasonality is likely to contribute significantly to the accuracy of the models. Additionally, properly conducted physiological experiments are needed to verify the associations reported here in relation to the ecology of the shrubs.

#### 5.0 Acknowledgements

This study was conducted under the auspices of the US National Science Foundation Grant No. 0120732. We thank P.S. Sarr of ISRA for helping out in data collection, Mactar Gueye of CSE for his assistance with the GIS databases, Julia Jones for her assistance with geostatistics and Joan Sandeno for manuscript editing.

### CHAPTER THREE

#### **Carbon stocks and patterns in native shrub communities of Senegal's Peanut Basin.**

A. Lufafa , I. Diedhiou, N.A.S. Ndiaye, M. Sene, M. Khouma, F. Kizito, R.P. Dick and J.S. Noller

Geoderma (in review)  
P.O. Box 1930  
1000 BX Amsterdam  
The Netherlands

## Abstract

Accurate and reliable estimates of carbon (C) storage in landscapes is critical to the development of effective policies and strategies to mitigate atmospheric and climate change. Carbon stocks of two native woody shrub (*Guiera senegalensis* J.F. Gmel and *Piliostigma reticulatum* (DC.) Hochst) communities and associated soils within Senegal's Peanut Basin were determined and the spatial structure of soil C quantified. These shrubs are of interest because they dominate semi-arid sub-Saharan Africa and commonly coexist with row crops but have been largely overlooked as a key vegetative component of this landscape. Peak-season shrub biomass C was measured in forty-five 0.81 ha plots at 8 locations using allometric relationships along with soil sampling (0 to 40 cm depth) and analysis for organic C and bulk density. Soil samples to a depth of 20 cm were taken every two meters in 24 m x 20 m grids and every 0.5 m in four nested 3 m x 3 m grids containing at least one shrub or tree canopy and geostatistical techniques used to quantify scale and degree of soil organic carbon (SOC) spatial dependence. Estimates of peak-season biomass C ranged from 0.9 Mg C ha<sup>-1</sup> to 1.4 Mg C ha<sup>-1</sup> with an overall mean of 1.12 Mg C ha<sup>-1</sup> (SEM = ±0.079) in the *G. senegalensis* sites and from 1.3 to 2.0 Mg C ha<sup>-1</sup> (mean = 1.57 Mg C ha<sup>-1</sup>; SEM = ±0.18) in the *P. reticulatum* communities. The overall mean of SOC to 40 cm was 17 and 17.2 Mg C ha<sup>-1</sup> respectively, at the *G. senegalensis* and *P. reticulatum* sites with 57% of that C residing in the top 20 cm. Semivariograms of soil organic carbon (SOC) showed moderate spatial dependence and spatial autocorrelation at distances of less than 0.56 and 1.34 m at the *G. senegalensis* and *P. reticulatum* sites, respectively. Comparison across the different grids showed that the presence of shrub canopies at either site had much closer relationship to SOC levels than trees.



## 1.0 Introduction

Assessment and improved understanding of total system carbon (C) stock and its individual components in Senegal's Peanut Basin is important for understanding biogeochemical processes in this ecosystem, improving soils for crop productivity, and practical strategies to sequester C in soils. Carbon lost from the Parkland agroforestry systems in this basin significantly contributes to atmospheric change, particularly increases in carbon dioxide concentrations (Houghton et al., 1993). Therefore a quantification of C stocks for different land management systems allows for better estimates of these C losses to the atmosphere as land degradation patterns are compared over time (Lal, 2002). Besides the global goal of mitigating elevated atmospheric carbon dioxide, sequestering C in soils would be of interest to land managers by improving soil properties (Woomer et al., 1997). These benefits could include increased land productivity, better yields and also contribute to improved overall soil quality and health, which in turn can help buffer these inherently fragile ecosystems (Bationo, and Mkwunye, 1991; Brouwer and Bouma, 1997) from abiotic stresses (Elliot et al., 1993; Woomer et al., 1994; Murage, et al., 2000).

The policies and scientific research/actions concerned with C cycling depend on accurate information about spatial distribution of C in vegetative and soil components of the terrestrial ecosystems. The Kyoto Protocol (1997) presents an internationally negotiated framework for guiding these policies. Lal et al. (1999), and Lal (2002) argue that Article 3.3 and Article 3.4 of the protocol provides rationale for the importance of managing drylands to sequester C via two key mechanisms, restoration of desertified lands (Lal, et al., 1998) and the promotion of perennial woody biomass (Manley et al., 1995).

The Peanut Basin of Senegal is located on the Sahel's north/south vegetative gradient between the sparsely wooded grasslands of the north and tree-dominated ecosystems to the south. It is characterized by intensively cultivated Parkland systems (Freeman, 1992) comprising mainly of randomly dispersed trees and woody shrubs in farmers' fields (Weber and Major, 1984). Depending on geological substrate, temperature and rainfall amounts, the woody shrub component covers the entire landscape in some areas and normalized difference vegetation indices (NDVI) show it is the single largest vegetation component in this region (A. Lufafa pers. comm., 2005).

*Guiera senegalensis* and *Piliostigma reticulatum* dominate this shrub component. In farmers' fields these shrubs are normally pruned back to the soil surface and the residue is burned in the spring of every year prior to cultivation but the shrubs will continue to grow if left uncut. Consequently, because of their dominance in the landscape and coexistence with cropping activities, these shrubs have the potential to be a significant source or sink for C within the global cycle, depending on land use and management. A number of studies of C stocks in the Peanut Basin have been done that have had a range of estimates for the vegetative component (Woomer, 1993; Bationo et al., 1998; Manlay, 2000; Bationo and Buerkert, 2001; Batjes, 2001; Manley et al., 2002; Liu et al., 2004; Tschakert et al., 2004; Woomer et al., 2004a; Woomer et al., 2004b). However, these studies have had various levels of resolution and generally neglected the importance of the shrub component. We hypothesized that the shrubs, because of their abundance, are more important than the tree component or other sources of C such as animal manure in regulating the C stocks of the Peanut Basin. The objective of this study was to quantify shrub biomass C and evaluate the relative influence of the shrubs and trees on the spatial dependence of soil C levels.

## **2.0 Materials and Methods**

### **2.1. Study area characteristics**

The study area is the Peanut Basin in Senegal, West Africa. Located east of Dakar, the area center lies approximately at 16°W, 14.7°N with a spatial coverage of ~44000sqkm. The climate is semiarid, with on average more than 85% of precipitation falling between August and October. Mean annual precipitation is approximately 540 mm, skewed towards the south and with high variability from year to year (Dacosta, 1989). The mean annual minimum and maximum temperatures are 20°C and 34°C respectively, with a marked seasonal variation. Geological substrates in the area include mainly aeolian deposits of Harmattan wind sand (Herrmann, 1996) of Quaternary age over sedimentary rocks of Cretaceous to Miocene age (Monciardini, 1966), and highly eroded colluvial-alluvial ferruginous sediments derived from paleosols (Neogene) and Precambrian bedrock (Renaud, 1961; Michel, 1973). Basin soils are sandy, classifying as *Psammets* and *Calcids* according to Soil Taxonomy (2003) and fall broadly into two indigenous types, i.e. Dior and Deck (Badiane et al., 2000). Basin vegetation is shrubland with scattered trees (Diouf and Lambin, 2001).

## 2.2. Carbon stock estimation

### 2.2.1. Biomass C estimation

The peak-season standing biomass of shrubs in farmers' fields occurs in late spring just before the shrubs are pruned back to prepare for the summer cropping season. Thus we assessed shrub C stocks at this time of year. The procedure was to develop allometric equations that use easily measured shrub properties to estimate biomass. In turn, plot scale inventories of shrub densities in combination with allometric equations were done to estimate shrub biomass.

#### 2.2.1.1. Allometric equations

The allometric equations were developed based on 75 shrubs (49 *G. senegalensis* and 26 *P. reticulatum*) sampled from eleven sites that captured the north-south rainfall gradient in the Peanut Basin. At each site, shrub samples were chosen to reflect the full range of heights, diameters, forms and shrub decadence levels encountered. Measurements of maximum height (*maxht*), mean canopy diameter (*mcd*), mean diameter of the shrub base (*mbd*), and total number of stems (*stems*) were recorded for individual shrubs at their peak growth in March (Breman and Kessler, 1995; Ker, 1995). The shrubs were clipped and roots were excavated to measure biomass in the above-and below-ground components. Samples (leaves, stems and roots) of the biomass components were oven dried at 105°C for 48 hours to determine biomass on a dry weight basis.

The data set obtained was randomly split to create an independent set of 12 *G. senegalensis* and 6 *P. reticulatum* samples that was used for validation of the biomass predictive models. Linear ( $\log Y = \beta_0 + \beta_1 X_1$ ), logarithmic ( $Y = \beta_0 + \beta_1 \log X$ ), exponential ( $\log Y = \log \beta_0 + X^{\beta_1}$ ), log-log ( $\log Y = \log \beta_0 + \beta_1 \log X$ ), and quadratic ( $Y = \beta_0 + \beta_1 X + \beta_2 X^2$ ) regression models ( $Y$  = dry weight of biomass in grams,  $X_1 \dots X_n$  are the respective explanatory variables in each model e.g. *maxht*, *mcd*, *mbd*) were fit to the obtained data. Optimal equations were selected based on adjusted  $R^2$  values and independent variables that maximized the significance ( $p$ -value) of the regression coefficients. The validation data set was used to evaluate the predictive capacity of the regression estimators (Neter et al., 1996) and to select the final equations.

### 2.2.1.2. Shrub Inventories

Shrub biomass inventories were performed in 90 m x 90 m plots that were originally designed to derive relationships between remotely-sensed shrub abundance and landscape level biomass C stocks (A.Lufafa pers. comm., 2005). Eight sites (6 for *G. senegalensis* and 2 for *P. reticulatum*) with varying number of plot replicates (more plots at sites with great variation in shrub density) were selected for the inventory. The location, topography and number of replicates at each of the sites are provided in Table 3.1. Measurements of biomass predictive variables (as adduced from the allometric equations) were recorded on all shrubs encountered in each plot and used to assign biomass to each shrub. The proportion of C in all biomass pools was assumed to be 0.47 (Skog and Nicholson, 1998).

### 2.2.2. Soil C estimates

Total soil organic C ( $\text{Mg ha}^{-1}$ ) to 40 cm depth in the plots was calculated from measurements of C concentrations ( $\text{g C kg}^{-1}$ ) of the 0-20 and 20-40 cm soil layers and soil bulk density at 15 and 30 cm depths. In each of the 90 m x 90 m plots, 15 sampling points were located along three transects positioned approximately 30 m apart. Soils were collected from these sampling points with four sub-samples bulked and mixed. Bulk density was measured as described by Okalebo et al. (2002) at 15 and 30 cm depths. Samples for soil C were air dried and analyzed for total C by combustion on a LECO C-144 C analyzer (LECO Inc., St. Joseph, Michigan). No attempts were made to correct for carbonate as soils in the study area are predominantly acidic ( $\text{pH} < 7$ ) (Tschakert, 2004).

Table 3.1. Location, elevation, number of plot replicates and average number of shrubs at the biomass inventory sites.

Shrub species	Location, Lat/Long	Elevation(m)	Replicates	Shrubs/ha†
<i>G. senegalensis</i>	Keur Asane Lo N14.78, W16.74	34.3	3	275 (59.7)
	Keur Mandiamba N14.75, W16.67	46.7	4	409 (35.7)
	Keur Matar Aram N14.77, W16.86	50.5	7	239 (74.6)
	Keur Ibra Fall N14.75, W16.76	25.1	2	312 (162.0)
	Ndiagne N14.76, W16.77	22.5	5	407 (36.0)
	Thilla Ounte N14.79, W16.68	27.3	3	228 (42.0)
<i>P. reticulatum</i>	Sikatrou N13.98, W15.99	24.0	8	134 (20.9)
	Sanguel N14.03, W16.04	23.5	13	288 (22.4)

† Standard error of the mean in parentheses

## 2.2. Soil C spatial structure

The goal of this sub-study was to quantify and determine the relative influence of the shrubs and trees on soil C spatial patterns. Across the landscape, shrubs are interspersed among different tree species and we hypothesized that tree type could be an influence on soil C. As a preface to this study, soil C under canopies of the most common trees coexisting with the shrubs was determined. Results (not presented) showed no statistical difference in soil C due to tree type enabling our study to be representative of any tree/shrub community in our study area. Two sites, one each for *G. senegalensis* and *P. reticulatum* were selected for this study. A 24 m x 20 m grid (selected to capture at least two trees and shrubs) was established at either shrub study site and points sampled every 2 m (for a total of 143 samples). Within this major grid, samples were taken at a finer scale of 0.5 m within four small nested 3 m x 3 m grids (198 fine scale samples, a total of 339 samples for the entire site). Each of the smaller grids contained at least one shrub or a tree canopy.

Soil samples were taken to a depth of 20 cm and analyzed for total C. Geostatistical analyses (Goovaerts, 1999) were conducted on the data to quantify the patterns of spatial variation in soil C across the “shrub-tree” sites. Semivariograms (Isaaks and Srivastava, 1990; Atteia et al., 1994; Wackernagel, 1994; Gotway and Hergert, 1997) were first calculated (in Splus) using all 339 points at a site to provide an unbiased description of the scale and pattern of soil C spatial variation (Oliver and Webster, 1986). Semivariograms were recalculated to examine the effect of the shrubs and trees on soil C patterning. First, the 98 data points from the two small grids around the shrubs were deleted and semivariograms calculated on the remaining data; then, the 98 points from the two small grids around the trees were deleted and semivariograms recalculated. The semivariance,  $\gamma(h)$ , is calculated for each specific lag distance  $h$  as:

$$\gamma(h) = \frac{1}{2N(h)} \sum_{i=1}^{N(h)} [z(x_i) - z(x_i + h)]^2 \quad (1)$$

where  $N(h)$  is the number of pairs of points separated by distances  $h$ ,  $z(x_i)$  is the measured sample value at point  $x_i$ , and  $z(x_i + h)$  is the sample value at point  $x_i + h$ . The minimum pair distance used was 0.5 m and the maximum was 12 m (roughly half the maximum distance available from the data). Least squares estimates of the parameters  $\theta = (C_0, C_0 + C_1, a)$ , where  $C_0$  is the nugget (variance that is not spatially dependent),  $C_0 + C_1$  is the sill (semivariance corresponding to the range), and  $a$  is the range (scale of spatial autocorrelation), were obtained by fitting the sample variogram  $\hat{\gamma}(h)$  to the variogram model  $\gamma(h; \theta)$ . The least squares estimator for  $\theta$  is obtained by finding the  $\hat{\theta}$  that minimizes the *ad hoc* criteria:

$$D(\theta) = \sum_{h=0}^{h_0} n(h) \{ \hat{\gamma}(h) - \gamma(h; \theta) \}^2 \quad (2)$$

where  $n(h)$  is the number of observations used in computing the sample variogram when using weighted least squares, and  $h_0$  is the maximum distance used in fitting the variogram model.

Depending on the best reduced sum of squares, semivariograms were fit to linear, spherical or exponential models (Cressie, 1985; McBratney and Webster, 1986; Zhang et al., 1992). Model parameters were used to evaluate the magnitude of spatial dependence and the scale of spatial autocorrelation for the variate (Robertson, 1987; Robertson et al., 1988; Jackson and Caldwell, 1993a).

### 3.0 Results and Discussion

#### 3.1. C stock estimation

##### 3.1.1. Allometric relationships

Studies elsewhere have found log-log and quadratic equations most useful for predicting biomass for a number of shrub species (Bentley et al., 1970; Rittenhouse and Sneva 1977; Bryant and Kothmann, 1979). In this study, log-linear relationships produced the highest coefficients of determination with random residuals and significant two-sided  $p$ -values ( $p > 0.05$ ). Table 3.2 is a summary of the regression coefficients and the corresponding mean square error of prediction (MSEP) for the selected models. The fit of *maxht* and *mcd* on *G. senegalensis* aboveground biomass yielded an  $R^2$  value of 0.90, while the same fit yielded a coefficient of determination of 0.87 for *P. reticulatum* aboveground biomass estimates. The best fit models for shrub belowground biomass were those that incorporated *mbd* for *G. senegalensis* ( $R^2 = 0.69$ ) and *mcd* for *P. reticulatum* ( $R^2 = 0.81$ ). Mean crown diameter (*mcd*) was equally the best predictor of *P. reticulatum* total biomass ( $R^2 = 0.83$ ) while for *G. senegalensis*, total biomass was best predicted by a combination of *mcd* and *mbd* ( $R^2 = 0.80$ ).

Significant correlations between observed and predicted biomass were observed and these relationships are plotted in Figure 3.1. While the regressions of observed and predicted biomass are significant, there are differences between the regression line and unity (1:1 relationship). The predictive models for the *G. senegalensis* belowground fraction and total biomass over estimated biomass and had considerably greater scatter in the medium-sized shrubs. The *G. senegalensis* aboveground fraction equation and all the equations for *P. reticulatum* were more accurate with regression of observed vs. predicted values yielding slopes close to 1 (although the intercepts are not quite zero). *G. senegalensis* total biomass predicted using the "total biomass" equation was consistently lower than total biomass from the summation of above-and belowground predictions, although the differences were not statistically significant (two-sided  $p$ -value = 0.72). The opposite was true for *P. reticulatum* with the "total biomass" equation consistently predicting more biomass as compared to the biomass prediction from the summation of the above and belowground predictions (two-sided  $p$ -value = 0.74).

Table 3.2. Selected model coefficients of determination ( $R^2$ ), MSEPs and Pearson correlation coefficients ( $r$ ) for observed and predicted biomass.

Shrub species	Biomass model	Regression		Prediction	
		MSE	$R^2$	MSEP	$r$
<i>G. senegalensis</i>					
	LogAbgm = 4.39+(0.0056* <i>maxht</i> )+(0.011* <i>mcd</i> )	0.08	0.90	0.03	0.98
	LogBgbm = 7.41+(0.015* <i>mbd</i> )	0.16	0.69	3.64	0.88
	LogTbm = 7.14+(0.0054* <i>mcd</i> )+(0.0083* <i>mbd</i> )	0.10	0.80	2.99	0.91
<i>P. reticulatum</i>					
	LogAbgm = 4.20+( 0.013* <i>maxht</i> )+(0.0059* <i>mcd</i> )	0.07	0.87	0.29	0.99
	LogBgbm = 7.14+(0.0099* <i>mcd</i> )	0.11	0.81	2.60	0.99
	LogTbm = 7.31+(0.0098* <i>mcd</i> )	0.09	0.83	2.09	0.99

Abgm = Aboveground dry weight biomass; Bgbm = Belowground dry weight biomass; Tbm = Total dry weight biomass; *maxht* = shrub maximum height; *mbd* = mean shrub basal diameter; *mcd* = mean shrub crown diameter.



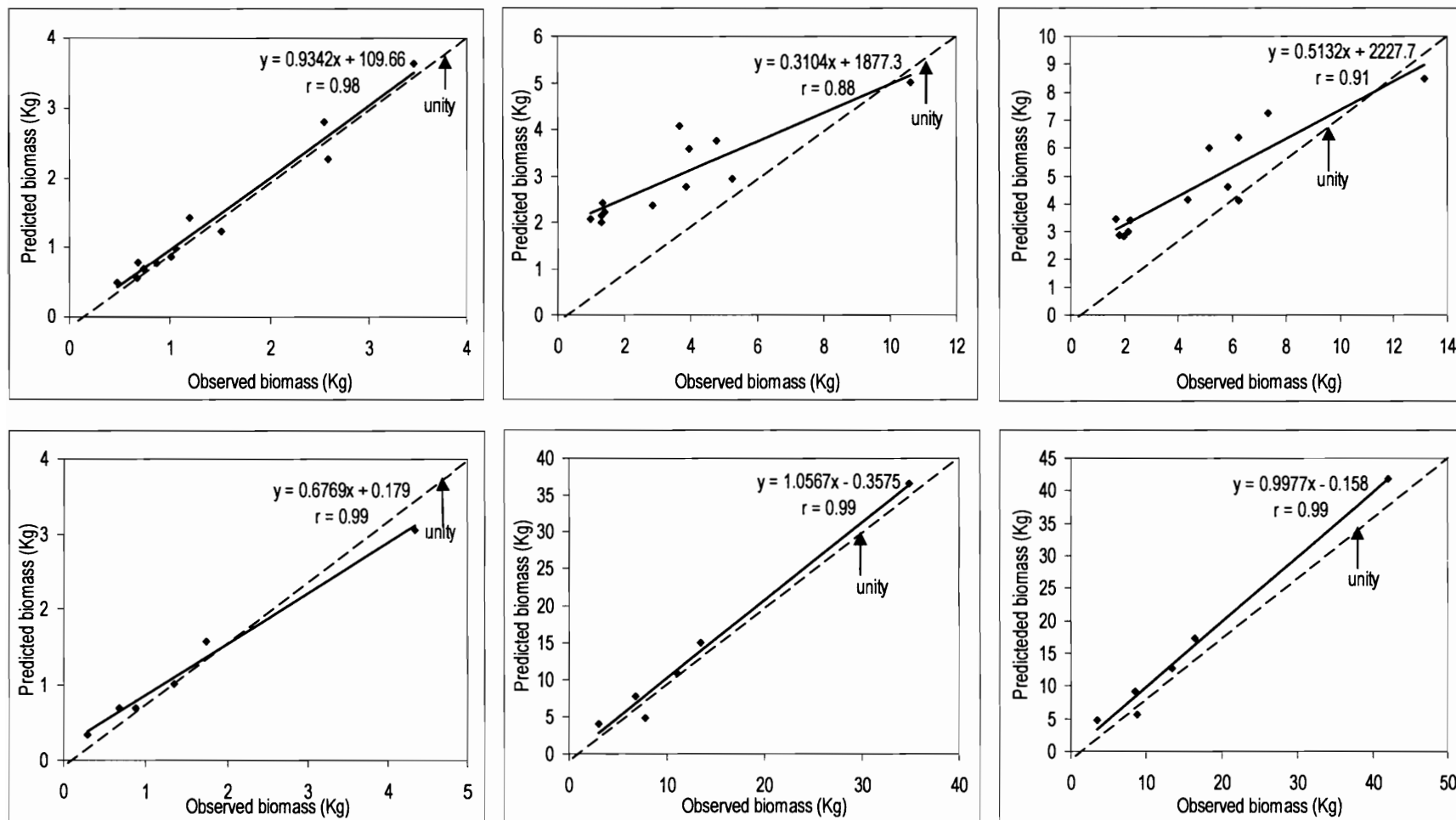


Figure 3.1. Unity plots of observed and predicted biomass. Top left to right: aboveground, belowground and total *G. senegalensis* biomass. Bottom left to right: aboveground, belowground and total *P. reticulatum* biomass.

### 3.1.2. Biomass C stocks

Total biomass C stock at the *G. senegalensis* sites ranged between 0.93 and 1.40 Mg ha<sup>-1</sup> (Table 3.3) with an overall mean of 1.12 Mg ha<sup>-1</sup> (SEM =  $\pm 0.079$ ). Approximately 82% of this C was belowground with only 18% allocated aboveground in leaves and stems. The belowground fraction constitutes 86% of total (mean = 1.57 Mg ha<sup>-1</sup>) biomass C stock at the *P. reticulatum* sites, whereas a relatively smaller proportion (0.15 Mg ha<sup>-1</sup>) resides in the aboveground fractions. Significant correlation (where belowground biomass C =  $2.2418 \times \text{aboveground C stock} + 0.4592$ ;  $r = 0.89$ ) was observed between averaged aboveground and belowground biomass C stocks across the *G. senegalensis* sites. Derivation of these relationships was not possible for the *P. reticulatum* sites because of sample-size limitations, but regressions of individual plot above- and below-ground biomass C stocks were significant (where belowground biomass C =  $10.72 \times \text{aboveground C stock} - 0.099$ ;  $r = 0.94$ ).

There was no statistical difference in above or belowground biomass C either within site or across site for *G. senegalensis*. Mean aboveground biomass C stocks were statistically similar within site but different across the two *P. reticulatum* sites (LSD = 0.061). No statistical difference was observed in belowground biomass C either within or across these two sites. Comparison across shrub type reveals a statistical difference between belowground biomass C ( $p$ -value = 0.038) and no differences between aboveground biomass C stock for the two shrub species. Number of shrubs significantly correlated ( $p < 0.001$ ) with C stocks in all *P. reticulatum* biomass fractions (Table 3.4) and warrants consideration as a surrogate indicator of C stocks. There were no significant relationships (data not presented) between *G. senegalensis* shrub numbers and any of the biomass C fractions, probably reflecting our tenuous ability to isolate single shrub entities without excavation.

To determine the amount of biomass that is pruned upon reclearing (when the regrowth is dense enough to impede crop growth), we selected and pruned shrubs in demarcated plots at peak-season and left them to regrow in line with farmers' practice. These were then clipped again at a time coincident with farmers pruning and the amount of biomass was measured. Table 3.5 shows the average aboveground biomass regrowth for the shrubs with average values of 12% and 22% as a proportion of peak-season biomass respectively, for *G. senegalensis* and *P. reticulatum*.

Table 3.3. Peak-season biomass, soil and system C stocks in native shrubs in Senegal's Peanut Basin. Ranges within each site are in parentheses.

Shrub type/Location	Biomass C		Soil C		Total C
	Aboveground	Belowground	0-20 cm	20-40 cm	
	-----Mg ha <sup>-1</sup> -----				
<i>G. senegalensis</i>					
Keur Asane Lo (n = 3)	0.184 (0.079)	0.842 (0.393)	9.86 (23.61)	6.63 (7.12)	17.516
Keur Mandiamba (n = 4)	0.181 (0.089)	0.861 (0.367)	9.50 (28.14)	6.99 (18.08)	17.532
Keur Matar Aram (n = 7)	0.244 (0.353)	1.131 (1.853)	8.37 (17.89)	6.84 (8.12)	16.585
Keur Ibra Fall (n = 2)	0.309 (0.192)	1.094 (0.278)	12.03 (30.59)	7.05 (9.88)	20.483
Ndiagne (n = 5)	0.217 (0.249)	0.925 (0.473)	9.95 (32.34)	7.25 (16.25)	18.342
Thilla Ounte (n = 3)	0.150 (0.017)	0.783 (0.473)	9.95 (43.48)	7.58 (30.39)	18.463
LSD	n.s.	n.s.	n.s	n.s.	
<i>P. reticulatum</i>					
Sikatrou (n = 8)	0.107 (0.219)	1.108(2.384)	11.27(68.56)	9.03 (60.27)	21.52
Sanguel (n = 13)	0.173 (0.179)	1.757(1.969)	8.07(28.05)	6.02 (11.31)	16.02
LSD	0.061	n.s.	1.320	0.987	

Table 3.4. Regression relationships between shrub numbers and *P. reticulatum* biomass C stocks as Mg ha<sup>-1</sup>.

Dependent variable	Regression equation	<i>r</i>	<i>p</i>
Aboveground biomass C	$y = 0.0312 + 0.0005 \times \text{number of shrubs}$	0.75	<0.001
Belowground biomass C	$y = 0.293 + 0.0053 \times \text{number of shrubs}$	0.74	<0.001
Total biomass C	$y = 0.334 + 0.0062 \times \text{number of shrubs}$	0.75	<0.001

We did not quantify surface litter C because movement of this component by wind across the landscape made it difficult to trace its source with reasonable accuracy. However, work conducted elsewhere in Senegal (Manlay et al., 2002; Woerner et al., 2004b) shows that this may be a relatively large C pool ranging from 493 – 980 kg ha<sup>-1</sup>.

Table 3.5. Peak-season and regrowth biomass for *G. senegalensis* and *P. reticulatum* at selected sites in Senegal's Peanut Basin.

Shrub type	Peak-season biomass -----kg-----	Regrowth biomass	Regrowth as a percentage of peak-season biomass
<i>G. senegalensis</i>			
Plot 1	12.31	1.32	10.75
Plot 2	7.43	1.51	20.33
Plot 3	8.91	0.78	8.78
Plot 4	11.83	0.80	6.74
Overall mean	10.12 (0.21) <sup>†</sup>	1.10 (0.035)	11.65
<i>P. reticulatum</i>			
Plot 1	13.97	3.37	24.12
Plot 2	8.96	3.12	34.82
Plot 3	13.46	1.94	14.41
Plot 4	18.19	2.39	13.14
Overall mean	13.65 (1.89)	2.71 (0.33)	21.62

<sup>†</sup>Standard error in parentheses

### 3.1.3 Soil C stocks

The 0-40 cm soil profile in the *G. senegalensis* sites contained an average of 17 (±0.528) Mg C ha<sup>-1</sup>, 58.3% of which resides in the 0-20 cm (Table 3.3). Despite ranging from 8.37 to 12.03 Mg ha<sup>-1</sup> and 6.63 to 7.58 Mg ha<sup>-1</sup> at the 0-20 cm and 20-40 cm depth respectively, there were no

statistical differences in SOC across the *G. senegalensis* sites at these depths. There was more variability observed in the 0-20 cm depth (average CV of 57.3% across the sites) as compared to the 20-40 cm depth (average CV of 39.4%) and this could be due to organic matter "hotspots" from animal manure or the "islands of fertility" phenomena (Vinton and Burke, 1995; Schlesinger et al., 1996; Kelly and Burke, 1997). Similar trends in variability were observed in the *P. reticulatum* sites with a CV of 55.8% at the 0-20 cm depth compared to 48 % at the 20-40 cm depth. There were statistical differences in soil C between the two *P. reticulatum* sites, with more soil C at the site with lower biomass C.

Significant correlations ( $r = 0.87$ ;  $p\text{-value} = 0.05$ ) were observed between SOC at the two depths in the *P. reticulatum* sites, whereas no correlations existed between SOC at the two depths in the *G. senegalensis* sites probably reflecting differences in C sequestration mechanisms in the two shrub communities. Carbon input and sequestration in the *P. reticulatum* sites is mostly driven by vegetation decomposition processes, whereas stochastic processes due to wind movement and dust entrapment (Elkins et al., 1986; Garner and Steinberger, 1989; Coppinger et al., 1991) could be an additional influence on C sequestration in the drier *G. senegalensis* sites.

### 3.2. Soil C spatial structure

Variability of soil C across the 24 m x 20 m grid and the small grids around the trees was higher at the *P. reticulatum* site, whereas variability around the shrubs was higher at the *G. senegalensis* site (Table 3.6). Mean total C was consistently higher under trees at both sites and lower under the shrubs. Semivariograms for all the 339 points at either site (Figure 3.2 and Table 3.7) showed a moderate spatial dependence among sampling locations with a proportion of structural variance (C) to total estimated variance (sill;  $C + C_0$ ) of 53% and 55% respectively, for *G. senegalensis* and *P. reticulatum* sites. The range of spatial autocorrelation was shorter for *G. senegalensis* (within 0.56 m) as compared to *P. reticulatum* (within 1.34 m) although semivariograms were essentially not flat beyond these distances; probably indicating patchiness or repeating soil C patterns. Excluding the 98 data points from the small grid around the shrubs produced variograms shown in Figure 3.3. The shapes of the resulting semivariograms were unchanged but there were changes in the structural and total variance, reduced spatial dependence (from 53 to 51% at the *G. senegalensis* site and 55% to 52% at the *P. reticulatum* site) and reduction in the scale of autocorrelation (Table 3.7).

Table 3.6. Descriptive statistics for the study grids of C structure at the two sites. Shown are means ( $\mu$ ), standard deviations ( $\sigma$ ), range and sample size ( $n$ ) for the soil C measurements.

Statistic	<i>G. senegalensis</i>			<i>P. reticulatum</i>		
	24 m x 20 m grid	Tree-grid	Shrub-grid	24 m x 20 m grid	Tree-grid	Shrub-grid
$\mu$	0.55 (0.017) <sup>†</sup>	0.72 (0.035)	0.47 (0.016)	0.58 (0.031)	0.75 (0.046)	0.46 (0.012)
$\sigma$	0.20	0.35	0.16	0.36	0.46	0.12
Range	1.13	1.99	0.83	2.15	2.63	0.57
CV (%)	36.4	48.6	34.0	62.1	61.3	26.1
$n$	143	98	98	143	98	98

<sup>†</sup>Standard error of the mean in parentheses

Tree-grid and shrub-grid refer to the small 3 m x 3 m grids around the tree and shrub, respectively.

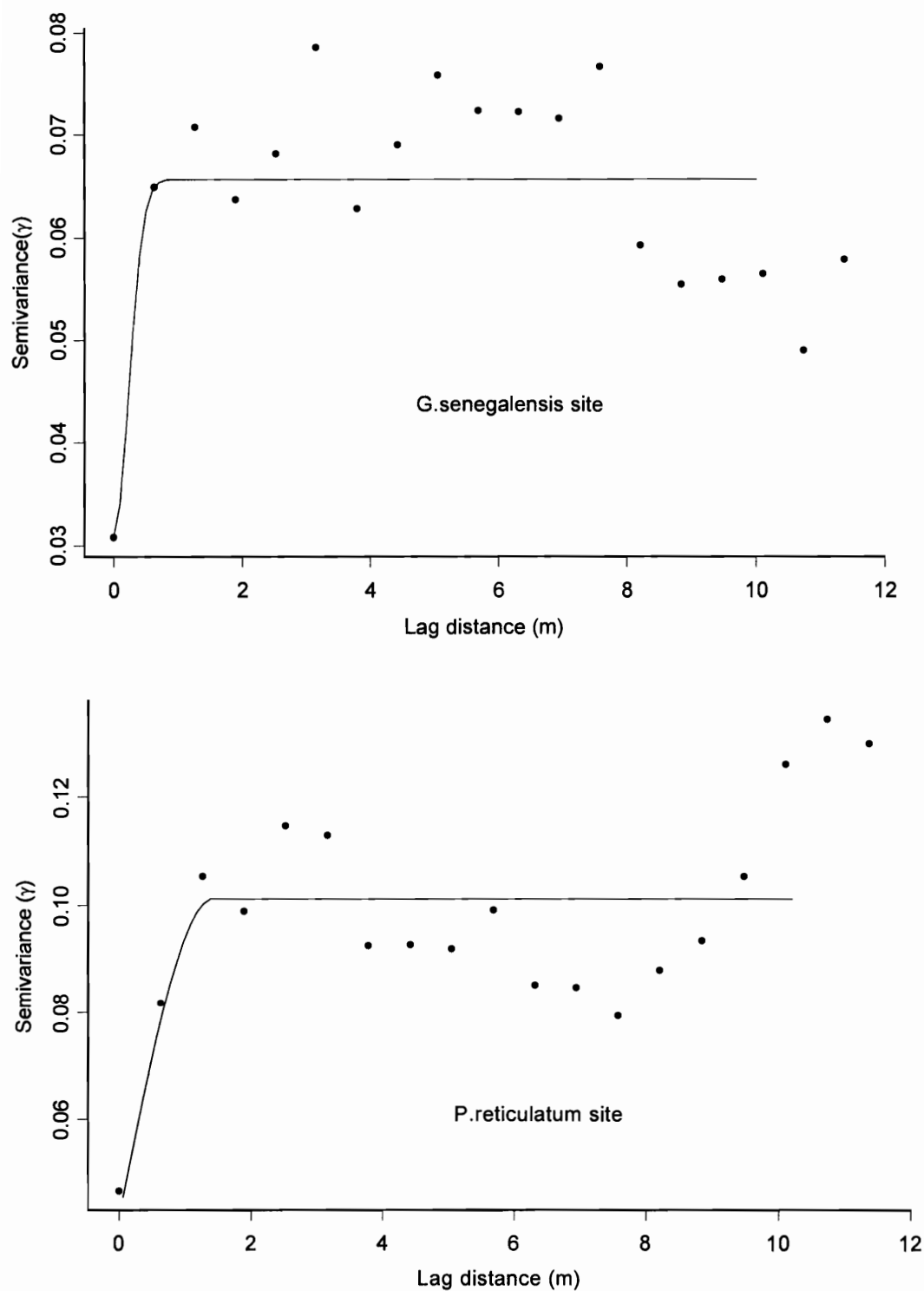


Figure 3.2. Semivariograms of soil C based on 339 samples from the 480 m<sup>2</sup> plot. Semivariograms stratify calculated variances by the distance (lag) separating each pair of points. Parameter values for the solid line models for each variogram are presented in Table 3.7.

Table 3.7. Summary of model parameters fitted through each of the semivariograms in Figure 3.2 through 3.4 (based on different grid combinations). The nugget is the y-intercept of the graph, the sill is the semivariogram value (y value) where each graph becomes a plateau, the range is the distance (x value) where the plateau begins and  $C/(C_0 + C)$  is the degree of spatial dependence, a ratio of the structural to population variance.

Semivariogram points	Model	Nugget ( $C_0$ )	Sill ( $C_0 + C$ )	Range (m)	$C/(C_0 + C)$
G339	Gaussian	0.0308	0.066	0.56	0.53
P339	Spherical	0.0462	0.102	1.34	0.55
Gshrub-grid	Spherical	0.017	0.038	7.46	0.56
Pshrub-grid	Spherical	0.0044	0.079	6.67	0.94
Gtree-grid	Spherical	0.0418	0.085	0.068	0.51
Ptree-grid	Spherical	0.0716	0.150	0.77	0.52

G339 and P339 represents semivariograms constructed based on all the 339 samples from the 24 m x 20 m plot at the *G. senegalensis* and *P. reticulatum* sites, respectively. Gshrub-grid and Pshrub refer to semivariograms constructed based on the 143 points that comprise the 24 m x 20 m grid plus the samples from the two 3 m x 3 m grids around the shrubs. Gtree-grid and Ptree-grid refer to semivariograms constructed based on the 143 points that comprise the 24 m x 20 m grid plus the samples from the two 3 m x 3 m grids around the trees.

Removal of the 98 data points from the small grids around trees in the plots had the opposite effect (variograms in Figure 3.4), changing variogram shapes, increasing the scale of autocorrelation (to ranges that corresponded to half the distance between shrub grids) and spatial dependence to 56 and 94% at the *G. senegalensis* and *P. reticulatum* sites, respectively.



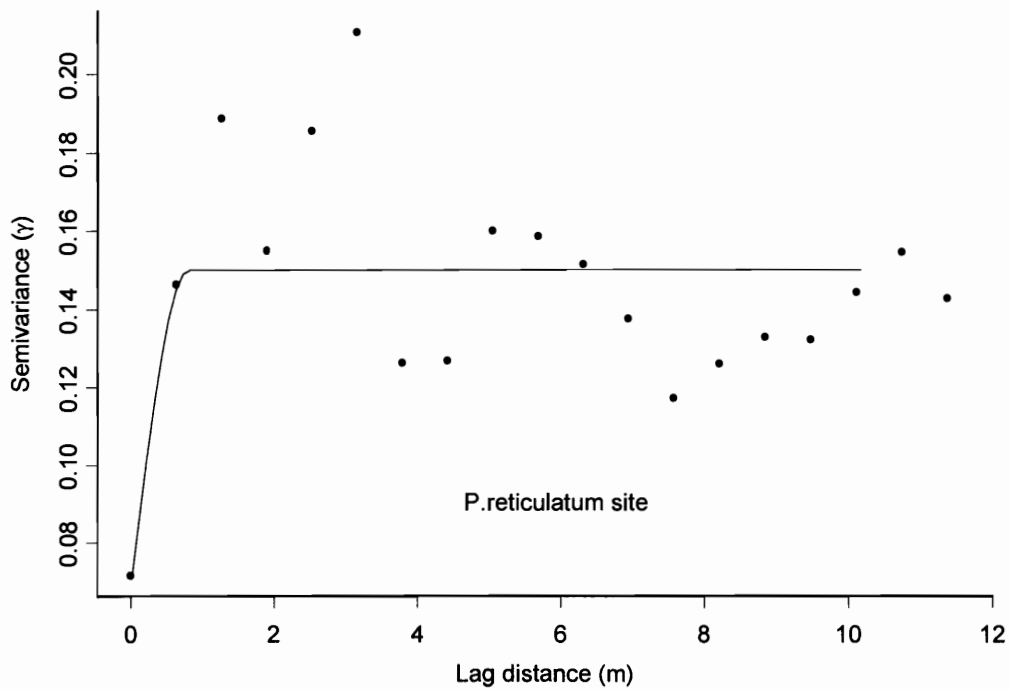
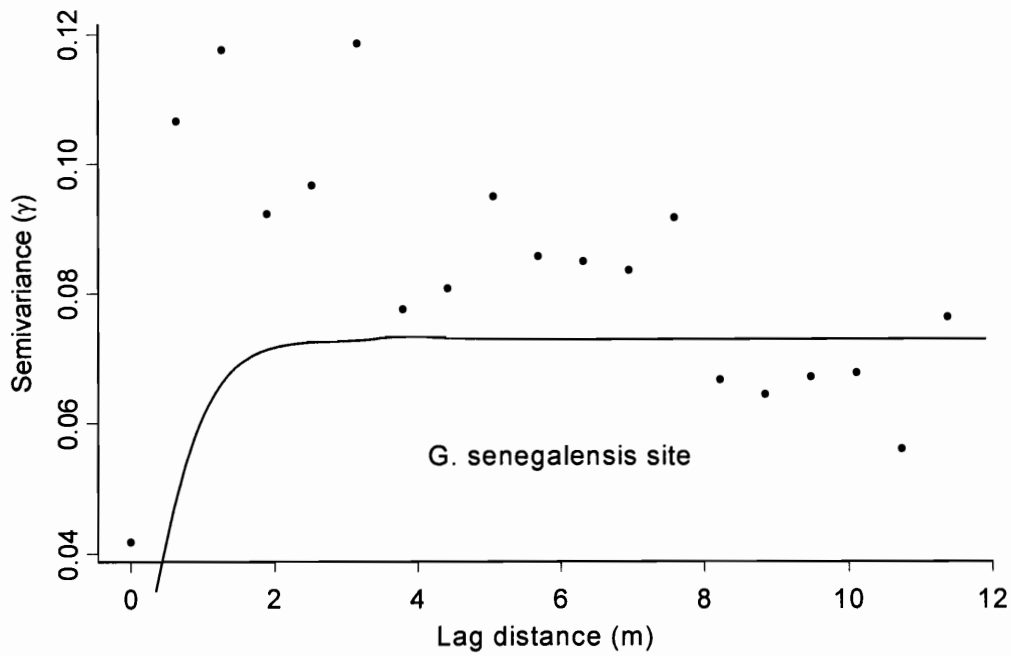


Figure 3.3. Separate semivariograms for soil C with the small-grid data around the shrubs removed. Semivariograms were calculated on the 143 points that comprise the 24 m x 20 m grid plus the samples from the two 3 m x 3 m grids around the trees.

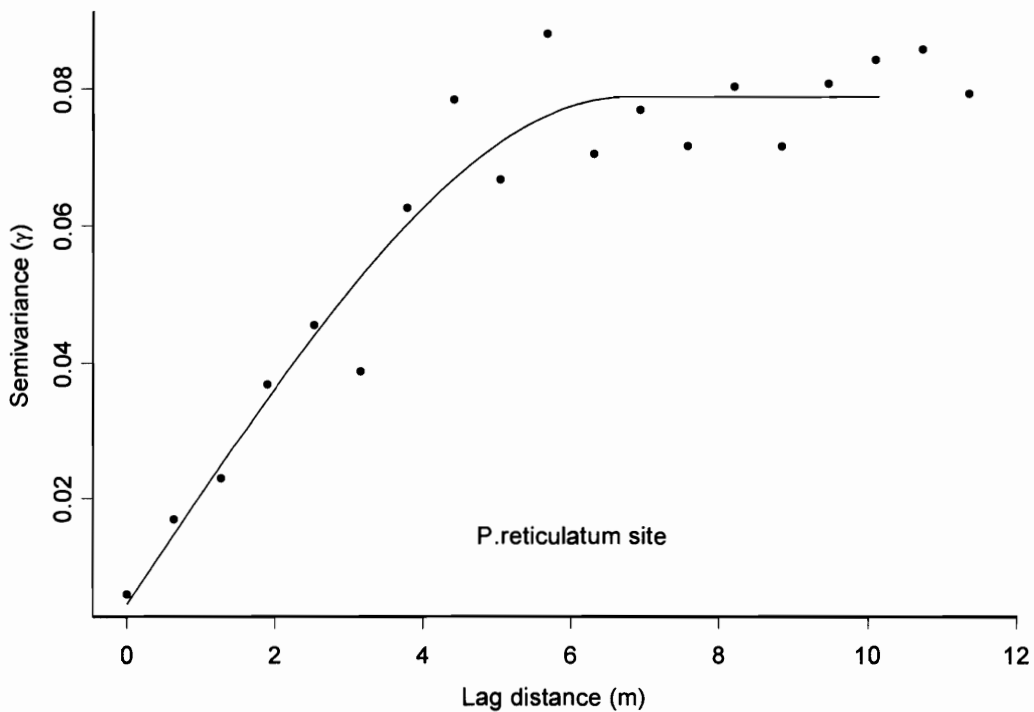
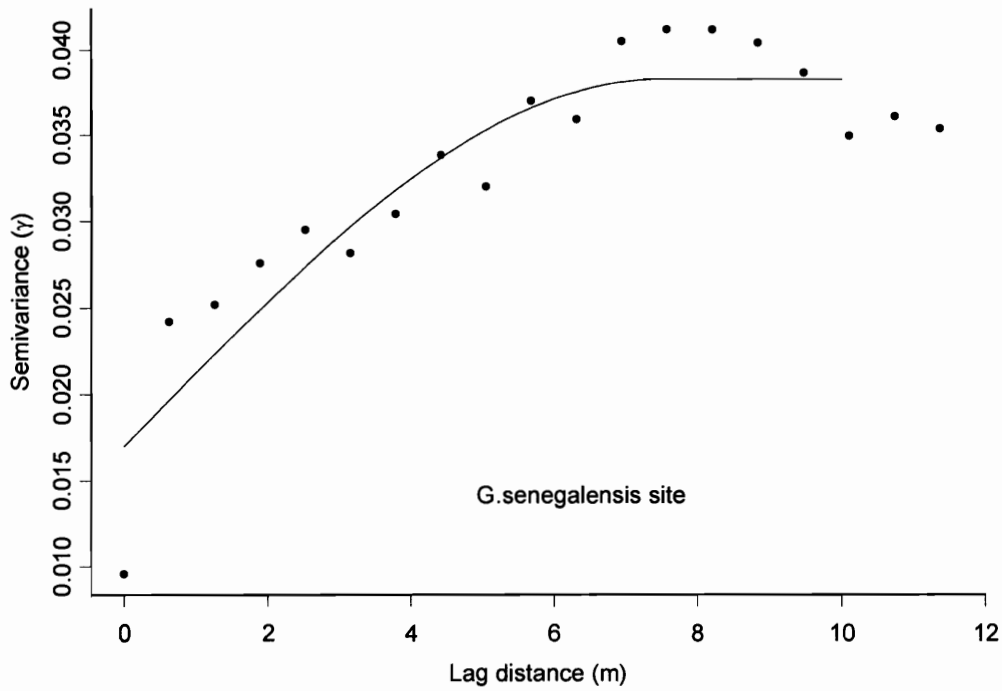


Figure 3.4. Separate semivariograms for soil C with the small-grid data around the trees removed. Semivariograms were calculated on the 143 points that comprise the 24 m x 20 m grid plus the samples from the two 3 m x 3 m grids around the shrubs.

Individual plants have been shown to influence soil properties in various ecosystems, including forest trees (Boerner and Koslowsky, 1989), dune grasslands and tussock grasses (Schlesinger et al., 1990, 1996). According to Hook et al. (1991), spatial variability in primary production and redistribution of surface soil are the two major processes implicated in generating the heterogeneity of soil properties associated with plant cover in dry areas. The spatial patterns of primary production directly affect the spatial pattern of C through plant litter inputs (Muneto et al., 2001) and decomposition of belowground biomass. There were increases in spatial dependence (relative to the 339 point grid) of soil C at both sites when the 98 points for the small-grid data around the trees were removed from the semivariograms and decreases in spatial dependence when the 98 points for the small-grid data around the shrubs were removed.

These changes in spatial dependence imply that the importance of plant species (shrub or tree) overshadows that of plant presence in structuring soil C patterns at the two sites. At either site, the shrub component has a stronger influence on soil C structure than do the trees although this influence was more pronounced at the *P. reticulatum* site (spatial dependence of 94%). The differences in shrub type influence on soil C patterning could be attributed to differences in biomass productivity, rainfall amounts and surface soil redistribution processes at either site.

The average above-and belowground biomass for *G. senegalensis* were about 0.46 and 2.0 Mg ha<sup>-1</sup>, and those of *P. reticulatum* were about 0.29 and 3.04 Mg ha<sup>-1</sup>, respectively (Table 3.5; dividing C stock values by 0.47 to convert to biomass). The greater total primary production of *P. reticulatum* may lead to higher accumulation of C and hence it's stronger influence on soil C variability. On the other hand, a proportion of variability in C measured at the *G. senegalensis* site is most likely due to accumulation of fine materials from the capture of windblown materials by the shrub canopy (Sterk et al., 1996), hence lowering the influence of the shrubs on C patterning. The higher rainfall amounts and low wind speed in the *P. reticulatum* site curtail this possibility and allow the accumulation and stabilization of fallen litter in areas proximal to the shrub. In addition, the long lifespan of *P. reticulatum* means the species can occupy a place for a longer time than the relatively short-lived *G. senegalensis*, and this may amplify the effect of *P. reticulatum* on the local soil properties.

In this study we hypothesized that at the landscape level, the shrubs in the parkland systems are greater controls on system C stocks than is the tree component. Decades of research have shown that soil variability increases with the area measured (Beckett and Webster, 1971; Palmer, 1990; Nolin et al., 1996). However, in absence of large-scale gradients in topography, soil depth and parent material (Jackson and Caldwell, 1993b) as is the case in our general study area, overall landscape C variability may not be different than our study grids. This is because “within-field” variance often does not vary very much with the size of the field (Beckett and Webster, 1971; Saldana, 1998; Conant and Paustian, 2002). Indeed, Ferrari and Vermeulen (1995) demonstrated that pooled soil samples from fields 0.33 to 2.5 ha in size had similar coefficients of variation lending credence to our assertion that at the landscape level we should expect the same C variability and controls in soil C patterning.

#### 4.0 Perspectives

White (1983) classifies these Parkland systems as shrubland with scattered trees and Woomeer et al. (2004b) estimated the total biomass C stocks in these systems at  $\sim 6.5 \text{ Mg C ha}^{-1}$ . A study by Tschakert et al. (2004) presents a mean value for tree C in these systems at  $6.3 \text{ Mg C ha}^{-1}$ . Subtracting these two values would result in an estimate of shrub C stocks of  $0.2 \text{ Mg ha}^{-1}$ , a modest 15% of the actual C stocks ( $1.35 \text{ Mg ha}^{-1}$ ) that we measured in the field for these shrubs. One approach that has been used to offset significant portions of previous C emissions is increasing woody vegetation density (Houghton, et al., 1999; Van Auken, 2000; Pacala et al., 2001; Lal, 2002) and these shrub systems could be a C repository. However, as several authors argue (e.g. Covington et al., 1994; Lal, 1995; Lal et al., 1998), depending on management, this strategy could have potential negative outcomes triggering large C losses hence exacerbating climate change. Our study is a case in point, indicating a precarious balance between potential gains and rapid losses in C depending on management. If “business as usual” with annual pruning and burning continues in the basin, these systems represent on average a loss of  $0.21$  and  $0.14 \text{ Mg C ha}^{-1} \text{ yr}^{-1}$  in cultivated soils for *G. senegalensis* and *P. reticulatum* sites, respectively. Conversely, using percent regrowth estimates in Table 3.5 shows that only  $0.024 \text{ Mg C}$  (*G. senegalensis*) and  $0.03 \text{ Mg C ha}^{-1}$  (*P. reticulatum*) is returned to the fields at the second clearing when farmers can not burn the biomass. This represents a modest 10 and 18% of

total annual aboveground biomass C respectively, for *G. senegalensis* and *P. reticulatum*.

Lufafa et al. (2005), showed the potential distribution of the shrubs across the Peanut Basin to have approximate areal coverages of  $2.34 \times 10^6$  and  $9.14 \times 10^5$  ha, respectively, for *G. senegalensis* and *P. reticulatum* (Figure 3.5). Tottrup and Rasmussen (2004), report an average cultivation density of 49% in sections of the Peanut Basin for 1999. Using this rather conservative cultivation density estimate and areal coverage of the shrubs reveals that these systems annually lose about  $3.51 \times 10^5$  Mg of biomass C, an equivalent of  $2.11 \times 10^5$  Mg of CO<sub>2</sub> per year (Brady and Weil, 1999).

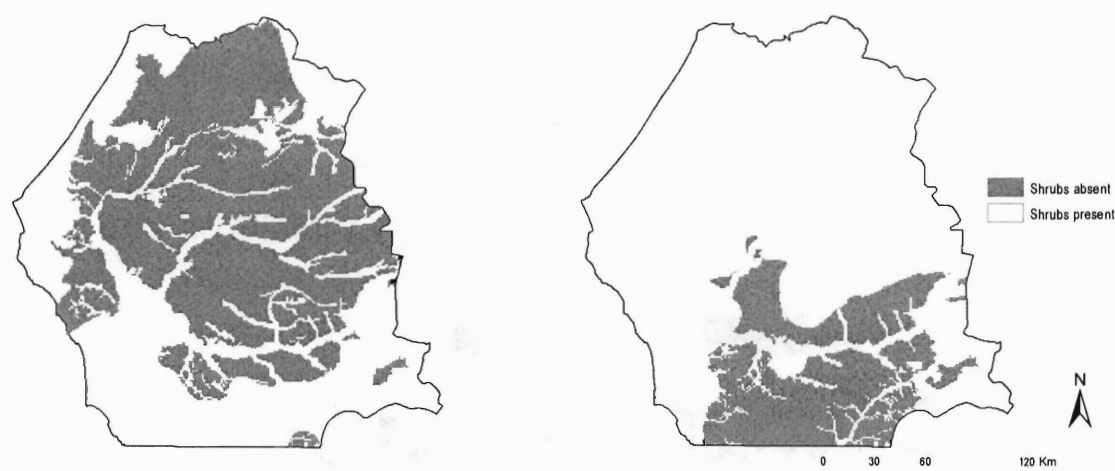


Figure 3.5. Potential distribution maps for *G. senegalensis* (left) and *P. reticulatum* (right) in the Peanut Basin of Senegal.

With 98.4% of the land area in the basin under cultivation (Ba et al., 2000), crop residues represent a potential source of C. The biggest percentage of the cultivated land however, is cropped to pearl millet (51.1 %) and peanut (38.2 %) (Ba et al., 2000) which produce 1.0-2.0 and 0.7-1.0 Mg crop residue ha<sup>-1</sup>, respectively (Badiane et al., 2000). All of the peanut residue and 50-100 % of millet residue is removed from fields and largely used for animal fodder (Badiane et al., 2000) leaving only animal manure as a source of organic inputs. Badiane et al. (2000) determined in field surveys that on average 0.5 Mg animal manure ha<sup>-1</sup> is applied only to millet fields and since the standard crop rotation is peanut–millet, this results in an annual input of 0.25 Mg animal manure ha<sup>-1</sup>. Furthermore, since only about half of the land in a typical village is under

millet, averaging across all fields results in long-term inputs of 0.12 Mg manure ha<sup>-1</sup> yr<sup>-1</sup> or 0.048 Mg manure-C ha<sup>-1</sup> yr<sup>-1</sup>. Although trees practically contain more biomass C than do the shrubs (Tschakert et al., 2004; Woomer et al., 2004b), the overall fraction of tree C applied to soils is only the leaf material whereas in theory, the entire shrub aboveground biomass could be returned to soils upon pruning. Consequently, our results showed that shrubs are the dominant source of C to soils in the Peanut Basin.

## 5.0 Acknowledgments

This study was conducted under the auspices of the US National Science Foundation Grant No. 0120732. We thank Papa Serigne Sarr, Younouss Tamba, Papa Omar Dieye and Moussa Diop of ISRA for helping in data collection, Julia Jones for her assistance with geostatistics and Joan Sandeno for the laboratory analyses and editing the manuscript.

## CHAPTER FOUR

### **Regional carbon stocks and dynamics in native woody shrub communities of Senegal's Peanut Basin.**

A. Lufafa, J. Bolte, D. Wright, M. Khouma, I. Diedhiou, R.P. Dick, F. Kizito, and J.S. Noller

Agriculture, Ecosystems and Environment  
P.O. Box 211  
1000 AE Amsterdam  
The Netherlands

## Abstract

Estimating regional carbon (C) stocks and understanding their dynamics is crucial, both from the perspective of sustainable landscape management and global change feedback. This study combines remote sensing techniques and a loose coupling between the CENTURY soil organic matter model and GIS to estimate regional C stocks and dynamics for *Guiera senegalensis* shrub communities in Senegal's Peanut Basin. A statistical model relating shrub aboveground biomass C to satellite image-derived shrub abundances at training plots was developed and used to scale-up biomass C across a major part of the Basin. i\_Century aided by Visual Basic Arc scripts and Microsoft Access® was used to facilitate the link between the CENTURY model and a GIS database. Significant correlation ( $r = 0.73$ ;  $p = 0.05$ ) was observed between satellite image-derived shrub abundance and biomass C at the training plots. However, the strength of correlations progressively declined with increasing distance between training plots; reflecting differences in shrub spectral reflectance. Aboveground C stocks ranged from 0.01 to 0.45 Mg ha<sup>-1</sup> with an approximate total of 247,000 Mg C for the 3060 km<sup>2</sup> study area. CENTURY model projections indicate that C sequestration in these systems is contingent on long-term effectiveness of non-thermal management of shrub residue and that the actual rates depend strongly on scenarios of future land use and soil type. Compared with the traditional "pruning-burned" management, returning prunings for 50 years would increase soil C sequestration by 200 to 350% under a no fertilization regime, and increase soil C sequestration by 270 to 483% under a low (35 kg ha<sup>-1</sup> N yr<sup>-1</sup>; 20 kg ha<sup>-1</sup> P yr<sup>-1</sup>) fertilization regime depending on soil type and climatic regime. These results indicate that altered land management could contribute to transforming these degraded semiarid agroecosystems from a source to a sink for atmospheric CO<sub>2</sub>.



## 1.0 Introduction

Understanding the spatial distribution of biomass carbon and sequestration potential thereof is essential for carbon (C) trading initiatives through the Clean Development Mechanism (CDM) of the Kyoto Protocol (UNFCCC, 2004). Although current project foci under the CDM exclusively target increasing C stocks in biomass, in the near future, soil C stocks are poised to start compensating fossil CO<sub>2</sub> emissions (IPCC, 2003). Indeed, as a flexibility mechanism, the Kyoto Protocol proposes that developing nations should receive “greenhouse gas credits” for increasing soil C stocks, opening up possibilities to obtain financing to support sustainable soil resource management. These trends require reliable methods for monitoring and verification of C sequestration in soil and biomass (Lal et al., 1999; Kirschbaum et al., 2001) as well as reasonable predictions of C sequestration potential across large areas.

Vegetation and soils in arid and semiarid regions control significant proportions of terrestrial C stocks and fluxes between the land-atmosphere interface (Verhoef et al., 1996; Lal, 2001; Lal, 2002) and it is estimated that these regions have lost two thirds of their C in areas affected by desertification (IPCC, 1996). Consequently, the Intergovernmental Panel on Climate Change (IPCC) estimates that these areas could resequenter 12 to 19 Pg C over a 40- to 50-year period (IPCC, 1996) and woody shrubs are a dominant consideration for rebalancing the C budgets of these areas (Goodale and Davidson, 2002).

In Senegal and throughout neighboring Sahelian countries, there is one potentially large but poorly quantified carbon sink—that of native woody shrub species (*Guiera senegalensis* J.F. Gmel) occurring in vast sections of the agricultural landscape. These shrubs are typically left to grow in farmers’ fields where they are coppiced at the end of the dry season (March to April) and subsequently are pruned at every weeding during the growing season of row crops. Following the growing season, the shrubs are allowed to regrow reaching > 2 m diameters and > 1 m heights. Depending on land use and management, these shrubs have the potential to be a significant source or sink for carbon within the global carbon cycle. Lufafa et al. (submitted), and Tschakert et al. (2004) using field assessments quantified C stocks of these shrub systems. While such focused individual field measurements are a step towards determining current vegetative C

stocks in these systems, their utility in assessing and quantifying the rather more meaningful regional scale C stock and sequestration potential (Paustian et al., 1997; Falloon et al., 2002; Ardo and Olsson, 2003) is limited because of high spatial heterogeneity and inability to have adequate sample frequency (due to excessive costs).

We exploit the synergy between remotely sensed data and field-based measurements to quantify landscape level C stocks in *G. senegalensis* communities for a section of the Senegalese Peanut Basin and use a loose coupling between the CENTURY model and Geographical Information System (GIS) databases to simulate current and project future soil organic carbon (SOC) levels in these systems under different land management and climate change scenarios.

## 2.0 Materials and Methods

### 2.1. Study area description

The study area is a section of the Thiés and Diourbel regions in Senegal's Peanut Basin (Figure 4.1). With an areal extent of 3,062 km<sup>2</sup>, study area boundaries extend from: upper left, 276070; 1650760 m and lower right 347500; 1590700 m. The climate is semiarid, with a mean annual precipitation and temperature of 540 mm and 27°C, respectively (Dacosta, 1989). Geological substrates in the area include mainly aeolian deposits of Harmattan wind sand (Herrmann, 1996) of Quaternary age over sedimentary rocks of Cretaceous to Miocene age (Monciardini, 1966), and highly eroded colluvial-alluvial ferruginous sediments derived from paleosols (Neogene) and Precambrian bedrock (Renaud, 1961; Michel, 1973). The soils are sandy, classifying as *Psammets* and *Calcids* according to Soil Taxonomy (Soil survey staff, 2003) and fall broadly into two indigenous types, i.e. *Dior* and *Deck* (Badiane et al., 2000). Study area vegetation is shrubland with scattered trees (Diouf and Lambin, 2001). The shrub layer is dominated by *G. senegalensis*, whereas the tree component is dominated by *Faidherbia albida*.

## 2.2. Regional biomass C estimation

The general approach used for upscaling C stock estimates to regional levels is based on utilizing spectral information in satellite images for target detection and classification (Puyou-Lascassies et al., 1994; Jensen, 2000; Gonzalez and Woods, 2002).

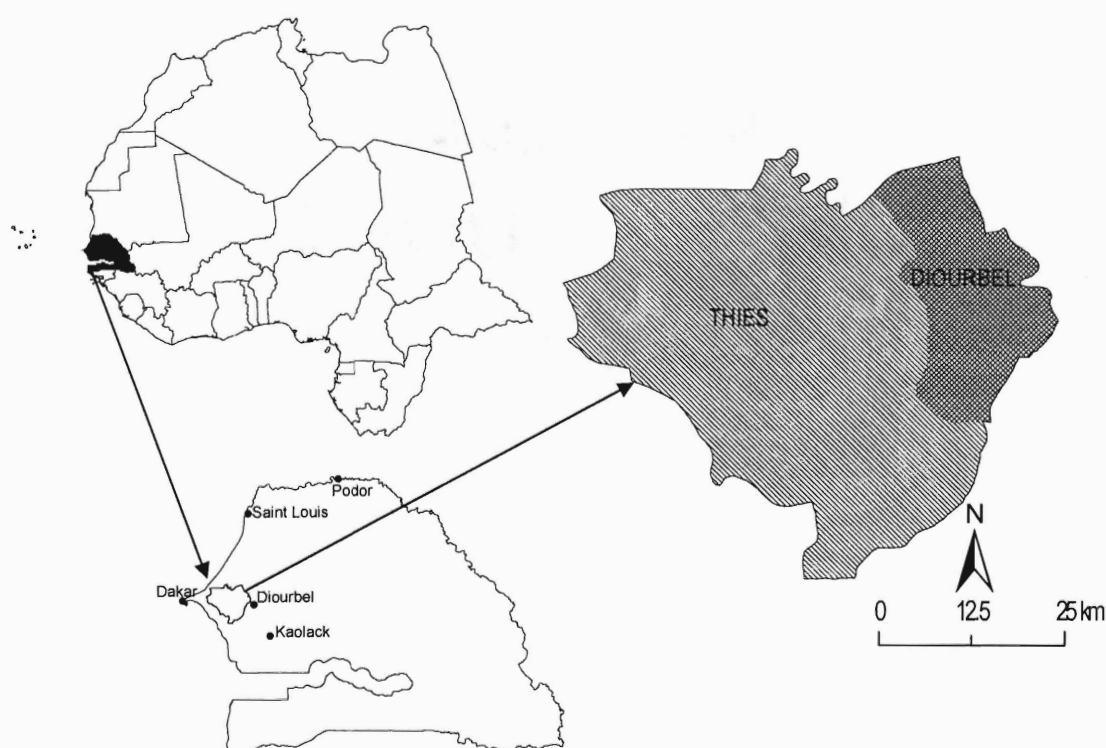


Figure 4.1. Study area location.

Detection of *G. senegalensis* was implemented by means of spectral unmixing or sub-pixel analysis (Adams et al., 1993; Boardman et al., 1995; Ichku and Karnieli, 1996; Elmore et al., 2000; Rosin, 2001). This approach is premised on the assumption that satellite image pixels contain materials (cover fractions) whose spectral signatures are linearly independent (and therefore linearly summed) and explain the spectral signature of a pixel as a whole (Guyot et al., 1989; Shimabukuru and Smith, 1991; Quarmby et al., 1992; Garcia-Haro et al., 1996; Small, 2001). The materials or cover fractions in the satellite image pixel are called endmembers (Smith et al., 1992; Okin et al., 1998; Roberts et al., 1998; Okin et al., 1999).

Spectral unmixing decomposes satellite image pixels into constituent spectrally pure signatures of endmembers and estimates the relative combinations of the endmembers which would produce a spectral signature similar to that of the "mixed" pixel (van der Meer, 1995; Ichku and Karnieli, 1996; Asner and Heidebrecht, 2002). This allows estimates to be made of the mixing proportions (or abundances) of the "mixed" pixel which is made up of the various spectral endmembers (Smith et al., 1990).

### 2.2.1. Selection of training plots for biomass C scaling-up

Within the study area, a georeferenced November, 1999 Landsat ETM image was used to select twenty-three 8,100 m<sup>2</sup> (equivalent to 3 x 3 Landsat pixels) training plots at six sites (Table 4.1). Sampling unit size was based on the need for a uniform spatial and radiometric plot size for comparison with Landsat data and the difficulty of identifying a single Landsat pixel (30 m x 30 m) in the field without introducing spatial errors. At each site, fields with the target shrub were identified and the center coordinates of each selected field were acquired using a Garmin Etrex global positioning system (GPS). Using the center coordinates as a reference point, the 9 pixels making up a 90 m x 90 m plot around the center coordinate were located on the Landsat image and the coordinates of their four outermost bounding corners recorded. Using the GPS unit, these four outermost corner points were located in the field and served as boundaries of the 8,100m<sup>2</sup> plots.

### 2.2.2. Shrub biomass inventory

Our goal was to quantify regional C stock in peak-season standing biomass of the shrubs and this occurs in late spring just before the shrubs are pruned back to prepare for the summer cropping season. Thus shrub biomass inventory for C stock estimation was done at this time of year. The procedure was to develop allometric equations that use easily measured shrub properties to estimate biomass. In turn, inventories of shrub densities in combination with allometric equations were done to estimate shrub biomass in all of the training plots. The C content of *G. senegalensis* biomass was 0.495 as determined on a Leco C analyzer (LECO Inc., St. Joseph, Michigan).

Table 4.1. Vegetation and elevation across the biomass C upscaling training plots.

Site	Plot	Plot center coordinates		Elevation	Other vegetation in plot
		Easting	Northing		
		------(m)-----			
Keur Asanulo (n = 3)	1	312360	1635450	103	<i>Acacia albida</i>
	2	312270	1635480	24	<i>Faidherbia albida</i>
	3	312360	1635300	29	<i>Acacia albida</i>
Keur Mandiamba (n = 4)	4	318270	1631760	25	<i>Balanites aegyptica</i>
	5	318390	1631640	47	<i>Balanites aegyptica</i>
	6	318480	1631850	23	none
	7	318180	1631430	16	none
Keur Matar Aram (n = 6)	8	300240	1633920	87	<i>Acacia occidentale</i>
	9	300330	1633920	66	none
	10	300420	1633950	74	none
	11	300030	1633830	65	<i>Combretum glutinosum</i>
	12	299700	1633590	82	<i>Acacia occidentale</i>
	13	299610	1633500	79	none
Ndiagne (n = 2)	14	309180	1632960	24	none
	15	309060	1632810	18	<i>Acacia albida</i>
Keur Ibra Fall (n = 5)	16	309930	1633050	16	<i>Azadirachta indica</i>
	17	310020	1633050	16	<i>Combretum glutinosum</i>
	18	310140	1633050	24	<i>Balanites aegyptica</i>
	19	310230	1632960	25	<i>Acacia albida</i>
	20	310170	1633260	17	<i>Azadirachta indica</i>
Thilla Ounte (n = 3)	21	318000	1636530	18	<i>Acacia albida</i>
	22	318120	1636590	27	<i>Acacia albida</i>
	23	318240	1636740	22	none

### 2.2.3. Field spectrometry

To enable evaluation of the satellite image with the shrubs as endmembers, a field analysis was conducted to measure the spectral properties of the shrubs in the study area. Field-based reflectance spectra of the shrub canopies were collected using a 0.35  $\mu\text{m}$  to 1.050  $\mu\text{m}$  (Fieldspec, 1997) range field-portable spectrometer (Fieldspec FR-Pro, Analytical Spectral Devices, Inc). The instrument has a spectral resolution of approximately 3 nm at around 700 nm and measures the Visible/Near Infrared (VNIR) portion of the spectrum using a 512-channel silicon photodiode array overlaid with an order separation filter. Data were collected coincident with the field biomass inventories and the date of acquisition of the target Landsat image.

At this time of year, the grass and most of the tree canopies were fully senescent while the shrub canopies were fully green. Spectra were consistently acquired from 0.4 to 0.5 m (nadir-looking) above the shrub canopies with a bare fiber optic having a canonical view subtending a full angle of about 25 degrees. The spectra were then divided by the near simultaneous (< 2 minutes) spectrum of an uncalibrated 99% reflective Spectralon panel (Labsphere, Inc) to yield reflectance. The obtained spectra measurements of the same target were averaged to improve their signal-to-noise ratios (De Jong et al., 2003). Due to severe noise in data in the distant portions of the spectrum, only the data gathered in the 0.350  $\mu\text{m}$  through 1.025  $\mu\text{m}$  was used, reducing the number of spectral channels to about 470.

#### 2.2.4. Landsat data

A Landsat ETM<sup>+</sup> (Path 205, Row 50) for 6<sup>th</sup> March, 2004 approximately coincident with the field measurement campaigns was acquired and georeferenced by image to image registration on the November, 1999 image used in selecting training plots. The image was then subsetting to the extents of the study area and radiometrically normalized (Price, 1987; Markham and Barker, 1988) for factors such as sun incidence angle, time of data gathering, earth-sun distance, and sensor degradation. The radiometric data were calibrated to apparent surface reflectance in ENVI<sup>TM</sup> which automatically uses the published Landsat post-launch gains and offsets (ENVI, 2004). Further corrections to the reflectance image included atmospheric normalization (Richter, 1997) using dark pixel subtraction (Elvidge et al., 1995) to remove haze (Kaufman, 1984; Du et al., 2002).

#### 2.2.5. Spectral unmixing for pixel shrub abundance estimation

The Landsat reflectance data were processed to a fractional shrub abundance image with the Mixture Tuned Matched Filtering (MTMF) algorithm (Boardman et al., 1995; Funk et al., 2001) in ENVI<sup>TM</sup>. MTMF is a technique designed and optimized to detect extremely weak signals that are essentially in the noise (Funk et al., 2001) and does not require knowledge of all the endmembers within an image scene. It works by separating spectral reflectance into “signal” and “noise” components. The signal is the desired spectrum scaled to represent its radiance in a pixel (Okin et al., 2001) and everything else is assumed to be noise.

First, a minimum noise fraction (MNF) was performed on the subsetted image to reduce and compress the data and to produce an image with isotropic unit variance noise that is needed in successful unmixing (ENVI, 2004). The MNF sequentially performs two cascaded principle component transformations on the data (Green et al., 1988; Boardman and Kruse, 1994): the first transformation, based on an estimated noise covariance matrix, separates white noise (i.e. uninformative data) resulting in transformed data in which the noise has a unit variance and no band-to-band correlations (Harsanyi and Chang, 1994). The second is a standard principal components transformation of the noise-whitened data and a recombination of the bands into new composite bands which account for most of the variance in the original data (Underwood et al., 2003). In the second step, the field spectral database of shrub canopies was high-frequency filtered to remove noise, resampled to Landsat wavelengths and used as the target endmember in MTMF to produce a shrub abundance image. Abundance values at known 3 x 3 pixel windows corresponding to the biomass C training plots were retrieved from the MTMF image, aggregated and used as a dependent variable in a regression against field measured biomass C.

### 2.3. Spatial modeling of SOC dynamics

The dynamics of soil organic carbon (SOC) for the shrub system were simulated using the CENTURY model (Parton et al., 1987, Metherell et al., 1993). CENTURY is a point-based (performs simulation of and predictions for one site at a time) ecosystem model simulating biogeochemical fluxes of C, N, P, and S. To spatially model SOC dynamics across our study area, a loose link was made between GIS datasets and the CENTURY model by exploiting the database handling capabilities of interactive Century (i\_Century). *"i\_Century is a model control system for the CENTURY model developed by the Natural Resource Ecology Laboratory at Colorado State University. It stores CENTURY input data in an Access database, feeds that data to CENTURY, runs CENTURY, and reads and stores the result. The basic philosophy of the i\_Century approach is to manage both the input and output data of a large set of CENTURY simulations within a single database (Iowa, 2004). By handling large sample populations, i\_Century provides a basis for managing spatially explicit units obtained from GIS and allows for spatially variable simulation of SOC dynamics.*

### 2.3.1. CENTURY model description

Version 4.0 of the CENTURY model (Metherell et al., 1993) was adopted for the shrub ecosystem and used for simulating biomass production and SOC dynamics. A detailed description of the model is presented by Parton et al. (1987, 1993) but briefly, CENTURY includes a grassland/crop production submodel that simulates plant production and a forest submodel that stimulates the growth of forests. Our shrub system was modeled as a savannah and to simulate a savannah, CENTURY uses both the grassland/crop and forest submodels with some additional code to simulate nutrient competition and shading effects. Using data from field studies and laboratory analyses, we changed CENTURY default parameters to suit our study species. We used a potential aboveground production of  $500 \text{ g C m}^{-2} \text{ yr}^{-1}$ , altered lignin content to 14.1% and 18% respectively for shoots and roots, and increased the range of plant tissue C to N ratio to 50 to 30 in line with our laboratory analyses. Based on our field biomass data and cognizant of the influence of annual pruning on biomass allocation (Pearsall, 1927; Troughton, 1960; Brouwer, 1962; Wilson, 1988), we arrived at a fixed C allocation; about 90% of annual plant production is allocated to aboveground growth and 10% to belowground production. CENTURY is an inherently transient rather than an equilibrium model (Parton et al., 1989), so in order to bring the C pools of our simulated system to levels consistent with our field measurements, the model was run for 1500 years with prescribed disturbance regimes obtained from anecdotal information. Due to a lack of reliable information about early system management, a generalized pattern was modified so that the 2005 year run yields a representation similar to current field observed C values.

### 2.3.2. CENTURY-GIS coupling procedure and model input parameterization

The conceptual framework of the CENTURY-GIS coupling procedure is illustrated in Figure 4.2. Climatic data (monthly precipitation, maximum and minimum temperatures) for stochastic climate generation within the CENTURY model was obtained at georeferenced climate stations from Direction de la Météorologie Nationale (DMN) in Dakar and are based on mean values during a 42-year period (1960-2002). A total of 18 climate stations (all collecting rainfall data with only four of them recording temperature) were used in this study. The soil map was obtained from Centre de Suivi Ecologie (CSE) and is based on work by Stancioff et al. (1986). With a cartographic scale of 1:250,000, soil unit delineations are classified according to major landscape



morphological formations and topography derived from Landsat MSS data (1973-1981). Four broad soil types (Torrox, Psammets, Torrents and Torrerts) with twelve varying map units are found in the study area.

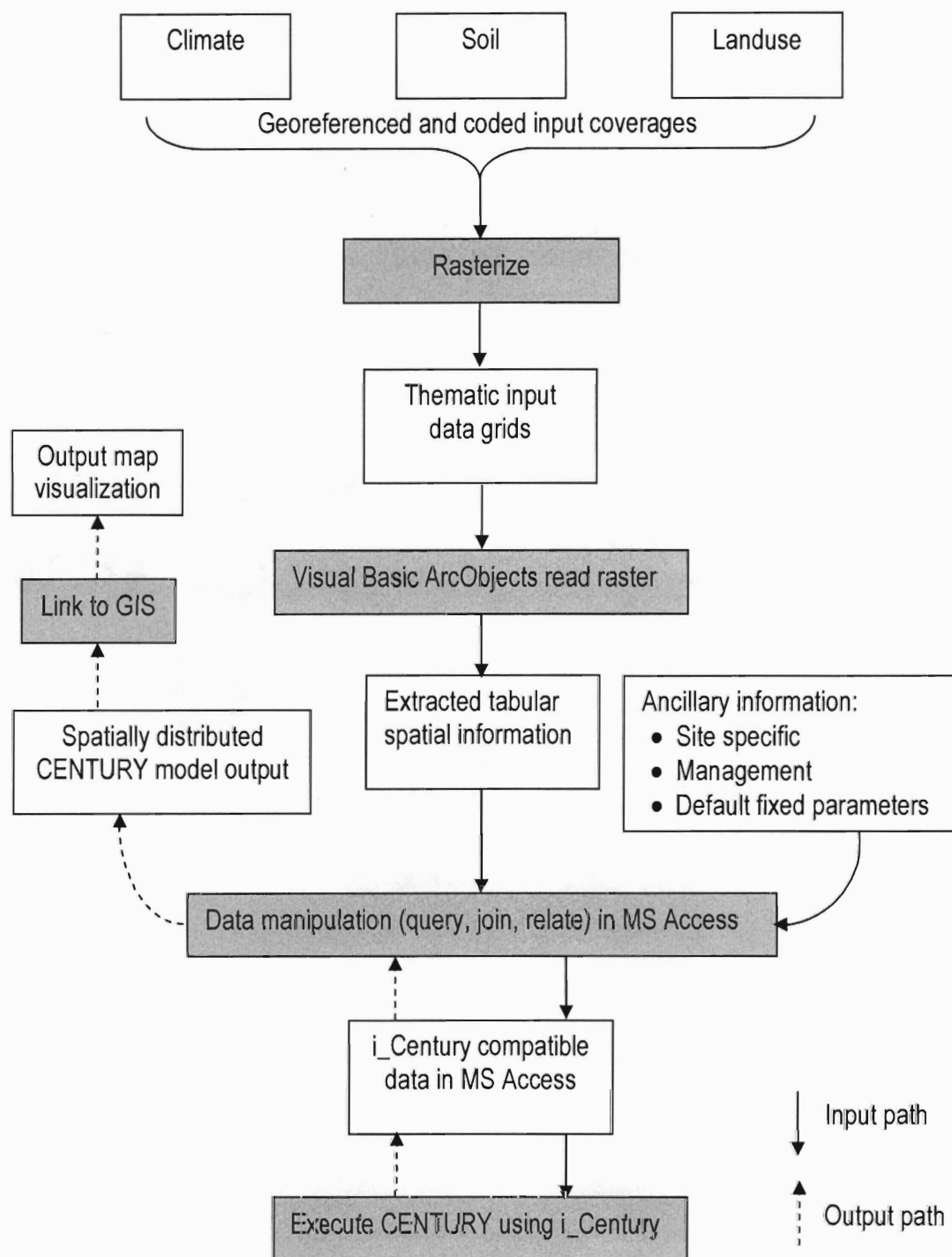


Figure 4.2. Conceptual framework for the loose coupling between GIS and the CENTURY model.

Through a combination of field measurements and literature evaluation, soil parameters: pH (Stancioff et al., 1986; Sagna-Cabral, 1989; Woomer et al., 2001), texture (Bonfils and Faure, 1956; Charreau and Vidal, 1965; Stancioff et al., 1986; Woomer et al., 2001; Elberling et al., 2003) and bulk density (Manlay et al., 2002; Tschakert et al., 2004) needed to drive the model for the encountered soil types were obtained and georeferenced. Land use information was derived from the satellite image through manual digitization of areas identified as having the shrubs in the MTMF image to obtain two land uses i.e. shrubland and nonshrub land. We assumed that all shrub areas are under agricultural production with no fallows.

The obtained soil-information records and climate-station points were interpolated using Thiessen polygons (Lam, 1983; Burrough, 1986; Mitas and Mitasova, 1999) to assign each point in space a value similar to that of the closest record. All of the thematic maps were converted into raster at a grid resolution of 500 m corresponding to the modeling spatial scale. A point file with individual points centered in the 500 m x 500 m grid was created and a Visual Basic Arcobjects read-raster program automatically extracted the climate, soil and landuse information from the thematic grids at these centered points into a table. Individual record or grid identifiers serving as the actual link between GIS and CENTURY and X/Y coordinates were added to the table using Visual Basic scripts. The obtained table was subsequently manipulated in Microsoft Access® and appended to additional information (management events, default parameters, etc.) needed by CENTURY to produce a data structure that is compatible with i\_Century. After running the model, the output was once again manipulated in Microsoft Access® to produce a table with SOC output at desired instances for display in GIS.

### 2.3.3. Model simulation

Under current management, the shrubs are pruned and the residue burned at peak-season in the spring to clear land for agricultural production. A two-year rotation with millet grown the first year, groundnut the second and grazing was run to estimate future C stocks and sequestration potential for this “prunings burned” management under contemporary and changed climate. To investigate the possibilities of increasing SOC over the period 2005-2100, we extended the simulations using hypothetical alternative management practices based on the same two-year

millet-groundnut rotation with grazing under current and future climate change scenarios.

The hypothetical management alternatives included a "prunings returned" scenario where the clipped aboveground biomass is returned to the soil as an amendment, a "prunings returned-low fertilization" regime where the prunings are returned and 35 kg ha<sup>-1</sup> yr<sup>-1</sup> N and 20 kg ha<sup>-1</sup> yr<sup>-1</sup> P added and a "prunings returned-high fertilization" with 75 kg ha<sup>-1</sup> yr<sup>-1</sup> N and 20 kg ha<sup>-1</sup> yr<sup>-1</sup> P added. Current climate conditions simulated are based on average precipitation and temperature observed from 1962 to 2002, whereas future climate scenarios are drawn from work on African climate change by Hulme et al. (2001).

### **3.0 Results and Discussion**

#### **3.1. Regional biomass C stocks**

Table 4.2 shows measured biomass C stock at the 23 training plots. There was a tenfold difference in observed aboveground C stock ranging from 0.035 to 0.35 Mg C with an overall average of 0.186 Mg C (SEM =  $\pm 0.0176$ ) and a more than tenfold difference in belowground C stock (range = 2.0; mean = 1.03; SEM =  $\pm 0.087$ ). The highest biomass C was observed at the Ndiagne site which was identified as a 2-year old fallow while the lowest biomass was recorded at Thilla Ounte an intensively managed field. On average, there is approximately 5.8 times more biomass C in the belowground fractions as compared to the aboveground fraction and there was a significant but rather weak correlation (Belowground C =  $4.4598 \times \text{Aboveground C}^{0.8753}$ ;  $r = 0.87$ ) between these two C fractions.

To avoid degradation of pixel reflectance that would occur upon resizing the image to a 90 m x 90 m resolution that would be consistent with the biomass C training plots, the MTMF analysis was conducted on the 30 m x 30 m Landsat pixels. The output abundances in the 9 contiguous pixels corresponding to the training plot were then summed-up (post classification) to yield a single value for the 9 pixels (Table 4.2). Results show a linear and significant relationship between measured biomass C in the plots and the satellite image-derived aggregated shrub abundances (Figure 4.3), although a considerable amount of variability remains unexplained by this.

Table 4.2. Carbon stocks and satellite image derived shrub abundances at the training plots.

Site	Plot	Biomass C (Mg)		Pixel abundance
		Aboveground	Belowground	
Keur Asanulo (n = 3)	1	0.205	1.165	0.544
	2	0.143	0.825	0.381
	3	0.135	0.740	0.343
Keur Mandiemba (n = 4)	4	0.173	1.051	0.486
	5	0.123	0.714	0.330
	6	0.201	1.110	0.513
	7	0.137	0.847	0.392
Keur Matar Aram (n = 6)	8	0.284	1.617	0.748
	9	0.344	2.165	0.790
	10	0.095	0.469	0.217
	11	0.035	0.162	0.075
	12	0.276	1.391	0.643
	13	0.246	1.528	0.707
Ndiagne (n = 2)	14	0.355	1.032	0.477
	15	0.187	1.332	0.616
Keur Ibra Fall (n = 5)	16	0.345	1.098	0.508
	17	0.171	1.009	0.467
	18	0.157	0.776	0.359
	19	0.127	1.288	0.597
	20	0.149	0.826	0.382
Thilla Ounte (n = 3)	21	0.137	0.599	0.277
	22	0.122	0.828	0.383
	23	0.135	1.110	0.514

The strength of this relationship progressively declined with increasing distance between training plots reflecting variations in shrub spectral signature which are likely due to differences in shrub moisture content or differences in optical properties of underlying soils. The considerable amount of unexplained variability could be due to a number of reasons including the band number limitations of multispectral data, inaccuracies in georeferencing, variation in shrub spectral signature and the inherent failure of our methods to capture a more robust representation of the shrub canopies.

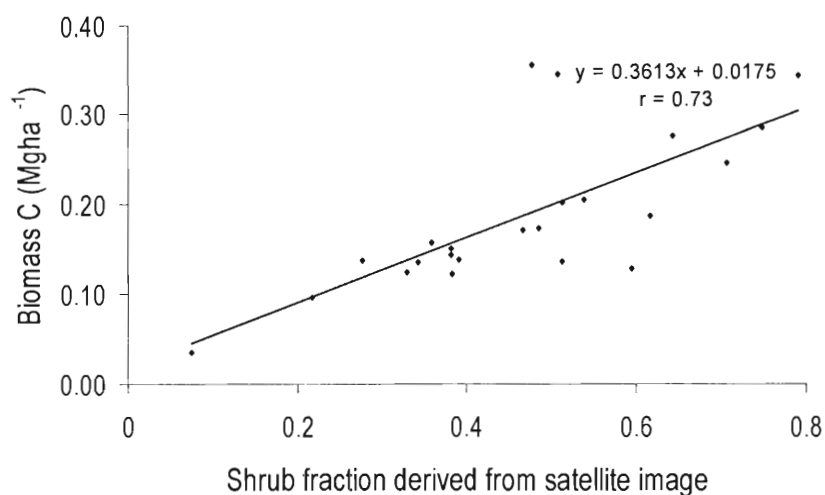


Figure 4.3. Relationship between biomass C stock and shrub abundance from spectral unmixing.

Applying the statistical model (Biomass C = 0.3613\*shrubs abundance + 0.0175) that describes the above relationship to the MTMF abundance image produced the predicted aboveground biomass C shown in Figure 4.4.

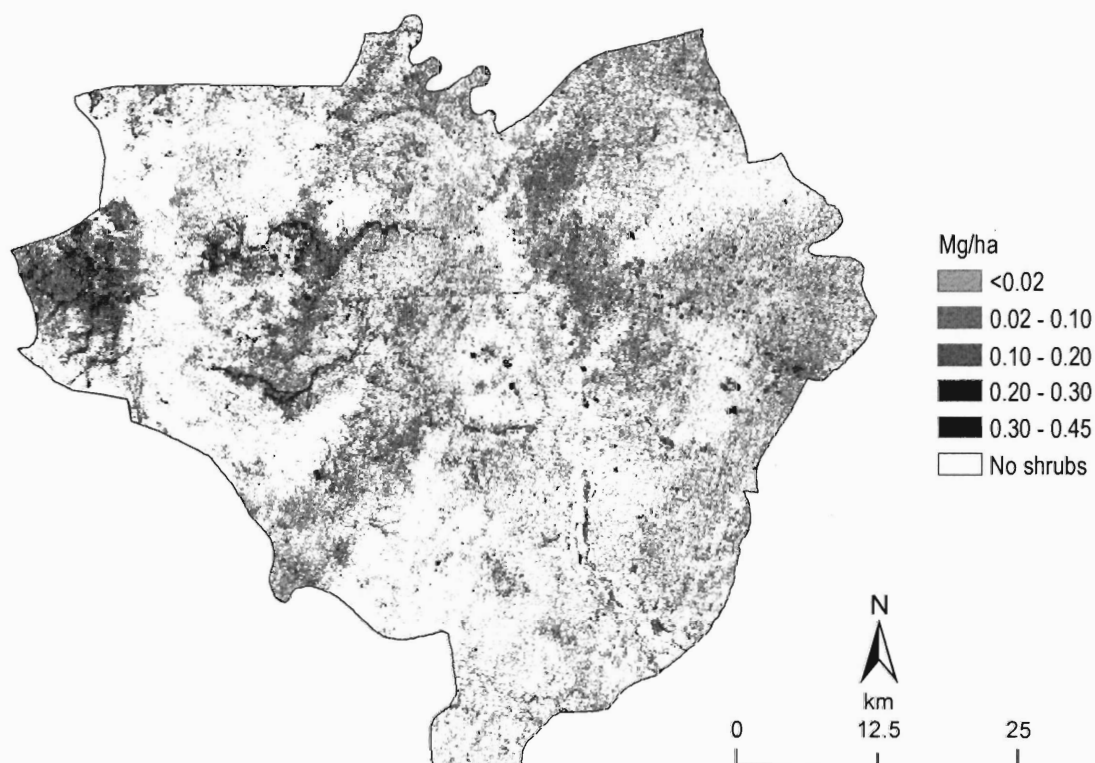


Figure 4.4. Predicted aboveground biomass C stocks in the study area. Shown values have been resampled to a 90 m resolution.

Model predicted aboveground biomass C ranged from 0.012 to 0.45 Mg C per 9 pixels (8,100m<sup>2</sup>) with no consistent pattern or trend across the study area although there were areas of high biomass C density that were consistent with our field observations. Either our statistical model or the general method underestimates biomass C because field observations show there are biomass C densities lower than 0.012 Mg and yet these are not captured in the output. This could once again be an artifact of the limitations of multispectral imagery in retrieving endmember fractions or a reflection of poor training plot selection. Although sampling plots were chosen to maximize variability and shrub biomass/cover did vary among these plots, we did not sample fields with very low shrub densities and hence did not get an excellent biomass C continuum for developing the model. The failure to retrieve low biomass C abundances could also be related to limitations of the MTMF algorithm in retrieving abundances where cover of the target is less than 5% of the pixel (McGwire et al., 2000). Estimated total aboveground biomass C in the 3062 sqkm study area is 36370 Mg and using the observed proportion of 1:5.8 for above-to-belowground biomass C, there is approximately 210,900 Mg of belowground C for a total of 247,000 Mg C in the study area at peak-season.

### 3.2. Spatial SOC dynamics

The simulation of the native shrub system under contemporary climate demonstrated that total annual shrub production ranged from 51 to 250 g C m<sup>-2</sup> yr<sup>-1</sup>, with an average of approximately 90 C m<sup>-2</sup> yr<sup>-1</sup> for the "prunings burned" situation. The "prunings returned" regime slightly reduced the average total annual shrub production by about 2.1% relative to the "prunings burned" scenario and we postulate that this reflects changes in shrub-grass/crop competition mediated by fire or stimulation of relatively higher production by fire. Increased nutrient availability through fertilization drastically improved ecosystem aboveground productivity and also increased belowground productivity to a less extent in both of the "prunings returned-fertilization" scenarios. Higher aboveground productivity (590 g C m<sup>-2</sup> yr<sup>-1</sup>) was predicted for the "prunings returned-high fertilization" scenario at equilibrium compared to the "prunings returned-low fertilization" regime (470 g C m<sup>-2</sup> yr<sup>-1</sup>). Average SOC at equilibrium ranged from 4.9 to 35 Mg ha<sup>-1</sup> depending on implemented management (Figure 4.5). The simulated total soil organic C of the 0-20 cm soil layer declined from 6.8 to ~4.9 Mg ha<sup>-1</sup> during the 64 yrs prior to equilibrium under the "prunings

burned" scenario. The model predicted dramatic changes in total SOC in the first 20 years after implementing the "prunings returned" and both of the "prunings returned-fertilization" management regimes and strongly suggested that significant C sequestration occurred after implementing these changes. These results under fertilization are contrary to Woerner et al. (2004) who proposed that C stocks could not be increased in this area through agricultural intensification because of the low rains and lack of access to irrigation.

Average total SOC at 0 to 20 cm increased from 6.8 to 27.2 Mg ha<sup>-1</sup> under the "prunings returned" scenario and from 6.8 to 33.4 Mg ha<sup>-1</sup> under the two "prunings returned-fertilization" regimes over approximately 25 yrs. Thereafter, SOC increased very slowly with average accumulation rates of 0.084 Mg ha<sup>-1</sup> yr<sup>-1</sup> and 0.11 Mg ha<sup>-1</sup> yr<sup>-1</sup> respectively for "prunings returned" and "prunings returned-fertilization" regimes over the remaining time of the simulation. The superior sequestration potential under the fertilization-prunings returned regimes is due to increased nutrient availability resulting from mineralization of shrub residues accompanied by increased substrate for microbial processes and the direct application of chemically recalcitrant residues.

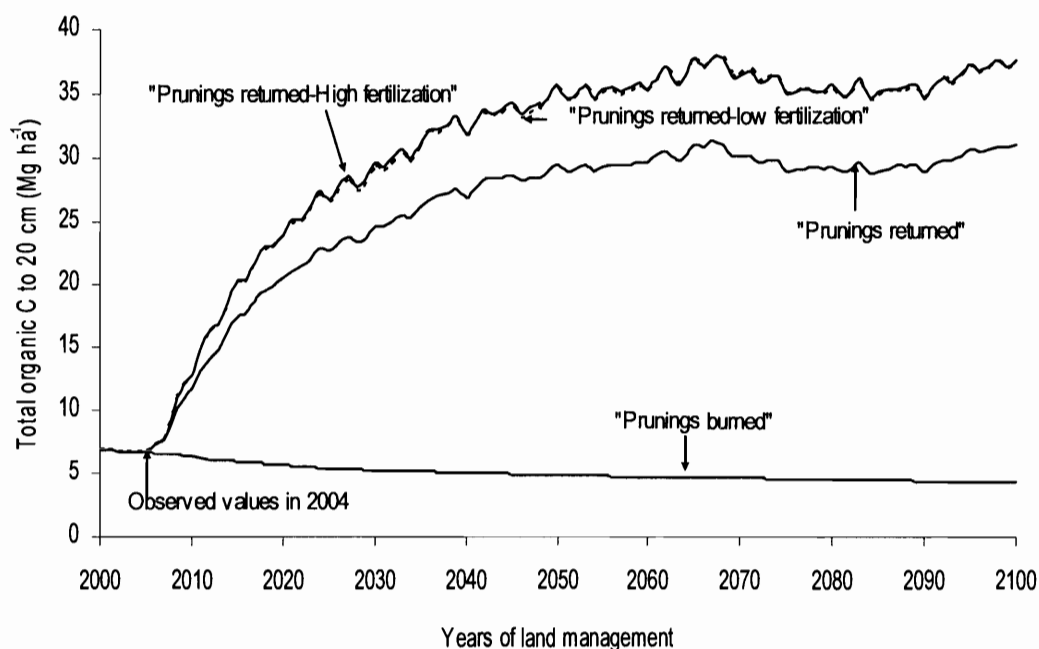


Figure 4.5. Simulation of total organic C in soils (0-20 cm depth) by CENTURY model under the four management scenarios.

Although fertilization at  $75 \text{ kg ha}^{-1} \text{ yr}^{-1}\text{N}$  and returning prunings produced more biomass, the SOC sequestered for this scenario closely mirrored the SOC sequestered under a lower biomass regime produced by  $35 \text{ kg ha}^{-1} \text{ yr}^{-1}\text{N}$  and returning prunings indicating that beyond a certain threshold, it is not possible to increase SOC by amending with chemical fertilizers or increased production of biomass. Under very fertile conditions ("prunings returned-high fertilization"), CENTURY predicts either the expression of a new hierarchical limitation (such as moisture availability) to C sequestration or an inherent insensitivity to SOC response under intense inputs. The "prunings returned-high fertilization" regime resulted in superior crop yields, i.e.  $2.1$  and  $2.5 \text{ Mg ha}^{-1}$ , respectively, for millet and peanut compared to  $1.8$  and  $2.2 \text{ Mg ha}^{-1}$  under the "prunings returned-high fertilization" regime. An economic analysis of the returns was beyond the scope of this study, but findings seem to corroborate findings by Woerner et al. (1998) who compared the C sequestration efficiency vs. the economic returns of six alternative SOC sequestration strategies in smallholdings in Kenya and showed that it was not possible to maximize both. It is also worth noting that the improved management strategies modeled here could easily conflict with farmers' major production and livelihood objectives because they involve taking economic risks to fertilize and may also require additional labor to manage the prunings.

The spatial distribution of simulated SOC in the study area at selected instances in time for the different management scenarios under the present climate is shown in figures 4.6 through 4.8. Depending on location, simulated time to reach a steady state ranged from about 45 to 60 years and sequestration rate ranged from  $0.09$  to  $0.25 \text{ Mg ha}^{-1}\text{yr}^{-1}$  for the "prunings returned" regime and from  $0.09$  to  $0.32 \text{ Mg ha}^{-1}\text{yr}^{-1}$  for both of the "prunings returned-fertilization" scenarios indicating variability in sequestration potential as a function of climate-soil type permutations. There is a north-south rainfall gradient in the area with precipitation increasing by at least  $1 \text{ mm km}^{-1}$  southwards (Camberlin and Diop, 1999). We therefore expected higher SOC values to the south as a consequence of greater plant growth (driven by precipitation) and thus superior litter inputs to soil C pools. However, equilibrium SOC values are surprisingly higher around the Thies region (northwestern part of the study area) compared to the southern parts of the area. It is highly likely that SOC in this area is more sensitive to differences in soil texture (there are heavier soils in that region) than it is to rainfall or the differences in rainfall amounts are not significant enough to elicit differences in net primary production across the study area.



Table 4.3 shows the study area averaged C stocks grouped by soil type. Under traditional management, the highest SOC loss rate is predicted for the Torrerts ( $5.28 \text{ g m}^{-2} \text{ yr}^{-1}$ ) and the lowest loss rate in Torrox ( $2.38 \text{ g m}^{-2} \text{ yr}^{-1}$ ). The fact that the soil with the lowest initial C loses the least amounts probably reflects shift towards an equilibrium at which further losses will not occur. As a proportion of 2005 stocks, the greatest SOC loss is predicted for Psammets (~38%) and the lowest loss is predicted for the Torrerts (~31%).

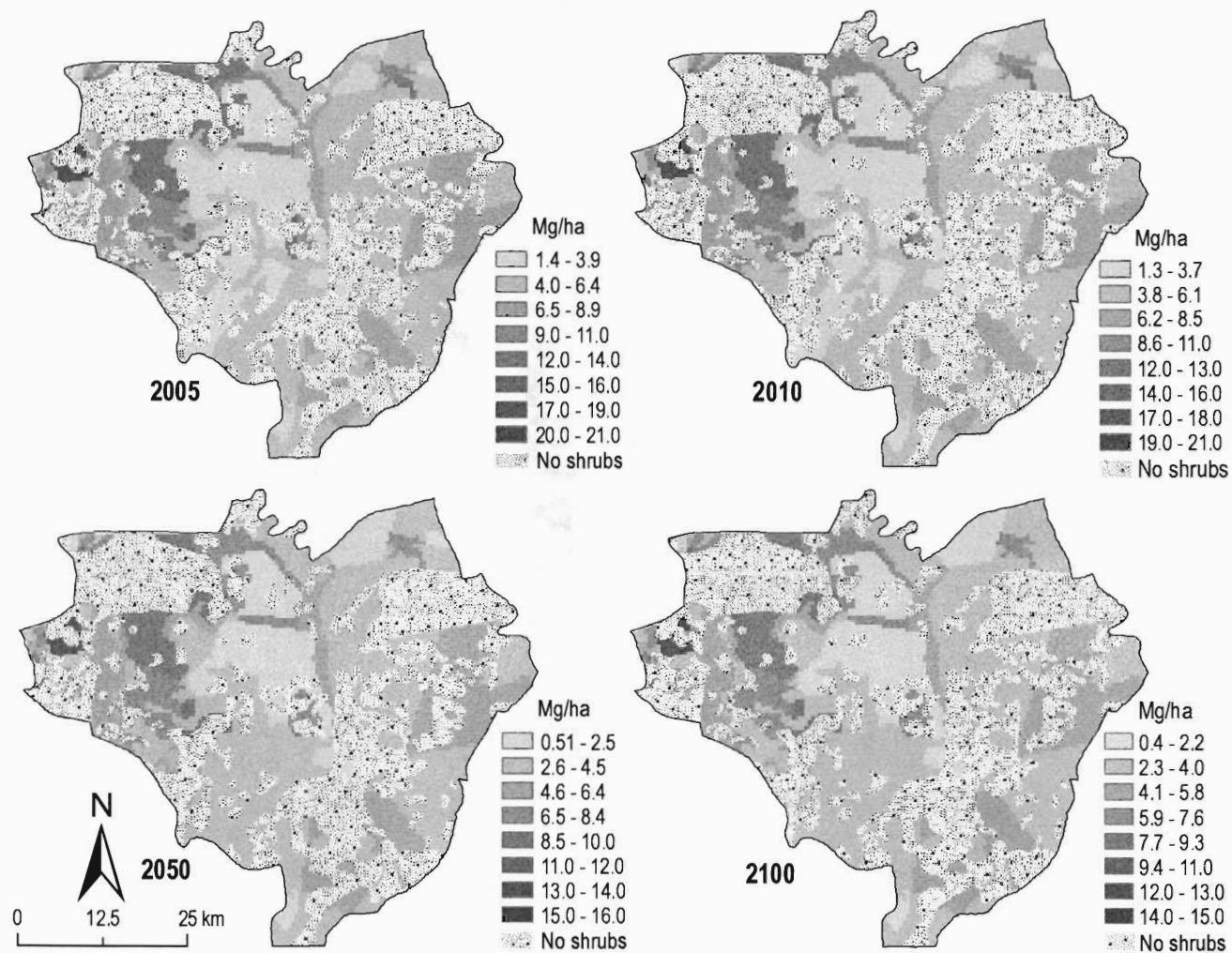


Figure 4.6. Spatial distribution of predicted SOC under the "prunings burned" regime in 2005, 2010, 2050 and 2100.

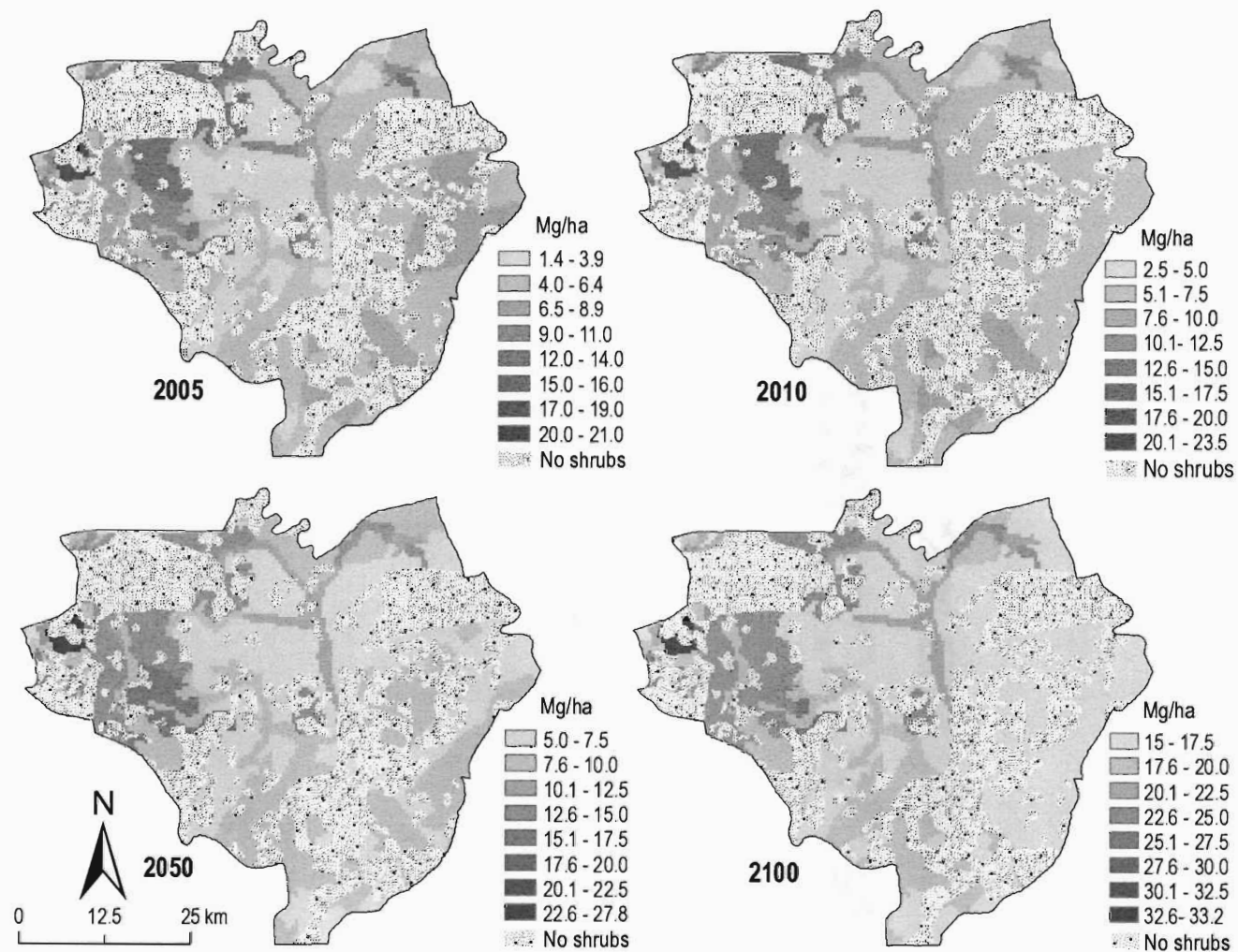


Figure 4.7. Spatial distribution of predicted SOC under the “prunings returned” regime in 2005, 2010, 2050 and 2100.

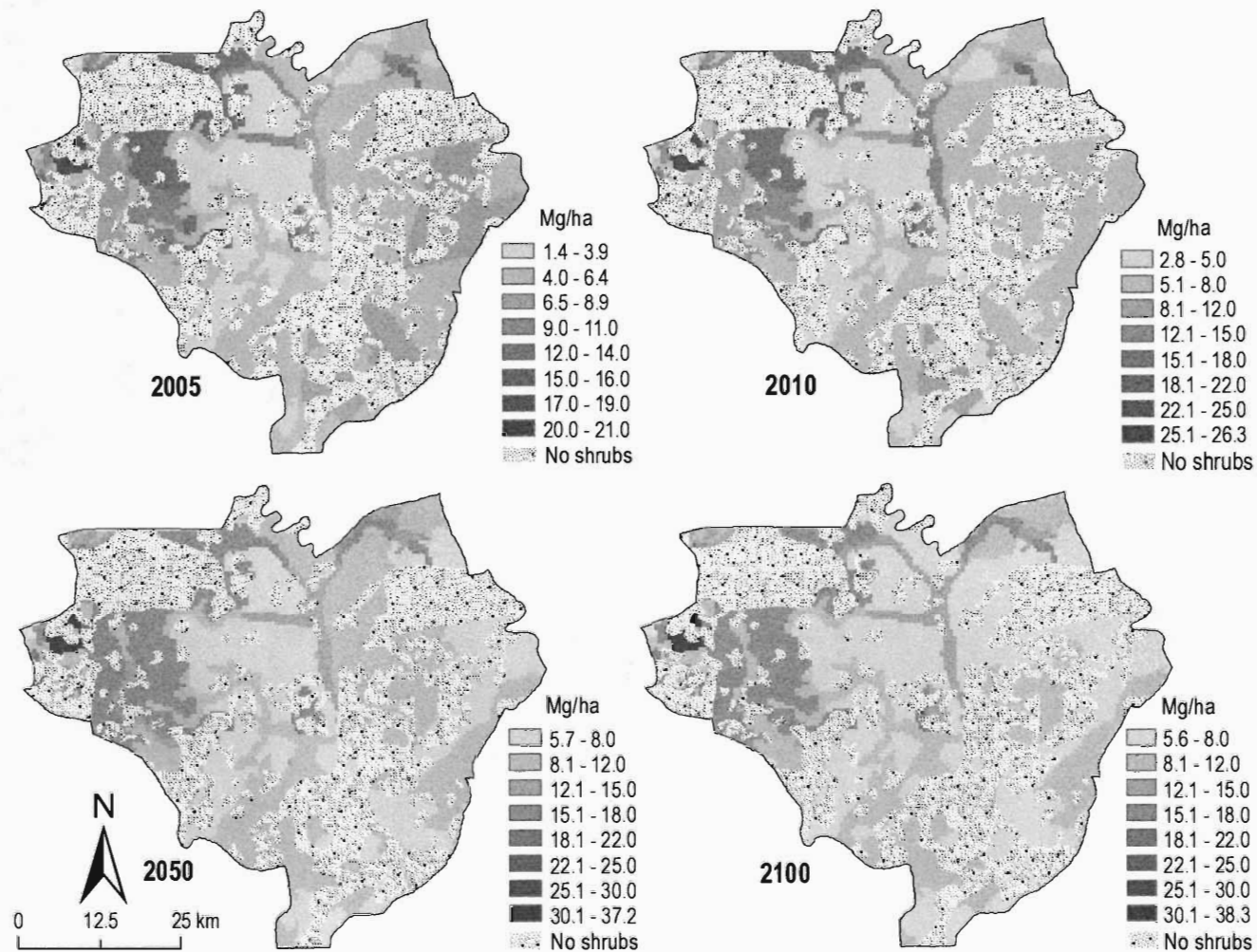


Figure 4.8. Spatial distribution of predicted SOC under the “prunings returned-low fertilization” regime in 2005, 2010, 2050 and 2100.

Table 4.3. Simulated average SOC in the different soils of the study area in 2010, 2050 and 2100. Initial C stock in the year 2005 was 15.8, 6.3, 8.1, 6.6 and 10.6 Mg ha<sup>-1</sup> respectively for Torrerts, Torrox, Psammets, Halosols and Torrents.

Soil type†/Scenario	Mean C stock (Mg ha <sup>-1</sup> )			Annual net C gain (Mg ha <sup>-1</sup> yr <sup>-1</sup> )
	2010	2050	2100	
<i>Prunings burned</i>				
Torrerts	15.38	12.16	10.92	-0.0528
Torroxs	5.97	4.47	4.04	-0.0238
Psammets	7.73	5.68	5.05	-0.0322
Halosols	6.27	4.74	4.29	-0.0244
Torrents	10.14	7.82	7.07	-0.0368
<i>Prunings returned</i>				
Torrerts	16.30	31.99	37.40	0.2270
Torroxs	6.27	16.86	17.99	0.1231
Psammets	8.74	23.44	26.35	0.1921
Halosols	7.53	18.02	19.41	0.1348
Torrents	11.40	23.66	26.24	0.1650
<i>Prunings returned-low fertilization</i>				
Torrerts	12.77	36.88	43.22	0.2882
Torroxs	6.63	19.79	21.09	0.1557
Psammets	9.94	26.85	30.15	0.2321
Halosols	7.93	21.25	22.92	0.1717
Torrents	12.32	27.20	30.19	0.2065

† The average proportion (%) of sand, silt and clay is as follows for the soils: Torrerts = 44.0, 18.9, 37.1; Torrox = 92.3, 3.2, 4.5; Psammets = 48.5, 24.2, 27.3; Halosols = 91.3, 4.4, 4.3; Torrents = 88.2, 4.5, 7.3.

Model predictions indicate that if the “prunings returned” and the “prunings returned-fertilization” regimes were implemented, highest absolute SOC gains are expected to occur in Torrerts (0.23 and 0.29 Mg ha<sup>-1</sup> yr<sup>-1</sup> respectively for “prunings returned” and “prunings returned-fertilization” regimes) and the lowest gains are expected to occur in Torrox (0.12 and 0.16 Mg ha<sup>-1</sup> yr<sup>-1</sup> respectively for the “prunings returned” and “prunings returned-fertilization” regimes). As a proportion of current C stocks, the Halosols are expected to have the highest C gains (225% and 274% respectively for “prunings returned” and “prunings returned-fertilization” regimes respectively) while the lowest gains are expected to occur for the Torrerts. SOC differences among the five soils indicate model sensitivity to soil texture. As expected, higher sequestration rates are registered in Torrerts because clay particles provide greater protection and sorption surface area for soil organic matter than do sand and silt (Anderson et al., 1988; Stevenson, 1994; Amelung et al., 1998; Balesdent et al., 1998).

Through a variety of flexibility mechanisms, the Kyoto Protocol allows trading in carbon emissions permits or credits where C sequestered by one party can be used to offset emissions in another sector of the national economy or traded internationally to fulfill national commitments. The improved management strategies in the simulation could result in the supply of marketable quantities of C sequestered in the soils, opening up possibilities for farmers in this region to obtain financing to support soil resource management. However, if carbon trading were to occur, the low sequestration rates and potential in Tropical Ferruginous soils (covering 81% of the area) will greatly reduce the amount of C credits that farmers could accumulate.

### **3.3. Impact of climate change on SOC sequestration**

Simulated response to changes in climate were complex, underscoring the potentially large impact of climate change on these ecosystems. Increasing mean monthly temperature by 1.5°C and decreasing precipitation by 1 mm (climate change scenario 1) reduced total SOC by an average of 8% for the “prunings burned” while an increase in mean monthly temperature by 3°C and a reduction in precipitation of 2 mm (climate change scenario 2) increased C loss by about 17% relative to the contemporary climate. Although total ecosystem C still increased under both climate change scenarios for the “prunings returned-fertilization” regimes, there was a more pronounced effect of both climate changes on total SOC (i.e. an average reduction of 21% and 23% respectively for climate change scenario 1 and 2) which could be due to reduced net primary productivity occasioned by a decline in precipitation and increased temperature regulation of microbial activity and mineralization of resident soil organic matter.

## **4.0 Acknowledgments**

This research was supported by the Geoscience Biocomplexity Program of the National Science Foundation (Award 0120732). We thank Papa Serigne Sarr, Younouss Tamba, Papa Omar Dieye and Moussa Diop of ISRA for helping in data collection, Analytical Spectral Devices (ASD) for lending us the Spectroradiometer and Joan Sandeno for editing the manuscript.

## CHAPTER FIVE

### Conclusions to the Dissertation

Land degradation in sub-Saharan Africa (SSA) has led to a decline in soil quality which is related to the reduction in C content due to oxidation of organic matter from intensification of land use, and emission of C from soil and ecosystems to the atmosphere. A major factor implicated in these trends is the loss of woody species and how they are managed in SSA. Land degradation amelioration and soil quality enhancement through restoration of degraded soils and ecosystems can lead to improved productivity, increase in soil and biomass C pools, and offset global climate change. Technologies to mitigate land degradation include improved cropping systems, pastures, fertility enhancement, crop residue management and establishment of appropriate xerophytic or halomorphic woody species (Lal and Dumanski, 1999) *inter alia*. In SSA, there is an unrecognized shrub component (comprising of *G. senegalensis* and *P. reticulatum*) that has potential to significantly impact C cycling and ecological status of the landscape. Currently, little is known about how these shrubs affect the functioning of SSA landscapes or how to manage them to optimize C return and sequestration to the soil and improve the sustainability of the cropping systems in this region. This study provided a better understanding of the factors that mediate the distribution and abundance of these shrub species and their potential roles in C sequestration and land degradation mitigation.

Formal Inference Recursive Modeling and logistic regression coupled with geostatistics were useful in examining factors and developing models of shrub distribution and abundance. Distribution of the shrubs is controlled by four factors: geological substrate, mean annual temperature, mean annual rainfall and terrain profile convexity. Relative abundance within the shrub communities is influenced by mean annual rainfall, maximum annual temperature and elevation; for *G. senegalensis* whereas for *P. reticulatum* it was mean annual rainfall, mean annual temperature, elevation and profile convexity. Shrub communities differed in their preference for geological substrate and overall, *G. senegalensis* exhibited a fairly more ubiquitous distribution while *P. reticulatum* is restricted to the relatively wetter areas. The derived distribution and abundance models were generally poor; probably highlighting the weakness of statistical models in analysis and quantification of the spatial structure of ecosystems or

limitations imposed by the quality and scale of input GIS data. It is suggested that recourse to the realm of the shrub physiology is the leeway to completely unravel the spatial structure and shrub-edaphic factor associations in these ecosystems and this can best be done by controlled experiment. Incorporation of finer-scale variables related to soil properties and site history are also expected to improve model performance.

Estimates of peak-season biomass C ranged from 0.9 Mg C ha<sup>-1</sup> to 1.4 Mg C ha<sup>-1</sup> with an overall mean of 1.12 Mg C ha<sup>-1</sup> (SEM =  $\pm 0.079$ ) in the *G. senegalensis* shrub communities and from 1.3 to 2.0 Mg C ha<sup>-1</sup> (mean = 1.57 Mg C ha<sup>-1</sup>; SEM =  $\pm 0.18$ ) in the *P. reticulatum* communities. The overall mean of SOC to 40 cm was 17 and 17.2 Mg C ha<sup>-1</sup> respectively, at the *G. senegalensis* and *P. reticulatum* sites with 57% of that C residing in the top 20 cm. A modest proportion (10- 18%) of the annual biomass C produced is returned to the soil while the rest is lost to burning leading these systems to annually lose about  $3.51 \times 10^5$  Mg of biomass C, an equivalent of  $2.11 \times 10^5$  Mg of CO<sub>2</sub> per year to the atmosphere. A significant finding of this study is that the shrubs have a stronger influence on overall observed landscape level SOC values than do the trees although the relative influence is greater in *P. reticulatum* communities which could be attributed to differences in biomass productivity, rainfall amounts and surface soil redistribution processes. Weighed against other sources of soil C (crop residues and tree leaves) in the study area, the study showed that shrubs are the dominant source of C to soils in the Peanut Basin.

In a third study, biomass carbon stocks measured at the plot-level were scaled-up to regional levels for a section of the Peanut Basin and their regional dynamics assessed using the CENTURY Model. The study presents a methodology for coupling the point-based CENTURY soil organic matter model, with a spatially explicit (GIS) database. The loose coupling between the model and GIS was assisted with existing and newly developed techniques for automating the spatial modeling. Results indicated that the selected section of the Peanut Basin contains approximately 247,000 Mg of biomass C and that C sequestration in these systems hinges on long-term non-thermal management of shrub biomass with actual sequestration rates depending strongly on scenarios of future land management and soil type. Compared with the contemporary "pruning-burned" management, returning prunings for 50 years would increase soil C sequestration by 200 to 350% under a no fertilization regime, and increase soil C sequestration by 270 to 480% under a low (35 kg ha<sup>-1</sup> N yr<sup>-1</sup>; 20 kg ha<sup>-1</sup> P yr<sup>-1</sup>) fertilization regime depending on soil type and climatic regime.



## Bibliography

- Abrahams, A.D., Parsons, A.J., Wainwright, J., 1994. Resistance to overland flow on semi-arid grassland and shrubland hillslopes, Walnut Gulch, Southern Arizona. *Journal of Hydrology* 156, 431-446.
- Adams, J.B., Smith, M.O., Gillespie, A.R., 1993. Imaging Spectroscopy: Interpretation based on spectral mixture analysis. In: Pieters, C.M., Englert, P. (Eds.), *Remote Geochemical Analysis: Elemental and Mineralogical Composition*, Cambridge University Press, New York, pp.145-166.
- Aguiar, M.R., Sala, O.E., 1999. Patch structure, dynamics and implications for the functioning of arid ecosystems. *Trends in Ecological Evolution* 14, 273-277.
- Albrecht, A., Kandji, T.S., 2002. Carbon sequestration in tropical agroforestry systems. *Agriculture, Ecosystems and Environment* 99,15-27.
- Amelung, W., Zech, W., Zhang, X., Follett, R.F., Tiessen, H., Knox, E., Flach, K.W., 1998. Carbon, nitrogen, and sulfur pools in particle size fractions as influenced by climate. *Soil Science Society of America Journal* 62, 172-181.
- Anderson, D.W., Sagggar, S., Bettany, J.R., Stewart, J.W.B., 1981. Particle size fractions and their use in studies of soil organic matter: I. The nature and distribution of forms of carbon, nitrogen, and sulfur. *Soil Science Society of America Journal* 45, 767-772.
- Archer, S., Scifres, C., Bassham, C.R., Maggio, R., 1988. Autogenic succession in a subtropical savanna: conversion of grassland to thorn woodland. *Ecological Monographs* 58,111-127.
- Ardo, J., Olsson, L., 2003. Assessment of soil organic carbon in semi-arid Sudan using GIS and the CENTURY model. *Journal of Arid Environments* 54, 633-651.
- Asner, G.P., Heidebrecht, K.B., 2002. Spectral unmixing of vegetation, soil and dry carbon cover in arid regions: comparing multispectral and hyperspectral observations. *International Journal of Remote Sensing* 23, 3939-3958.
- Atteia, O., Dubois, J.P., Webster, R., 1994. Geostatistical analysis of soil contamination in the Swiss Jura. *Environmental Pollution* 86, 315-327.
- Ba, M., Mbaye, M., Ndao, S., Wade, A., Ndiaye, L., 2000. Drylands Research Working Paper 21 Région de Diourbel: Volet cartographique des changements dans l'occupation et l'utilisation des sols. Dryland Research, Somerset, England.
- Badiane, A.N., Khouma, M., Sene, M., 2000. Region de Diourbel: Gestion des sols. Drylands Research Working Paper 15, Drylands Research, Somerset, UK.
- Baillie, I.C., Ashton, P.S., Court, M.N., Anderson, J.A.R., Fitzpatrick, E.A., Tinsley, J., 1987. Site characteristics and the distribution of tree species in mixed dipterocarp forest on tertiary sediments in central Sarawak, Malaysia. *Journal of Tropical Ecology* 3, 210-220.
- Balesdent, J., Besnard, E., Arrouays, D., Chenu, C., 1998. The dynamics of carbon in particle-size fractions of soil in a forest cultivation sequence. *Plant and Soil* 201, 49-57.

- Balesdent, J., Chenu, C., Balabane, M., 2000. Relationship of soil organic matter dynamics to physical protection and tillage. *Soil and Tillage Research* 53, 215-220.
- Barbier, E.B., 2000. The Economic Linkages between Rural Poverty and Land Degradation: Some Evidence from Africa. *Agriculture, Ecosystems and Environment* 82, 355-370.
- Barrow, C.J., 1991. *Land Degradation: Development and Breakdown of Terrestrial Environments*, Cambridge University Press, United Kingdom.
- Bartolome, J.W., Kosco, B.H., 1982. Estimating browse production by deerbrush (*Ceanothus integerimus*). *Journal of Range Management* 35,671-672.
- Bationo, A., Buerkert, A., 2001. Soil organic carbon management for sustainable land use in Sudano-Sahelian West Africa. *Nutrient Cycling in Agroecosystems* 61, 131-142.
- Bationo, A., Lompo, F., Koala, S., 1998. Research on nutrient flows and balances in West Africa: state of the art. *Agriculture Ecosystems and Environment* 71, 19-35.
- Bationo, A., Mokwunye, A.U., 1991. Alleviating soil fertility constraints to increased crop production in West Africa: The experience in the Sahel. *Fertilizer Research* 29, 95-115.
- Batjes, N.H., 1996. Total carbon and nitrogen in the soils of the world. *European Journal of Soil Science* 47, 151-163.
- Batjes, N.H., 1999. Management options for reducing CO<sub>2</sub>- concentrations in the atmosphere by increasing carbon sequestration in the soil, ISRIC, Wageningen, The Netherlands, 114 pp.
- Batjes, N.H., 2001. Options for increasing carbon sequestration in West African soils: an explanatory study with special focus on Senegal. *Land Degradation and Development* 12, 131-142.
- Beckett, P.H.T., Webster, R., 1971. Soil variability: a review. *Soils and Fertilizers* 34, 1-15.
- Belsky, A.J., 1986. Population and community processes in a mosaic grassland in the Serengeti, Tanzania. *Journal of Ecology* 74, 841-856.
- Bentley, J.R., Seegrist, D.W., Blakemen, D.A., 1970. A technique for sampling low shrub vegetation by crown volume classes. USDA Forest Service Research Note. PSW-215.
- Blaikie, P., 1989. Explanation and Policy in Land Degradation and Rehabilitation for Developing Countries. *Land Degradation and Rehabilitation* 1, 23-37.
- Blaikie, P., Brookfield, H., 1987. *Land Degradation and Society*, Methuen, London.
- Boaler, S.B., Hodge, C.A.H., 1962. Vegetation stripes in Somaliland. *Journal of Ecology* 50,465-474.
- Boardman, J.W., Kruse, F.A., Green, R.O., 1995. Mapping target signatures via partial unmixing of AVIRIS data. Summaries of the 5<sup>th</sup> JPL Airborne Earth Science Workshop, JPL Publication 95-1, 1, 23-26.
- Boardman, J.W., Kruse, F.A., 1994. Automated spectral analysis: a geological example using AVIRIS data, north Grapevine Mountains, Nevada: In *Proceedings, ERIM Tenth*

Thematic Conference on Geologic Remote Sensing, Environmental Research  
Institute of Michigan, Ann Arbor, MI, pp.1407-1418.

- Boerner, R.E.J., Koslowsky, S.D., 1989. Microsite variations in soil chemistry and nitrogen mineralization in a beech-maple forest. *Soil Biology and Biochemistry* 21, 795-801.
- Bonfils, P., Faure, J., 1956. Les sols de la region de Thies: Annales du Centre de Recherches Agronomiques de Bambey au Senegal. *Bulletin Agronomique* 16, 6-92.
- Boots, B., Getis, A., 1978. Point pattern analysis, London, Sage Publications.
- Bowers, M.A., Lowe, C.H., 1986. Plant-form gradients on Sonoran desert bajadas, *Oikos* 46, 284-291.
- Brady, N.C., Weil, R.R., 1999. The nature and properties of soils, Twelfth edition, Prentice Hall, New Jersey.
- Brand, G.J., Smith, W.B., 1985. Evaluating allometric shrub biomass equations fit to generated data. *Canadian Journal of Botany* 63, 64-67.
- Breiman, L., Friedman, J.H., Olshen, R.A., Stone, C.J., 1984. Classification and regression trees, Wadsworth, Belmont.
- Breman, H., Kessler, J.J., 1995. Woody plants in agro-ecosystems of semiarid regions. In: Advanced series in agricultural sciences, Volume 23, Springer, Berlin, Germany, 340 pp.
- Bridge, S.R.J., Johnson, E.A., 2000. Geomorphic principles of terrain organization and vegetation gradients. *Journal of Vegetation Science* 11, 57-70.
- Bromley, J., Brouwer, J., Barker, A.P., Gaze, S.R., Valentin, C., 1997. The role of surface water redistribution in an area of patterned vegetation in a semi-arid environment, southwest Niger. *Journal of Hydrology* 198, 1-29.
- Brouwer, J., Bouma, J., 1997. Soil and crop growth variability in the Sahel: highlights of research (1990-1995) at ICRISAT Sahelian Centre. Information Bulletin 49, ICRISAT, Patancheru, Andhra Pradesh, India.
- Brouwer, R., 1962. Distribution of dry matter in the plant. *Netherlands Journal of Agricultural Sciences* 10, 399-408.
- Brown, J.K., 1976. Estimating shrub biomass from basal stem diameters. *Canadian Journal of Forest Research* 6, 153-158.
- Bryant, F.C., Kothmann, M.M., 1979. Variability in predicting edible browse from crown volume. *Journal of Range Management* 32, 144-146.
- Buckman, R.E., 1966. Estimation of cubic volume of shrubs (*Corylus spp.*). *Ecology* 47, 858-860.
- Buech, R.R., Rugg, D.J., 1989. Biomass relations of shrub components and their generality. *Forest Ecology and Management* 26, 257-264.
- Buresh, R.J., Tian, G., 1998. Soil improvement by trees in sub-Saharan Africa. *Agroforestry Systems* 38, 51-76.
- Burrough, P.A., 1986. Principles of Geographical Information Systems for land Resources Assessment, Clarendon, Oxford.

- Cain, S.A., 1944. Foundations of Plant Geography, Harpers and Brothers, London.
- Callaway, R.M., Davis, F.W., 1998. Recruitment of *Quercus agrifolia* in central California: the importance of shrub-dominated patches. *Journal of Vegetation Science* 9, 647-656.
- Camberlin, P., Diop, M., 1999. Inter-relationships between groundnut yield in Senegal, interannual rainfall variability and sea-surface temperatures. *Theoretical and Applied Climatology* 63, 163-181.
- Chambers, J.M., Hastie, T.J., 1992. Statistical Models in S, Chapman and Hall, London.
- Charreau, C., Vidal, P., 1965. Influence de l'*Acacia albida* Del. sur le sol, la nutrition minerale et les rendements des mils *Pennisetum* au Senegal. *L'Agronomie Tropicale* 7, 601-626.
- Conacher, A.J., 2001. Land degradation, Kluwer, Dordrecht, Netherlands.
- Conant, R.T., Paustian, K., 2002. Spatial variability of soil organic carbon in grasslands: implications for detecting changes at different scales. *Environmental Pollution* 116, 127-135.
- Coppinger, K.D., Reiners, W.A., Burke, I.C., Olson, R.K., 1991. Net erosion on a sagebrush steppe landscape as determined by cesium-137 distribution. *Soil Science Society of America Journal* 55, 254-258.
- Covington, W.W., Everett, R.L., Steele, R., Irwin, L.L., Daer, T.A., Auclair, A.N.D., 1994. Historical and anticipated changes in forest ecosystems of the inland west of the United States. *Journal of Sustainable Forestry* 2, 13-63.
- Cressie, N., 1985. Fitting variogram models by weighted least squares, *Mathematical Geology* 17, 563-586.
- Cressie, N., 1993. Statistics for spatial data, Wiley and Sons, New York.
- Dacosta, H., 1989. Precipitations et ecoulements sur le bassin de la Casamance, Ph.D. Thesis, University of Cheikh Anta Diop, Dakar.
- De Gier, A., Sakouhi, A., 1996. Woody biomass mapping, using field data and SPOT-satellite imagery. In: Päivinen, R., Vanclay, J., Miina, S. (Eds.), *New Thrusts in Forest Inventory*, EFI Proceedings No 7, pp. 205-211.
- De Jong, S.M., Pebesma, E.J., Lacaze, B., 2003. Above-ground biomass assessment of Mediterranean forests using airborne imaging spectrometry: the DAIS Payne experiment. *International Journal of Remote Sensing* 24, 1505-1520.
- Decoursey, D.G., 1992. Developing models with more detail: do more algorithms give more truth? *Weed Technology* 6, 709-715.
- Diggle, P.J., 1983. Statistical analysis of spatial point patterns, Academic Press, London, England.
- Diouf, A., Lambin, E.F., 2001. Monitoring land-cover changes in semi-arid regions: remote sensing data and field observations in the Ferlo, Senegal. *Journal of Arid Environments* 48, 129-148.
- Dixon, R.K., 1995. Agroforestry systems: sources or sinks of greenhouse gas? *Agroforestry systems* 31, 99-116.

- DOE, 1999. Carbon Sequestration: State of the Science. United States Department of Energy (DOE), Washington, DC.
- Drechsel, P., Gyiele, L., Kunze, D., Cofie, O., 2001. Population Density, Soil Nutrient Depletion and Economic Growth in sub-Saharan Africa. *Ecological Economics* 38, 251-258.
- Du, Y., Guindon, B., Cihlar, J., 2002. Haze detection and removal in high resolution satellite image with wavelet analysis. *IEEE Transactions on Geoscience and Remote Sensing* 40, 210-217.
- Elberling, B., Touré, A., Rasmussen, K., 2003. Changes in soil organic matter following groundnut-millet cropping at three locations in semi-arid Senegal, West Africa. *Agriculture, Ecosystems and Environment* 96, 37-47.
- Elkins, N.Z., Sabol, G.V., Ward, T.J., Whitford, W.G., 1986. The influence of subterranean termites on the hydrological characteristics of a Chihuahuan desert ecosystem. *Oecologia* 68, 521-528.
- Ellenberg, H., 1988. *Vegetation ecology of Central Europe*, Cambridge University Press, Cambridge.
- Elliot, E.T., Cambardella, C.A., Cole, C.V., 1993. Modification of ecosystem processes by management and mediation of soil organic matter. In: Mulungoy, K., Merckx, R. (Eds.), *Soil organic matter dynamics and sustainability of tropical agriculture*, John Wiley and Sons, Chichester, UK, pp. 257-268.
- Elmore, J.E., Mustard, J.F., Manning, S.J., Lobell, D.B., 2000. Quantifying vegetation change in semi-arid environments: precision and accuracy of spectral mixture analysis and the normalized difference vegetation index. *Remote Sensing of Environment* 73, 87-102.
- Elvidge, C.D., Yuan, D., Weerackoon, R.D., Lunetta, R.S., 1995. Relative radiometric normalization of Landsat Multispectral Scanner (MSS) data using an automatic scattergram controlled regression. *Photogrammetric Engineering and Remote Sensing* 61, 1255-1260.
- ENVI, 2004. *The Environment for Visualization of Images. ENVI™ User Guide. ENVI™ version 4.1*. Research System Inc., 864 pp.
- Falloon, P., Smith, P., Szabó, J., Pásztor, L., 2002. Comparison of approaches for estimating carbon sequestration at the regional scale. *Soil Use and Management* 18, 164-174.
- FAO, 1998. Food and Agriculture Organization of the United Nations. *World Soil Resources Reports 84*. ISBN 92-5-104141-5, FAO, Rome, Italy.
- FAO, 2001. Food and Agriculture Organization of the United Nations. *Soil carbon sequestration for improved land management, World Soil Resources Report 96*, FAO, Rome, Italy.
- Fassbender, H.W., Beer, J., Henveldop, J., Imbach, A., Enriquez, G., Bonnemann, A., 1991. Ten year balance of organic matter and nutrients in agroforestry systems at CATIE Costa Rica. *Forest Ecology and Management* 45, 173-183.
- Ferrari, T.J., Vermeulen, F.H.B., 1995. Soil heterogeneity and soil testing. *Netherlands Journal of Agricultural Science* 3, 265-275.

- FieldSPEC, 1997. User's Guide, manual release, Analytical Spectral Devices, Inc. Boulder, Colorado, USA.
- Franklin, J., 2002. Enhancing a regional vegetation map with predictive models of dominant plant species in chaparral. *Applied Vegetation Science* 5, 135-146.
- Freeman, P.H., 1992. Land regeneration and agricultural intensification in Senegal's Groundnut Basin. United States Agency for International Development (USAID), Dakar, Senegal, 29 pp.
- Fuentes, E.R., Otaiza, R.D., Alliende, M.C., Hoffmann, A., Poiani, A., 1984. Shrub clumps of the Chilean matorral vegetation: structure and possible maintenance mechanisms. *Oecologia* 62,405-411.
- Funk, C.C., Theiler, J., Roberts, D.A., Borel, C.C., 2001. Clustering to improve matched filter detection of weak gas plumes in hyperspectral thermal imagery. *IEEE Transactions on Geoscience and Remote Sensing* 39, 1410-1420.
- Garcia-Haro, F.J., Gilabert, M.A., Melia, J., 1996. Linear spectral mixture modeling to estimate vegetation amount from optical spectral data. *International Journal of Remote Sensing* 17, 3373-3400.
- Garner, W., Steinberger, Y., 1989. A proposed mechanism for the formation of 'fertile islands' in the desert ecosystem. *Journal of Arid Environments* 16, 257-262.
- Gastellu, J.M., 1981. L'égalitarisme économique des Serers du Sénégal. Travaux et documents, No. 128. ORSTOM, Paris.
- Gerard, B., Buerkert, A., 1999. Aerial photography to determine fertilizer effects on pearl millet and *Guiera senegalensis* growth. *Plant and Soil* 210, 167-177.
- Getis, A., Franklin, J., 1987. Second-order neighborhood analysis of mapped point patterns. *Ecology* 68, 473-477.
- Gonzalez, R.C., Woods, R.E., 2002. Digital image processing, Second edition, New Jersey, Prentice Hall.
- Good, R., 1953. *The Geography of the Flowering Plants*, Longman, London.
- Goodale, C.L., Davidson, E.A., 2002. Carbon cycle: Uncertain sinks in the shrubs. *Nature* 418, 593-594.
- Goodall, D.W., 1952. Quantitative aspects of plant distribution. *Biological Review* 27,194-245.
- Goovaerts, P., 1999. Geostatistics in soil science: state-of-the-art and perspectives. *Geoderma* 89, 1-45.
- Gordon, H., White, T.C., 1994. Ecological guide to southern California chaparral plant series. USDA Forest Service, Pacific Southwest Region, San Diego, California.
- Gotway, C.A., Hergert, G.W., 1997. Nutrient management and soil and plant analysis. Incorporating spatial trends and anisotropy in geostatistical mapping of soil properties. *Soil Science Society of America Journal* 61, 298-309.

- Grace, J.B., 1999. The factors controlling species density in herbaceous plant communities: an assessment. *Perspectives in Plant Ecology, Evolution and Systematics* 2, 1-28.
- Green, A. A., Berman, M., Switzer, P., Craig, M.D., 1988. A transformation for ordering multispectral data in terms of image quality with implications for noise removal. *IEEE Transactions on Geoscience and Remote Sensing* 26, 65-74.
- Gregorich, E.G., Greer, K.J., Anderson, D.W., Liang, B.C., 1998. Carbon distribution and losses: erosion and deposition effects. *Soil and Tillage Research* 47, 291-302.
- Greig-Smith, P., 1983. *Quantitative plant ecology*, Studies in Ecology, Vol. 9, Blackwell Scientific Publications, Oxford, England.
- Grubb, M., Vrolijk, C., Brack, D., 1999. *The Kyoto Protocol-A guide and assessment*, Earthscan Publications, London.
- Guisan, A., Zimmermann, N.E., 2000. Predictive habitat distribution models in ecology. *Ecological Modeling* 135, 147-186.
- Guyot, G., Baret, F., Jacquemoud, S., 1989. Imaging spectroscopy for vegetation studies. In: Toselli, F., Bodechtel, J. (Eds.), *Imaging Spectroscopy: Fundamentals and Prospective Applications*, Kluwer Academic Publishers, Dordrecht, The Netherlands, pp.145-165.
- Hacker, R.B., 1984. Vegetation dynamics in a grazed mulga shrubland community. I. The mid-storey shrubs. *Australian Journal of Botany* 32, 239-249.
- Harrison, P., 1987. *The Greening of Africa: Breaking through in the Battle for Land and Food*, Paladin, Grafton Books, United Kingdom, 380pp.
- Harsanyi, J.C., Chang, C.I., 1994. Hyperspectral image classification and dimensionality reduction: An orthogonal subspace projection approach. *IEEE Transactions on Geoscience and Remote Sensing* 32, 779-785.
- Hasse, P., Pugnaire, F.L., Clark, S.C., Incoll, L.D., 1996. Spatial patterns in a two-tiered semi-arid shrubland in southeastern Spain. *Journal of Vegetation Science* 7, 527-534.
- Hawkins, D.M., 1995. *Formal inference-based recursive modelling*. Department of Applied Statistics, University of Minnesota, St Paul.
- Herrmann, L., 1996. *Staubdeposition auf Böden West-Afrikas. Eigenschaften und Herkunftsgebiete der Stäube und ihr Einfluß auf Boden und Standortseigenschaften*. Hohenheimer Bodenkundliche Hefte 36, 239 pp.
- Hook, P.B., Burke, I.C., Lauenroth, W.K., 1991. Heterogeneity of soil and plant N and C associated with individual plants and openings in North American shortgrass steppe. *Plant and Soil* 138, 247-256.
- Hosmer, D.W., Lemshow, S., 1989. *Applied Logistic Regression*, John Wiley and Sons, New York.
- Houghton, R.A., Hackler, J.L., Lawrence, K.T., 1999. The US carbon budget: contributions from land-use change. *Science* 285, 574-578.
- Houghton, R.A., Unruh, J.D., Lefebvre, P.A., 1993. Current landuse in the tropics and its potential for sequestering carbon. *Global Biogeochemical Cycles* 7, 305-320.

- Hubbell, S.P., 1979. Tree dispersion, abundance and diversity in a tropical dry forest. *Science* 230, 1299-1309.
- Huete, A.R., Jackson, R.D., 1987. Suitability of spectral indices for evaluating vegetation characteristics on arid rangelands. *Remote Sensing of Environment* 23, 213-232.
- Huete, A.R., Jackson, R.D., Post, D.F., 1985. Spectral response of a plant canopy with different soil backgrounds. *Remote Sensing of Environment* 17, 37-53.
- Hulme, M., Doherty, R., Ngara, T., New, M., Lister, D., 2001. African climate change: 1900-2100. *Climate Research* 17, 145-168.
- Ichku, C., Karnieli, A., 1996. A review of mixture modeling techniques for sub-pixel land cover estimation. *Remote Sensing Reviews* 13, 161-186.
- Insightful., 2001. S-PLUS 6 for Windows User's Guide, Insightful Corporation, Seattle, Washington.
- Iowa, 2004. i Century, [http://www.public.iastate.edu/~elvis/i\\_century\\_main.html](http://www.public.iastate.edu/~elvis/i_century_main.html). Iowa State University.
- IPCC, 1996. Climate Change 1995: The Science of Climate Change. In: Houghton, J.T., Meira Filho, L.G., Callander, B.A., Harris, N., Kattenberg, A., Maskell, K. (Eds.), Contribution of Working Group 1 to the second assessment report of the Intergovernmental Panel on Climate Change, Cambridge University Press, Cambridge.
- IPCC, 2003. Good practice guidance for land use, land-use change and forestry. Subject to final copyedit. [www.ipcc-nggip.iges.or.jp/lulucf/gpglulucf\\_unedit.html](http://www.ipcc-nggip.iges.or.jp/lulucf/gpglulucf_unedit.html). Intergovernmental Panel on Climate Change.
- Isaaks, E.H., Srivastava, R.M., 1990. An Introduction to Applied Geostatistics, Oxford University Press, New York.
- Iverson, L.R., Prasad, A.M., 1998. Predicting abundance of 80 tree species following climate change in the eastern United States. *Ecological Monographs* 68, 465-485.
- Iyamuremye, F., Gewin, V., Dick, R.P., Diack, M., Sene, M., Badiane, A.N., Diatta, M., 2000. Carbon, nitrogen, and phosphorus mineralization of agroforestry plant residues in soils of Senegal. *Journal of Arid Soil Research and Rehabilitation* 14, 359-371.
- Jackson, R.B., Caldwell, M.M., 1993a. Geostatistical patterns of soil heterogeneity around individual perennial plants. *Journal of Ecology* 81, 683-692.
- Jackson, R.B., Caldwell, M.M., 1993b. The scale of nutrient heterogeneity around individual plants and its quantification with geostatistics. *Ecology* 74, 612-614.
- Jensen, J.R., 2000. Remote Sensing of the Environment: An Earth Resource Perspective, Prentice Hall, Upper Saddle River, New Jersey.
- Johnson, D.L., Lewis, L.A., 1995. Land Degradation: Creation and Destruction, Oxford, Blackwell.
- Karjalainen, T., 1996. Dynamics and potentials of carbon sequestration in managed stands and wood products in Finland under changing climatic conditions. *Forest Ecology and Management* 80, 113-132.



- Kaufman, Y.J., 1984. Atmospheric effects on remote sensing of surface reflectance. *SPIE* 475, 20-33.
- Kellman, M., Kading, M., 1992. Facilitation of tree seedling establishment in a sand dune succession. *Journal of Vegetation Science* 3, 679-688.
- Kellner, K., Bosch, O.J., 1992. Influence of patch formation in determining the stocking rate for southern African grasslands. *Journal of Arid Environments* 22, 99-105.
- Kelly, R.H., Burke, I.C., 1997. Heterogeneity of soil organic matter following death of individual plants in shortgrass steppe. *Ecology* 78, 1256-1261.
- Kelly, V., Diagana, B., Reardon, T., Gaye, M., Crawford, E., 1996. Cash Crop and Food grain Productivity in Senegal: Historical View, New Survey Evidence, and Policy Implications. Michigan State University Department of Agricultural Economics, International Development Paper No. 20.
- Ker, A., 1995. Farming systems in the African savanna: A continent in crisis, International Development Research Center (IDRC), Ottawa.
- Kie, J.G., 1985. Production of deerbrush and mountain whitehorn related to shrub volume and overstory crown closure. USDA Forest Service Research Note PSW-377.
- Kirschbaum, M.U.F., Schlamadinger, B., Cannell, M.G.R., Hamburg, S.P., Karjalainen, T., Kurz, W.A., Prisley, S., Schulze, E.D., Singh, T.P., 2001. A generalized approach of accounting for biospheric carbon stock changes under the Kyoto Protocol. *Environmental Science and Policy* 4, 73-85.
- Klausmeier, C.A., 1999. Regular and irregular patterns in semiarid vegetation. *Science* 284, 1826-1828.
- Korzukhin, M.D., Ter-Mikaelian, M.T., Wagner, R.G., 1996. Process versus empirical models: which approach for forest ecosystem management? *Canadian Journal of Forestry Resources* 26, 879-887.
- Kyoto Protocol, 1997. Kyoto Protocol to the United Nations Framework Convention on Climate Change. FCCC/CP/ 1997/7/Add.1, Decision 1/CP.3, Annex, 7., UN, 39 pp.
- Lal, R., 1990. Soil erosion and land degradation: the global risks. *Advances in Soil Science* 11, 129-172.
- Lal, R., 1995. Global soil erosion by water and carbon dynamics. In: Lal, R., Kimble, J., Levine, E., Stewart, B.A. (Eds.), *Soils and global change*. CRC Lewis Publishers, Boca Raton, Florida, pp.131-142.
- Lal, R., 1997. Residue management, conservation tillage and soil restoration for mitigating greenhouse effect by CO<sub>2</sub>- enrichment. *Soil and Tillage Research* 43, 81-107.
- Lal, R., 1999. Potential soil C sequestration in sub-Saharan Africa. Executive Summaries of the workshop "Carbon Sequestration in Soils and Carbon Credits: Review and development of Options for Semi-Arid and Sub-Humid Africa. EROS Data Center, U.S. Geological Survey, Sioux Falls, South Dakota, April 1999(see <http://edcintl.cr.usgs.gov/ip/carbonseq/executive.html>).

- Lal, R., 2000. Soil quality and soil erosion, CRC Press, Boca Raton, Florida, 352 pp.
- Lal, R., 2001. Potential of desertification control to sequester carbon and mitigate the greenhouse effect. *Climatic Change* 51, 35-72.
- Lal, R., 2002. Carbon sequestration in dryland ecosystems of West Asia and North Africa. *Land Degradation and Development* 13, 45-59.
- Lal, R., Hassan, H.M., Dumanski, J., 1999. Desertification control to sequester carbon and mitigate the greenhouse effect. In: Rosenberg, R.C., Izaurralde, E.L. (Eds.), *Carbon sequestration in soils: Science, monitoring and beyond*. Malone Battelle Press, Columbus, Ohio pp. 83-107.
- Lal, R., Kimble, J., Follett, R., 1998. Knowledge gaps and reserable priorities. In: Lal, R., Kimble, J.M., Follett, R.F., Stewart, B.A. (Eds.), *Soil Processes and the Carbon Cycle*, Advances in Soils Science Series, CRC Press, Boca Raton, Florida, pp. 595-604.
- Lal, R., Kimble, J.M., Follett, R.F., Cole, C.V., 1998. The potential of U.S. cropland to sequester carbon and mitigate the greenhouse effect, CRC/Lewis Publishers, Boca Raton, Florida, 128pp.
- Lal, R., Stewart, B.A., 1994. *Land Degradation*, Advances in Soil Science Volume 11, New York, Springer.
- Lam, N., 1983. Spatial Interpolation Methods: A Review. *American Cartographer* 10,129-149.
- Larson, B.A., Frisvold, G.B., 1996. Fertilizers to Support Agricultural Development in Sub-Saharan Africa: What is needed and Why? *Food Policy* 21, 509-525.
- Leprun, C.J., 1967. Les sols de la region de Gourdiry (Senegal Oriental) : Inventaire, relations genetiques et mise en place de materiaux, ORSTOM, Center de Dakar-Hann. 133pp.
- Lericollais, A., 1972. SOB: Etude géographique d'un terroir sérèr (Sénégal). Paris, La Haye: Mouton and Company.
- Lischke, H., Guisan, A., Fischlin, A., Bugmann, H., 1998. Vegetation responses to climate change in the Alps-Modeling studies. In: Cebon, P., Dahinden, U., Davies, H., Imboden, D., Jaeger, C. (Eds.), *A View from the Alps: Regional Perspectives on Climate Change*, MIT Press, Boston, pp. 309-350.
- Liu, S., Kairé, M., Wood, E., Diallo, O., Tieszen, L.L., 2004. Impacts of land use and climate change on carbon dynamics in south-central Senegal. *Journal of Arid Environments* 59, 583-604.
- Ludwig, J.A., Reynolds, J.F., Whitson, P.D., 1975. Size-biomass relationships of several Chihuahuan desert shrubs. *American Midland Naturalist* 94,451-461.
- Ludwig, J.A., Tongway, D.J., 1995. Spatial organization of landscapes and its function in semi-arid woodlands, Australia. *Landscape Ecology* 10, 51-63.
- Lufafa, A., Diédhiou, I., Ndiaye, N.A.S., Kizito, F., Dick, R.P., Noller, J.S., 2005. Determinants of distribution and abundance of two shrub species; *Guiera senegalensis* and *Piliostigma reticulatum* in Senegal's Peanut Basin (In press, *Acta Oecologica*)

- Lyon, L.J., 1970. Length and weight-diameter relations of service berry twigs. *Journal of Wildlife Management* 34,456-460.
- Mabbutt, J.A., Fanning, P.C., 1987. Vegetation banding in arid western Australia. *Journal of Arid Environments* 12, 41-59.
- Manlay, R.J., 2000. Organic matter dynamics in mixed-farming systems of the West African savanna: a village case study from south Senegal. Ph.D. Thesis, ENGREF, Paris,
- Manlay, R.J., Kaire, M., Masse, D., Chotte, J., Ciomei, G., Floret, C., 2002. Carbon, nitrogen and phosphorus allocation in agro-ecosystems of a West African savanna. I. The plant component under semi-permanent cultivation. *Agriculture, Ecosystems and Environment* 88, 215-232.
- Manley, J.T., Schuman, G.E., Reeder, J.D., Hart, R.H., 1995. Rangeland soil C and N responses to grazing. *Journal of Soil and Water Conservation* 50, 294-298.
- Markham, B.L., Barker, J.L., 1988. Radiometric properties of U.S. processed Landsat MSS data. *Remote Sensing of Environment* 22, 39-71.
- Mauchamp, A., Rambal, S., Lepart, J., 1994. Simulating the dynamics of a vegetation mosaic: a spatialized functional model. *Ecological Modeling* 71,107-130.
- McArthur, R.H., 1972. *Geographical Ecology: Patterns in the Distribution of Species*, Harper and Row, New York.
- McAuliffe, J.R., 1994. Landscape evolution, soil formation, and ecological patterns and processes in Sonoran Desert Bajadas. *Ecological Monographs* 64, 111-148.
- McBratney, A.B., Webster, R., 1986. Choosing functions for semivariograms of soil properties and fitting them to sampling estimates. *Soil Science* 37, 617-639.
- McGwire, K., Minor, T., Fenstermaker, L., 2000. Hyperspectral mixture modeling for quantifying sparse vegetation cover in arid environments. *Remote Sensing Environment* 72, 360-374.
- McPherson, G.R., Wright, H.A., Wester, D.B., 1988. Patterns of shrub invasion in semiarid Texas grasslands. *American Midland Naturalist* 120, 391-397.
- Metherell, A.K., Harding, L.A., Cole, C.V., Parton, W.J., 1993. CENTURY soil organic matter model environment. Technical documentation. Agroecosystem Version 4.0. Great Plains System Research Unit Technical Report 4. USDA Agricultural Research Services, Fort Collins, CO.
- Michel, P., 1973. Le Bassins des fleuves Senegal et Gambie: Etude geomorphologique. Tomes I a III, ORSTOM, 365 pp.
- Mitas, L., Mitasova, H., 1999. Spatial Interpolation. In: Longley, P., Goodchild, M.F., Maguire, D.J., Rhind, D.W. (Eds.), *Geographical Information Systems: Principles, Techniques, Management and Applications*, GeoInformation International, Wiley, pp. 481-492.
- Monciardini, C., 1966. La sedimentation eocene au Senegal. *Memoir de Bureau de Recherche en Geologie et Mines* No. 43.

- Montagnini, F., Nair P.K.R., 2004. Carbon sequestration: An underexploited environmental benefit of agroforestry systems: New vistas in Agroforestry. *Agroforestry Systems* 61, 281-295.
- Montana, C., 1992. The colonization of bare areas in two phase mosaics of an arid ecosystem. *Journal of Ecology* 80,315-327.
- Muneto, H., Nobuhito, O., Nanae, K., Guo-sheng, Z., Lin-he, W., Ken, Y., 2001. Plant species effect on the spatial patterns of soil properties in the Mu-us desert ecosystem, Inner Mongolia, China. *Plant and Soil* 234,195-205.
- Murage, E.W., Karanja, N.K., Smithson, P.C., Woomer, P.L., 2000. Differences in soil properties between productive and non-productive fields identified by smallhold farmers in the Central Highlands of Kenya. *Agriculture Ecosystems and Environment* 79, 1-8.
- Murry, R.B., Jacobson, M.Q., 1982. An evaluation of dimension analysis for predicting shrub biomass. *Journal of Range Management* 35,451-454.
- Neter, J., Kutner, M.H., Nachtsheim, C.J., Wasserman, W., 1996. *Applied Linear Regression Models*, Third edition, Richard, D. Irwin, Inc., Chicago.
- Newbery, D.M., Alexander, I.J., Thomas, D.W., Gartian, J.S., 1988. Ectomycorrhizal rain-forest legumes and soil phosphorus in Korup National Park, Cameroon. *New Phytologist* 109, 433-450.
- Newbould, P.J., 1967. *Methods for estimating the primary production of forests*. IBP Handbook No. 2, Blackwell Scientific Publications, Oxford, England.
- Neyman, J., Scott, E.L., 1957. On a mathematical theory of populations conceived as conglomerations of clusters. In: *Cold Spring Harbor symposia on Quantitative Biology*. Vol. XXII. 109-120.
- Nolin, M.C., Guertin, S.P., Wang, C., 1996. Within-field spatial variability of soil nutrients and corn yield in a Montreal lowlands clay soil. *Proceedings of the third international conference on precision agriculture*. ASA, CSSA and SSSA, Madison, WI, pp. 257-270.
- Nyberg, G., Hogberg, P., 1995. Effects of young agroforestry trees on soils in on-farm situations in western Kenya. *Agroforestry Systems* 32, 45-52.
- Ohmann, L.F., Grigal, D.F., Brander, R.B., 1976. Biomass estimation for five shrubs from northeastern Minnesota. Research Paper NC-133. St. Paul, MN: U.S. Department of Agriculture, Forest Service, North Central Forest Experiment Station, 11pp.
- Okalebo, J.R., Gathua, K.W., Woomer, P.L., 2002. *Laboratory Methods of Soil and Plant Analysis: A Working Manual*, Second edition. SACRED-Africa, Nairobi, 128pp.
- Okin, G.S., Okin, W.J., Murray, B., Roberts, D.A., 2001. Practical limits on hyperspectral vegetation discrimination in arid and semiarid environments. *Remote Sensing of Environment* 77, 212-225.
- Okin, G.S., Okin, W.J., Roberts, D.A., Murray, B., 1998. Multiple Endmember Spectral Mixture Analysis: Application to an arid/semi-arid landscape. *Summaries of the 7<sup>th</sup> JPL Airborne Earth Science Workshop*, JPL Publication 97-21, 1, 291-299.

- Okin, W.J., Okin, G.S., Roberts, D.A., Murray, B., 1999. Multiple Endmember Spectral Mixture Analysis: Endmember choice in an arid shrubland. Summaries of the 8<sup>th</sup> JPL Airborne Earth Science Workshop, JPL Publication 99-17, 1,323-331.
- Oldeman, L.R., 1994. The global extent of soil degradation. In: Greenland, D.J., Szabolcs, I. (Eds.), *Soil Resilience and Sustainable Land Use*, CAB International, Wallingford, UK, pp. 99-117.
- Oliver, M.A., Webster, R., 1986. Combining nested and linear sampling for determining the scale and form of spatial variation of regionalized variables. *Geographical Analysis* 18, 227-242.
- Olsson, L., Ardo, J., 2002. Soil carbon sequestration in degraded semiarid agro-ecosystems - perils and potentials. *Ambio* 31, 471-477.
- Olsvig-Whittaker, L., Shachak, M., Yair, A., 1983. Vegetation patterns related to environmental factors in a Negev desert watershed. *Vegetatio* 54, 153-165.
- ORSTOM, Section de Pedologie, 1984. Inventaire, de etudes pedologiques effectuees a l'ORSTOM au Senegal de 1946 a 1984. ORSTOM, Paris. 41 pp.
- Pacala, S.W., Hurtt, G.C., Baker, D., Peylin, P., Houghton, R.A., Birdsey, R.A., Heath, L., Sundquist, E.T., Stallard, R.F., Ciais, P., Moorcroft, P., Caspersen, J.P., Shevliakova, E., Moore, B., Kohlmaier, G., Holland, E., Gloor, M., Harmon, M.E., Fan, S.M., Sarmiento, J.L., Goodale, C.L., Schimel, D., Field, C.B., 2001. Consistent land- and atmosphere-based U.S. carbon sink estimates. *Science* 292, 2316-2320.
- Palm, C.A., 1995. Contribution of agroforestry trees to nutrient requirements of intercropped plants. *Agroforestry Systems* 30, 105-124.
- Palmer, M.W., 1990. Spatial scale and patterns of species-environment relationships in hardwood forest of the North Carolina Piedmont. *Coenoses* 5, 79-87.
- Pandey, D.N., 2002. Carbon sequestration in agroforestry systems. *Climate Policy* 2, 367-377.
- Parker, K.C., 1991. Topography, substrate, and vegetation patterns in the northern Sonoran Desert. *Journal of Biology* 18, 151-163.
- Parton, W.J., Cole, C.V., Stewart, J.S.B., Ojima, D.S., Schimel, D.S., 1989. Simulating regional patterns of soil C, N, and P dynamics in the U.S. Central grassland region. In: Clarholm, M., Bergstrom, L. (Eds.), *Ecology of Arable Land*, Kluwer Academic Publishers, pp.99-108.
- Parton, W.J., Schimel, D.S., Cole, C.V., Ojima, D.S., 1987. Analysis of factors controlling soil organic matter levels in Great Plains grasslands. *Soil Science Society of America Journal* 51, 1137-1179.
- Parton, W.J., Scurlock, J.M.O., Ojima, D.S., Gilmanov, T.G., Scholes, R.J., Schimel, D.S., Kirchner, T., Menaut, J.C., Seastedt, T., Garcia Moya, E., Kamnalrut, A., Kinyamario, J.I., 1993. Observations and modeling of biomass and soil organic matter dynamics for the grassland biome worldwide. *Global Biogeochemical Cycles* 7,785-809.

- Paustian, K., Andr  n, O., Janzen, H.H., Lal, R., Smith, P., Tian, G., Tiessen, H., van Noordwijk, M., Woomer, P.L., 1997. Agricultural soils as a sink to mitigate CO<sub>2</sub> emissions. *Soil Use and Management* 13, 1-15.
- Paustian, K., Levine, E., Post, W.M., Ryzhova, I.M., 1997. The use of models to integrate information and understanding of soil C at the regional scale. *Geoderma* 79, 227-260.
- Paustian, K., Six, J., Elliott, E.T., Hunt, H.W., 2000. Management options for reducing CO<sub>2</sub> emissions from agricultural soils. *Biogeochemistry* 48, 147-163.
- Pearsall, W.H., 1927. Growth studies. VI. On the relative sizes of growing plant organs. *Annals of Botany* 41, 549-556.
- Peek, J.M., 1970. Relation of canopy area and volume to production of three woody species. *Ecology* 51, 1098-1101.
- Pelissier, P., 1953. Les paysans S  r  res. Essai sur la formation d'un terroir au S  n  gal. *Les Cahiers d'Outre-Mer*, 22-6i  me Ann  e, Avril-Mai: 8, 106-127.
- Phillips, P.L., Barnes, P.B., 2003. Spatial asymmetry in tree-shrub clusters in a Subtropical savanna. *American Midland Naturalist* 149, 59-70.
- Pieri, C., 1969. Etude pedologique de la region de Nioro du Rip. Volume I. Center National de Recherches Agronomique de Bambey, Senegal, 134 pp.
- Pontie, G., Lericollais, A., 1995. Relations    distance des migrants sereer. In: Philippe, A., Bara Diop, A. (Eds.), *La ville    guichets ferm  s?* Dakar, IFAN/Ch.A.D. et Paris:ORSTOM.
- Poss, R., 1991. Transferts de l'eau et des   l  ments chimiques dans les terres de barre du Togo. Th  se Univ. Paris VI. Editions ORSTOM. 355 pp.
- Prentice, I.C., Cramer, W., Harrison, S.P., Leemans, R., Monseruds, R.A., Solomon, A.M., 1992. A global biome model based on plant physiology and dominance, soil properties and climate. *Journal of Biogeography* 19, 117-134.
- Price, J.C., 1987. Special Issue on radiometric calibration of satellite data. *Remote Sensing of Environment* 22, 1-158.
- Puyou-Lascassies, P., Flouzat, G., Gay, M., Vignolles, C., 1994. Validation of the use of multiple linear regressions as a tool for unmixing coarse spatial resolution images. *Remote Sensing of Environment* 49, 155-166.
- Quarmby, N.A., Townshend, J.R., Settle, J.J., White, K.H., Milnes, M., Hindle, T.L., Silleos, N., 1992. Linear mixture modeling applied to AVHRR data for crop area estimation. *International Journal of Remote Sensing* 13, 415-425.
- Ramsey, F.I., Schafer, D.W., 2002. *The statistical sleuth: A course in methods of data analysis*, Duxbury Thomson Learning, Pacific Grove, California.
- Reardon, T., Barrett, C., Kelly, V., Savadogo, K., 1999. Policy Reforms and Sustainable Agricultural Intensification in Africa. *Development Policy Review* 17, 375-395.
- Reed, R.A., Peet, R.K., Palmer, L.W., White, P.S., 1993. Scale-dependence on vegetation environment correlation: a case study of a North Carolina piedmont woodland. *Journal of Vegetation Science* 4, 329-340.

- Reich, R.M., Davis, R., 2000. Quantitative Spatial Analysis. Lecture notes. Colorado State University, Fort Collins, Colorado.
- Reichard, S.H., Hamilton, C.W., 1997. Predicting invasions of woody plants introduced to North America. *Conservation Biology* 11, 193-203.
- Reinier, H., Rietkerk, M., Van Den Bosch, F., Herbert H.T., Prins, H.H.T., De Kroons, H., 2001. Vegetation pattern formation in semi-arid grazing systems. *Ecology* 82, 50-61.
- Renaud, L., 1961. Le precambrian du sud-ouest de la Mauritanie et du Senegal Oriental. Memoir, de Bureau de Recherche en Geologie et Mines No. 5. 134 pp.
- Rhoades, C.C., 1997. Single-tree influences on soil properties in agroforestry: lessons from natural and savanna ecosystems. *Agroforestry Systems* 35, 71-94.
- Richter, R., 1997. Correction of atmospheric and topographic effects for high spatial resolution satellite imagery. *International Journal of Remote Sensing* 18, 1099-1111.
- Rietkerk, M., Ketner, P., Burger, J., Hoorens, B., Olf, H., 2000. Multiscale soil and vegetation patchiness along a gradient of herbivore impact in a semi-arid grazing system in West Africa. *Plant Ecology* 148, 207-224.
- Rietkerk, M., Van de Koppel, J., 1997. Alternate stable states and threshold effects in semi-arid grazing systems. *Oikos* 79, 69-76.
- Ringius, L., 2002. Soil carbon sequestration and the CDM: opportunities and challenges for Africa. *Climate Change* 54, 471-495.
- Ripley, B.D., 1977. Modeling spatial patterns. *Journal of the Royal Statistical Society, Series B* 39 172-212.
- Ripley, B.D., 1979. Tests of "randomness" for spatial pattern. *Journal of the Royal Statistical Society, Series B*, 41,368-374.
- Ripley, B.D., 1981. *Spatial statistics*, Wiley, New York.
- Rittenhouse, L.R., Sneva, F.A., 1977. A technique for estimating big sagebrush production. *Journal of Range Management* 27, 97-100.
- Robert, M., 1996. Aluminum toxicity a major stress for microbes in the environment. In: Huang, P.M. (Ed.), *Environmental Impacts*. Vol. 2, Soil component interactions, CRC press, pp. 227-242.
- Robert, M., Chenu, C., 1991. Interactions between soil minerals and microorganisms. In: Bollag, J.M., Stotzky, G. (Eds.), *Soil Biochemistry* 7, Marcel Dekker, New York, pp.307-393
- Roberts, D.A., Gardner, M., Church, R., Ustin, S., Scheer, G., Green, R.O., 1998. Mapping chaparral in the Santa Monica Mountains using multiple endmember spectral mixture models. *Remote Sensing of Environment* 65, 267-279.
- Robertson, G.P., 1987. Geostatistics in ecology: interpolating with known variance. *Ecology* 68, 744-748.
- Robertson, G.P., Huston, M.A., Evans, F.C., Tiedje, J.M., 1988. Spatial variability in a successional plant community: patterns of nitrogen availability. *Ecology* 69, 1517-1524.

- Rosin, P.L., 2001. Robust pixel unmixing. *IEEE Transactions on Geoscience and Remote Sensing* 39, 1978-1983.
- Rouget, M., Richardson, D.M., Milton, S.J., Polakow, D., 2001. Predicting invasion dynamics of four alien *Pinus* species in a highly fragmented semi-arid shrubland in South Africa. *Plant Ecology* 152, 79-92.
- Ruthenberg, H., 1971. *Farming systems in the Tropics*, Clarendon Press, Oxford.
- Rutherford, M.C., 1979. Plant-based techniques for determining available browse and browse utilization: a review. *Botany Review* 45, 203-228.
- Rykiel, E.J., Cook, T.L., 1986. Hardwood-red cedar clusters in the post oak savanna of Texas. *Southwest Naturalist* 31, 73-78.
- Sagna-Cabral, M.A., 1989. Utilisation et gestion de la matiere organique d'origine animale dans un terroir du centre nord du Senegal. *Memoire d'Etude, Centre National d'Etudes Agronomiques des Regions Chaudes (CNEARC), Montpellier, France.*
- Saldana, A., Stein, A., Zinck, J.A., 1998. Spatial variability of soil properties at different scales within three terraces of the Henare River (Spain). *Catena* 33, 139-153.
- Salisbury, E.J., 1926. The geographical distribution of plants in relation to climatic factors. *Geography Journal* 57, 312-335.
- Sanchez, P.A., 1995. Science in agroforestry. *Agroforestry Systems* 30, 5-55.
- Sanchez, P.A., Buresh, R.J., Leakey R.R.B., 1999. Trees, soils and food security. *Philosophical Transactions of the Royal Society of London, Series B* 353, 949-961.
- Sanchez, P.A., Salinas, I.G., 1982. Low-input technology for managing oxisols and ultisols in tropical America. *Advances in Agronomy* 34, 279-406.
- SAS, 1999. *SAS user's guide: statistics*, 1999. SAS Institute Inc., Cary, N.C. USA.
- Schenk, H.J., Holzapfel, C., Hamilton, J.G., Mahall, B.E., 2003. Spatial ecology of a small desert shrub on adjacent geological substrates. *Journal of Ecology* 91, 383-395.
- Schindler, D.W., 1999. The mysterious missing sink. *Nature* 398, 105-107.
- Schlesinger, W.H., Abrahams, A.D., Parsons, A.J., Wainwright, J., 1999. Nutrient losses in run-off from grassland and shrubland habitats in Southern New Mexico: I. rainfall simulation experiments. *Biogeochemistry* 45, 21-34.
- Schlesinger, W.H., Raikes, J.A., Hartley, A.E., Cross, A.F., 1990. On the spatial pattern of soil nutrients in desert ecosystems. *Ecology* 77, 364-374.
- Schlesinger, W.H., Reynolds, J.H., Cunningham, G.L., Huenneke, L.F., Jarell, W.M., Virginia, R.A., Whitford, W.G., 1996. Biological feedbacks in global desertification. *Science* 271, 1043-1048.
- Schmitt, M.D.C., Grigal, D.F., 1981. Generalized biomass estimation equations for *Betula papyrifera* Marsh. *Canadian Journal of Forest Research* 11, 837-840.
- Schroeder, P., 1993. Agroforestry systems: integrated land use to store and conserve carbon. *Climate Research* 3, 53-60.



- Schroeder, P., 1994. Carbon storage benefits of agroforestry systems. *Agroforestry Systems* 27, 89-97.
- Shimabukuru, Y.E., Smith, J.A., 1991. The least squares mixing models to generate fraction images derived from remote sensing multispectral data. *IEEE Transactions on Geoscience and Remote Sensing* 29, 16-20.
- Skarpe, C., 1991. Spatial patterns and dynamics of woody vegetation in an arid savanna. *Journal of Vegetation Science* 2, 565-572.
- Skog, K.E., Nicholson, G.E., 1998. Carbon cycling through wood products: the role of wood and paper products in carbon sequestration. *Forest Products Journal* 48, 75-83.
- Small, C., 2001. Estimation of urban vegetation abundance by Spectral Mixture Analysis. *International Journal of Remote Sensing* 22, 1305-1334.
- Smith M.O., Ustin S.L., Adams J.B., Gillespie A.R., 1990. Vegetation in deserts: I. a regional measure of abundance from multispectral images. *Remote Sensing of Environment* 31, 1-26.
- Smith, M.O., Adams J.B., Ustin S.L., Roberts, D.A., 1992. Using endmembers in AVIRIS images to estimate changes in vegetative biomass. *Summaries of the 4<sup>th</sup> Annual JPL Airborne Geoscience Workshop*, JPL Publication 92-14, 1, 69-71.
- Smith, W.B., Brand, G.J., 1983. Allometric biomass equations for 98 species of herbs, shrubs, and small trees. *Research Paper*. Forest Service, North Central Forest Experiment Station, 8pp.
- Soil Survey Staff., 2003. *Soil Taxonomy*, Third edition, USDA/NRCS Agriculture Handbook 436.
- Stancioff, A.M., Staljanssens, M., Tappan, G., 1986. Mapping and remote sensing of the resources of the Republic of Senegal: a study of the geology, hydrology, soils, vegetation and land use potential. South Dakota State University, Remote Sensing Institute, SDSU-RSI-86 -01, South Dakota, USA.
- Sterk, G., Herrmann, L., Bationo, A., 1996. Wind-blown nutrient transport and soil productivity changes in Southwest Niger. *Land Degradation and Development* 7, 325-335.
- Stevenson, F.J., 1994. *Humus Chemistry: Genesis, Composition, Reactions*, John Wiley and Sons, New York, 496 pp.
- Stocking, M.A., Murnaghan, N., 2001. *Field assessment of land degradation*, Earthscan Publications, London.
- Stott, P., 1981. *Historical Plant Geography, An Introduction*, George Allen and Unwin, London.
- ter Steege, H., Jetter, V.G., Polak, A.M., Werger, M.J.A., 1993. Tropical rainforest types and soil factors in a watershed area in Guyana. *Journal of Vegetation Science* 4, 705-716.
- Thiery, J.M., D'Herbes, J.M., Valentin, C., 1995. A model simulating the genesis of banded vegetation patterns in Niger. *Journal of Ecology* 83, 497-507.
- Tisdall, J.M., Oades, J.M., 1982. Organic matter and water stable aggregates. *Journal of Soil Science* 33, 141-163.

- Tongway, D.J., Ludwig, J.A., 1990. Vegetation and soil patterning in semi-arid mulga lands of eastern Australia. *Australian Journal of Ecology* 15, 23-34.
- Tottrup, C., Rasmussen, M.S., 2004. Mapping long-term changes in savanna crop productivity in Senegal through trend analysis of time-series of remote sensing data. *Agriculture, Ecosystems and Environment* 103, 545-560.
- Troughton, A., 1960. Further studies on the relationship between shoot and root systems of grasses. *Journal of the British Grassland Society* 15, 41-47.
- Tschakert, P., 2001. Human dimensions of carbon sequestration: A political ecology approach to soil fertility management and desertification control in the Old Peanut Basin of Senegal. *Arid Lands Newsletter* 49, May-June.
- Tschakert, P., Khouma, M., Sene, M., 2004. Biophysical potential for soil carbon sequestration in agricultural systems of the Old Peanut Basin of Senegal. *Journal of Arid Environments* 59, 511-533.
- Underwood, E., Ustin, S., DiPietro, D., 2003. Mapping nonnative plants using hyperspectral imagery. *Remote Sensing of Environment* 86, 150-161.
- UNECA, 2001. United Nations Economic Commission for Africa. State of the Environment in Africa. ECA/FSSDD/01/06, Addis Ababa, Ethiopia.
- UNECA, 2002. United Nations Economic Commission for Africa. 2002. Harnessing Technologies for Sustainable Development. ECA Policy Research Report, Addis Ababa, Ethiopia.
- UNFCCC, 2004. UN Framework Convention on Climate Change, <http://unfccc.int/index.html>. UNFCCC.
- Urbanska, K.M., 1997. Reproductive behaviour of arctic/alpine plants and ecological restoration. Disturbance and Recovery in Arctic Lands: In: Crawford, M.M.M. (Ed.), *An Ecological Perspective*, Kluwer Academic, Boston, pp. 481-502.
- Valentin, C., d'Herbès, J.M., Poesen, J., 1999. Soil and water components of banded vegetation pattern. *Catena* 37, 1-24.
- Van Auken, W., 2000. Shrub invasions of North American semiarid grasslands. *Annual Review of Ecology and Systematics* 31, 197-215.
- van der Meer, F., 1995. Spectral unmixing of landsat thematic mapper data. *International Journal of Remote Sensing* 16, 3189-3194.
- van Noordwijk, M., Hairiah, K., Sitompul, S.M., 2000. Reducing uncertainties in the assessment at national scale of C stock impacts of land use change. Proceedings of the IGES/NIES workshop on GHG Inventories for Asia-Pacific Region Institute for Global Environmental Strategies (IGES, Hayama, Japan).
- Verdu, M., Garcia-Fayos, P., 1996. Nucleation processes in a Mediterranean bird-dispersed plant. *Functional Ecology* 10, 275-280.
- Verhoef, A., Allen, S., De Bruin, H.A.R., Jacobs, C.M.J., Heusinkveld, B.G., 1996. Fluxes of carbon dioxide and water vapour from a Sahelian savanna. *Agricultural and Forest Meteorology* 80, 231-248.

- Vinton, M.A., Burke, I.C., 1995. Interactions between individual plant species and soil nutrient status in shortgrass steppe. *Ecology* 76, 1116-1133.
- Wackernagel, H., 1994. Cokriging versus kriging in regionalized multivariate data analysis. *Geoderma* 62, 83-92.
- Walter, H., 1985. *Vegetation of the Earth and Ecological Systems of Geobiosphere*, Heidelberg.
- Webb, L.J., 1959. A physiognomic classification of Australian rain forests. *Journal of Ecology* 47, 551-570.
- Webb, L.J., 1968. Environmental relationships of the structural types of Australian rain forest vegetation. *Ecology* 49, 296-311.
- Weber, A.W., Major, J.T., 1984. Report and recommendation on soil conservation and regeneration in the Groundnut Basin of Senegal (draft). International Institute for Environment and Development, 30pp.
- Wezel, A., 2000. Scattered shrubs in pearl millet fields in semiarid Niger: Effect on millet production. *Agroforestry Systems* 48, 219-228.
- White, F., 1983. Chapter XVI, The Sahel regional transition zone. In: *The Vegetation of Africa*. UNESCO, Paris, pp.203-215.
- Wijesinghe, D., Handel, S.N., 1994. Advantages of clonal growth in heterogeneous habitats: an experiment with *Potentilla simplex*. *Journal of Ecology* 82, 495-502.
- Wilson, J.B., 1988. A review of the evidence on the control of shoot:root ratio in relation to models. *Annals of Botany* 61, 433-449.
- Winterbottom, R., Hazlwood, P.T., 1987. Agroforestry and sustainable development: making the connection. *Ambio* 16, 100-110.
- Wondzell, S.M., Cunningham, G.L., Cornelius, J.M., 1990. Fine grained environmental patterns and the vegetation of a Chihuahuan desert playa. *Journal of Vegetation Science* 1, 403-410.
- Wood, J., 1996. *The Geomorphological Characterization of Digital Elevation Models*. Ph.D. Thesis, University of Leicester, Department of Geography, Leicester, UK.
- Woodward, F.I., 1987. *Climate and plant distribution*, Cambridge University Press, Cambridge.
- Woomer, P.L., 1993. The impact of cultivation on carbon fluxes in woody savannas of Southern Africa. *Water Air and Soil Pollution* 70, 403-412.
- Woomer, P.L., Karanja, N.K., Murage, E.W., 2001. Estimating total system C in smallhold farming of the east African Highlands. In: Lal, R., Kimble, J.M., Follett, R.F., Stewart B.A. (Eds.), *Assessment Methods for Soil Carbon*. Advances in Soil Science (Series), Lewis Publishers, Boca Raton, USA, pp.147-166.
- Woomer, P.L., Martin, A., Albrecht, A., Resck, D.V.S., Scharpenseel, H.W., 1994. The importance and management of soil organic matter in the tropics. In: Woomer, P.L., Swift, M.J. (Eds.), *The Biological Management of Tropical Soil Fertility*, Wiley, Chichester, UK, pp. 47-80.

- Woomer, P.L., Palm, C.A., Qureshi, J.N., Kotto-Same, J., 1998. Carbon sequestration and organic resource management in African smallholder agriculture. In: Lal, R., Kimble, J.M., Follet, R.F., Stewart, B.A. (Eds.), Management of carbon sequestration in soils, Advances in Soil Science, CRC Press, Florida, pp. 58-78.
- Woomer, P.L., Tieszen, L.L., Tappan, G., Touré A., Sall, M., 2004a. Land use change and terrestrial carbon stocks in Senegal. *Journal of Arid Environments* 59, 625-642.
- Woomer, P.L., Tieszen, L.T., Tschakert, P., Parton, W.J., Touré, A., 2001. Landscape Carbon Sampling and Biogeochemical Modeling: A two-week skills development workshop conducted in Senegal. SACRED Africa, Nairobi, Kenya.
- Woomer, P.L., Touré, A., Sall, M., 2004b. Carbon stocks in Senegal's Sahel Transition. *Journal of Arid Environments* 59, 499-510.
- World Bank, 1996. Toward Environmentally Sustainable Development in Sub-Saharan Africa. A World Bank Agenda, Washington, D.C., USA, pp.40.
- Yeaton, R.I., 1978. A cyclical relationship between *Larrea tridentata* and *Opuntia leptocaulis* in the northern Chihuahuan Desert. *Journal of Ecology* 66, 651-656.
- Zewge, T., 2003. Contributions to e-discussion on "Rehabilitation of Degraded Lands in Sub-Saharan Africa and TACCDA." UCNW, Bangor, UK.
- Zhang, R., Myers, D.E., Warrick, A.W., 1992. Estimation of the spatial distribution of soil chemicals using pseudo-cross-variograms. *Soil Science Society of America Journal* 56, 1444-1452.

## APPENDICES

## Appendix 1

### Analysis of the spatial patterning of two native shrubs in Senegal's Peanut Basin.

(This appendix is a detailed presentation of shrub pattern indices used in Chapter 2)

#### Introduction

The spatial structure of a plant community observed at any point in time is the product of many processes in its past (Schenk et al., 2003). These include biotic processes, such as plant dispersal, growth, mortality and herbivory, as well as other factors and processes, such as substrate, topography, climate, fire, disturbance, or land-use history. More often than not, spatial patterns in the structure of a plant community will be the product of complex and interacting processes. The processes that are most important for shaping community structure are likely to leave observable imprints in spatial community structure. It should be possible therefore to use spatial pattern analyses in order to generate testable hypotheses about the processes that shape community structure. In this study, we studied the spatial patterning of two native shrubs (*Guiera senegalensis* and *Piliostigma reticulatum*) in Senegal's Peanut Basin in order to determine the influence of anthropogenic factors on shrub abundance.

#### Materials and methods

Six villages (3 with *Guiera senegalensis* and 3 with *Piliostigma reticulatum*) lying along the North-South rainfall gradient were selected from a set of twelve villages using random numbers. The climate in the selected villages (from now on referred to as sites) is arid, with on average more than 85% of precipitation falling between August and October. Mean annual precipitation is approximately 540 mm, skewed towards the *Piliostigma reticulatum* sites and with high variability from year to year. Once in a selected site, a table of random degrees (45° to 360° in 45° increments) was used to identify a direction in which a transect cutting across the three village production rings was located starting 50 meters away from the homesteads. A 100 m x 100 m quadrat was located well inside each of the three rings (to avoid edge effects) and in cases where the quadrat would not fit, it was oriented perpendicular to the transect. The position of individual shrubs in the quadrat was mapped along an X-Y plane and where shrubs were very close that suckering was suspected, they were considered as one individual.

Second order neighborhood analysis (Ripley, 1977; Ripley, 1981; Diggle, 1983) was used to infer shrub spatial pattern using Ripley's k-function (a linear expression of expected number of points within distance  $d$  of all points  $i$ ):

$$L(d) = \left[ \frac{ASK(i, j)}{\pi N(N-1)} \right]^{0.5}$$

and  $S$  is:

$$S = \sum_{i=1}^N \sum_{j=1, j \neq i}^N$$

where  $A$  is the size of the study area and  $S k(i, j)$  is the number of points  $j$  within distance  $d$  of all  $i$  points.

The function is designed to test randomness hypotheses, often based on the Poisson distribution, by examining the proportion of total possible pairs of points in Euclidean space whose pair members are within a specified distance of each other (Getis and Franklin, 1987). The test of the hypothesis is: If,  $L(d) = d$ , randomness (hence a plot of  $L(d)$  against distance is linear); if  $L(d) > d$ , clustering and if  $L(d) < d$ , regular (Boots and Getis, 1978; Cressie, 1993). Statistical significance is ascertained by comparing observed pattern to the confidence limits of a Monte Carlo simulation. The confidence limits can also be approximated as  $\pm 1.42 A^{0.5}/N$ , where  $A$  = area and  $N$  = number of plants (Ripley, 1977; Ripley, 1979; Ripley, 1981; Diggle, 1983; Getis and Franklin, 1987). In this case, observed values occurring above the upper 95% confidence interval indicate aggregation (i.e. shrubs closer to each other than expected) and values occurring below the lower 95% limit indicate a regular pattern (i.e. shrubs farther from each other than expected).

Both methods of estimating confidence limits yielded the same conclusions about the patterns of spatial dispersion of these shrubs. Furthermore, this graphical analysis provides an estimate of cluster radius. The distance at which the observed k-function deviates the most from the expectation of randomness provides an estimate of the maximum cluster radius (read from the graphs as the corresponding distance,  $d$ ).

First the data for each species were examined to determine if they fitted a random distribution. Then a Neyman-Scott cluster process was used as a model for the spatial distribution of the shrubs (Neyman and Scott, 1957). In a Neyman-Scott process it is assumed that the clusters are randomly distributed, and that the number of individuals per cluster follows a Poisson distribution. The k-function for a Neyman-Scott point process is given by:

$$L(d; \sigma^2, \rho) = \pi d^2 + \rho^{-1} \{1 - \exp(-d^2 / 4\sigma^2)\}$$

where  $\rho$  is the intensity of the parent process which is assumed to be a homogeneous Poisson process, and  $2\sigma^2$  is the mean squared distance from the cluster center to a given shrub. Values of cluster radius,  $d$  (i.e. distance over which the model is fitted) were used to compute number of clusters in the population and the average number of shrubs per cluster.

## Results

Figure A.1.1 is an excerpt of representative mapped distributions of *Guiera senegalensis* (a) and *Piliostigma reticulatum* (b) over 100 x 100 m quadrats in bush rings at Keur Matar Aram (for *Guiera senegalensis*) and Nioro (for *Piliostigma reticulatum*). Figure A.1.2 shows the observed and expected  $L(d)$  values;  $L(d)$  represents the average distance relationships for the total number of shrubs in the 100 x 100 m quadrats shown in Figure A.1.1.

The curves (Figure A.1.2 a and b) for both shrubs (excepting at 1 m for *Guiera senegalensis* and at 1.2 m for *Piliostigma reticulatum*) extend above the simulation envelopes for a Poisson process suggesting that both shrub species are aggregated.



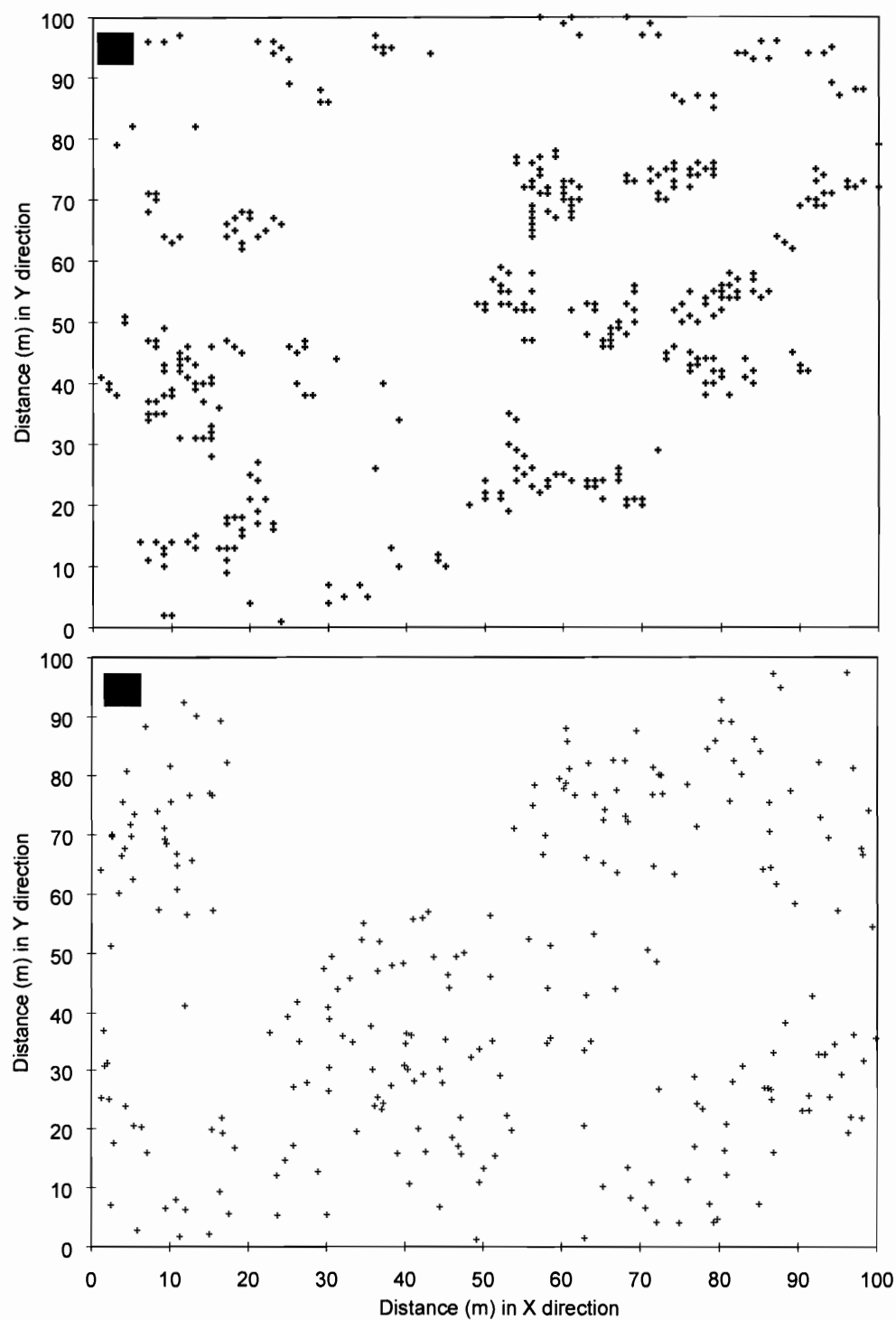


Figure A.1.1. Point pattern representation of the shrubs (a) *G. senegalensis*, (b) *P. reticulatum*.

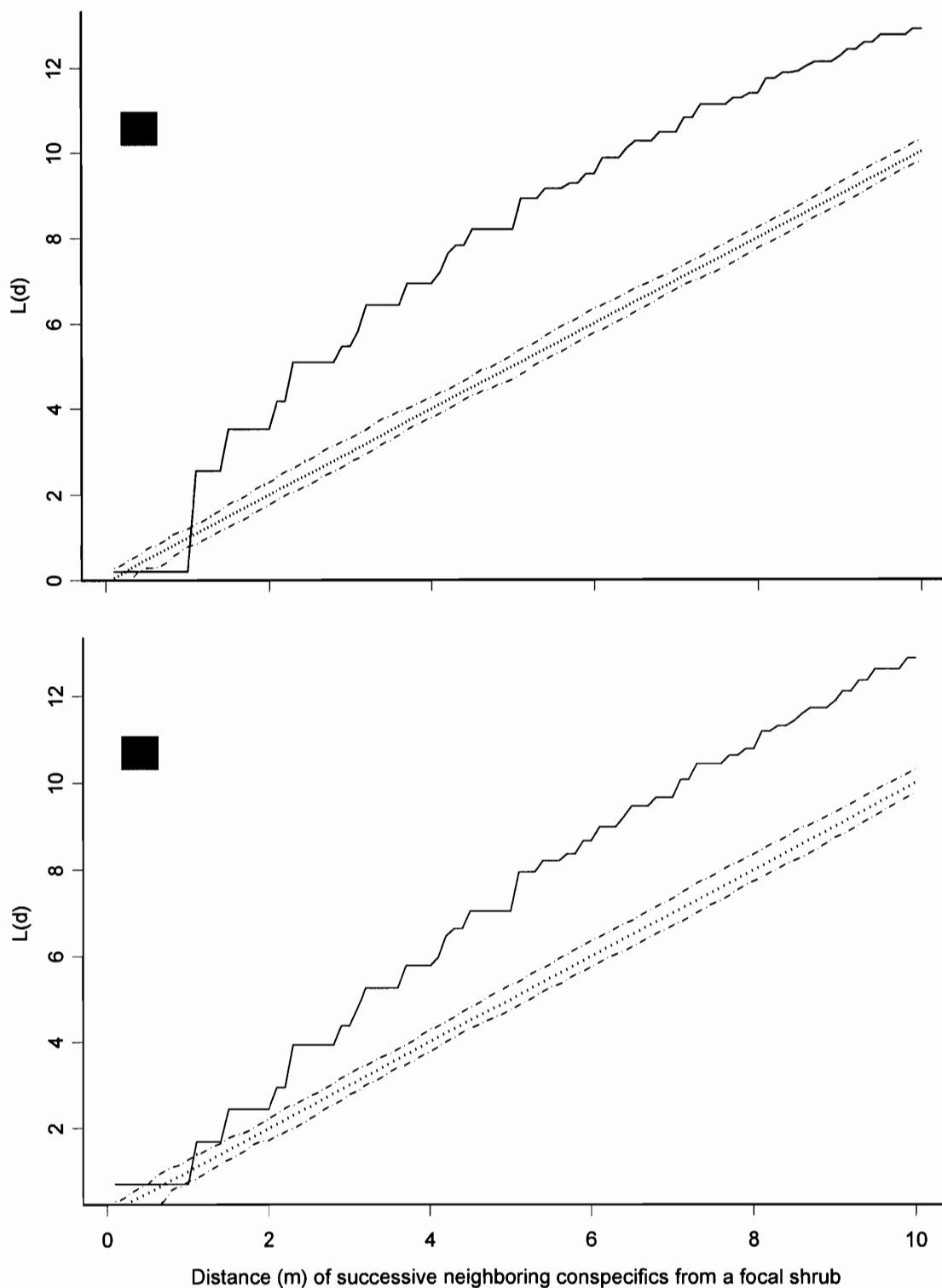


Figure A.1.2. Shrub spatial patterns (a) *G. senegalensis* and (b) *P. reticulatum*. Ripley's  $L(d)$  plotted against  $d$  (distance of successive neighboring shrub from "focal shrub"). (.....) is the approximate 95% confidence limits (simulation envelopes; see text) about the expected value (.....) for complete spatial randomness, which = 0.

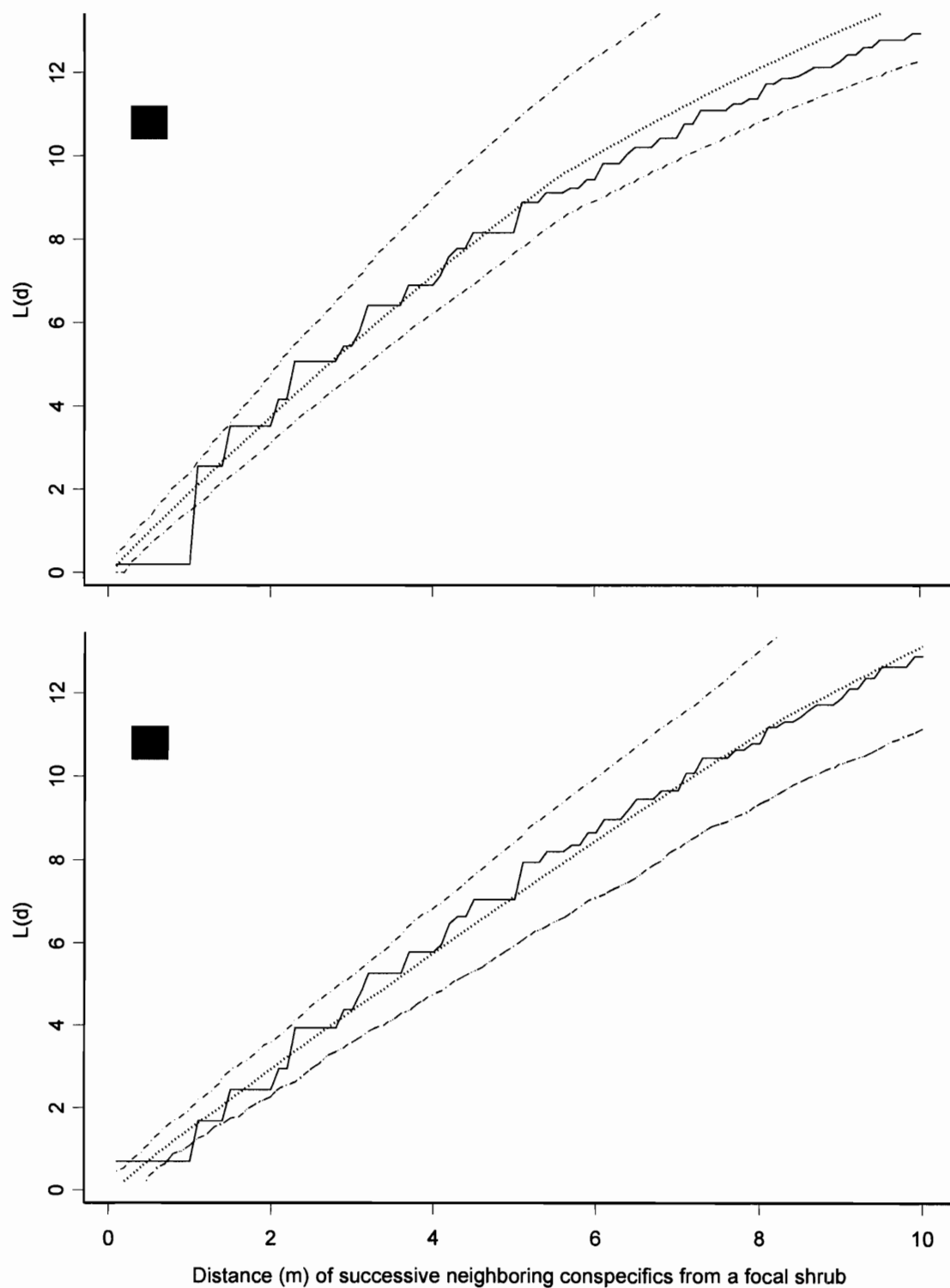


Figure A.1.3. Ripley's  $L(d)$  for the shrubs (a) *G. senegalensis* and (b) *P. reticulatum* with simulation envelopes (---) based on 100 replications of a Neyman-Scott cluster process (—) is observed pattern and (.....) is the expected pattern.

These results are consistent with those of Yeaton (1978), Fuentes et al. (1984), Hacker (1984), Rykiel and Cook (1986), Archer et al. (1988), McPherson et al. (1988), Tongway and Ludwig (1990), Kellman and Kading (1992), Kellner and Bosch, (1992), Hasse et al. (1996), Verdu' and Garcya-Fayos (1996), Callaway and Davis (1998), Valentin et al. (1999), Rietkerk et al. (2000) and Phillips and Barnes (2003) in showing that clustering is typical of arid plant communities. These spatial patterns are the product of a suite of complex and interacting processes (Schenk et al., 2003). Although data are not available to allow examination of the processes generating this clustering, a number of possible putative causes (including drainage relations, some variation in soil factors, and reproductive strategies) is probable.

A common hypothesis for the cause of clustering is a positive feedback between plant density and water infiltration (Olsvig-Whittaker, et al., 1983; Belsky, 1986; Bowers and Lowe, 1986; Mabbutt, and Fanning, 1987; Wondzell, et al., 1990; Parker, 1991; Montana, 1992; Abrahams et al., 1994; Mauchamp et al., 1994; McAuliffe, 1994; Ludwig and Tongway, 1995; Thiery et al., 1995; Bromley et al., 1997; Rietkerk and Van de Koppel, 1997; Aguiar and Sala, 1999; Schlesinger et al., 1999; Rietkerk et al., 2000; Reinier et al., 2001). The nature of this positive feedback is that at higher plant densities more water infiltrates into the soil than at lower plant densities. In vegetation mosaics, rain falling on bare patches of soil will barely infiltrate and run off. This runoff water subsequently accumulates in the vegetated patches, where it can infiltrate more easily and therefore enhance vegetation growth. Boaler and Hodge (1962) attribute clustering to the variation in texture of soil parent material. Belsky (1986) suggested that the initiation of the vegetation clusters was caused by soil sodicity and salinity through differential leaching of salts from the sodic soils.

The aggregated spatial pattern of the species could be also related to their reproductive strategy and the capability for vegetative spread, which is stimulated by the burial of the stems by sand particles from wind erosion such as, was observed in *Guiera senegalensis* at Keur Matar Aram and Tabakali. This growth of stems found around the site of the "mother plant" has often been held responsible for over dispersion (Goodall, 1952; Hubbell, 1979; Greig-Smith, 1983; Skarpe, 1991; Wijesinghe and Handel, 1994; Urbanska, 1997; Grace, 1999; Klausmeier, 1999). Seed dispersion seems to play a minor role in determining clustering, because our observations show that these species are widely dispersed by animals and wind.

Table A.1.2 is a summary of cluster parameters for the entire set of quadrats studied. We expected the overall distribution of cluster radius and distance to nearest neighbor to be biased towards smaller clusters and larger distances to nearest neighbors in areas proximal to the village center. This difference was expected on the basis of intensified continuous cultivation in the compound rings (Manlay et al., 2002) as opposed to longer periods of fallowing as one moves from the bush ring to the savanna ring (Manlay, 2000). However, there was no consistent pattern that differentiated between the cluster radius and distances to nearest neighbor distributions of shrubs in the different village production rings. Even though the cluster density estimates for the shrubs differed between village production rings, patch radii within each species was not significantly different ( $p$ -value from one way ANOVA = 0.335 and 0.817 for *Guiera senegalensis* and *Piliostigma reticulatum* respectively). Thus the estimates of patch parameters appear to reflect the actual spatial pattern of these shrub populations in contrast to a different view that they can be a consequence of management intensity. In fact, it seems very likely that the manner of management in these Parkland systems is such that it preserves the shrubs in the landscapes (the annual pruning notwithstanding) due to the importance the farmers attach to them in replenishing the fertility of soils (A. Sene pers. comm., 2004). This notion is buoyed by the lack of a significant difference ( $p$ -value = 0.324 and 0.065 respectively for *Guiera senegalensis* and *Piliostigma reticulatum* from a one way ANOVA) in number of shrubs in the different village production rings.

Table A.1.2. Comparison of cluster parameters for the shrubs in the different sampling locations.

Shrub species	Site name	Distance from village (m)	No. of shrubs	Cluster density m <sup>-2</sup>	Cluster radius(m)	No. of shrubs/ cluster	Nearest neighbor distance (m)
<i>G. senegalensis</i>	Thilla Ounte	50	129	0.000806	5.8	16	1.05
		464	234	0.000585	6.6	40	1.21
		655	206	0.000665	3.3	31	1.10
	Tabakali	50	386	0.001608	5.9	24	0.90
		281	561	0.001145	8.4	49	1.00
		543	566	0.001204	7.5	47	1.16
	Keur Matar Aram	50	164	0.000586	5.6	28	1.30
		212	479	0.001261	6.1	38	1.07
		487	487	0.001218	4.8	40	1.22
<i>P. reticulatum</i>	Boubname	50	102	0.000785	6.2	13	1.15
		433	189	0.000591	4.7	32	1.00
		802	167	0.000642	7.4	26	1.00
	Babanene	50	78	0.00078	3.8	10	0.98
		398	201	0.000558	6.4	36	1.11
		676	237	0.000608	4.2	39	1.00
	Nioro	50	124	0.000729	7.3	17	1.20
		378	344	0.000905	4.8	38	1.05
		708	314	0.000766	6.7	41	1.15

## Appendix 2

### Allometric relationships for estimating biomass in annually pruned native shrubs in Senegal's Peanut Basin.

(This appendix is a detailed presentation of allometric work for equations used in Chapter 3 and 4)

#### Introduction

Rapid, nondestructive methods are needed to make biomass estimates because of the labor and expense necessary to clip and weigh large plants, and the need to preserve the ecosystem (Ludwig et al., 1975). Methods to establish a relationship between easily obtained plant measurements and plant biomass include a technique termed dimension analysis (Whittaker, 1970) where shrub biomass is estimated using regression analysis (Whittaker, 1966; Newbould, 1967; Rutherford, 1979). Using dimension analysis, shrub biomass can be estimated nondestructively by one of two basic methods: high correlation between stem diameters and weights of various shrub parts (Whittaker, 1965; Buckman, 1966; Lyon, 1970) and a method that relies on the relationships between biomass, canopy area, and canopy volume as described for semi desert shrubs in New Mexico (Ludwig et al., 1975), sagebrush (*Artemisia tridentata*) (Rittenhouse and Sneva, 1977), and low shrubs in California (Bently et al., 1970). This method requires measurements of crown diameters and shrub height.

Biomass estimates of various shrub species and plant fractions have been developed using stem diameter, crown diameter, crown volume, crown cover, and height X circumference as independent variables (Murray and Jacobson, 1982). Bartolome and Kosco (1982) used basal diameter of branches arising from the primary stem (2<sup>nd</sup> order stems) to predict leaf and branch weights of deerbrush (*Ceanothus interrimus*.) using an allometric transformation in linear regression. Crown diameters and heights were used to estimate biomass of 3 tip sagebrush (*Artemisia tripartita* Rydb.) and Douglas rabbitbrush (*chrysothamnus viscidiflorus*) in Idaho (Murray and Jacobson, 1982). Rittenhouse and Sneva (1977) used log transformations of width and height to predict biomass of Wyoming big sagebrush (*Artemisia tridentata*). Crown volume was used to predict dry weights of greenleaf manzanita (*Arctostaphylos patula*) and snowbrush or tobacco brush (*Ceanothus velutinus*) on the west slopes of the southern Cascade-Siskiyou (Bentley et al., 1970). Other examples abound e.g. Telfer (1969), Peek (1970), Brown (1976),

Ohmann et al. (1976), Schmitt and Grigal (1981), Smith and Brand (1983), Brand and Smith (1985), Kie (1985) and Buech and Rugg (1989). The goal of this study was to develop above and belowground biomass predictive equations for 2 shrub species: *Piliostigma reticulatum* and *Guiera senegalensis* for use in characterizing shrub biomass C stocks.

## Materials and methods

This study was conducted in a total of eleven sites (7 for *G. senegalensis* and 4 for *P. reticulatum*) that were purposively selected to capture the north-south rainfall gradient in the Peanut basin (Table A.2.1). Of the total sample of 147 shrubs (99 *G. senegalensis* and 46 *P. reticulatum*), seventy two (50 *G. senegalensis* and 22 *P. reticulatum*) were studied in the first year while 49 *G. senegalensis* and 26 *P. reticulatum* were studied in the second year. The exact number of shrub samples in each site was chosen to reflect the full range of heights, diameters, forms and shrub decadence levels encountered.

Table A.2.1. Number of samples, their location, average number of stems and average diameter for the sampled shrubs.

Samples	Location	Average no. of stems	Mean diameter (cm)
<i>G. senegalensis</i>			
12	Keur Matar Aram	41.8	239.8
12	Thilla Ounte	36.5	202.9
16	Tabakali	18.5	218.6
13	Keur Asanulo	30.9	256.0
17	Ndiaye Mbidiene	29.7	233.5
16	Taibar Mar	24.0	239.8
13	Keur Mandie Mbar	28.3	192.7
<i>P. reticulatum</i>			
6	Boubname	23.6	219.1
6	Medina Ndiobène	22.8	240.8
28	Sikatrou	42.7	247.7
8	Sanguel	35.1	242.1



Measurements (to the nearest cm) of maximum height, 2 canopy diameters (taken at right angles to each other across the canopy of the plant, one of which was the maximum shrub diameter), mean diameter of the shrub base, and total number of stems were taken on each shrub during the months of March-April when the shrubs are at their peak growth (Breman and Kessler, 1995; Ker, 1995) before being pruned to clear fields for agricultural production. Clipping and excavation of the specimens was then done to measure biomass in the above and belowground components. Samples (leaves, stems and roots) of the biomass components were taken to the laboratory and oven dried at 105°C for 48 hours to enable calculation of dry weight biomass. The dataset obtained in the second year was randomly split to create an independent set of 12 *G. senegalensis* and 6 *P. reticulatum* that was used for validation of the obtained biomass predictive models.

Dimensional analysis (Whittaker, 1966; Newbould, 1967; Telfer, 1969; Peek, 1970; Whittaker, 1970; Ohmann et al., 1976; Rutherford, 1979; Smith and Brand, 1983; Brand and Smith, 1985) using ordinary least squares regression (Whittaker, 1965; Buckman, 1966; Bently et al., 1970; Lyon, 1970; Ludwig et al., 1975; Brown, 1976; Rittenhouse and Sneva, 1977; Schmitt and Grigal, 1981; Bartolome and Kosco, 1982; Murray and Jacobson, 1982; Kie, 1985; Buech and Rugg, 1989) was used to establish relationships between the obtained shrub measurements and shrub biomass. The independent variables used were mean crown diameter (*mcd*), mean base diameter (*mbd*), maximum height (*maxht*) and number of stems (*stems*) on the shrub. The dependent variables were dry weight of aboveground (leaves and stems), belowground and total biomass and were tested for normality of distribution using the Shapiro-Wilk statistic, W (SAS, 1999) following which, log transformations were done to obtain normal distribution of the residuals from regression analyses.

Linear ( $\log Y = \beta_0 + \beta_1 X_1$ ), logarithmic ( $Y = \beta_0 + \beta_1 \log X$ ), exponential ( $\log Y = \log \beta_0 + X^{\beta_1}$ ), log-log ( $\log Y = \log \beta_0 + \beta_1 \log X$ ), and quadratic ( $Y = \beta_0 + \beta_1 X + \beta_2 X^2$ ) regression models ( $Y$  = dry weight of biomass in grams,  $X_1 \dots X_n$  are the respective explanatory variables in each model) were used to assess choice of independent variables and predictive equations selected based on adjusted  $R^2$  values and the significance ( $p$ -value) of the regression coefficients. The validation data set was used to evaluate the predictive capacity of the various regression estimators and to

select the final equations for use in estimation of biomass. The Mean Square Error of the Prediction ( $MSEP = (y_i - \bar{y}_i)^2/m$ , where  $m$  is the number of observations used in the validation set, and the  $y_i$ 's are observed and predicted biomass for observation  $i$ ) was calculated based on the prediction equations applied to the independent validation data set. The MSEP measures the predictive ability of the models and serves as a basis for comparing different model predictions (Neter et al., 1996).

## Results

Tables A.2.2 through A.2.4 show the results of regression analyses of biomass fractions on the independent variables for the two study years. In all the models presented, the two-sided  $p$ -values for the coefficients (Tables A.2.7 to A.2.10 at the end of this appendix) were significant ( $p > 0.05$ ). Coefficients of determination were on average high, with log-linear relationships exhibiting the highest  $R^2$  values. Other studies on shrubs have found log-log and quadratic equations useful for a number of species (Bentley et al., 1970, Rittenhouse and Sneva 1977, Bryant and Kothmann, 1979). The equally good performance of *G. senegalensis* regression equations is surprising given the methodological difficulty of distinguishing and isolating individual shrubs for sampling. As measured by  $R^2$ , shrub maximum height (*maxht*) and number of stems (*stems*) were the best predictors of *G. senegalensis* aboveground biomass (leaves + stems) in 2003 while mean crown diameter (*mcd*) was the best predictor of aboveground biomass for *P. reticulatum* although the  $R^2$  (0.38) was very modest (Table A.2.2).

Table A.2.2. Biomass predictive equations and their adjusted  $R^2$  values for *G. senegalensis* and *P. reticulatum* in 2003.

Biomass fraction	Predictive equation	MSE	$R^2$
<i>G. senegalensis</i> (n = 50)			
Aboveground			
	$\log \text{ dry wt.} = 6.58 + (0.0036 * \text{maxht}) + (0.010 * \text{stems})$	0.07	0.70
Belowground			
	$\log \text{ dry wt.} = 6.95 + (0.0048 * \text{maxht}) + (0.015 * \text{mbd})$	0.12	0.55
Total			
	$\log \text{ dry wt.} = 8.13 + (0.017 * \text{stems})$	0.08	0.62
<i>P. reticulatum</i> (n = 22)			
Aboveground			
	$\log \text{ dry wt.} = 5.72 + (0.0076 * \text{mcd})$	0.17	0.38
Belowground			
	$\log \text{ dry wt.} = 8.57 + (0.013 * \text{mbd})$	0.05	0.75
Total			
	$\log \text{ dry wt.} = 8.75 + (0.012 * \text{mbd})$	0.05	0.73

Maximum height and mean base diameter (*mbd*) explained 55% of the total variation in the measured belowground biomass for *G. senegalensis*, whereas it explained 75% of belowground biomass variation for *P. reticulatum* in 2003. The number of *stems* and *mbd* explained 62% and 73% of the variation in observed total biomass for *G. senegalensis* and *P. reticulatum* respectively.

There were conspicuous changes in the performance and hence choice of predictor variables for biomass estimation in 2004 and these were accompanied by general improvements in  $R^2$  values over those obtained in 2003 (Table A.2.3).

Table A.2.3. Biomass predictive equations and their adjusted  $R^2$  values for *G. senegalensis* and *P. reticulatum* in 2004.

Biomass fraction	Predictive equation†	MSE	$R^2$
<i>G. senegalensis</i> (n =38)			
Aboveground	log dry wt. = $4.39+(0.0056*maxht)+(0.011*mcd)$	0.08	0.90
	Alt: log dry wt. = $4.77 +(0.013*mcd)$	0.09	0.88
Belowground	log dry wt. = $7.18+( 0.0094*mbd)+(0.025*stems)$	0.09	0.82
	Alt: log dry wt. = $7.41+(0.015*mbd)$	0.16	0.69
Total	log dry wt. = $6.85+( 0.0069 *mcd)+(0.023*stems)$	0.07	0.87
	Alt: log dry wt. = $7.14+(0.0054*mcd)+(0.0083*mbd)$	0.10	0.80
<i>P. reticulatum</i> (n = 20)			
Aboveground	log dry wt. = $4.20+( 0.013*maxht)+(0.0059*mcd)$	0.07	0.87
	Alt: log dry wt. = $4.89+(0.009*mcd)$	0.12	0.78
Belowground	log dry wt. = $7.14+(0.0099*mcd)$	0.11	0.81
	Alt: log dry wt. = $7.92+(0.016*mbd)$	0.14	0.76
Total	log dry wt. = $7.31+( 0.0098*mcd)$	0.09	0.83
	Alt: log dry wt. = $8.10+(0.015*mbd)$	0.14	0.76

† Alt = Alternative model

The fit of maximum height (*maxht*) and *mcd* on *G. senegalensis* aboveground biomass yielded an  $R^2$  value of 0.90, while the same fit gave a coefficient of determination of 0.87 for *P. reticulatum* aboveground biomass (Table A.2.3) estimates. The best fit models for shrub belowground biomass were those that incorporated *mbd* and *stems* for *G. senegalensis* ( $R^2 = 0.82$ ) and only *mcd* for *P. reticulatum* ( $R^2 = 0.81$ ). Mean crown diameter (*mcd*) was equally the best predictor of *P. reticulatum* total biomass ( $R^2 = 0.83$ ) while for *G. senegalensis*, total biomass was best predicted by a combination of *mcd* and *stems* ( $R^2 = 0.87$ ).

Better model performance in 2004 compared to 2003 presumably reflects the effect of timing of sampling (early-to-mid March in 2004 and late March to early April in 2003). Although plants have a remarkable capacity to coordinate the growth of their organs leading to a tight balance between biomass invested in different organs (Pearsall, 1927; Brouwer, 1962; Wilson, 1988), this practicable proportion and consequently performance of allometric relationships could have been compromised by the late sampling in 2003. Under extended periods of dryness, the shrubs drop their leaves in a “drought deciduous” behavior and this coupled with senescence, browsing by animals and the ubiquitous use of the stems for firewood leads to relatively higher plant decadence with time and hence poor performance of allometric relationships. Validating this notion is a shoot:root ratio analysis where log transformed shoot:root ratio plots (whose slopes are not expected to deviate much from unity for all plants species) shown in Figure A.2.1 reveal that the slopes were closer to unity in 2004 (slope = 0.70 and 0.94 for *G. senegalensis* and *P. reticulatum* respectively) compared to 2003 (slope = 0.58 and 0.47 for *G. senegalensis* and *P. reticulatum* respectively). Another plausible explanation for changes in allometric relationships across the study years as espoused by Peek (1970) is that stable relationships between shrub diameters/heights and weight only occur when a stable overstorey canopy has developed, or when a shrub community has matured to where further growth is minimal and decadence is not appreciable. This was not the case across the years hence the observed changes in the regression relations. van Noordwijk et al. (2000) surmise that relationships between diameters/heights of shrubs and their corresponding biomass have not stood the test of time; varying widely between sites, species and years and propose incorporation of wood density as a factor in the developed equations. Gerard and Buerkert (1999) attribute the differences in *G. senegalensis* regression coefficients across years to inter-annual variability in regrowth phenology but data are not available to confirm their postulate.

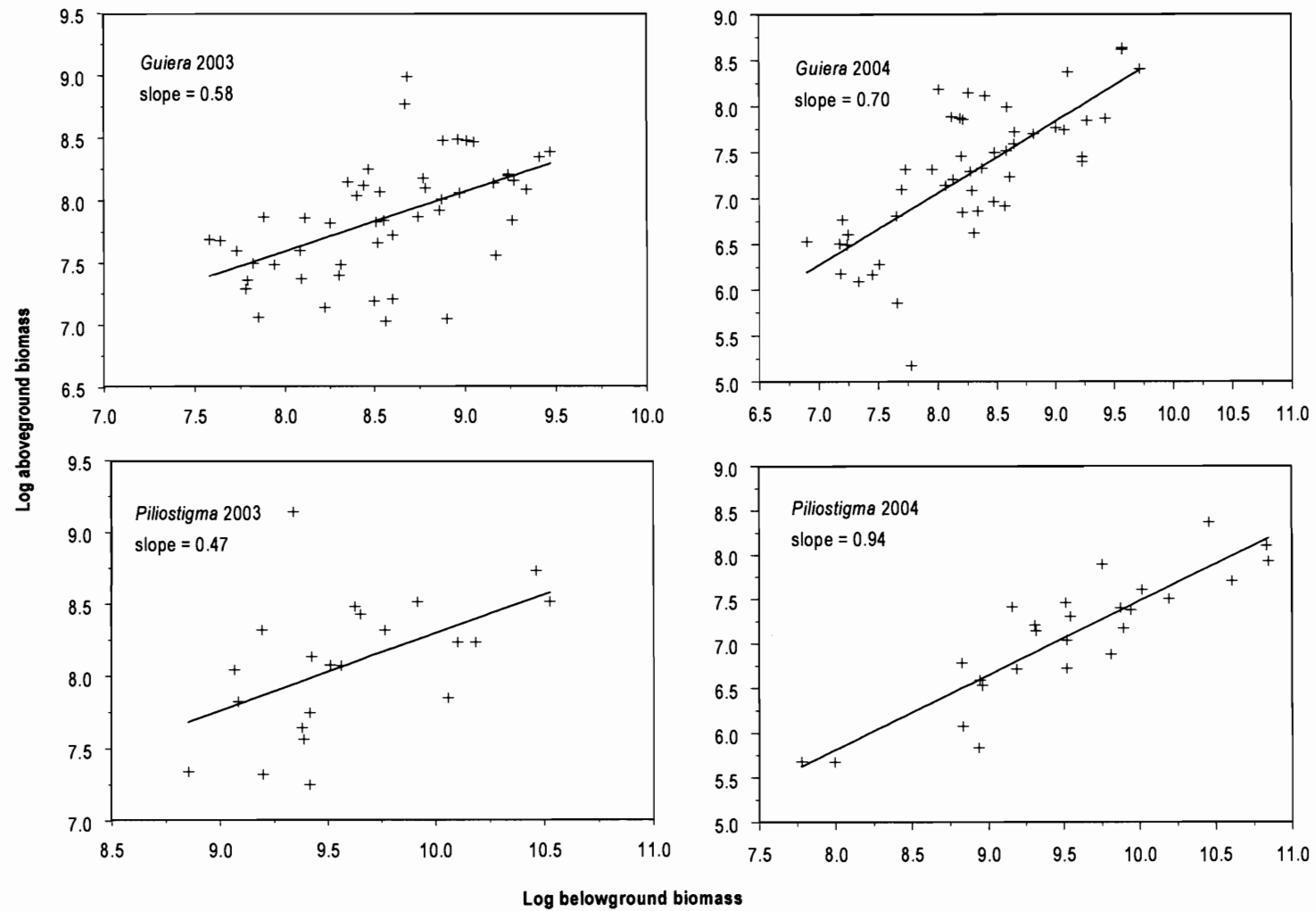


Figure A.2.1. Biomass allocation expressed as shoot:root ratio for the sampled shrubs.

Merging individual species data over the two years produced results shown in Table A.2.4. In general this analysis led to an improvement in coefficients of determination of the best fit models (average  $R^2 = 0.73$  for both *G. senegalensis* and *P. reticulatum*) over those obtained in 2003 (average  $R^2 = 0.62$  for both *G. senegalensis* and *P. reticulatum*) for the shrubs but a decline in  $R^2$  values when compared to those obtained in 2004 - average  $R^2 = 0.83$  for *G. senegalensis* and 0.80 for *P. reticulatum* (see Tables A.2.2, A.2.3 and A.2.4). This could reflect undersampling or the influence of the 2004 data on the relationships.

Table A.2.4. Biomass predictive equations and their adjusted  $R^2$  values for the shrubs over the two study years.

Biomass fraction	Predictive equation†	MSE	$R^2$
<i>G. senegalensis</i>			
Aboveground	$\log \text{ dry wt.} = 5.11 + (0.011 * mcd) + (0.0083 * stems)$	0.09	0.81
	Alt: $\log \text{ dry wt.} = 5.12 + (0.012 * mcd)$	0.10	0.79
Belowground	$\log \text{ dry wt.} = 7.42 + (0.011 * mbd) + (0.015 * stems)$	0.12	0.65
	Alt: $\log \text{ dry wt.} = 7.52 + (0.015 * mbd)$	0.15	0.59
Total	$\log \text{ dry wt.} = 7.15 + (0.005 * mcd) + (0.005 * mbd) + (0.013 * stems)$	0.07	0.78
	Alt: $\log \text{ dry wt.} = -3685 + (26.34 * mcd) + (50.54 * mbd) + (111 * stems)$	56E <sup>6</sup>	0.73
<i>P. reticulatum</i>			
Aboveground	$\log \text{ dry wt.} = 3.88 + (0.023 * maxht) + (0.003 * mcd)$	0.20	0.69
Belowground	$\log \text{ dry wt.} = 7.64 + (0.005 * mcd) + (0.0075 * mbd)$	0.10	0.75
Total	$\log \text{ dry wt.} = 7.76 + (0.006 * mcd) + (0.0057 * mbd)$	0.10	0.76
	Alt: $\log \text{ dry wt.} = 7.60 + (0.0089 * mcd)$	0.12	0.70

† Alt = Alternative model

Although some allometric work indicates that relationships are not heavily species-specific (de Gier and Sakouhi, 1996), general models for the two shrubs performed relatively poorly compared to species-specific models (Table A.2.5).

Table A.2.5. General allometric models for estimating shrub biomass.

Fraction	Predictive equation	MSE	R <sup>2</sup>
Aboveground	$\log \text{ dry wt.} = 4.58 + (0.011 * \text{maxht}) + (0.008 * \text{mcd}) - (0.005 * \text{stems})$	0.17	0.68
Belowground	$\log \text{ dry wt.} = 7.24 + (0.002 * \text{mcd}) + (0.012 * \text{mbd}) + (0.008 * \text{stems})$	0.21	0.66
Total	$\log \text{ dry wt.} = 7.32 + (0.005 * \text{mcd}) + (0.008 * \text{mbd}) + (0.006 * \text{stems})$	0.14	0.72

### Final model selection and comparison of observed and predicted biomass values

The Mean Square Error of Prediction (MSEP) for the best performing models and the corresponding correlations between observed and predicted biomass values are presented in Table A.2.6.

Table A.2.6. Selected model MSEPs and their Pearson correlation coefficients for observed and predicted biomass.

Model	MSEP	r
<i>G. senegalensis</i>		
Aboveground biomass: $\log \text{ dry wt.} = 4.39 + (0.0056 * \text{maxht}) + (0.011 * \text{mcd})$	0.03	0.98
Belowground biomass: $\log \text{ dry wt.} = 7.41 + (0.015 * \text{mbd})$	3.64	0.88
Total biomass: $\log \text{ dry wt.} = 7.14 + (0.0054 * \text{mcd}) + (0.0083 * \text{mbd})$	2.99	0.91
<i>P. reticulatum</i>		
Aboveground biomass: $\log \text{ dry wt.} = 4.20 + (0.013 * \text{maxht}) + (0.0059 * \text{mcd})$	0.29	0.99
Belowground biomass: $\log \text{ dry wt.} = 7.14 + (0.0099 * \text{mcd})$	2.60	0.99
Total biomass: $\log \text{ dry wt.} = 7.31 + (0.0098 * \text{mcd})$	2.09	0.99



In our selection of final models for use in biomass prediction, we deliberately disregarded equations with stems as a predictor variable owing to the extra cost and time needed in collecting the data and the uncertainty its inclusion would occasion given the fact that the stems are commonly harnessed by farmers for firewood.

Significant correlations between observed and predicted biomass were observed and these relationships are plotted in Figure A.2.2. While the regressions of observed and predicted biomass are significant, there are differences between the regression line and unity (1:1 relationship). The predictive models for the *G. senegalensis* belowground fraction and total biomass over estimated biomass with considerably higher scatter in the medium-sized shrubs. The *G. senegalensis* aboveground fraction equation and all the equations for *P. reticulatum* were more accurate, regression of observed vs predicted values yielding slopes close to 1, although the intercepts are not quite zero (Figure A.2.2). *G. senegalensis* total biomass predicted using the “total biomass” equation was consistently lower than total biomass from the summation of above and belowground predictions although the differences were not statistically significant (two-sided  $p$ -value = 0.72). The opposite was true for *P. reticulatum* with the “total biomass” equation consistently predicting more biomass as compared to the biomass prediction from the summation of the above and belowground predictions (two-sided  $p$ -value = 0.74).

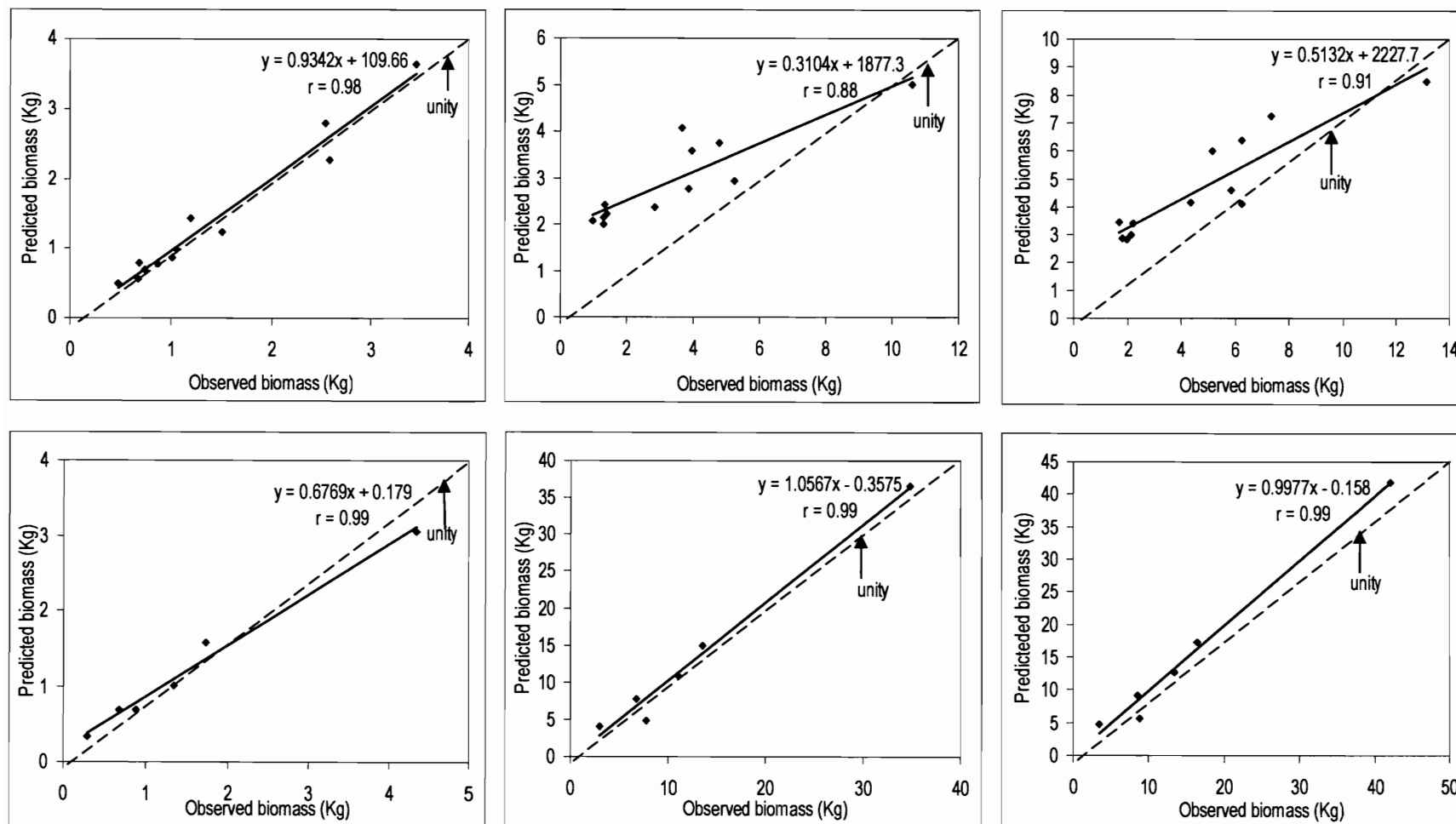


Figure A.2.2. Unity plots of observed and predicted biomass. Top left to right: aboveground, belowground and total *G. senegalensis* biomass. Bottom left to right: aboveground, belowground and total *P. reticulatum* biomass.

Table A.2.7. Regression statistics for predicting biomass fractions of the shrubs species in 2003. Parameters are from the relationship  $\log Y = \beta_0 + \beta_1(X_1) + \dots + \beta_n(X_n)$ , where  $Y$  = dry weight of biomass in grams,  $X_1 \dots X_n$  are the respective explanatory variables in each model.

Species	Component	Parameter	Estimate	Std. Error	t-value	Pr(> t )
<i>G. senegalensis</i>	Aboveground biomass	<i>constant</i>	6.5777	0.2276	28.8944	0.0000
		<i>maxht</i>	0.0036	0.0018	2.0580	0.0453
		<i>stems</i>	0.0101	0.0027	3.7225	0.0005
	Belowground biomass	<i>constant</i>	6.9468	0.3082	22.5432	0.0000
		<i>maxht</i>	0.0048	0.0022	2.1954	0.0331
		<i>mbd</i>	0.0152	0.0022	6.7829	0.0000
	Total biomass	<i>constant</i>	8.1313	0.1063	76.4711	0.0000
		<i>stems</i>	0.0172	0.0029	5.9246	0.0000
<i>P. reticulatum</i>	Aboveground biomass	<i>constant</i>	5.7216	0.5198	11.0081	0.0000
		<i>mcd</i>	0.076	0.0022	3.5116	0.0022
	Belowground biomass	<i>constant</i>	8.5673	0.1413	60.6498	0.0000
		<i>mbd</i>	0.0129	0.0017	7.7658	0.0000
	Total biomass	<i>constant</i>	8.7471	0.1411	62.0024	0.0000
		<i>mbd</i>	0.0122	0.0017	7.3873	0.0000

Table A.2.8. Regression statistics for predicting biomass fractions of the shrubs in 2004. Parameters are from the relationship  $\log Y = \beta_0 + \beta_1(X_1) + \dots + \beta_n(X_n)$ , where  $Y$  = dry weight of biomass in grams,  $X_1 \dots X_n$  are the respective explanatory variables in each model.

Species	Component	Parameter	Estimate	Std. Error	t-value	Pr(> t )
<i>G. senegalensis</i>	Aboveground biomass	<i>constant</i>	4.3912	0.2276	19.2945	0.0000
		<i>maxht</i>	0.0056	0.0022	2.5536	0.0166
		<i>mcd</i>	0.0113	0.0010	11.0289	0.0000
		<i>constant</i>	4.7692	0.1891	25.2165	0.0000
		<i>mcd</i>	0.0128	0.0009	14.0327	0.0000
	Belowground biomass	<i>constant</i>	7.1843	0.1307	54.9847	0.0000
		<i>mbd</i>	0.0094	0.0021	4.5453	0.0001
		<i>stems</i>	0.0251	0.0059	4.2864	0.0002
		<i>constant</i>	7.4143	0.1517	48.8878	0.0000
		<i>mbd</i>	0.0154	0.0019	7.8882	0.0000
	Total biomass	<i>constant</i>	6.8490	0.1648	41.5583	0.0000
		<i>mcd</i>	0.0069	0.0010	6.6326	0.0000
		<i>stems</i>	0.0219	0.0048	4.5794	0.0001
		<i>constant</i>	7.1356	0.2413	29.5685	0.0000
		<i>mcd</i>	0.0054	0.0021	2.5281	0.0176
		<i>mbd</i>	0.0083	0.0035	2.4086	0.0231
<i>P. reticulatum</i>	Aboveground biomass	<i>constant</i>	4.2036	0.2601	16.1619	0.0000
		<i>maxht</i>	0.0125	0.0031	4.0056	0.0006
		<i>mcd</i>	0.0059	0.0011	5.2618	0.0000
		<i>constant</i>	4.8874	0.2503	19.5271	0.0000
		<i>mcd</i>	0.0091	0.0010	9.1132	0.0000
	Belowground biomass	<i>constant</i>	7.1376	0.2413	29.5777	0.0000
		<i>mcd</i>	0.0099	0.0010	10.2674	0.0000
		<i>constant</i>	7.9184	0.1978	40.0399	0.0000
		<i>mbd</i>	0.0156	0.0018	8.7511	0.0000
	Total biomass	<i>constant</i>	7.3087	0.2253	32.4329	0.0000
		<i>mcd</i>	0.0098	0.0009	10.9346	0.0000
		<i>constant</i>	8.1046	0.1952	41.5239	0.0000
		<i>mbd</i>	0.0154	0.0018	8.7116	0.0000

Table A.2.9. Regression statistics for predicting biomass fractions of the two shrubs with individual species combined over the two years. Parameters are from the relationship  $\log Y = \beta_0 + \beta_1(X_1) + \dots + \beta_n(X_n)$ , where  $Y$  = dry weight of biomass in grams,  $X_1 \dots X_n$  are the respective explanatory variables in each model.

Species	Component	Parameter	Estimate	Std. Error	t-value	Pr(> t )
<i>G. senegalensis</i>	Aboveground biomass	<i>constant</i>	5.1061	0.1437	35.5341	0.0000
		<i>mcd</i>	0.0105	0.0007	14.7170	0.0000
		<i>stems</i>	0.0083	0.0027	3.0460	0.0032
	Belowground biomass	<i>constant</i>	5.1171	0.1511	33.8701	0.0000
		<i>mcd</i>	0.0115	0.0007	17.2622	0.0000
		<i>constant</i>	7.4193	0.1002	74.0713	0.0000
		<i>mbd</i>	0.0106	0.0019	5.7159	0.0000
		<i>stems</i>	0.0149	0.0039	3.7964	0.0003
		<i>constant</i>	7.5176	0.1047	71.7681	0.0000
		<i>mbd</i>	0.0154	0.0015	10.5245	0.0000
		<i>constant</i>	7.1505	0.1326	53.9099	0.0000
		<i>mcd</i>	0.0049	0.0008	6.3866	0.0000
		<i>mbd</i>	0.0046	0.0017	2.7071	0.0084
		<i>stems</i>	0.0132	0.0030	4.4130	0.0000
	Total biomass	<i>constant</i>	3.8828	0.3703	10.4866	0.0000
		<i>maxht</i>	0.0229	0.0034	6.8147	0.0000
		<i>mcd</i>	0.0031	0.0014	2.2670	0.0282
		<i>constant</i>	7.6421	0.2065	36.9991	0.0000
		<i>mcd</i>	0.0051	0.0012	4.1361	0.0002
		<i>mbd</i>	0.0075	0.0018	4.1071	0.0002
		<i>constant</i>	-3685.03	1168.046	-3.1549	0.0023
		<i>mcd</i>	26.3399	6.7948	3.8765	0.0002
<i>P. reticulatum</i>	Aboveground biomass	<i>mbd</i>	50.5393	14.9928	3.3709	0.0012
		<i>stems</i>	111.1660	26.3924	4.2120	0.0001
		<i>constant</i>	7.7610	0.1999	38.8175	0.0000
	Belowground biomass	<i>mcd</i>	0.0061	0.0012	5.0941	0.0000
		<i>mbd</i>	0.0057	0.0018	3.2152	0.0024
		<i>constant</i>	7.5994	0.2123	35.8041	0.0000
	Total biomass	<i>mcd</i>	0.0089	0.0009	10.3708	0.0000
		<i>mbd</i>				
		<i>constant</i>				
		<i>mcd</i>				
		<i>mbd</i>				
		<i>constant</i>				

Table A.2.10. Regression statistics for the general biomass predictive models. Parameters are from the relationship  $\log Y = \beta_0 + \beta_1(X_1) + \dots + \beta_n(X_n)$ , where  $Y$  = dry weight of biomass in grams,  $X_1 \dots X_n$  are the respective explanatory variables in each model.

Component	Parameter	Estimate	Std. Error	t-value	Pr(> t )
Aboveground biomass	<i>constant</i>	4.5828	0.1994	22.9873	0.0000
	<i>maxht</i>	0.0108	0.0018	6.1325	0.0000
	<i>mcd</i>	0.0079	0.0009	8.5704	0.0000
	<i>stems</i>	-0.0051	0.0022	-2.3284	0.0215
Belowground biomass	<i>constant</i>	7.2373	0.1799	40.227	0.0000
	<i>mcd</i>	0.0023	0.0011	2.1934	0.0301
	<i>mbd</i>	0.0118	0.0021	5.5026	0.0000
	<i>stems</i>	0.0077	0.0031	2.4437	0.0159
Total biomass	<i>constant</i>	7.3222	0.1475	49.6567	0.0000
	<i>mcd</i>	0.0047	0.0009	5.4063	0.0000
	<i>mbd</i>	0.0083	0.0018	4.7134	0.0000
	<i>stems</i>	0.0062	0.0026	2.4116	0.0173

## Appendix 4

### Tutorial to using i\_Century for coupling the CENTURY Model and GIS

This appendix is a step by step description of procedures to undertake in order to replicate and transfer the CENTURY-GIS coupling method developed in this study to other study areas. ArcMap is used as the GIS domain while Microsoft Access® is used in database management and therefore, application of this method demands a working knowledge of the two.

#### Introduction

i\_Century is not a spatially distributed version of CENTURY but it makes a spatial approach to carbon assessment more manageable by providing possibilities for handling and running large sample populations. The basic philosophy of the i\_Century approach is to manage both the input and output data of a large set of CENTURY simulations within a single database. This requires converting all existing data from ASCII files and other file formats into Microsoft Access®, the database program that was selected for the i\_Century system. Thus, it is incumbent upon the user to develop methods of processing desired input data into the proper database structure required for i\_Century. Figure A.4.1 is a summary of the procedures described in this tutorial to enable a loose coupling between the CENTURY model and GIS.

#### Preparation of GIS data

1. Load the Century-GIS “.mxd” file provided on the inserted CD into ArcMap. This brings up an additional toolbar called Century/GIS modeling. The loaded map contains a point layer called Extractions. This is not in anyway related to the procedure but rather it is just used to carry the application.
2. Install the ET Geowizards program provided on the CD using the accompanying instructions.
3. Load your landuse (whatever number of years of known landuse available), soil, precipitation and temperature layers. Precipitation and temperature layers should be point files of climate station location. Also load a polygon layer delimiting your study area and make sure that all the layers have a common coordinate system.
4. Convert the precipitation and temperature point layers into Thiessen polygons using the ET Geowizards.

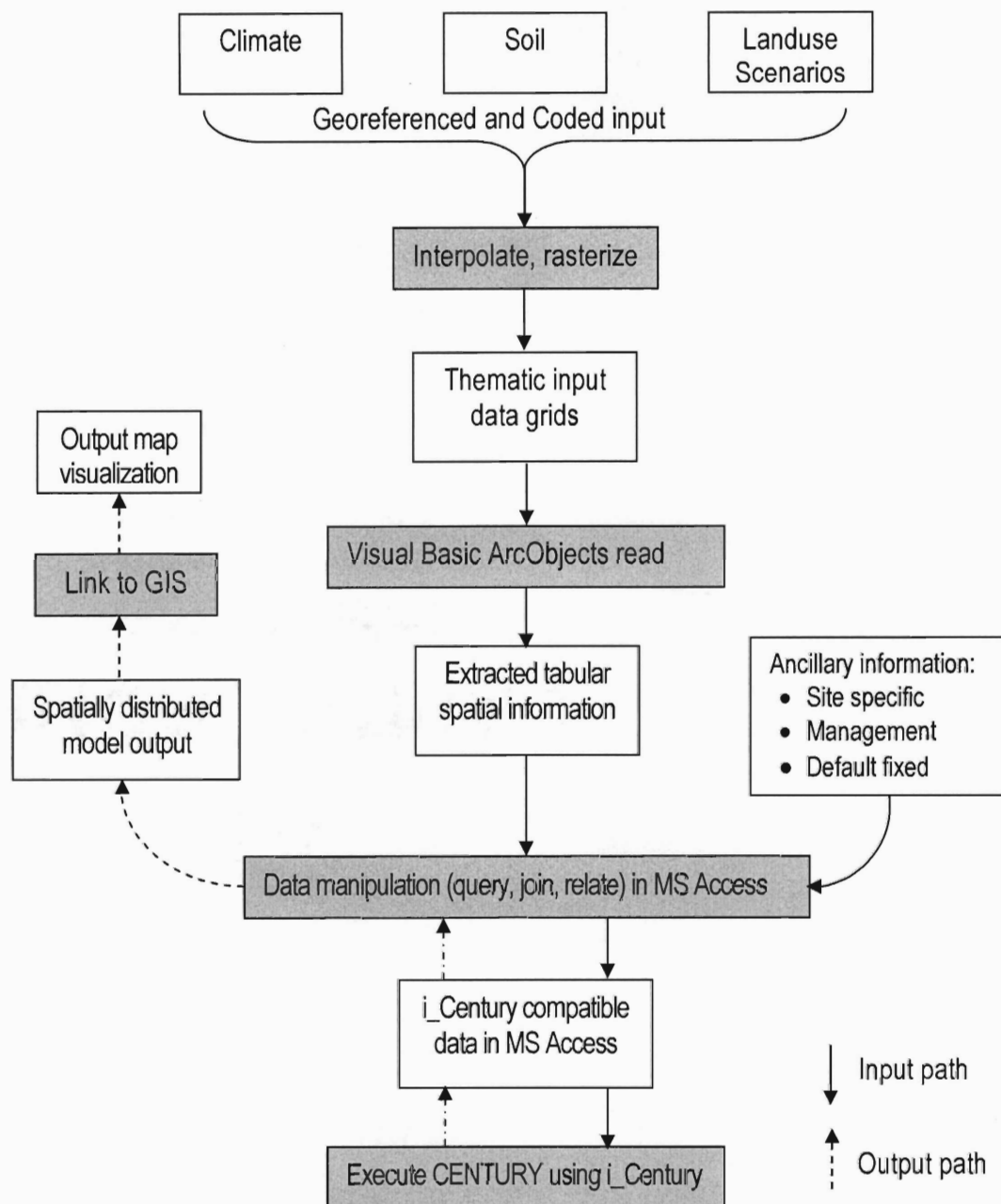


Figure A.4.1. Conceptual framework of the loose coupling between GIS and the CENTURY model



5. Convert all the thematic polygon layers into raster layers.
  - (a) Decide upon cell size (this is synonymous to the resolution of your study). Note that changes on cell size have an effect on the quality of data. The bigger the size, the greater the level of errors and uncertainties
  - (b) Load Spatial Analyst if not already done.
  - (c) In the Spatial Analyst menu that appears click on Spatial Analyst—►Options and fill out the options in the three banners. Choose for instance your study area as analysis extent, and the cell size should be the cell size you have decided upon as your resolution.
  - (d) Choose Spatial Analyst—►Convert—►Features to Raster. Be sure to choose the correct field. The field of interest is the ID field.
  - (e) Repeat the above steps for each polygon layer.
6. Using ET Geowizards, create a point for each center of the cells generated in the step above. Choose the Point banner—►Create point grid. Choose your study area as initial grid extent; define the grid extent by subtracting half the size of the cell size from the extents shown in the banner or else the point will be placed in the corners of the cell. Delete the points that are outside the study area through selection by location. Export the data and add it as a layer in ArcMap.

### Extraction of data from GIS

1. Use the Extract GIS Data routine under the Century-GIS toolbar to extract data from the multiple layers containing land use data, precipitation, temperature and soil data. (Note that the data layers only contain the IDs of the maps and no other attributes. The additional attributes are stored in the Access database and linked to the IDs by the use of queries in the Access database). In the dialogue box that appears, fill in the names of the appropriate grids containing the appropriate information. If you do not have enough data layers for the available options, just fill out with other raster themes and delete the columns subsequently. The names specified as column names should be maintained.
2. Assign IDs to the extracted points using the AssignID function under the Century-GIS toolbar.
3. Add X and Y coordinates to the attribute table using the Add XY coordinates tool in Arc Toolbox (Under data management tools). This step is based on the assumption that you are using ArcGIS 9. If you are running version 8, you can download a script from <http://arcscripts.esri.com> (search for "add XY") and follow the instructions provided to add coordinates to the table.
4. Export the data in the table to a text file (\*.txt): Options—►Export.

### In the Microsoft Access® database

Open the 'tutorial.mdb' on the CD in Microsoft Access® and read in the \*.txt file exported from GIS using File—→Get external data. Name it 'From GIS Samples'. The table imported from GIS contains columns as listed in Table A.4.1. The columns contain codes for soil, climate and landuse. The SoilID indicates which record to use from the Soils table. The WeatherID specifies the ID number of the cell from which to use the time series of precipitation and temperature. The time series of the cells are listed in the Weather Station Data table. The PrecID and TempID specify the ID number of the Thiessen polygon used for the average precipitation and temperature data outside the time series. Each column of LuXX represents a year of known land use or a created land use scenario. The LuXX contains codes for different landuses which in the database must be hooked up to tables containing information on land use practice and history.

Table A.4.1. Output information from GIS.

Column name	Content
ID	Unique ID of cell
SoilID	ID for soil characteristics (sand, silt and clay)
WeatherID	Name of the file name from which to use time series of minimum and maximum temperature and precipitation
PrecID	ID of climate station for precipitation
TempID	ID of climate station for minimum and maximum temperature
LuccXX	Land use code for year XX of known landuse cover

Running i\_Century requires the creation of 8 different tables to hold the CENTURY model initialization parameters. Theses tables and their CENTURY model equivalent are shown in Table A.4.2 below.

Table A.4.2. Overview of i\_Century tables

i_Century table	CENTURY equivalent	Comments	Key inputs
<u>Site specific data</u>			
Site data	Site specific	If not known, model can be set to automatically simulate these	Initial nutrients and SOM values
Soil layers		A record for each layer	Layer depth
Soils		Average of upper 20 cm	Fractions of sand, silt and clay, bulk density, pH
Monthly weather		Monthly averages	Precipitation, maximum & minimum temperature
<u>Management</u>			
Event blocks	Management	Each block represents a time span of a certain land use	First year of block, last year of block
Events		Specification of land use system	For each block is given: year, month and event
<u>Fixed parameters</u>			
Control records	Default fixed	Contains fixed parameters: intercept, slope, constants, factors etc.	Default values
<u>Event files</u>			
Crops	crop.100	Crop and grass parameterization	Default values
Cultivation	cult.100	Parameterization of cultivation methods	Default values
Fertilizer	fert.100	Parameterization of fertilization methods	Default values
Fire	fire.100	Parameterization of fire options	Default values
Grazing	graz.100	Parameterization of grazing options	Default values
Harvest	harv.100	Harvest methods Parameterization	Default values
Irrigation	irr.100	Irrigation options parameterization	Default values
Omad	omad.100	Parameterization of OM addition methods	Default values
Tree removal	trem.100	Parameterization of tree removal methods	Default values
Trees	tree.100	Parameterization of tree types	Default values
<u>Output</u>			
Output Annual		Out put parameters provided annually	Default values
Output		Output parameters in last year	Default values

To convert the columns exported from GIS into the desired tables, a number of queries detailed in appendix 5 must be performed. The first step is to assign a unique number to each of the possible land use trajectories. In our study, there was one landuse instance and therefore only one map is possible. If you have four periods of known landuse,  $16 (4^4)$  different land use trajectories are possible, which should be assigned in a column in the 'From GIS Samples' table.

1. Enter the Design View of the 'From GIS Samples' table (right click the table).
2. Add a new Field name called LuUnique (Unique landuse) with the field type "Number".
3. Go to Queries and run the 'LuXX' queries (double click each of them). You can add other queries of the same type depending on the exact number of landuse trajectories in the study.
4. Run the 'Delete Samples' query. This will delete records in the 'Samples' table, so that it is ready to run with a new set of samples.
5. Open the 'Samples' table in Design View and make sure the ID field is defined as Autonumbering type.
6. Run the 'Samples' query. This query will add all of the unique samples from the 'From GIS Samples' table to the 'Samples' table.
7. Open the 'Samples' table in Design View again and change the ID field to Number type. This is needed in later queries as one query cannot handle autonumbering in two tables simultaneously.
8. Run the 'Update Samples' query. The fields Default01 and Default99 are filled out in the 'Samples' table. Default01 is used in the 'Control Records' query and the 'Site Data' query whereas Default99 is to be used in the 'Block' queries.
9. Run the 'Monthly Temperature' query. This extracts the average temperature data for each ID from the 'Monthly Weather Average' table. The 'Monthly Weather Average' table contains the monthly precipitation, minimum and maximum temperature averages for each of the climate Thiessen polygons in your study area.
10. Run the 'Monthly Precipitation' query. This extracts the average precipitation data for each ID from the 'Monthly Weather Average' table.
11. Run the 'Monthly Weather' query. This will combine the 'Monthly Weather Temperature' and the 'Monthly Precipitation' queries into a table called 'Monthly Weather' which is used in i\_Century.

12. Run the 'Control Records' query. This will create the 'Control Records' table, which is used in i\_Century. The 'Control Records Default' table is used in this query and this particular method is possible because the same Control Records were applied in our study area. It is possible to have more than one Control Records Default table by an additional query.
13. Run the 'Site Data' query. This will create the 'Site Data' table, which is used in i\_Century. The Site 'Data Default' table is used in this query and this particular method is possible because the same 'Site Data' are applied in the whole area. Build a similar query if you have more than one set of Site Data in your study area.
14. Run the 'Event Blocks Site Delete' query. This will delete records in the 'Event Blocks Site' table.
15. Run each of the 'Block xx – xx' queries. This will create the 'Event Blocks Site' table. For each query the IDs are assigned a Block number and the WeatherID.
16. Run the 'Event Block query1'. Here each block number from the 'Event Blocks Site' table is combined with the 'Sequence Number Default' table, which assigns the years and weather choice. Furthermore it also assigns which sequence to use depending on the land use. In this example only shrub and nonshrub sequences exist.
17. Run the 'Event Blocks query2'. This query creates the 'Sequence Events' table. It assigns the events taking place for each sequence.
18. Run the 'Events' query. This query combines the 'Event Blocks query1' with the block years and weather choices with the 'Events Blocks' table. The 'Event' table is created, which is used in i\_Century.

### **Install and run the i\_Century**

Before installing i\_Century, version 4 of the CENTURY SOM model should be installed on the computer. The default century.exe file distributed with the CENTURY model is not compatible with i\_Century and therefore has to be replaced by the century.exe file provided on the CD insert.

1. Install i\_Century and through configuration (File—> Configuration), point i\_Century to the location of the century executable file (century.exe) as shown in Figure A.4.2.

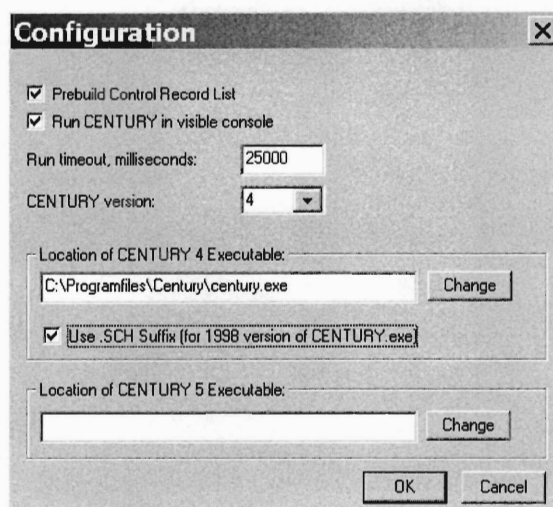


Figure A.4.2. i\_Century configuration dialog box

2. Open the tutorial.mdb and run the records. i\_Century will now execute the CENTURY model for each record.

### Extraction and visualization of results

After the simulations have run, the results can be viewed in the Access database in the tables Output and Output Annual. You may run the Test01 query to check if the simulations have been performed correctly.

1. Run the 'Output Year xxxx' queries for the years you please. These queries extract the SOMTC (total belowground carbon) for the specified year from the Output Annual table. These queries can be changed to other years or output variables by entering the Design View.
2. Run the 'Output to GIS' query. This query hooks up the extracted values to the 'Samples' table, which again is connected to the 'From GIS Samples' table thereby associating the values with each cell in the map. The query also creates the table 'Output to GIS'. The 'Output to GIS' table should be exported (File → Export) as \*.dbf file.
3. Open the \*.dbf file in ArcMap. Right click it and choose Display XY data. Load Spatial Analyst and convert the point data (Feature to raster) into raster by choosing the field containing the year of interest. Specify the same cell size as the resolution of your study.

To modify the Access database to be valid in your own area you should adjust the following tables:

- a) 'Monthly Weather Average', according to weather data in your study area.
- b) 'Weather Station Data': Historical weather data which should at least contain average monthly precipitation, minimum temperature and maximum temperature for each year.
- c) 'Soil Layers', according to the soil map: The 'Soil Layers' table contains the depth, nutrient contents, wilting point and field capacity for each horizon.
- d) 'Soils', according to the soil map. It contains the texture fractions, bulk density, pH and other soil characteristics.
- e) 'Sequence Event Default'. The 'Sequence Event Default' table contains the definitions of different blocks. A block consists of sequences of events and may be several years e.g. four years of agricultural practice, 10 years of fallow of which there is bush fire every fourth year. A sequence defines the events whereas the block defines the actual time period in which the sequences are taking place.
- f) 'Sequence Data Default', according to the management methods in the area: This table contains information about the different sequences that can be used in the blocks. There is a short description and an indication on how many years the sequence covers. This table also contains information about what time of year the output data is wanted and how often the output should be extracted.
- g) 'Sequence Number Default', according to the management history or modeled future management in the area.
- h) 'Control Records Default', according to the chosen control parameters for the area.
- i) 'Site Data Default', according to the site specific data for the area.

A number of queries should be changed according to the data set that you are putting into Microsoft Access®. For instance if you have only three years with land use/land cover data the LuXX queries should be adjusted and the number of LuXX queries should be changed from 16 to 9. Furthermore both the 'Samples' query and the 'Samples' table should be adjusted to suit three and not four land use/land cover layers. Doing this in turn necessitates modification of the 'Block' queries which are all based on one of the LuccXX fields. Output year queries should be adjusted to your wishes and the 'Output To GIS' query should be modified according to the number of LuccXX fields.

## Appendix 5

### Preprocessing i\_Century input data

This appendix is a detailed presentation of the routines undertaken to process data from GIS and other input data into tables that are compatible with the i\_Century structure to enable coupling of GIS and the CENTURY model. Presented flow diagrams show which tables are related and the queries performed to relate them. Figure A.5.1 is a key to the symbols used.







Symbol	Description	Symbol	Description
	Table exported from GIS	<u>Site data</u>	Name of table
	Table with additional information	 Query	Query of table(s) above
	Table with default fixed information	————	Relation
	Query table	<b>Temp</b>	Bold words are columns transferred to the new table after a query
	Resulting table ready for use in i_Century	..	More columns in the table

Figure A.5.1. Symbols used in the flow diagrams showing the database routines used

### Site Specific data

In i\_Century, site specific data is stored in four tables (Site Data, Monthly Weather, Soil Layers, and Soils). The Site Data table is created by the routine in Figure A.5.2. The Monthly Weather table is created through the routine presented in Figure A.5.3. This table is needed if the CENTURY model is to be initialized using average climate data. If the model is to be run using time series of climatic data, tables with the time series data for each station are used. The 'Soil Layers' and 'Soils' tables are additional information tables that are set up in the database. No queries are needed on these tables and they are just related to the other tables using the routine presented in Figure A.5.4.



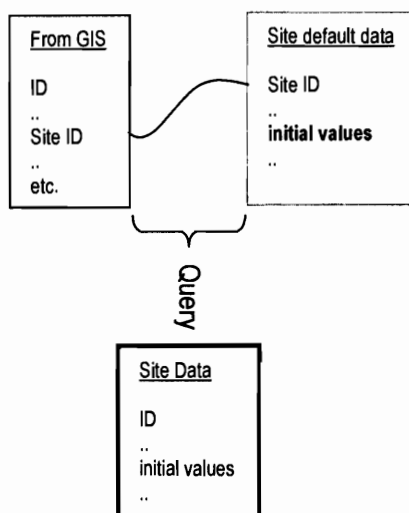


Figure A.5.2. Routine used to create the Site Data table.

### Default Fixed Data

Fixed Default Data in i\_Century is provided in the Control Records table. The Control Records table is thus made from default values distributed with the CENTURY Model as shown in Figure A.5.4.

### Management Data

Management data comprises two tables; the Events and Event Blocks tables and these contain codes for management practices and land use history. The routines used in creating these tables are outlined in Figures A.5.5 and A.5.6.

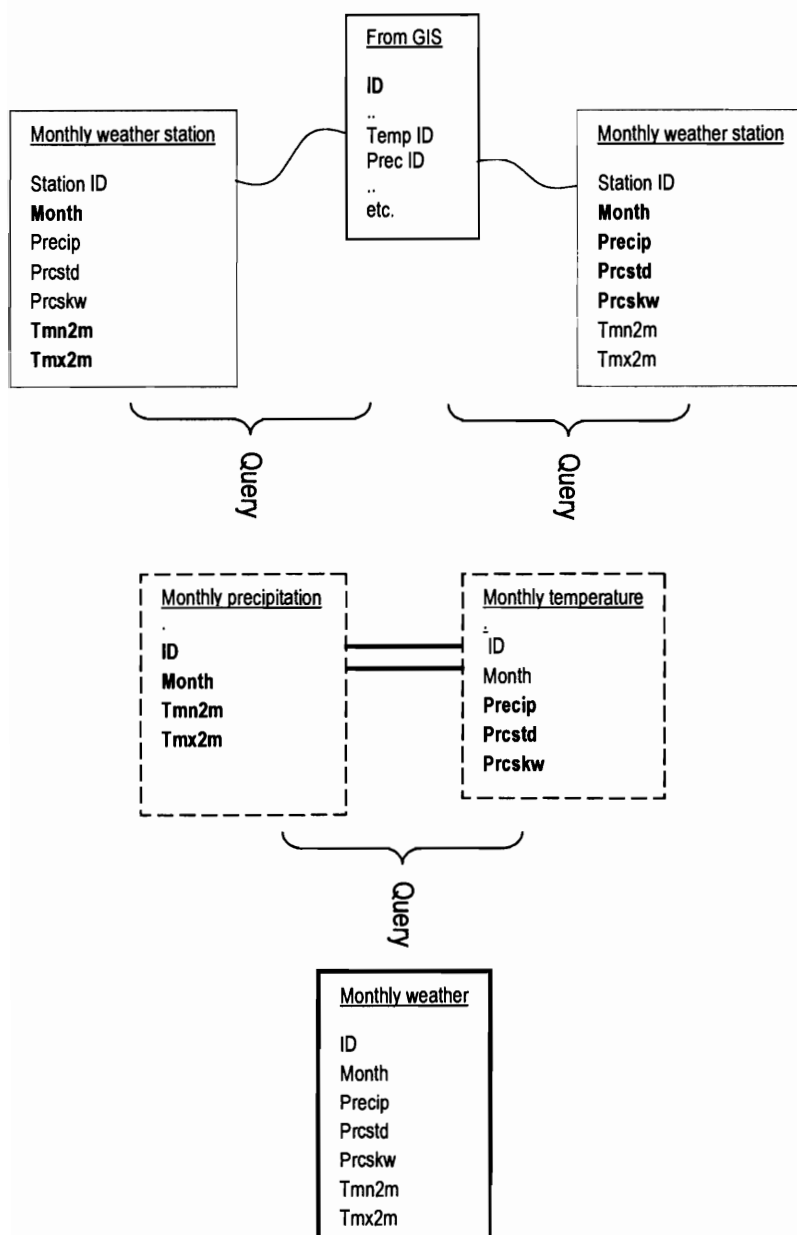


Figure A.5.3. Routine for creating the Monthly Weather table.

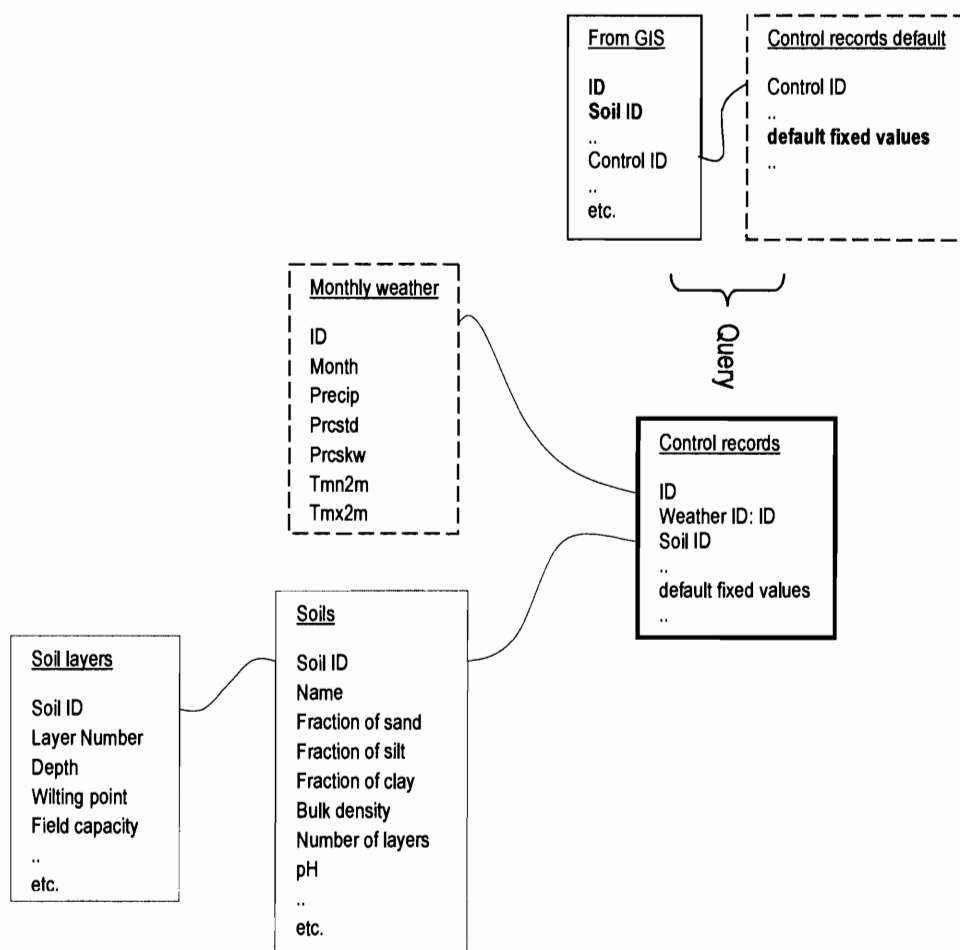


Figure A.5.4. Routine for creating the Control Records table and relating the Control Records table to the Monthly Weather and Soils tables.

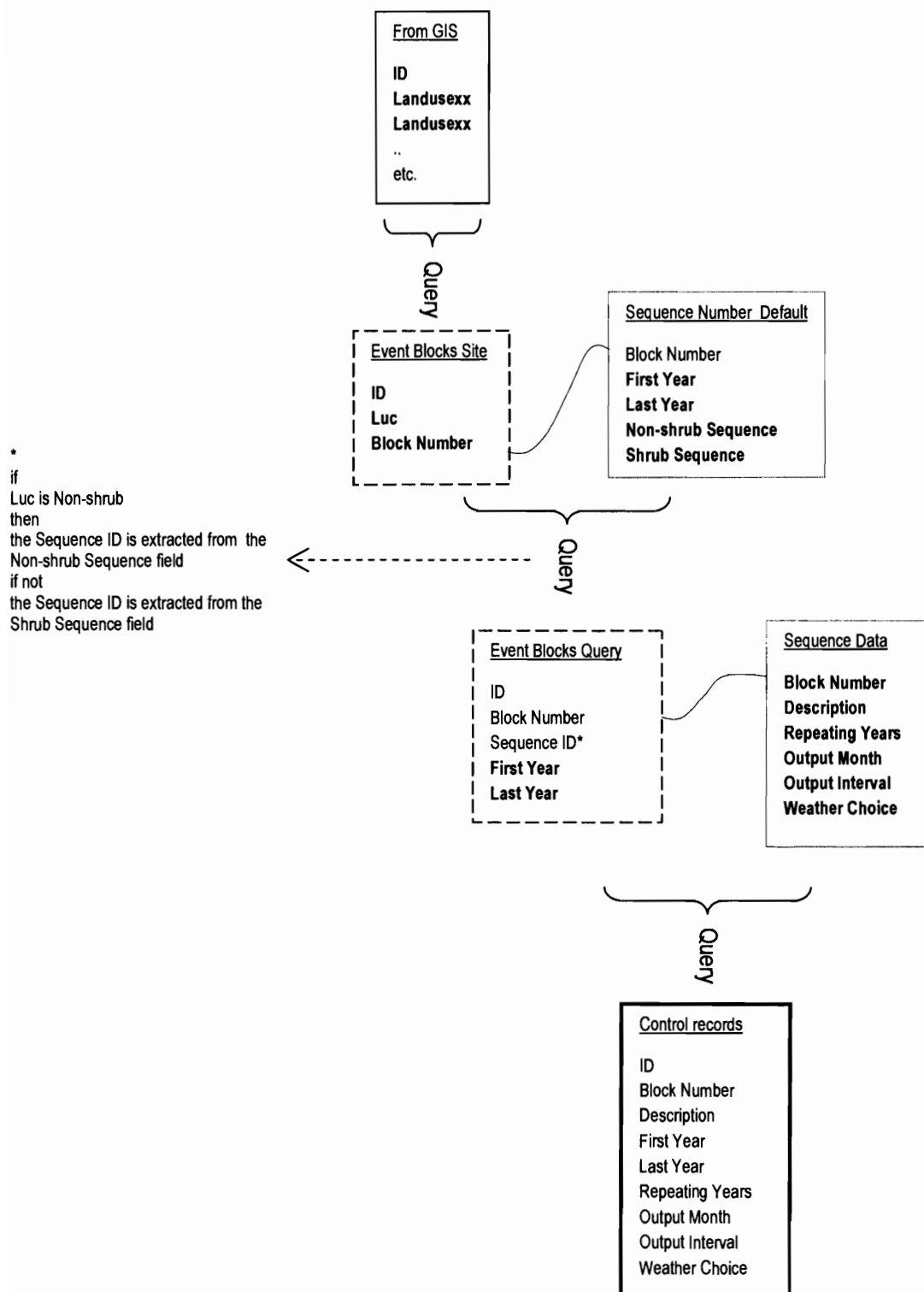


Figure A.5.5. Routine for creating the Event Blocks table.

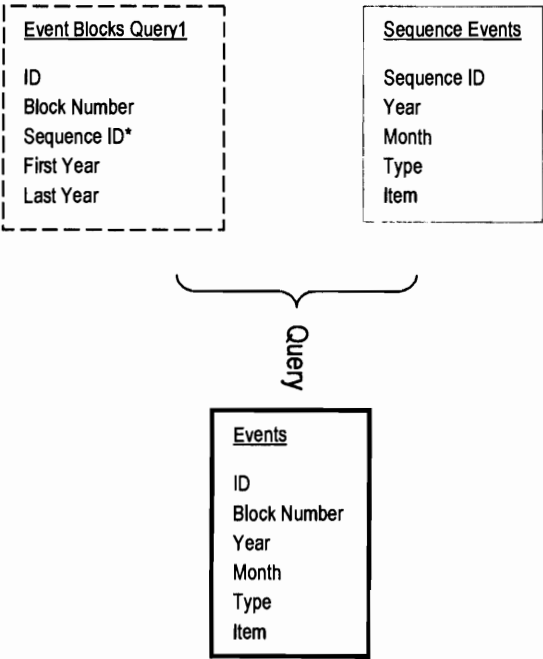


Figure A.5.6. Routine for creating the Events table.