The effects of management and canopy tree functional type on nutrient pools, retention, and loss from coffee agroforestry systems in Costa Rica

Katherine Lynn Tully
Danville, CA

Bachelor of Arts, Kenyon College, 2004
Master of Science, University of Virginia, 2007

A Dissertation presented to the Graduate Faculty
of the University of Virginia in Candidacy for the Degree of
Doctor of Philosophy

Department of Environmental Sciences

University of Virginia
May, 2011

Deborah Lawrence

Edmund Russell

Todd Scanlon

Herman H. Shugart

Abstract

This dissertation examines the effects of management techniques on nutrient dynamics in coffee farms in Costa Rica's Central Valley. I describe how fertilizer management and species alter the storage, movement, and release of nutrients. Coffee can be cultivated in monoculture or under the shade of trees (agroforest). While monocultures are always amended with mineral fertilizer, fertilizer is applied to agroforests in two forms: mineral and organic, with some farms receiving no fertilizer. A major finding of this study was the systematic difference in nutrient pools among management systems. Unfertilized agroforests had the largest aboveground nitrogen (N) and phosphorus (P) pools while fertilized agroforests had the largest soil nutrient pools. An approach based on nutrient balances indicated that mineral-fertilized agroforests are more likely to lose nutrients than organically-fertilized or unfertilized agroforests. Further, mineral-fertilized agroforests had higher concentrations of nitrate-N (but not phosphate-P) concentrations in leachate (at 15cm) than organically-fertilized agroforests, supporting the hypothesis that mineral-fertilized agroforests are more likely to lose N. Despite greater nutrient inputs in mineral- compared to organically-fertilized agroforests, calculated N losses were similar. However, N losses were three times higher in the coffee monoculture. Deep soil N pools (50-80cm) were larger in mineral- compared to organically-fertilized agroforests, but correspondingly large soil carbon (C) pools appear to be capable of immobilizing N. High concentrations of iron and aluminum oxides provide a mechanism for P retention. Finally, over a year, more N is released under a mixed-species canopy than a mono-specific canopy suggesting that increased canopy diversity may accelerate nutrient

release to the crop. Further, N and P release from *Erythrina* pruning residues may be enhanced during periods high nutrient demand if *Musa* is simultaneously pruned. In sum, this dissertation suggests that nutrient optimization may be best achieved by a management strategy in which coffee is cultivated under the shade of several shade species. As the price organic coffee farmers receive for their beans continues to drop, this research shows that by farming under the shade of trees and applying moderate amounts of inorganic fertilizer, small-scale farmers may be able to enhance profitability while maintaining a low environmental impact.

Table of Contents

List of Tables

Chapter 2

Appendix (Chapter 2)

Chapter 3

Chapter 4

Chapter 5

Chapter 6

Appendix (Chapter 6)

List of Figures

Chapter 1

Chapter 2

Chapter 3

Chapter 4

Chapter 5

Chapter 6

Acknowledgements

I have been fortunate to receive a great deal of financial, field, lab, and intellectual support over the five years it took to complete this work. First, I would like to thank my advisor, Deborah Lawrence, and my committee members: Ed Russell, Todd Scanlon, and Hank Shugart for their support and advice. I would also like to thank Manuel Lerdau for his aid in improving the nutrient balances.

I am grateful to the Jefferson Scholars Foundation and in particular Kenneth L. Bazzle who provided funding for five years of my graduate study at the University of Virginia. The Bankard Fund for Public Economy and the Graduate School of Arts and Sciences Dissertation Year Fellowship provided additional funding. The Jefferson Scholars Foundation, University of Virginia Double Hoo award, several Raven awards, CLAS grant, and the Department of Environmental Sciences supplied research funding.

I want to thank Laura Bonner and Amanda Schwantes for their tremendous and tireless support in the field and laboratory. There are also many people at CATIE (*Centro Agronómico Tropical de Investigación y Enseñaza*) who made this research possible, but none more than Gabriela Soto, who fed, housed, and encouraged me. Blanca Salguero-Lodoño, Mauricio Scheelje, and Romain Fievez provided field support while I was in the U.S., and Patricia Leandro aided with all laboratory analysis at CATIE. A large number of undergraduates have helped me in the lab over the past years, and to them I am indebted. Meg Miller helped with elemental analysis, as well as the analysis of the water samples. I would also like to thank my lab mates, Rishi Das and Karen Vandecar, for their moral support and for troubleshooting cranky machines.

Megan McGroddy deserves her own acknowledgements section for all of her conceptual, technical, and emotional support. I am so fortunate to have worked with this brilliant woman.

Aside from their boundless encouragement, my parents, Barbara and Frank Tully helped construct litterbags and lysimeters, visited me in the field, and proof read innumerable drafts of this dissertation. I would like to thank Ty Landrum, Katie Touchton-Leonard, Kiley Dunlap, Abi Bhattachan, Nicole Fitzwater, Katie Burke, and the yogis of Ashtanga Yoga Charlottesville for helping me maintain a semblance of sanity throughout this process.

Finally, I am ever grateful to the coffee farmers of Turrialba for letting me take over their farms (at times) and for graciously letting me into their homes and hearts (always). I would especially like to thank the families of Don Héctor Fuentes and Doña Maria Luisa Araya Fuentes. *Gracias a ustedes.*

Chapter 1: Introduction and summary of major findings

Abstract

This dissertation examines the effects of management techniques on nutrient dynamics in coffee farms in Costa Rica's Central Valley. I describe how fertilizer management and species alter the storage, movement, and release of nutrients. Coffee can be cultivated in monoculture or under the shade of trees (agroforest). While monocultures are always amended with mineral fertilizer, fertilizer is applied to agroforests in two forms: mineral and organic, with some farms receiving no fertilizer. A major finding of this study was the systematic difference in nutrient pools among management systems. Unfertilized agroforests had the largest aboveground nitrogen (N) and phosphorus (P) pools while fertilized agroforests had the largest soil nutrient pools. An approach based on nutrient balances indicated that mineral-fertilized agroforests are more likely to lose nutrients than organically-fertilized or unfertilized agroforests. Further, mineral-fertilized agroforests had higher concentrations of nitrate-N (but not phosphate-P) concentrations in leachate (at 15cm) than organically-fertilized agroforests, supporting the hypothesis that mineral-fertilized agroforests are more likely to lose N. Despite greater nutrient inputs in mineral- compared to organically-fertilized agroforests, calculated N losses were similar. However, N losses were three times higher in the coffee monoculture. Deep soil N pools (50-80cm) were larger in mineral- compared to organically-fertilized agroforests, but correspondingly large soil carbon (C) pools appear to be capable of immobilizing N. High concentrations of iron and aluminum oxides provide a mechanism for P retention. Finally, over a year, more N is released under a mixed-species canopy than a mono-

specific canopy suggesting that increased canopy diversity may accelerate nutrient release to the crop. Further, N and P release from *Erythrina* pruning residues may be enhanced during periods high nutrient demand if *Musa* is simultaneously pruned. In sum, this dissertation suggests that nutrient optimization may be best achieved by a management strategy in which coffee is cultivated under the shade of several shade species. As the price organic coffee farmers receive for their beans continues to drop, this research shows that by farming under the shade of trees and applying moderate amounts of inorganic fertilizer, small-scale farmers may be able to enhance profitability while maintaining a low environmental impact.

1.0 Agroforestry - Optimizing nutrient use in farming systems with a focus on coffee

Northern Latin America is home to seven of the ten countries with the highest deforestation rates in the world (Rice 1999). Once a vast tropical forest, only 43% of this original land cover remains (FAO 2008). Land has been and continues to be cleared for agriculture and pasture, especially the cultivation of coffee, banana, and sugarcane and the production of cattle (Myster 2004). My research will focus specifically on the biogeochemical consequences of coffee cultivation in Costa Rica.

Coffee (*Coffea arabica*) is native to the Horn of Africa, and was already widely cultivated by the 14th century (Topik 2000). The Dutch brought it to the New World in the 1700s where it thrived in the mid- and high-elevation mountainous regions of tropical America (Rice 1999). Currently, coffee is the top exported commodity in Central America by dollar value ($2.4 billion; FAO 2008), and its cultivation and production support over 8.5 million people in the region (Vedenov et al. 2007). About 2.7 million hectares of land are devoted to the cultivation of coffee across northern Latin America (FAO 2009), comprising the majority of the agricultural land cover between 500 and 2000 meters above sea level (Perfecto et al. 1996).

Coffee was traditionally cultivated as an understory shrub in an otherwise intact tropical forest (in other words, in an agroforest). In the 1950s attempts were made to "modernize" coffee farms by removing trees in hopes of improving yields. Coffee producers largely ignored these efforts since coffee exports continued to boom and demands steadily rose during the years immediately following World War II. However,

in the 1970s, a fungal disease, *la roya* (*Hemileia vastatrix*), devastated coffee crops across Latin America. The blight was exacerbated by the moister, shady conditions of agroforests, and as it swept across Central America and the Caribbean, efforts to control the disease became subsumed in a multi-national push to "technify" coffee farms (Rice 1999). Technified coffee farms differ tremendously from traditional coffee agroforests as overstory trees (30+ meters) are either removed entirely or replaced by regularly-spaced, short (5-8m) service trees. Service species such as *Erythrina poeppigiana,* have symbiotic associations with nitrogen (N)-fixing bacteria which provide a source of nutrients to the system. However, their wood is of an extremely low quality, and they offer little in the form of habitat for local species. Further, they are regularly pruned such that for the majority of year, the landscape essentially behaves as a monoculture. Although some agroforests remain, currently 80-85% of coffee in Central America is cultivated in monoculture or highly technified conditions (FAO 2008).

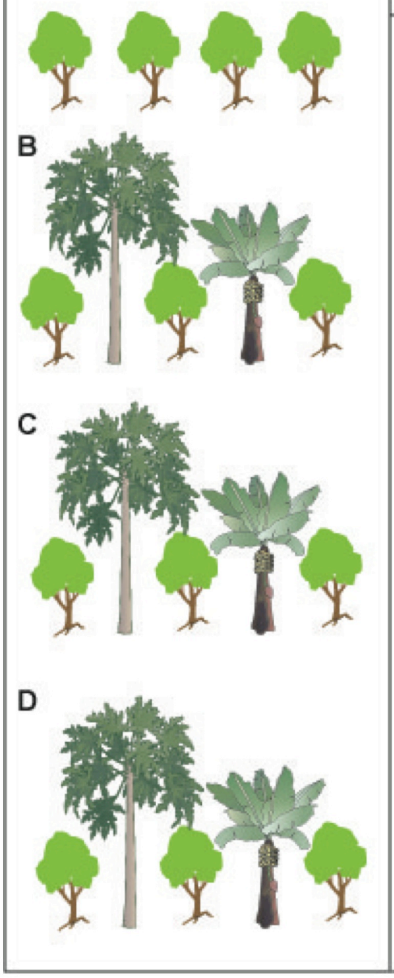
The cultivation of crops under the shade of trees is an ancient practice. A spectrum of systems exist- ranging from subsistence livestock and pastoral systems to shaded coffee, tea, and cacao plantations and homegardens. Over half of the farmed landscapes worldwide include some form of tree cover, however Central America has the highest density of trees on farms (Zomer et al. 2009).

In the past 40 years these agroforests have received much attention from land managers, international development agencies, and researchers for their potential to improve environmental and economic sustainability (Hassan et al. 2005; UNFCCC 2008). Agroforests store more carbon (Nair et al. 2009), recycle greater quantities of

nutrients (Palm 1995; Schroth et al. 2001), maintain higher levels of biodiversity (Perfecto et al. 1996), and have a greater potential to improve regional economic stability by diversifying livelihoods (Leakey et al. 2005; Amekawa et al. 2010) than industrialized agriculture. Nevertheless, in most research, the benefits of agroforests are assessed in comparison to regional extremes such as mature forests on the one hand and heavily fertilized monocultures on the other. Little research has investigated how management within the spectrum of agroforestry can alter ecosystem processes.

Specifically, I examine how nutrient dynamics are altered under different management systems. Farmers may influence nutrient cycling through the application of fertilizers, and by their selection of the number, identity and relative abundance of species present. In this research, I examine four different coffee management systems: (1) mineral-fertilized monocultures (MONO); (2) mineral-fertilized agroforests (MAF); (3) organically-fertilized agroforests (OAF), and (4) unfertilized agroforests (ZAF, for 'zero' fertilizer; Figure 1). In unshaded monocultures, coffee is grown in the absence of trees and amended with large quantities of mineral fertilizers (in excess of 200 kg N ha^{-1} yr^{-1}; Harmand et al. 2007). In this study, mineral-fertilized agroforests receive moderate quantities of mineral fertilizer (between 50-120 kg N ha^{-1} yr^{-1}). However, in efforts to compensate for the lower yields in shaded systems, it is not uncommon for farmers in this region to amend their shade-grown coffee with 300kg N ha^{-1} yr^{-1} (Harmand et al. 2007). Further, shade trees are pruned frequently (2-3 times a year) and aggressively (nearly all branches removed) in mineral-fertilized agroforests. Synthetic fertilizers and pesticides are excluded in organically-fertilized agroforests, and manure and compost are applied to

maintain crop productivity. The nutrient concentration of these substrates is low compared to mineral fertilizers (Chapter 3, Table 1), therefore total N inputs are typically less than 85 kg ha^{-1} yr^{-1}. In addition, pruning is less frequent and less intense (fewer branches removed) in organically-fertilized agroforests leading to smaller quantities of recycled nutrients transferred to the soil, and also less sun exposure for the coffee plants. In 2008, fertilizer prices rose to a historic high for both organic and mineral forms, and many farmers were forced to reduce their annual amendments. Still others found themselves unable to afford fertilizer at all, and relied solely on N fixed by leguminous shade trees and internal nutrient cycling to nourish the crop (referred to here as 'unfertilized agroforests'). Surprisingly, these farms had the lowest pruning frequency among agroforests averaging once per year (typically in January or February). Clearly, different farm management strategies will have consequences for the coupled cycling of nutrients, water, and energy. My dissertation research is the first to investigate how nutrient dynamics are altered under these different forms of management.



Management System	Fert N kg/ha/yr	Fert P kg/ha/yr	Prunes per year
Mineral Fertilized Monoculture	200-300	30-60	0
Mineral Fertilized Agroforest	50-120	8-30	2-3
Organically Fertilized Agroforest	2-85	0.2-30	2
Unfertilized Agroforest	0	0	1-2

Figure 1: Characteristics of four coffee management systems examined in Costa Rica. Average fertilizer N and P and typical pruning frequency in (a) mineral-fertilized monocultures, (b) mineral-fertilized agroforests, (c) organically-fertilized agroforests, and (d) unfertilized agroforests.

2.0 Research Objectives

As the tropical landscape matrix continues to shift, it is important to identify strategies that optimize nutrient use in agroecosystems. In an effort to identify some of these strategies, my research asks the following questions:

How does management affect nutrient pools, fluxes, and balances? (Chapter 2)

How does management affect nutrient loss? (Chapters 3 and 4)

How does management affect soil nutrient retention? (Chapter 5)

How does management affect patterns of nutrient release? (Chapter 6)

3.0 Synthesis of Results

In **Chapter 2** I examine N and P balances in 21 coffee agroforests across the Central Valley of Costa Rica. I identified agroforests receiving (a) mineral (n=9), (b) organic (n=7), and (c) no fertilizer (n=5), in which *Coffea* is grown under a combination of *Erythrina poeppigiana*, *Musa acuminata,* and *Cordia alliodora*. In this chapter, I address two main questions:

1. *Does management affect nutrient pools?*
2. *Does management affect nutrient fluxes?*

To estimate the size of aboveground biomass and soil nutrient pools, I collected soils, live, and senesced leaf tissues for nutrient analysis. I used allometric equations to calculate nutrient pools in aboveground biomass and bulk density to calculate the size of soil nutrient pools. I interviewed farmers to capture the specific management practices employed on their farm. Specifically, I gathered information on (a) fertilizer inputs and

(b) harvest outputs. I used a series of models to estimate other nutrient fluxes including N-fixation, N deposition, and gaseous N losses. By subtracting N and P in outputs (e.g. harvest and gas losses) from nutrient inputs (e.g. harvest, N-fixation, and deposition), I estimated nutrient "excess" or potential leaching losses. *I found management affected both nutrient pools and nutrient fluxes.* Unfertilized agroforests had larger aboveground N and P pools compared to mineral- and organically-fertilized agroforests. On the other hand, fertilized farms had larger soil N and P pools than unfertilized farms. Mineral-fertilized agroforests had more excess N and P, suggesting that they have a greater potential to lose nutrients than either organically-fertilized or unfertilized agroforests.

In **Chapter 3,** I design a leaching experiment to empirically measure N and P concentrations in soil water in coffee farms (MAF=4; OAF=4; MONO=1) in order to test the hypothesis that agroforests amended with mineral fertilizer were more prone to nutrient loss than agroforests amended with organic fertilizer. In this chapter, I address the question; *does management affect potential nutrient loss?* I used gravity and tension lysimeters to collect soil water at 15cm and 100cm below the soil surface, and measured nitrate, phosphate, and ammonium concentrations in soil water. *I found management affected potential nutrient loss.* Surface (at 15cm) nitrate concentrations were significantly higher in mineral-compared to organically-fertilized agroforests, with the highest surface nitrate concentrations in the monocultures. This lent further support to the hypothesis that agroforests amended with mineral fertilizers would have higher N losses than agroforests amended with organic fertilizers. However, nitrate concentrations at depth (100cm) were not significantly different among agroforests. Nitrate

concentrations in soil water (at depth) collected in monocultures were three times higher than concentrations in agroforests, suggesting that perhaps the most important mechanism preventing N loss is the presence of trees. P concentrations in soil water were not significantly different among management systems at either 15cm or 100cm.

In **Chapter 4**, I develop a water balance to estimate the quantity of water drainage in each coffee farm instrumented with lysimeters (MAF=4; OAF=4; MONO=1). In this chapter I address two questions,

1. *Does management affect actual nutrient loss?*
2. *What mechanisms prevent nutrient loss?*

I couple this water balance with the nutrient concentrations measured in **Chapter 3** to calculate annual N and P losses from each management system. *I found management affected actual nutrient loss.* I show that annual N and P losses (at both 15cm and 100cm) are similar between mineral and organically fertilized agroforests, but that N and P loss is enhanced in the absence of trees. *I also identified mechanisms that may prevent nutrient loss.* I show that N losses decline with increasing aboveground biomass, while P losses do not. Rather, P losses decline with increasing soil iron pools as this pool provides binding sites for dissolved P. The size of the biomass pool may be a useful metric for estimating on-farm N loss as it is easily estimated from simple field surveys.

In **Chapter 5,** I examine the effect of fertilizer type on soil characteristics in agroforests to determine whether inherent soil properties or management confer nutrient retention capacity. In this chapter, I address two questions,

1. Does management affect soil pools?

2. *How are nutrients retained?*

Three soil pits (to 80cm) were dug in each agroforest (MAF=4; OAF=4; total of 24 pits). I measured soil bulk density, and analyzed soils for total N, C, and P concentrations. I also measured cation exchange capacity (CEC), and fractions of the aluminum (Al) and iron (Fe) oxide pools. I used bulk density to calculate the size of the soil pools. Surface (0-15cm), mid (15-50cm), and deep (50-80cm) soil pools were examined. *I found management affects soil pools*, but only deep soil pools. Soil N pools were significantly larger in agroforests amended with mineral fertilizer compared to agroforests amended with organic fertilizer likely due to their larger N amendments and subsequent transport of N to deeper soil layers. *I also identified mechanisms for nutrient retention.* Given large N inputs and low N losses at 100cm, it appears that pruning of shade trees is critical in mineral-fertilized agroforests. Frequent, heavy pruning helps maintain soil C pools, which in turn, are capable of immobilizing nutrients, especially N. Even at depth, where soil N pools are significantly larger in mineral fertilized soils, leaching is buffered by a large C pool. The C:N ratio is not significantly different in organically and mineral fertilized agroforests, but the total pool sizes are. High soil Al and Fe oxide concentrations and high bulk density at depth provide more binding sites for both N and P, further enhancing nutrient retention. Trees not only prevent against nutrient loss by actively assimilating and recycling nutrients (direct effect), they also help maintain certain chemical and physical properties of the soil which enhance the capacity of the system to retain nutrients (indirect effect).

In **Chapter 6,** I examine how management affects patterns of nutrient release from decomposing leaves. The transfer of nutrients in litterfall and pruning residues can exceed those added even via mineral fertilizers. Decomposition rates of these materials can constrain the release of nutrients to the crop, which in turn are limited by environmental variables and substrate quality. Therefore, in this chapter I addressed two questions,

1. *Does canopy composition affect nutrient release? (environmental variable)*
2. *Does leaf composition affect nutrient release? (substrate quality)*

To address the first question, I tracked decomposition rates and patterns of nutrient release in leaves under two different canopy types: a mono-specific canopy and mixed-species shade canopy. To address the second question, I examined decomposition rates and patterns of nutrient release in leaves in leaf tissues representative of those found in pruning residues throughout the Central Valley of Costa Rica. *I found both canopy and leaf composition affected nutrient release.* A larger proportion of tissue N is released under the mixed canopy after a year, but canopy composition did not significantly affect patterns of P release. Over the course of the growing season, maintaining a diverse canopy may provide greater nutritional benefits to the crop than could be conferred by a single-species canopy. Further, I found that N and P release is facilitated in when leaf mixtures include *Musa*. While *Erythrina* is systematically pruned in January-February and June-July *Erythrina*, there is no regular *Musa* pruning schedule. This research shows that N and P release may be facilitated during periods of enhanced nutrient demand (e.g. fruit set and maturation if *Musa* is simultaneously pruned.

4.0 Conclusions

This dissertation shows that despite a wide range in the form and quantity of nutrient inputs, trees are capable of minimizing nutrient losses. Annual nutrient losses are similar among mineral- and organically-fertilized agroforests and significantly lower than those observed under sun-grown monocultures. Furthermore, N losses decline linearly with increasing shade tree biomass, further supporting the claim that trees minimize nutrient loss. In heavily-fertilized agroforests, intensive pruning of shade trees allows for a build up of soil organic matter that effectively buffers nitrogen losses. Thus, the incorporation of trees on farms is the best way to mitigate the negative effects of heavy fertilizer application. Further, this research shows that farming organically does not enhance the capacity of the agroforest system to retain nutrients.

The price premiums for organic coffee continue to drop, and farmers are forced to consider other management strategies and crops. Therefore, what management strategy offers the greatest potential to optimize nutrient use and sustain farmer livelihoods? Plant trees, and plant several species of them. This strategy would be effective in a shifting economic climate where organic coffee prices continue to drop and these farmers are being quickly shouldered out of the market. By farming under the shade of trees and applying moderate amounts of mineral fertilizer, small-scale farmers can improve profitability while maintaining a low environmental impact.

5.0 Glossary

Mineral-fertilized agroforest (MAF). A management system where coffee is cultivated under the shade of overstory trees and amended with mineral fertilizers.

Mineral-fertilized monoculture (MONO). A management system where coffee is cultivated in the absence of trees and amended with mineral fertilizers.

Nutrient Excess. Defined as nutrient inputs (e.g. as fertilizer or fixed N) minus nutrient outputs (e.g. as harvest or gaseous N). Units are in kg of N or P ha^{-1} yr^{-1}. See equations 2 and 3 in Chapter 2.

Nutrient Recovery. The difference between outputs on fertilized versus unfertilized farms divided by the difference between inputs on fertilized versus unfertilized farms. Gaseous losses are not included in the estimation of N recovery. See equations 4 and 5 in Chapter 2.

Organically-fertilized agroforest (OAF). A management system where coffee is cultivated under the shade of overstory trees and amended with organic fertilizers such as manure, compost, bocashi, and broza.

Pruning frequency. The number of times a year shade trees are pruned.

Pruning intensity. Relates to the number of branches removed from shade trees per pruning. Selective pruning refers to the removal of all but 4 or 5 branches while total pruning refers to the removal of all branches.

Sustainability. In ecology, it is the capacity of a system to maintain the diversity of functional groups, productivity, and rates of biogeochemical cycling over time and within a normal disturbance regime. In this research, I focus specifically on the biogeochemical aspect of sustainability. That is, the capacity of ecosystems to retain and recycle nutrients and prevent against nutrient loss.

Sustainable agriculture. An agroecosystem that integrates environmental health, economic profitability, and social and economic equity.

Technification (technified). The process by which traditional coffee agroforests became modernized (mostly during the 1970s). Included the removal of native overstory trees and increased inputs of fertilizers. Coffee rows are spaced a little father apart than in the agroforest setting, but coffee plants in each row are much closer together. Trees (if included at all) are typically planted at a low density (150 trees ha^{-1}) in comparison to high density agroforests (up to 1000 trees ha^{-1}) and aggressively pruned.

Unfertilized agroforest (ZAF, for ‘zero fertilizer’ agroforests). A management system where coffee is cultivated under the shade of overstory trees and to which no amendments (mineral or organic) are made.

References

Amekawa, Y., H. Sseguya, S. Onzere, and I. Carranza. 2010. Delineating the multifunctional role of agroecological practices: Toward sustainable livelihoods for smallholder farmers in developing countries. Journal of Sustainable Agriculture 34:202-228.

Food and Agriculture Organization of the United Nations (FAO). 2008. FAO Statistical Databases < http://www.fao.org/corp/statistics/en/>

Food and Agriculture Organization of the United Nations (FAO). 2009. FAO Statistical Databases < http://www.fao.org/corp/statistics/en/>

Harmand, J. M., H. Avila, E. Dambrine, U. Skiba, S. de Miguel, R. V. Renderos, R. Oliver, F. Jimenez, and J. Beer. 2007. Nitrogen dynamics and soil nitrate retention in a Coffea arabica-Eucalyptus deglupta agroforestry system in Southern Costa Rica. Biogeochemistry 85:125-139.

Hassan, R., R. Scholes, and N. Ash. 2005. Ecosystems and human well-being: current state and trends, Volume 1. Findings of the Conditions and Trends Working Group of the Millennium Ecosystem Assessment Island Press: Washington.

Leakey, R. R. B., Z. Tchoundjeu, K. Schreckenberg, S. E. Shackleton, and C. M. Shackleton. 2005. Agroforestry tree products (AFTPs): targeting poverty reduction and enhanced livelihoods. International Journal of Agricultural Sustainability 3:1-23.
Myster, R. W. 2004. Post-agricultural invasion, establishment, and growth of neotropical trees. The Botanical Review 70:381-402.

Nair, P. K., V. D. Nair, B. M. Kumar, and S. G. Haile. 2009. Soil carbon sequestration in tropical agroforestry systems: a feasibility appraisal. Environmental Science & Policy 12:1099-1111.

Palm, C. A. 1995. Contribution of agroforestry trees to nutrient requirements of intercropped plants. Agroforestry Systems 30:105-124.

Perfecto, I., R. A. Rice, R. Greenberg, and M. E. VanderVoort. 1996. Shade coffee: A disappearing refuge for biodiversity. BIOSCIENCE 46:598-608.

Rice, R. A. 1999. A place unbecoming: the coffee farm of northern Latin America. Geographical Review 89:554-579.

Schroth, G., E. Salazar, and J. P. Da Silva. 2001. Soil nitrogen mineralization under tree crops and a legume cover crop in multi-strata agroforestry in Central Amazonia: Spatial and temporal patterns. Experimental Agriculture 37:253-367.

Topik, S. C.2000. III.4. Coffee. In K. F. Kiple, and K. C. Ornelas., Eds. The Cambridge world history of food, New York, NY.

UNFCCC (United Nations Framework Convention on Climate Change). 2008. Challenges and opportunities for mitigation in the agricultural sector. Technical Paper, UNFCCC, Bonn, DE. (Document FCCC/TP/2008/8).

Vedenov, D., J. Houston, and G. Cardenas. 2007. Production Efficiency and Diversification in Mexican Coffee-Producing Districts. Journal of Agricultural and Applied Economics 39:749.

Zomer, R. J., A. Trabucco, and R. Coe. 2009. Trees on farm: analysis of global extent and geographical patterns of agroforestry. ICRAF Working Paper-World Agroforestry Centre

Chapter 2: Closing the loop – The effect of management systems on nutrient balances in coffee agroforests

Abstract

Agroforests are a primary example of ecologically sustainable agroecosystems due to their efficient use of natural resources and ability to buffer against ecological and socio-economic stresses. I constructed nitrogen (N) and phosphorus (P) balances to examine the ecological sustainability of mineral-fertilized, organically-fertilized and unfertilized coffee agroforests. A similar percent of applied nutrients were recovered in mineral- and organically-fertilized coffee yields. However, nutrient excess (inputs minus outputs) was higher in mineral-fertilized agroforests, suggesting they may be more prone to nutrient loss. Nutrient pools were large in these agroforests, and unfertilized farms tended to store nutrients aboveground, and fertilized farms belowground. Future research should investigate the fate of excess nutrients to develop specific strategies promoting nutrient optimization in agroforests.

1.0 Introduction

Agroforestry is a land-use practice in which a subsistence or cash crop is grown under the canopy of over-story trees, and in the 1970's it gained recognition as a sustainable form of land management that combined the best aspects of both forestry and agriculture (Bene et al. 1977; Steppler and Nair 1987). Agroforests are complex systems with interacting components in which ecological structures and functions are strongly modified by management. Controlled field experiments have described how certain components (e.g. shade tree species, size, or density) can alter nutrient cycling (Glover and Beer 1986; Beer 1988; Harmand et al. 2007). However, in working farms, many components are simultaneously manipulated since management techniques respond to rising and falling prices of fertilizer, labor, equipment, and gasoline (Sick 1997). Controlled field experiments examine the direct effects of a few variables at a given time. On-farm studies examine the effects of the same variables, but within the socio-economic context of an active farm. By including the inherent variability among and within coffee agroforests, I capture the nitrogen (N) and phosphorus (P) trade-offs of farm management regimes. In this study, I examine how specific management practices affect on-farm biogeochemical sustainability by altering nutrient pools and fluxes.

Coffee (*Coffea arabica*) was traditionally grown in an agroforestry setting. However, in efforts to maximize yields, it is now often cultivated in monoculture and amended with large quantities of N fertilizer (often exceeding 200 kg N ha^{-1} yr^{-1}; FAO 1997; Harmand et al. 2007). Recently, these high fertilization rates have been linked to nitrate loading in aquifers underlying coffee-producing regions such as Costa Rica's

Central Valley (Reynolds-Vargas and Richter 1995; Reynolds-Vargas et al. 2006). However, coffee agroforests persist, and despite reduced yields, they provide a much more ecologically sustainable alternative to unshaded monoculture plantations. For example, in contrast to unshaded farms where nutrients located beyond the coffee root-zone are essentially lost to the system, the incorporation of shade trees in an agroforest mediates nutrient loss as tree roots can access nutrients stored beyond the reach of coffee plants (Berendse 1979; Berendse 1982; Fassbender 1987; Chapin et al. 1996; Ataroff and Monasterio 1997). Several studies have shown N loss to be significantly reduced in coffee agroforests compared to unshaded monocultures (Babbar and Zak 1995; Udawatta et al. 2002; Harmand et al. 2007). This research examines nutrient balances in agroforests under different management regimes in order to determine if nutrient optimization can be further enhanced in these systems.

Nutrient inputs may vary both in their quantity and form. For example, mineral fertilizers are applied in ratios (N:P:K) that mimic plant requirements, and are therefore readily assimilated by the crop. On the other hand, organic fertilizers, such as compost, manure, or worm castings (vermicompost) must first undergo microbial transformations before they are available for plant uptake. The inclusion of leguminous shade trees such as *Erythrina poeppigiana* can provide an additional source of N as atmospheric N_2 is fixed into a bioavailable form. Although not an (external) input, the periodic pruning of shade trees transfers nutrients stored in shade tree biomass to the mineral soil (Aranguren et al. 1982; Glover and Beer 1986). Similar to organic fertilizers, nutrients stored in this pool must first be broken down into an inorganic form before they can be utilized by the

crop. Nutrients are exported (outputs) primarily in the harvest, where as much as eight metric tons of coffee berries are removed per hectare per year (Lyngbæk et al. 2001). However, yields vary greatly from farm to farm. In this study, I examine differences in nutrient inputs and outputs among agroforests to determine patterns among farms under similar management regimes.

The diversification of management regimes can be linked, at least in part, to the coffee crisis of the late 1990s, when prices plunged to a historic low. In response, niche markets for specialty coffees began to emerge (Goodman 1999). Farmers who are Certified Organic®, for example, receive a higher price premium per unit than their non-organic neighbors. Such incentives are meant to financially promote sustainable practices, however to my knowledge, no scientific evidence indicates organically-fertilized coffee agroforests are more sustainable, from a nutrient cycling perspective, than mineral-fertilized coffee agroforests. In fact, due to the global rise in the cost of fertilizers, some Certified Organic farmers have reduced or ceased fertilizer application altogether, suggesting these farms are neither financially nor ecologically sustainable. The sustained export of nutrients from these farms in coffee yields without any additional nutrient inputs can ultimately deplete long-term soil fertility (Stoorvogel and Smaling 1990). Recent economic forcings provided us with a unique opportunity to compare the biogeochemical advantages and disadvantages of mineral and organic fertilizer management to "baseline" or unfertilized conditions.

Nutrients are mobile, and may also be retained in plant biomass and soil pools. Therefore, I was not only interested in examining how nutrients moved in and out of

these systems, but also the quantity and distribution of nutrient reserves. For example, agroforests (in their broadest definition) play a central role in the global C cycle and may contain roughly 12% of the total terrestrial carbon (Smith et al. 1993; Dixon et al. 1994; Dixon 1995; Nair et al. 2009). Furthermore, soil C tends to be maintained and is often enhanced under organic fertilizer management in both temperate (Reganold et al. 1988; Su et al. 2006; Fließbach et al. 2007) and tropical systems (Goyal et al. 1999; Manna et al. 2005). Therefore, this research also aimed to examine biogeochemical sustainability in terms of the effect of management on nutrient pools, both above and below ground.

2.0 Materials and Methods

2.1 Site Selection and Characteristics

This study was conducted at four locations near Turrialba, Costa Rica (9°53'N 83°40'W; Figure 1) between June 22 and July 12 (rainy season) of 2006. Soils in this region are characterized as Andisols, Ultisols and Inceptisols (Table 1). Farms were located between 9°86'N and 9°96'N (from Grano de Oro to Tres Equis) and between 83°46'W and 83°71'W (from Grano de Oro to San Juan Norte). The altitude of sites ranged from 728 to 1100 masl (from Tres Equis to Grano de Oro). The year prior to sampling (July 2005-June 2006) received 2945mm of rainfall and had a mean annual temperature of 22.0°C. Before initial field sampling, farmers were extensively interviewed to acquire information regarding annual fertilizer inputs, yields, and pruning regimes (Table 2). Twenty-one farms were selected for field sampling. Fertilizers were applied to agroforests in two forms: mineral (9 farms) and organic (7 farms), and others

received no fertilizer at all (5 farms). I may refer to agroforests receiving mineral fertilizers as MAF, organic fertilizers as OAF, and no/zero fertilizer as ZAF.

Table 1: Site characteristics of four sampling locations in Central Valley of Costa Rica. Soil samples were collected at 0-15cm. Values represent means across all farms in that location and parenthetical values represent standard error of the mean.

	Colorado	Grano de Oro	San Juan Norte	Tres Equis
Latitude	9.914	9.86	9.889	9.96
Longitude	-83.7	-83.46	-83.71	-83.57
Altitude (masl)	858	1100	989	728
Silt (%)	41 (1.1)	48 (1.3)	44 (1.5)	42 (1.2)
Clay (%)	32 (3.0)	35 (3.1)	28 (2.0)	36 (1.9)
Sand (%)	27 (2.8)	17 (2.6)	27 (0.7)	22 (2.1)
SOM (%)	17 (0.7)	15 (1.0)	25 (1.1)	18 (1.4)
Bulk Density (g cm^{-3})	0.7	0.7	0.9	0.8
Soil C (Mg ha^{-1})	37.7 (4.7)	36.9 (2.1)	84.0 (6.6)	53.6 (9.7)
Soil N (Mg ha^{-1})	3.1 (0.4)	3.1 (0.1)	5.5 (0.4)	4.5 (0.8)
Soil P (Mg ha^{-1})	1.4 (0.2)	1.3 (0.1)	1.6 (0.2)	1.4 (0.2)

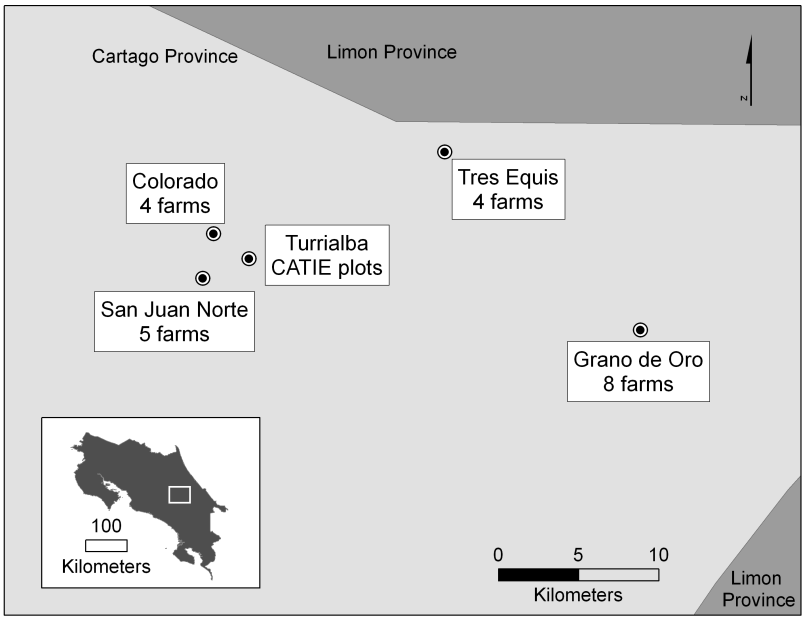


Figure 1: Map of locations in Central Valley of Costa Rica. Map of the locations of the 21 farms sampled between June 22 and July 12, 2006. All farms are located between 728 and 1100masl. Nine farms are mineral-fertilized agroforests, 7 are organically-fertilized agroforests, and 5 are unfertilized agroforests.

Agroforests ranged in age from 4 to 50 years old, and previous land uses consisted of agriculture (52.3%), pasture (9.5%), secondary forests (28.6%), and mature forest (9.5%; Table 2). For the purposes of statistical analysis, I grouped previously grazed and cultivated farms into one category (called agriculture), and farms previously in secondary and mature forest into another category (called forest). All field data (with the exception of bulk density) were collected in a single visit to the farm, and in most cases several neighboring farms were sampled the same day.

In the Central Valley of Costa Rica, the most common shade trees grown in agroforests are *Erythrina poeppigiana* (a nitrogen fixing species), *Musa spp*. (banana and plantain, an herbaceous plant), and *Cordia alliodora* (laurel, a timber tree), as well as a variety of fruit trees (Albertin and Nair 2004). I sampled the following canopy covers: *Erythrina* (3 farms); *Musa* (2 farms); *Erythrina-Musa* (12 farms); and *Erythrina-Cordia* (4 farms; Table 2).

Table 2: Farm characteristics and nutrient fluxes from coffee agroforests in Costa Rica. Farm characteristics, nutrient inputs, and nutrient outputs on each of the 21 agroforests represented in the study. MAF refers to mineral-fertilized agroforests, OAF to organically-fertilized agroforests, and ZAF to unfertilized agroforests.

Farm Characteristics								Inputs (kg/ha/yr)			Outputs (kg/ha/yr)			Excess (kg/ha/yr)	
Mgmt	Shade Species	Location	Prior land use	Age (yrs)	Area (ha)	Coffee Plants/ ha	Shade Trees/h a	N-fix	Fert N	Fert P	Coffee Yield	Harvest N	Harvest P	N	P
MAF	*Erythrina* + *Musa*	Colorado	pasture	10	4	5700	900	90	19	5	1260	25	3	78	2
MAF	*Erythrina* + *Musa*	Grano de Oro	secondary forest	25	4.5	6200	2260	70	142	39	2520	50	6	131	33
MAF	*Erythrina* + *Musa*	Grano de Oro	secondary forest	6	4	5700	570	70	104	29	2025	41	5	110	24
MAF	*Erythrina* + *Musa*	Grano de Oro	secondary forest	20	3	4200	1015	70	104	29	2025	41	5	110	24
MAF	*Erythrina*	San Juan Norte	sugar cane	10	2	6000	130	70	223	62	2753	55	7	191	55
MAF	*Erythrina* + *Musa*	San Juan Norte	pasture	30	2	5000	1200	70	107	12	2700	54	7	100	55
MAF	*Erythrina* + *Musa*	San Juan Norte	pasture	6	2	5200	830	70	48	13	2250	45	6	62	5
MAF	*Erythrina* + *Cordia*	Tres Equis	agriculture	14	0.5	6000	390	90	166	46	2700	54	7	166	7
MAF	*Erythrina* + *Cordia*	Tres Equis	sugar cane	4	4.2	7000	210	70	83	23	2665	53	7	81	39
OAF	*Erythrina*	Colorado	mature forest	15	0.7	6000	575	90	18	7	1547	31	4	72	3
OAF	*Erythrina* + *Musa*	Colorado	mature forest	15	0.7	6500	1432	90	18	7	768	15	2	87	5
OAF	*Erythrina* + *Musa*	Colorado	mature forest	40	4.6	6500	730	90	89	10	2000	34	5	125	5
OAF	*Erythrina* + *Musa*	San Juan Norte	agriculture	50	1	6200	1820	70	14	6	900	15	2	64	4

Table 2: continued

	Farm Characteristics							Inputs (kg/ha/yr)			Outputs (kg/ha/yr)			Excess (kg/ha/yr)	
Mgmt	Shade Species	Location	Prior land use	Age (yrs)	Area (ha)	Coffee Plants/ ha	Shade Trees/h a	N-fix	Fert N	Fert P	Coffee Yield	Harvest N	Harvest P	N	P
OAF	*Erythrina*	San Juan Norte	agriculture	50	1	6500	390	70	14	6	900	15	2	64	4
OAF	*Erythrina* + *Cordia*	Tres Equis	secondary forest	6	2	6300	600	70	57	12	2000	40	5	74	7
OAF	*Erythrina* + *Cordia*	Tres Equis	mature forest	20	2	3280	180	90	24	11	2000	40	5	68	6
ZAF	*Musa*	Grano de Oro	secondary forest	6	5	6000	935	70	0	0	720	14	2	68	-2
ZAF	*Musa*	Grano de Oro	secondary forest	20	2	4300	1500	90	0	0	1200	24	3	64	-3
ZAF	*Erythrina* + *Musa*	Grano de Oro	secondary forest	15	1	3800	550	93	0	0	1200	24	3	67	-3
ZAF	*Erythrina* + *Musa*	Grano de Oro	secondary forest	20	2	5600	1500	90	0	0	1200	24	3	64	-3
ZAF	*Erythrina* + *Musa*	Grano de Oro	secondary forest	20	2	5600	1040	90	0	0	540	11	1	77	-1

2.2 Vegetation Density

Within each farm, I established a 16m by 24m plot (Figure 2). I measured diameter at breast height (dbh) for all shade trees in the plot. I also measured mean coffee tree height and density. Shade tree density was calculated by scaling up plot totals to a hectare. In the case of banana plants, only pseudostems ≥10cm dbh were counted. Bananas produce clonally, and these small offshoots were not tabulated for this study.

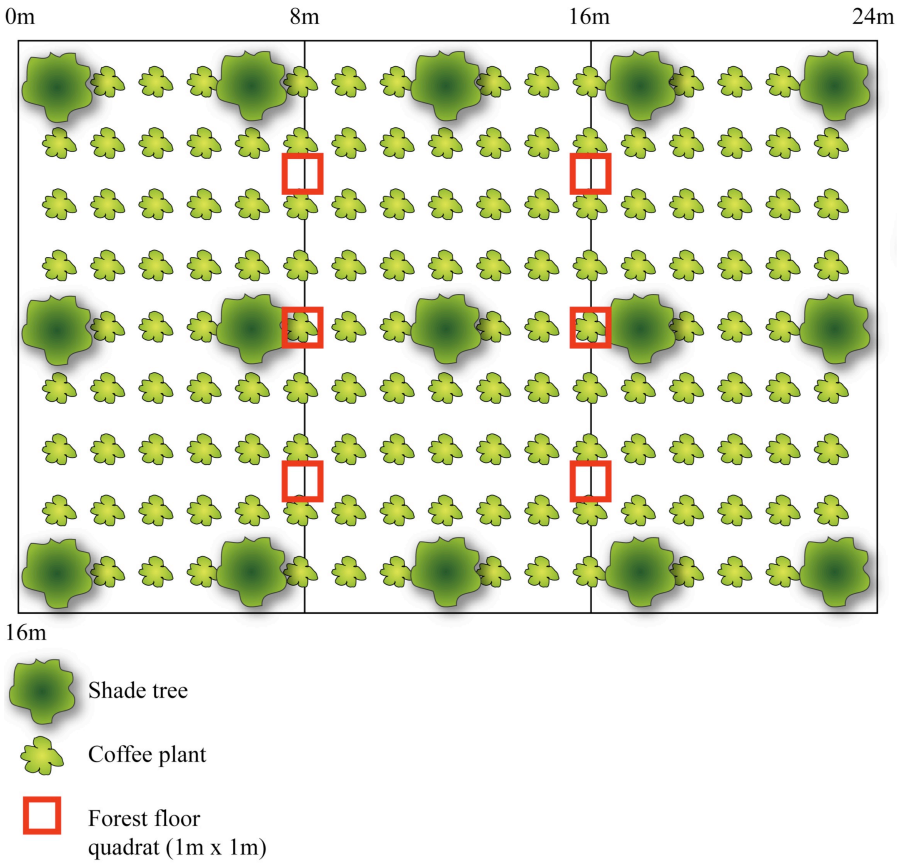


Figure 2: Field sampling diagram for 21 coffee farms in Central Valley of Costa Rica. Field plots of 24m by 16m were established with two transects running widthwise at 8m and 16m. Three $1m^2$ forest floor samples were taken every 3m on each transect. Eight soil samples (to 15cm depth) were taken across each transect (4 in plant rows and 4 in the interrows).

2.3 Sampling and Nutrient Analysis of Green Leaves and Forest Floor Litter

I established two transects perpendicular to the length of the plot at 8m and 16m (Figure 2). All forest floor litter (leaves, reproductive parts, and stems <8mm in diameter) was collected within a $1m^2$ area every three meters along the transect (total of 6 samples per plot) and weighed in the field. A sub-sample (approximately 50g) of the composite litter from each transect was collected and reserved for nutrient analysis. This sub-sample was dried in an air-conditioned room for five to six days and then re-weighed to obtain a rough wet-to-dry conversion. I collected mature sun leaves from *Coffea* plants along each transect as well from each of the shade trees in the plot. In some cases, due to recent pruning, no mature shade tree leaves were available for collection. Green leaves were pressed and air-dried, and leaves and forest floor litter were transported to the University of Virginia for nutrient analysis.

Sub-samples from the green leaves and forest floor litter samples were dried to a constant mass (<65°C) and then ground in a 20-mesh Wiley mill (Thomas Scientific, Inc., Swedesboro, NJ, USA). Ground material was digested using a modified Kjeldhal protocol and the resulting solutions analyzed for total P on an Alpkem Flow Solution IV Autoanalyzer (OI Analytical; College Station, Texas, USA). Total N and C were measured in leaf and litter samples that were dried and ground by dry combustion on an elemental analyzer (Carlo Erba, Model NA 2500; Milan, Italy).

2.4 Soil Sampling and Nutrient Analysis

Eight soil samples (to 15cm depth) were taken across each transect (4 in plant rows and 4 in the interrows). Following collection, soil samples were homogenized, air-

dried, passed through a 2mm sieve, and transported to the University of Virginia for analysis. Sub-samples of the processed soils were digested using a modified Kjeldhal protocol, and resulting solutions analyzed for total P on an Alpkem. Total soil C and N were determined by dry combustion, as above, on soils that had been dried and ground (<149μm). Soil texture analysis was performed by Brookside Laboratories, Inc. (New Knoxville, OH). At a later date (in June 2008), bulk density (0-15cm) measurements were taken in replicates of three per farm with a steel cylinder.

2.5 Calculating Nutrient Fluxes

Information of the type, dose, and yearly application schedule of fertilizers was acquired from each farmer. Nutrient inputs via fertilizer were estimated by multiplying the quantity of fertilizer in kilograms by the known (or estimated) nutrient concentration (Appendix Table 1). The typical mineral fertilizer has a nutrient content of 18:5:15 (N:P:K). Organic fertilizers consisted of chicken manure, vermicompost, and/or fermented coffee hulls. Although I verified all ancillary data when possible, as data came from farmer interviews, they should be read as estimates, not absolute quantities. Nutrient export through the coffee harvest was estimated using yield data (kg ha^{-1}) and published values of nutrient removal in coffee beans (20 kg N and 2.5 kg P per 1000kg of coffee beans; Harmand et al. 2007; Malavolta 1990). The contribution of fixed N by leguminous shade trees (e.g. *Erythrina*) is difficult to estimate based on large spatial and temporal variability in N fixation rates. For example, *Erythrina* root nodules disappear for 10 weeks following pruning, suggesting that N fixation may occur only 7 months per year (Nygren and Ramirez 1995). *Erythrina* pruning residues may transfer 237, 227 and

173 kg N ha^{-1} yr^{-1} to the mineral soil in farms with one, two and three annual prunings (Russo and Budowski 1986). I assume a third of foliar N is derived from the soil during periods of suspended N fixation (Nygren and Ramirez 1995). Further, based on measured N fixation rates, Leblanc et al. (2007) extrapolated that of the remaining two thirds of leaf N, 59% is derived from the atmosphere. Therefore, *Erythrina* may contribute 93, 90, and 70 kg N ha^{-1} yr^{-1} of fixed N for farms with one, two and three annual prunings. Based on pruning regimes in the three different management types (average of 2.8, 2.3, and 1.6 annual prunings in mineral-fertilized, organically-fertilized, and unfertilized agroforests), I estimate N fixation rates of 74, 83, and 87 kg N ha^{-1} yr^{-1} in mineral-fertilized, organically-fertilized, and unfertilized agroforests, respectively. These estimations are reasonably larger than N fixation rates in tropical forest stands (20-60 kg N ha^{-1} yr^{-1}; Houlton et al. 2008), but well below maximum N fixation rates of tropical legume plantations (>200 kg N ha^{-1} yr^{-1}; Binkley and Giardina 1997).

The measurement of atmospheric N deposition and emission were beyond the scope of this study. I estimated 7 kg N ha^{-1} yr^{-1} and 1 kg P ha^{-1} yr^{-1} is deposited in bulk precipitation and cloud water (Clark et al. 1998). Based on Houlton et al. (2006), I estimated background N emissions of 3 kg N ha^{-1} yr^{-1} (1 kg N_2O+NO_x, 2 kg N_2) at 2945mm of annual rainfall. A portion of N fertilizer is lost in gaseous N fluxes. Based on emission rates in nearby fertilized coffee plantations, I calculated that approximately 5% of added N is lost as N_2O (Hergoualc'h et al. 2008). Including the other gaseous N species, I estimated that 20% of the total added N is lost via N gas emissions (Veldkamp and Keller 1997). Therefore total farm gaseous N losses (kg ha^{-1} yr^{-1}) are estimated as,

$$N_{Gas} = Fert_N * 0.2 + 3 \quad \text{(Eq. 1)}$$

Finally, I calculated annual farm-level nutrient *excess* by subtracting nutrients exported in the coffee harvest from the sum of inputs (fertilizer, N-fixation, and deposition in the case of N, and just fertilizer in the case of P; Eq. 2 and 3).

$$\text{Annual Excess}_N = Fert_N + Fix_N + Dep_N - Harvest_N - Gas_N \quad \text{(Eq. 2)}$$

$$\text{Annual Excess}_P = Fert_P + Dep_P - Harvest_P \quad \text{(Eq. 3)}$$

Using the unfertilized farms as a baseline, I estimated the percent recovery of those nutrients added through fertilizer. I divided the difference between outputs on fertilized versus unfertilized farms by the difference between inputs (e.g. fertilizer and N-fixation) on fertilized versus unfertilized farms (Eq. 4). Gaseous losses were not included in the estimation of N recovery. Their inclusion increases farm nutrient recovery, when in fact, the volatilization of fertilizer N denotes inefficiency and constitutes a loss. In the case of P, I assumed no inputs on unfertilized farms (Eq. 5).

$$\text{N Recovery (\%)} = (Harvest_{Fert} - Harvest_{UnF}) / (Fix_{Fert} + Fert_{Fert} - Fix_{UnF}) * 100 \quad \text{(Eq. 4)}$$

$$\text{P Recovery (\%)} = (Harvest_{Fert} - Harvest_{UnF}) / (Fert_{Fert}) * 100 \quad \text{(Eq. 5)}$$

2.6 Calculating Nutrient Pools

To calculate coffee nutrient pools, first total aboveground coffee biomass (kg plant^{-1}) was estimated using allometric equations described by Segura et al. (2006). They found the proportion of woody and foliar tissue in a coffee plant to be 83% and 17%, respectively (Appendix A.1). Shade tree nutrient pools were estimated using allometric equations derived for *Erythrina* (Frank and Eduardo 2003; Appendix A.2), *Cordia* (Segura et al. 2006; Appendix A.3) and *Musa* (Hairiah et al. 2001; Appendix A.4).

Pruning affects tree biomass, and the equation used to calculate *Erythrina* biomass included a term for the level of pruning (Appendix A.2). Pruning intensity was categorized into three levels: *mild:* only the low-lying branches removed; *partial*: all but one or two of the tallest branches removed; *total*: all branches removed. I used measured leaf tissue nutrient concentrations to estimate leaf nutrient pools (for shade and coffee plants) and estimated woody nutrient pools in shade trees using N and P concentrations of wood derived from the literature (Heuveldop et al. 1988; Dossa et al. 2008; Appendix C); carbon concentration in wood was assumed to be 50%.

Forest floor litter N and P concentrations (mg g^{-1}) were scaled up to forest floor nutrient pools (kg N and kg P ha^{-1}) using dry mass-per-area relationships derived during sampling (kg litter m^{-2}). Soil N, P and C pools (to 15cm depth) were calculated using soil nutrient concentrations (mg g^{-1}) and bulk density (g soil cm^{-3}) to scale up to kg N, kg P, and kg C ha^{-1}.

2.7 Statistical Approach

Agroforests were divided into three categories: mineral-fertilized (n=9), organically-fertilized (n=7), and unfertilized (n=5). Initially I determined whether fertilizer amendments differed among management types using a one-way analysis of variance (ANOVA) with management type as the main effect. I used a contingency table and Pearson's Chi-square analysis to determine whether pruning regime differed by management type. Once these differences were established, I investigated how management regimes constrained nutrient pools.

I analyzed variation in nutrient pools and fluxes by management type and prior land use (main effects) with farm age as a covariate. Statistical analyses were performed in JMP 8.0 for Macintosh (2009, SAS Institute, Inc.).

3.0 Results

3.1 Nutrient fluxes vary with management type

Annually, mineral-fertilized farms received significantly more N (110 kg compared to 34 kg of N ha^{-1} yr^{-1}) and P (28 kg compared to 8 kg of P ha^{-1} yr^{-1}) than organically-fertilized farms. (Table 2A and D). Five farmers (all in Grano de Oro) claimed to apply no fertilizer at all, and relied solely on biological N-fixation and the recycling of leaf litter and pruning residues to supply the crop with necessary nutrients. Quantities of applied nutrients varied more widely among mineral-fertilized agroforests than among organically-fertilized agroforests (18-223 compared to 15-90 kg N ha^{-1} yr^{-1}).

Despite the range of inputs, yields in mineral-fertilized agroforests (ca. 2200 kg ha^{-1} yr^{-1}) were nearly twice as high as those in organically-fertilized agroforests, and two and a half times as high as unfertilized yields (Figure 3). Thus, nutrient outputs (via the harvest) were greatest in mineral-fertilized agroforests at 46±3 kg N and 6±0.4 kg P per ha per year (Figure 4C and D). Finally, annual nutrient excess (difference between inputs and outputs) differed among management types. Annual excess N was significantly greater in mineral-fertilized (113±14 kg N ha^{-1} yr^{-1}) compared to organically-fertilized (80±8 kg N ha^{-1} yr^{-1}) and unfertilized (65±2 kg N ha^{-1} yr^{-1}) farms (Figure 4E and F). Excess P was also significantly greater in mineral-fertilized agroforests (22±5 vs. 4±0.5 vs. -2±0.6 kg P ha^{-1} yr^{-1}).

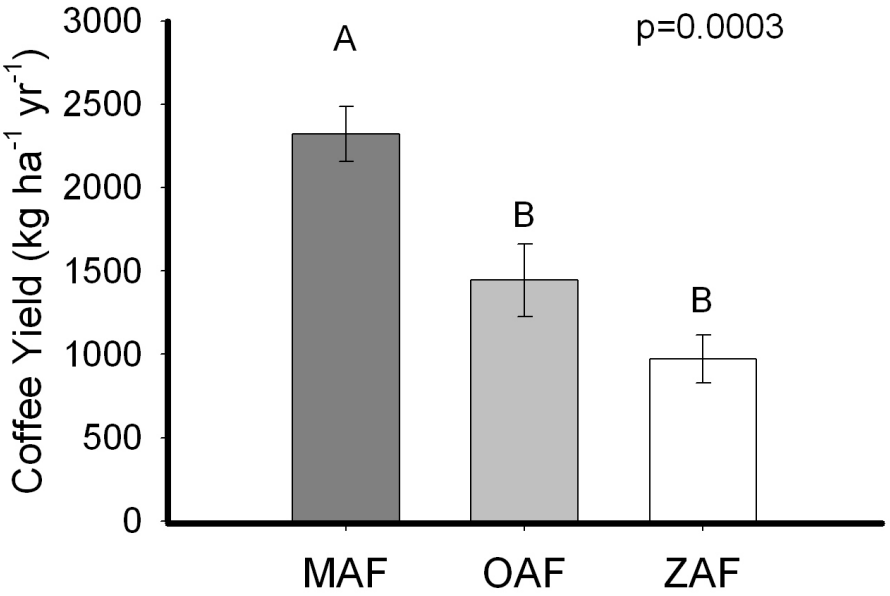


Figure 3: 2005-2006 Coffee yields among agroforest management systems in Costa Rica. Mean yield of fresh coffee berries of mineral-fertilized (MAF; n=9), organically-fertilized (OAF; n=7), and unfertilized (ZAF; n=5) coffee agroforests. Error bars represent standard error of the mean. Differences in yields among the three management types were tested by ANOVA. Values were significantly different at $p<0.05$ are indicated by different letters. Yield data were garnered from farmer interviews and converted to kg ha^{-1} yr^{-1} where 1 metric ton of fresh weight coffee berries is equivalent to 3.92 Costa Rican fanegas or approximately 180 kg of green coffee beans with 11% moisture (255 kg fresh berries/fanega or 46 kg green coffee/fanega; Carvajal 1984).

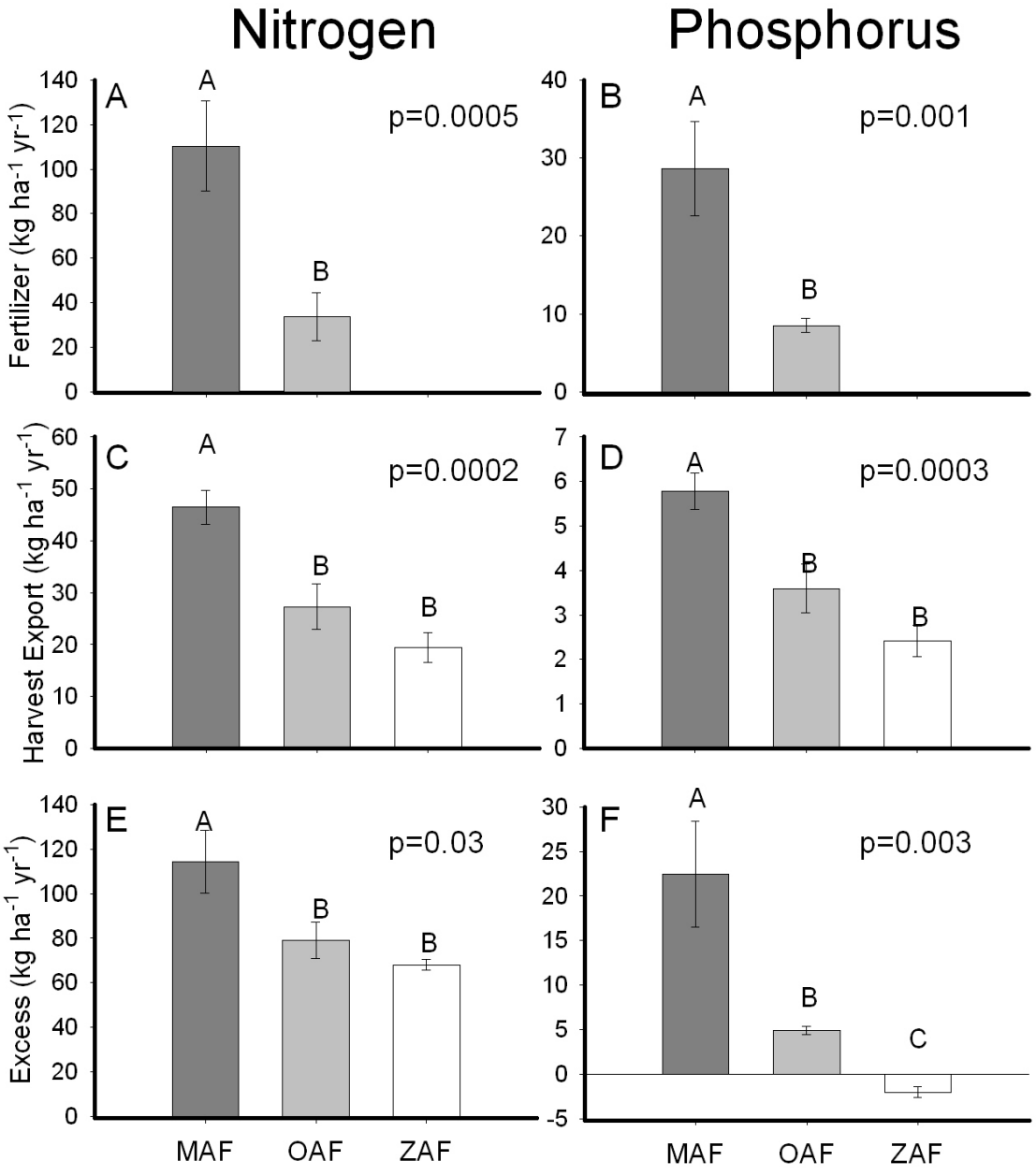


Figure 4: Nitrogen and phosphorus inputs and outputs across management systems in coffee agroforests in Costa Rica. Fertilizer N and P inputs (A and B); N and P exports through harvest (C and D); and excess N and P (E and F) in mineral-fertilized (MAF; n=9), organically-fertilized (OAF; n=7), and unfertilized (ZAF; n=5) coffee agroforests. Fertilizer and harvest nutrients are estimated based on data gathered from farmer interviews. Nutrient excesses are estimated based on input minus output equations. Error bars represent standard error of the mean. Differences in fertilizer inputs, harvest export, and excess among the three management types were tested by ANOVA. Values that were significantly different at $p<0.05$ are indicated by different letters.

3.2 Nutrient pools vary with management type and previous land use

Although *Erythrina* densities were roughly equivalent among farms, *Erythrina* biomass was 50% higher in unfertilized (238±42 Mg ha^{-1}) compared to fertilized farms (124±27 Mg ha^{-1}, Figure 5A). Thus, unfertilized farms held larger N and P pools in *Erythrina* biomass than fertilized farms (Figure 5C and E). *Erythrina* biomass was twice as great in farms that were previously in forest (208±25 Mg ha^{-1}) as in farms that were previously in agriculture (109±27 Mg ha^{-1}) despite a wide range of agroforest ages (Figure 5B). Thus, N and P pools were larger in those farms that were forested prior to cultivation than in farms that were previously under agricultural management (Figure 5D and F). Neither *Cordia* nor *Musa* biomass differed by management or previous land use, which is not surprising, as farmers do not manage these species as carefully as they do *Erythrina*.

Despite variation in inputs among management types, N, P, and C pools in surface soils (0-15cm) were similar across all agroforests. Further, soil pools did not vary with fertilization rates, farm age, or previous land use. However, soil N and C pools differed significantly by location ($p=0.0004$ and $p<0.0001$, respectively), with the largest N and C pools in San Juan Norte and the smallest in Grano de Oro.

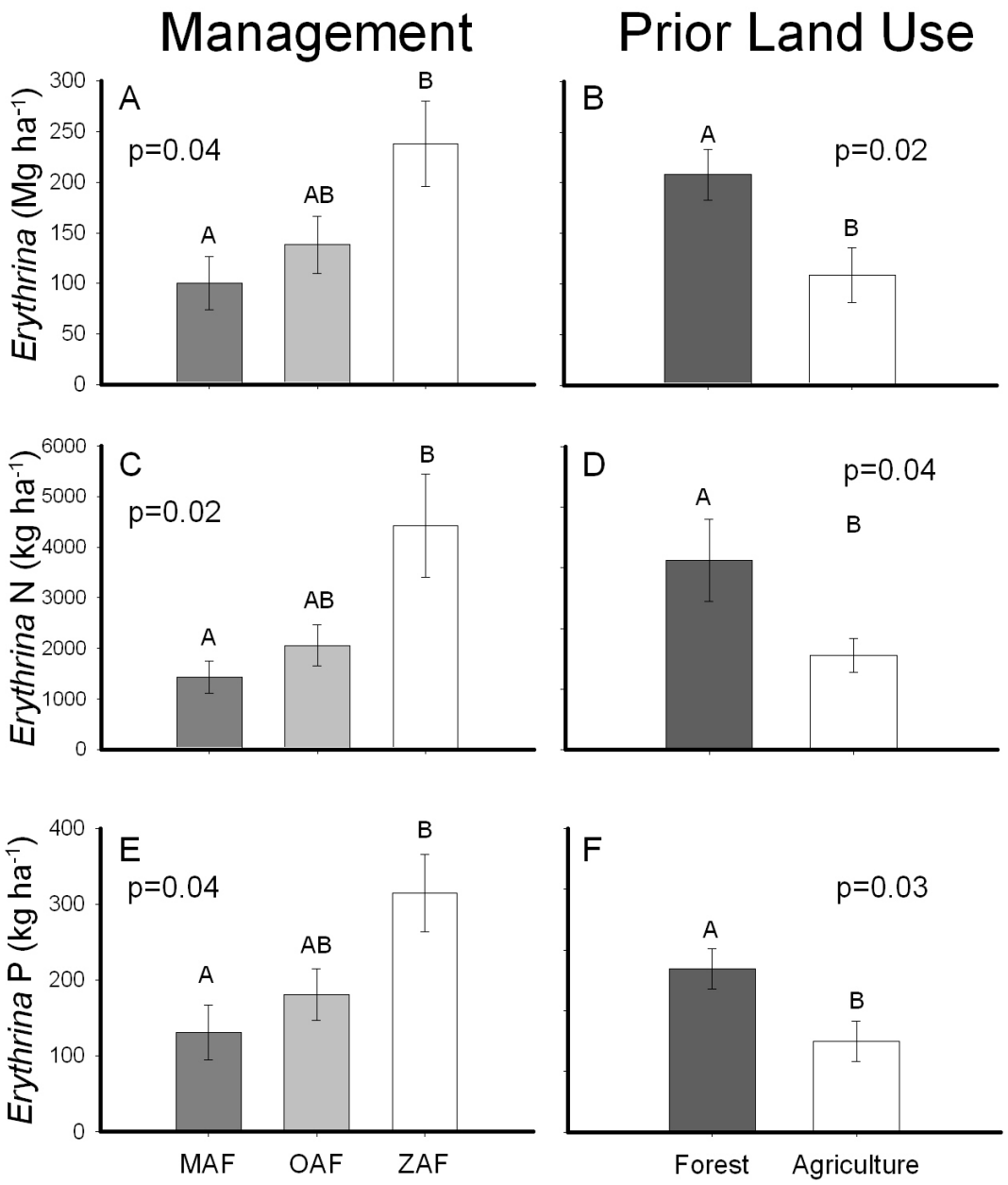


Figure 5: Aboveground *Erythrina* biomass, N and P pools by fertilizer management and by prior land use in coffee agroforests in Costa Rica. *Erythrina* biomass (A and B); N pools in *Erythrina* biomass (C and D); P pools in *Erythrina* biomass (E and F) by management (e.g. mineral-fertilized, organically-fertilized and unfertilized) and prior land use (e.g. forest or agriculture). Aboveground *Erythrina* biomass calculated using allometric equations (Frank and Eduardo 2003) and multiplied by tissue concentrations to calculate N and P pools. Error bars represent standard error of the mean. Differences in biomass, aboveground N and P pools among the three management types and two prior land uses were tested by ANOVA. Values were significantly different at $p<0.05$ are indicated by different letters.

4.0 Discussion

4.1 Similar N and P recovery in organically- and mineral-fertilized agroforests

Reported values of mineral fertilizer N recovery range from 30% to 40% for near-by coffee agroforests (Sommer 1978). I estimated that about 30% of applied N was recovered (Figure 6), corroborating the need for strategies that optimize on-farm nutrient use even in more sustainable systems.

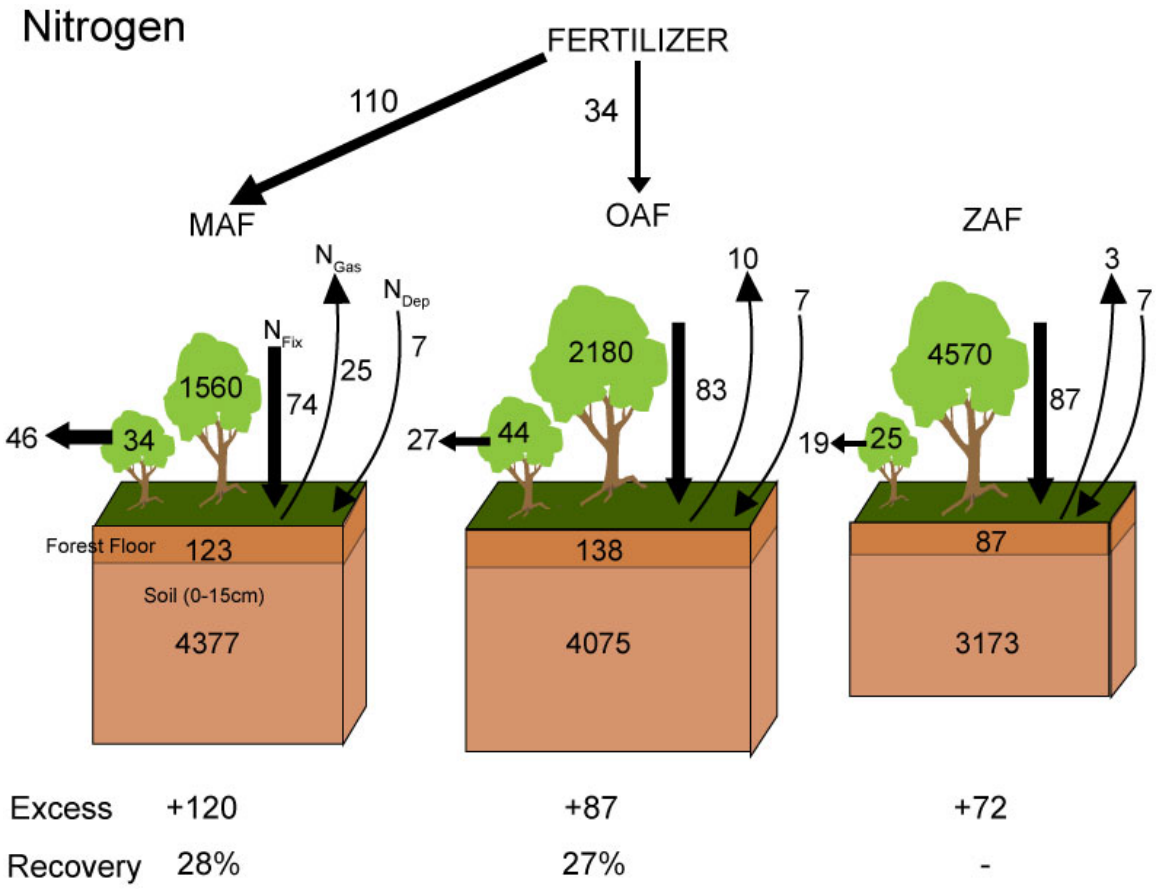


Figure 6: Schematic of nitrogen balances across management systems in coffee agroforests in Costa Rica. Nitrogen balances in mineral-fertilized (MAF; n=9), organically-fertilized (OAF; n=7), and unfertilized (ZAF; n=5) coffee agroforests. All pools are in kg ha^{-1}; all fluxes are in kg ha^{-1} yr^{-1}. Nitrogen excess and recovery are estimated as described in section 2.6.

At present, no other studies have estimated P recovery in coffee farms, which I found to be marginally higher in organically- compared to mineral-fertilized agroforests (25% and 14%; Figure 7). Low P (compared to N) recovery is likely due to the fact that P uptake and export is N-limited given the relative amount of each added, and the ratio in which they are exported in harvest. Similar estimates of nutrient recovery among farms suggest that regardless of fertilizer type, the crop utilizes a relatively fixed proportion of the available nutrients and that nutrients are cycled at similar rates in organically- and mineral-fertilized agroforests. Initially, this seems surprising considering that compared to organic fertilizers, mineral fertilizers are more readily incorporated into plant (e.g. crop) biomass. One might expect farms receiving mineral fertilizers to cycle N more rapidly and therefore for the crop to recover a greater proportion of the added N. However, all farms received large quantities of organic material in the form of litterfall and pruning residues, which creates a high demand for N by microbial biomass. In an agroecosystem where organic material can outweigh fertilizer inputs by 20 times (Russo and Budowski 1986), nutrient recovery may be constrained, in part, by pruning schedules and tissue decomposition rates (Chapter 6).

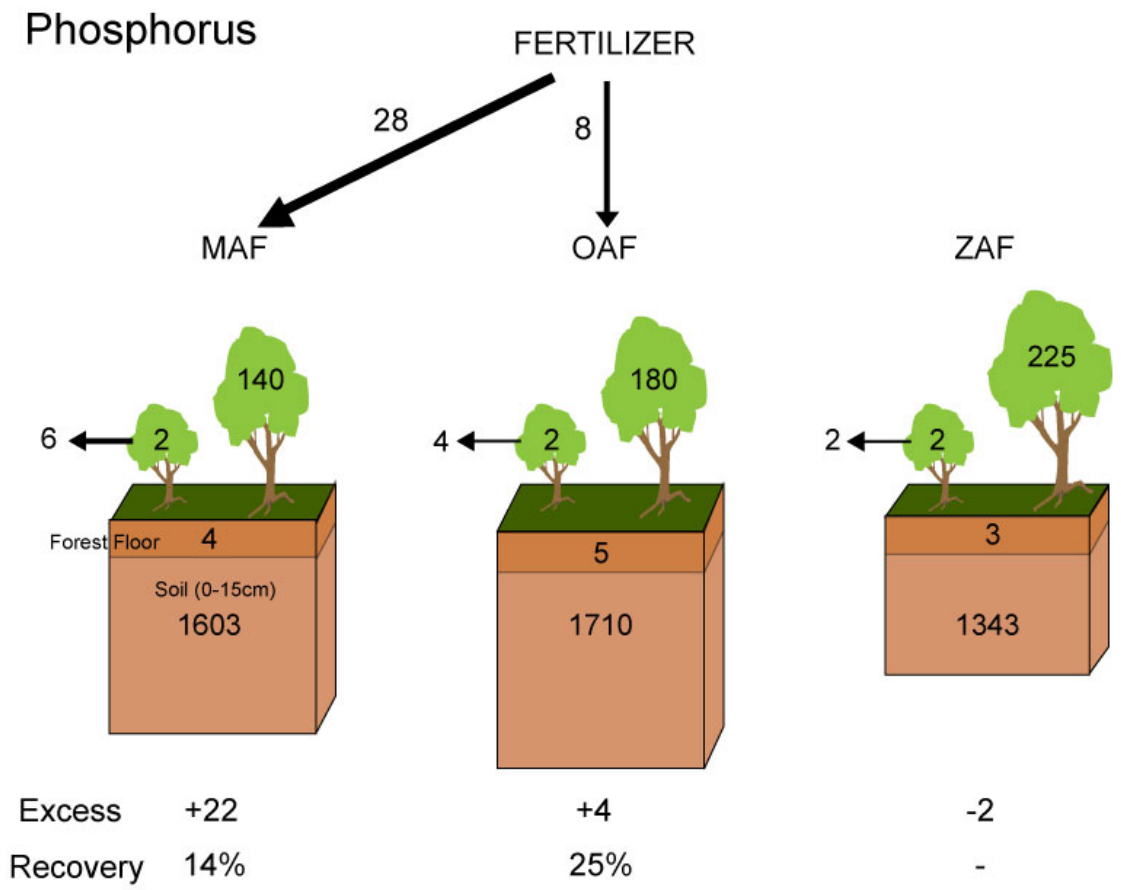


Figure 7: Schematic of phosphorus balances across management systems in coffee agroforests in Costa Rica. Phosphorus balance in mineral-fertilized (MAF; n=9), organically-fertilized (OAF; n=7), and unfertilized (ZAF; n=5) coffee agroforests. All pools are in kg ha^{-1}; all fluxes are in kg ha^{-1} yr^{-1}. Phosphorus excess and efficiency are estimated as described in section 2.6.

4.2 Mineral-fertilized agroforests show a greater potential to leach nutrients than organically-fertilized agroforests

Mineral-fertilized farms employ three times the fertilizer but produce only twice the yield compared to organically-fertilized agroforests. The additional N suggests greater N excess and potential loss from mineral-fertilized farms. Nitrogen and P not recovered in harvest may either (a) accumulate in soil and/or biomass pools or (b) leach to deeper soil layers, below the rooting zone and potentially out of the system. Nitrogen

excess estimations (113 kg N ha^{-1} yr^{-1}; Figure 6) were higher than measured nitrate loss (70 kg NO_3^--N ha^{-1} yr^{-1} at 120cm) in nearby agroforests (Harmand et al. 2007). However, some of this difference may be accounted for by other fractions of N (e.g. dissolved organic N) not measured in the aforementioned study (e.g. dissolved organic N:dissolved inorganic N ratio of 1:9 in tropical forests; Schwendenmann and Veldkamp 2005). In contrast to mineral-fertilized soils, years of organic fertilization can enhance soil N pools (Pimentel et al. 2005), which suggest that excess N in the mineral-fertilized farms may be more prone to N loss than excess N in the organically-fertilized farms.

The estimates of N excess in unfertilized farms were higher (65 kg N ha^{-1} yr^{-1}) than reported fluxes in tropical forests (8-14 kg NO_3^--N ha^{-1} yr^{-1}; Radulovich and Sollins 1991; Bigelow and Ewel *in press*). Again, total N fluxes are likely 10% higher than reported nitrate fluxes, but still these estimates exceed those measured elsewhere. Further, the estimate of N excess in unfertilized farms is driven primarily by the annual rate of N-fixation, and typically, N leaching losses are higher under legumes than non-legumes (Zhu 2003). Thus, it is reasonable to assume that higher densities of leguminous trees may elevate N losses above forest (background) levels even in unfertilized farms.

Phosphorus leaching is typically very low in tropical forests (1.2 kg P ha^{-1} yr^{-1}; Radulovich and Sollins 1991), however the model predicted annual P excess of 22 and 4 kg P ha^{-1} yr^{-1} in mineral-fertilized and organically-fertilized agroforests, respectively (there was negative P balance in unfertilized agroforests). Tropical soils have a high capacity to retain anions such as phosphate, and therefore a substantial portion of excess P in fertilized agroforests will likely be adsorbed onto clay minerals and remain in the

system. Potential P accumulation in soils is supported by a trend towards larger soil P pools in fertilized agroforests (Figure 7). Nevertheless, much of this P not be available to plants. Once it is chemically bound, it can be removed from the P cycle for decades (Walker and Syers 1976). Further, mineral fertilizers tend to acidify soils, and pH is significantly lower in mineral- compared to organically-fertilized soils (Chapter 3, Figure 5). As soils acidify, the available P pool may decline even further, which suggests that over time, mineral-fertilized systems may become P-limited. Finally, negative P balances in unfertilized farms indicate that farming without P amendments may reduce soil P fertility in the long-term. For example, there is little potential for trees to utilize P below the crop-rooting zone. Even deeply-rooted trees tend to take up the majority of plant available P from surface soils and then recycle this P to crops through the decomposition of litter, pruning residues, and root decay (Buresh and Tian 1998). Therefore, agroforestry does not eliminate the need for P amendments (Sanchez and Palm 1996; Buresh et al. 1997), and farming without some form of P fertilization is unsustainable.

Where past research has shown that the inclusion of trees can significantly reduce nutrient losses compared to unshaded monocultures, this research indicates that among shaded plantations, the form of fertilizer may be a less important determinant of nutrient losses compared to the total quantity of nutrients applied. That is, the model predicts that mineral-fertilized agroforests are no more efficient at recovering fertilizer nutrients than organically-fertilized systems likely due to the buffering effects of adding large quantities of organic material during annual prunings. Nevertheless, more added N leads to more excess N in mineral-fertilized agroforests. Thus, mineral-fertilized systems

may pose a greater threat to water quality than either organically-fertilized or unfertilized farms, however actual measurements of fluxes from study farms would be necessary to validate the proposed model. Nevertheless, such tools can be incredibly useful to farmers and land planners interested in optimizing on-farm nutrient management.

4.3 Larger nutrient pools in biomass on unfertilized farms

The allocation of nutrient pools differed among management types. Although not statistically significant, soil N and P pools tended to be largest in fertilized farms (Figure 6 and 7). Organically-fertilized and unfertilized farms tended to store more N and P in plant biomass than mineral-fertilized agroforests. Shade trees were pruned more intensely and more frequently in mineral-fertilized coffee farms (2.6 times a year on average), and less intensely and less frequently in unfertilized agroforests ($p=0.06$). Therefore, in mineral-fertilized farms, nutrients that otherwise would have been stored in plant tissues were transferred to the soil pool, while remaining in tree biomass in unfertilized agroforests (Figure 6 and 7). In farms receiving smaller (or no) nutrient amendments via fertilization, the transfer of nutrients from shade tree biomass to the mineral soil is a crucial determinant of crop productivity. In other words, unless shade trees are regularly pruned, these nutrients are essentially "locked" in this pool. Without human intervention, this pool has a long turnover time, and nutrients will remain unavailable to the crop. Studies have shown that not only the quantity, but also the timing of pruning events can play an important role in providing nutrients during critical crop growth periods (Chapter 6, Figure 11). Thus, in farms predominated by organic

inputs, careful management of shade trees (i.e. a regular pruning schedule) is necessary to promote crop productivity.

4.4 Smaller nutrient pools in biomass on farms previously in agriculture

Although the effect of land use history on nutrient pools was largely trumped by the effect of current farm management, nutrients pools in tree biomass were larger in previously forested farms compared to those converted to coffee farms from cultivated or pasture lands. It has been demonstrated that repeated cycles of cultivation can impede forest re-growth in the tropics (Lawrence 2005; Eaton and Lawrence 2009), which may explain why tree biomass is diminished in those farms that were previously managed.

5.0 Conclusions and Recommendations

Management techniques will alter a farm's nutrient balance both in terms of its productivity and its overall ecological sustainability. That is, while nutrients are recovered in roughly the same proportion in both organically- and mineral-fertilized agroforests, mineral-fertilized farms have larger inputs, and therefore, may lose more nutrients to groundwater supplies. Due to smaller inputs, organically-fertilized farms pose a smaller threat to groundwater but have significantly depressed yields. Farming without fertilizer is not sustainable in the long-term because soil P pools will be depleted. Future research should determine actual N and P losses from agroforests under different managements. Strategies for increasing N and P efficiency on agroforests should also be explored.

References

Albertin, A., and P. K. R. Nair. 2004. Farmers' perspectives on the role of shade trees in coffee production systems: an assessment from the Nicoya peninsula, Costa Rica. Human Ecology 32:443-463.

Aranguren, J., G. Escalante, and R. Herrera. 1982. Nitrogen cycle of tropical perennial crops under shade trees. I. Coffee. Plant and Soil 67:247-258.

Ataroff, M., and M. Monasterio. 1997. Soil erosion under different management of coffee plantations in the Venezuelan Andes. Soil Technology 11:95-108.

Babbar, L. I., and D. R. Zak. 1995. Nitrogen loss from coffee agroecosystem in Costa Rica: leaching and denitrification in the presence and absence of shade trees. Journal of Environmental Quality 24:227-233.

Beer, J. 1988. Litter production and nutrient cycling in coffee (*Caffea arabica*) or cacao (*Theobroma cacao*) plantations with shade trees. Agroforestry Systems 7:103-114.

Bene, J. G., H. W. Beall, and A. Cote. 1977. Trees, Food and People. IDCR, Ottawa, Canada.

Berendse, F. 1979. Competition between plant populations with different rooting depths I. Theoretical considerations. Oecologia 43:19-26.

Berendse, F. 1982. Competition between plant populations with different rooting depths III. Field experiments. Oecologia 53:50-55.

Bigelow, S. W. and J. J. Ewel. Plant diversity affects nutrient leaching from model tropical ecosystems. *In press.*

Binkley, D., and C. Giardina. 1997. Nitrogen fixation in tropical forest plantations. ACIAR Monograph Series 43:297-338.

Buresh, R. J., P. C. Smithson, D. T. Hellums, P. A. Sanchez, and F. Calhoun. 1997. Building soil phosphorus capital in Africa. Replenishing soil fertility in Africa 111-149.

Buresh, R. J., and G. Tian. 1998. Soil improvement by trees in sub-Saharan Africa. Agroforestry Systems 38:51-76.

Chapin, F. S. III, H. Reynolds, C. D'Antonio, and V. Eckhart. 1996. The functional role of species in terrestrial ecosystems. In B. Walker, Editor. Global Change in Terrestrial Ecosystems, Cambridge, UK.

Dixon, R. K. 1995. Agroforestry systems: sources of sinks of greenhouse gases? Agroforestry systems 31:99-116.

Dixon, R. K., S. Brown, R. A. Houghton, A. M. Solomon, M. C. Trexler, and J. Wisniewski. 1994. Carbon pools and flux of global forest ecosystems. Science 263:185-190.

Dossa, E. L., E. C. M. Fernandes, W. S. Reid, and K. Ezui. 2008. Above- and belowground biomass, nutrient and carbon stocks contrasting an open-grown and a shaded coffee plantation. Agroforestry Systems 72:103-117.

Eaton, J. M., and D. Lawrence. 2009. Loss of carbon sequestration potential after several decades of shifting cultivation in the Southern Yucatán. Forest Ecology and Management 258:949-958.

Food and Agriculture Organization (FAO). 1997. FertiStat. Food and Agriculture Organization of the United Nations, Rome.

Fassbender, H. W.1987. Nutrient cycling in agroforestry systems of coffee (*Coffea arabica*) with shade trees in the Central Experiment of CATIE. *In* J. Beer, H. W. Fassbender, and I. Heuveldop, editors. Advances in Agroforestry Research, Turrialba, Costa Rica, pp. 155-165.

Fließbach, A., H. -R. Oberholzer, L. Gunst, and M. Mader. 2007. Soil organic matter and biological soil quality indicators after 21 years of organic and conventional farming. Agriculture, Ecosystems and Environment 118:273-284.

Frank, B., and S. Eduardo. 2003. Biomass dynamics of *Erythrina lanceolata* as influenced by shoot-pruning intensity in Costa Rica. Agroforestry Systems 57:17-26.

Glover, N., and J. Beer. 1986. Nutrient cycling in two traditional Central American agroforestry systems. Agroforestry Systems 4:77-87.

Goodman, D. 1999. Agro-food studied in the 'Age of Ecology': Nature, corporeality, bio-Politics. Sociologio Ruralis 19:17-38.

Goyal, S., K. Chander, M. C. Mundra, and K. K. Kapoor. 1999. Influence of inorganic fertilizers and organic amendments on soil organic matter and soil microbial properties under tropical conditions. Biology and Fertility of Soils 29:196-200.

Hairiah, K., S. M. Sitompul, M. van Noordwijk, and C. Palm. 2001. ASB Lecture Note 4B. International Center for Research in Agroforestry (ICRAF), Bogor.

Harmand, J. M., H. Avila, E. Dambrine, U. Skiba, S. de Miguel, R. V. Renderos, R. Oliver, F. Jimenez, and J. Beer. 2007. Nitrogen dynamics and soil nitrate retention in a *Coffea arabica-Eucalyptus deglupta* agroforestry system in Southern Costa Rica. Biogeochemistry 85:125-139.

Hergoualc'h, K., U. Skiba, J. M. Harmand, and C. Hénault. 2008. Fluxes of greenhouse gases from Andosols under coffee in monoculture or shaded by *Inga densiflora* in Costa Rica. Biogeochemistry 89:329-345.

Heuveldop, J., H. W. Fassbender, G. Alpizar, G. Enriquez, and H. Folster. 1988. Modelling agroforestry systems of cacao (*Theobroma cacao*) with laurel (*Cordia alliodora*) and poró (*Erythrina poeppigiana*) in Costa Rica II. Cacao and wood production, litter production and decomposition. Agroforestry Systems 6:37-48.

Houlton, B. Z., D. M. Sigman, and L. O. Hedin. 2006. Isotopic evidence for large gaseous nitrogen losses from tropical rainforests. Proceedings of the National Academy of Sciences 103:8745.

Houlton, B. Z., Y. P. Wang, P. M. Vitousek, and C. B. Field. 2008. A unifying framework for dinitrogen fixation in the terrestrial biosphere. Nature 454:327-330.

Lawrence, D. 2005. Biomass accumulation after 10-200 years of shifting cultivation in Bornean rain forest. Ecology 86:26-33.

Leblanc, H. A., R. L. McGraw, and P. Nygren. 2007. Dinitrogen-fixation by three neotropical agroforestry tree species under semi-controlled field conditions. Plant and Soil 291:199-209.

Lyngbæk, A. E., R. G. Muschler, and F. L. Sinclair. 2001. Productivity and profitability of multi-strata organic versus conventional coffee farms in Costa Rica. Agroforestry Systems 53:205-213.

Malavolta, E. 1990. Nutricao mineral e adubacao do cafeeiro. Associacao Brasileira para Pesquisa da Potassa e do Fosfato (Piracicaba) and Editora Agromica Cerees Ldta, Sa Paulo.

Manna, M. C., A. Swarup, R. H. Wanjari, H. N. Ravankar, B. Mishra, M. N. Saha, Y. V. Singh, D. K. Sahi, and P. A. Sarap. 2005. Long-term effect of fertilizer and manure application on soil organic carbon storage, soil quality and yield sustainability under sub-humid and semi-arid tropical India. Field Crops Research 93:264-280.

Nair, P. K., V. D. Nair, B. M. Kumar, and S. G. Haile. 2009. Soil carbon sequestration in tropical agroforestry systems: a feasibility appraisal. Environmental Science & Policy 12:1099-1111.

Nygren, P., and C. Ramirez. 1995. Production and turn-over of N_2 fixing nodules in relation to foliage development in periodically pruned Erythrina poeppigiana (Leguminosae) trees. Forest Ecology and Management 73:59-73.

Pimentel, D., P. Hepperly, J. Hanson, D. Douds, and R. Seidel. 2005. Environmental, energetic, and economic comparisons of organic and conventional farming systems. Bioscience 55:573-582.

Radulovich, R., and S. Sollins. 1991. Nitrogen and phosphorus leaching in zero-tension drainage from a humid tropical soil. Biotropica 23:84-47.

Reganold, J. P., L. F. Elliott, and Y. L. Unger. 1988. Long-term effects of organic and conventional farming on soil erosion. Nature 330:370-372.

Reynolds-Vargas, J., J. Fraile-Merino, and R. Hirata. 2006. Trends in nitrate concentrations and determination of its origin using stable isotopes (18O and 15N) in groundwater of the western Central Valley, Costa Rica. Ambio 35:229-236.

Reynolds-Vargas, J. S., and D. D. Richter. 1995. Nitrate in groundwaters of the Central Valley, Costa Rica. Environment International 21:71-79.

Russo, R. O., and G. Budowski. 1986. Effect of pollarding frequency on biomass of *Erythrina poeppigiana* as a coffee shade tree. Agroforestry systems 4:145-162.
Sanchez, P. A., and C. A. Palm. 1996. Nutrient cycling and agroforestry in Africa. Unasylva (FAO).

Schwendenmann, L., and E. Veldkamp. 2005. The role of dissolved organic carbon, dissolved organic nitrogen, and dissolved inorganic nitrogen in a tropical wet forest ecosystem. Ecosystems 8:339-351.

Segura, M., M. Kanninen, and D. Suarez. 2006. Allometric models for estimating aboveground biomass of shade trees and coffee bushes grown together. Agroforestry Systems 68:143-150.

Sick, D. 1997. Coping with crisis: Costa Rican households and the international coffee market. Ethnology 36:255-275.

Smith, T. M., W. P. Cramer, R. K. Dixon, R. Leemans, R. P. Neilson, and A. M. Solomon. 1993. The global terrestrial carbon cycle. Water, Air, & Soil Pollution 70:19-37.

Sommer, K. 1978. Use of radioisotopes in agriculture. Report to the Government of Costa Rica. International Atomic Energy Agengy, Vienna, Austria.

Steppler, H.A., and P. K. R. Nair. 1987. Agroforestry: A Decade of Development. ICRAF, Nairobi, Kenya.

Stoorvogel, J. J., and E. M. A. Smaling. 1990. Assessment of soil nutrient depletion in sub-Saharan Africa: 1983-2000. The Winand Staring Centre, Wageningen.

Su, Y., F. Wang, D. Suo, Z. Zhang, and M. Du. 2006. Long-term effect of fertilizer and manure application on soil-carbon sequestration and soil fertility under the wheat-wheat-maize cropping system in northwest China. Nutrient Cycling in Agroecosystems 75:285-295.

Udawatta, R. P., J. J. Krstansky, G. S. Henderson, and H. E. Garrett. 2002. Agroforestry practices, runoff, and nutrient loss: a paired watershed comparison. Journal of Environmental Quality 31:

Veldkamp, E., and M. Keller. 1997. Nitrogen oxide emissions from a banana plantation in the humid tropics. Journal of Geophysical Research 102:15889-15898.

Walker, T. W., and J. K. Syers. 1976. The fate of phosphorus during pedogenesis. Geoderma 15:1-19.

Zhu, Y. F. 2003. Corn-soybean rotation effects on nitrate leaching. Agronomy Journal 95:1028.

Appendix Chapter 2

Appendix Table 1: Average nutrient concentrations in commonly applied fertilizers in coffee farms in Costa Rica. Nutrient concentrations and cost of mineral and organic fertilizers used to amend coffee farms in Costa Rica.

Fertilizer	Form	%N	%P	%K	%Ca	%Mg	US$/ metric ton
Formula Completa, Gran Manzón	Mineral	18	5	15	0	0	5750
Broza (fermented coffee hulls)*	Organic	3.2	0.3	0.4	4.3	0.4	-
Vermicompost (product of composting with worms)*	Organic	2	1.5	1.2	3	0.7	285
Chicken manure*	Organic	3.0	1.4	2.5	2.6	0.75	125
Rock phosphate	Organic	0	15	0	0	0	100
K-Mg	Organic	0	0	18.3	0	11	1540

* Gabriela Soto, personal communication

Appendix A: Allometric equations used to calculate total aboveground biomass (B_{total}) of coffee plants and shade trees.

A.1. Coffea arabica allometry

To estimate *Coffea* biomass I used allometric equations described in Segura et al. (2006).

$$Log_{10}(B_{plant}) = a+b *Log_{10}(h) \quad \text{(Eq. A.1.1)}$$

Where B_{plant} is the biomass in kg/plant, a and b are parameters with the values -0.799 and 2.338, respectively, and h is coffee height. To estimate the total aboveground biomass (B_{total}) in Mg ha^{-1} I used the following equation:

$$B_{total}=(B_{plant}*D)/10^4 \quad \text{(Eq. A.1.2)}$$

Where D is the density of coffee plants per hectare. Segura et al. (2006) estimated that the distribution of aboveground biomass in coffee plants is the following: stem 63.1%, branches 19.8% and foliage 17%.

To estimate leaf biomass (B_{leaf}) multiplied B_{total} by 0.17, and to estimate nutrient pools I calculated woody biomass (B_{wood}) as the sum of stem and branch biomass (83%) in Mg ha^{-1}. I then multiplied B_{leaf} by foliar nutrient concentrations (e.g. mg of P or N per g) measured on that farm. As I did not measure woody nutrient content, I used values found in the literature (Appendix Table 2) to estimate nutrient pools in wood I used nutrient concentrations of woody tissues literature values.

Appendix Table 2: Measured and literature values used for wood and leaf nutrient concentrations used in aboveground biomass nutrient calculations. The measured values here (indicated by the absence of a asterisk) are averages across mineral-fertilized (MAF), organically-fertilized (OAF), and unfertilized (ZAF) agroforests. I used farm-specific numbers to calculate nutrient pools as that was done on a per farm basis.

		Nitrogen (%)		Phosphorus (%)		Carbon
Mgmt	Species	Leaves	Wood	Leaves	Wood	Leaves
MAF	*Musa*	2.70	-	0.26	-	44.91
	Coffea	3.04	0.5**	0.18	0.02**	47.75
	Cordia	2.12*	1.31*	0.28*	0.12*	45.00*
	Erythrina	4.45	1.31*	0.60	0.09*	45.72
OAF	*Musa*	2.09	-	0.21	-	45.83
	Coffea	2.56	0.5**	0.19	0.02**	47.32
	Cordia	2.12*	1.31*	0.28*	0.12*	45.00*
	Erythrina	3.79	1.31*	0.40	0.09*	47.19
ZAF	*Musa*	2.52	0.5**	0.22	0.02**	44.39
	Coffea	2.96	1.31*	0.44	0.09*	47.36
	Erythrina	4.43	-	0.26	-	46.34

* Heuveldop et al., 1988
**Dossa et al., 2008

A.2. Erythrina poeppigiana allometry

To estimate *Erythrina* biomass I used allometric equations described in Frank and Eduardo (2003).

$$B_{leaf}=(8.95+a_s+a_{tr})DBH^{2.60}+^{bS} \quad \text{(Eq. A.2.1)}$$

Where a_{tr} is the pruning treatment effect (-0.37 for no pruning, 1.19 for partial pruning, and 0.42 for total pruning), a_s is the seasonal effect of sampling in June (1.11), and b_s is the seasonal effect of the nonlinearity of the curve (-0.081 for June). DBH is the diameter at breast height of the tree in centimeters.

$$B_{branch}=9.44DBH^{3.311} \quad \text{(Eq. A.2.2)}$$

Where DBH is the diameter at breast height of the tree in centimeters. Both B_{leaf} and B_{branch} are measured in g per plant, therefore to calculate B_{total} in Mg ha^{-1} I used the following equation:

$$B_{total}= (B_{leaf}+B_{branch}) * D/10^6 \quad \text{(Eq. A.2.3)}$$

To calculate the nutrient pools in woody and foliar biomass, nutrient concentrations were multiplied by B_{branch} and B_{leaf}, respectively. Nutrient pools were scaled up to Mg ha^{-1}.

A.3. Cordia alliodora allometry

To estimate the *Cordia* biomass I used the following equation described in Segura et al. (2006) to calculate B_{branch}, $B_{branch+foliage,}$ $B_{stem,}$ and B_{plant} using the following equations.

$$Log_{10}\ B_{branch} =-1.62 + 2.257 * Log_{10}DBH\ (cm) \quad \text{(Eq. A.3.1)}$$

$$Log_{10}\ B_{branch+foliage} =-1.121 + 1.932* Log_{10}DBH\ (cm) \quad \text{(Eq. A.3.2)}$$

$$Log_{10}\ B_{stem} =-0.942 + 2.062 * Log_{10}DBH\ (cm) \quad \text{(Eq. A.3.3)}$$

$$Log_{10}\ B_{plant} =-0.755 + 2.072* Log_{10}DBH\ (cm) \quad \text{(Eq. A.3.4)}$$

To calculate B_{total} I used the following equation,

$$B_{total}=B_{plant}*D/10^4 \quad \text{(Eq. A.3.5)}$$

When calculating nutrient pools, I assumed woody biomass to be the sum of B_{branch} and B_{stem}. I subtracted B_{branch} from $B_{branch+foliage}$ to estimate the amount of foliar biomass. These quantities were then multiplied by nutrient concentrations and scaled up by density to Mg ha^{-1}.

A.4. Musa spp. allometry

To estimate *Musa* biomass I used allometric equations derived from Hairiah et al. (2001),

$$B_{plant}= 0.030DBH^{2.13} \quad \text{(Eq. A.4.1)}$$

Where DBH is the diameter of the pseudostem at breast height (cm). To estimate the total aboveground biomass (B_{total}) in Mg ha^{-1}, I used the following equation:

$$B_{total}=(B_{plant}*D)/10^4 \quad \text{(Eq. A.4.2)}$$

Where D is the density of banana plants per hectare. The banana plant is a pseudostem with spirally arranged leaves, and therefore I multiplied the nutrient concentrations measured in banana leaves by B_{total} to estimate aboveground pools of N, P, and C. That is, there is no woody component to banana biomass.

Chapter 3: Managing nutrient leaching in coffee plantations: Effects of fertilizer type and species on nutrient losses

Abstract

Nutrient optimization is a critical component in the design of sustainable agroecosystems. The application of fertilizers (especially nitrogen) enhances farm productivity, but can also lead to the eutrophication of streams and estuaries and groundwater contamination. The incorporation of trees on farms (agroforestry) has been shown to reduce nutrient loss in comparison to monoculture plantations, however little research has examined how different management regimes and shade species can further alter the potential of an agroforest to retain or lose nutrients. I installed gravity lysimeters (at 15cm) in four mineral-fertilized and four organically-fertilized coffee (*Coffea arabica*) agroforests to determine losses beneath the active root zone. In each farm, I collected monthly soil samples and soil solutions from under two shade species as well as *Coffea* plants. Tension lysimeters (at 15 and 100cm) were also installed under *Coffea* plants in mineral-fertilized agroforests, organically-fertilized agroforests, and in one mineral-fertilized monoculture farm. Samples were collected between September 2008 and October 2009. Nitrogen concentrations in soils (0-10cm) and nitrate (NO_3^--N) in soil solutions (15cm) were higher in mineral- compared to organically-fertilized agroforests. Phosphate concentrations in leachate (at 15 and 100cm) did not differ significantly among management types. Despite similar P inputs among farms, organically-fertilized soils had higher bioavailable P concentrations. In mineral-fertilized monocultures, NO_3^--N concentrations were three times as high as agroforest leachate concentrations. Deep leachate concentrations did not differ between organically- and mineral-fertilized

agroforests. Thus, regardless of their origin (manure or mineral) applied nutrients are effectively retained in this agroforestry system. The choice of shade species also modified nutrient dynamics and soil characteristics in agroforests. I observed the highest soil N and NO_3^--N concentrations in surface soil solutions under N-fixing *Erythrina poeppigiana*. However, deep concentrations were not higher under N-fixers. This demonstrates the capacity of the system to utilize the extra N made available by biological N fixation. Both management and species effects influenced nutrient availability and leachate in agroforests.

1.0 Introduction

Coffee (*Coffea arabica*) is one of the most economically important crops in Central America. Over 4 million people depend directly on its production for their livelihoods (Tucker et al. 2010), and over 8.5 million people in the region are supported by the commodity if one accounts for all aspects of production (e.g. purchasing and processing; CEPAL 2002; Vedenov et al. 2007). Coffee was traditionally cultivated under the shade of a wide variety of overstory tree species (agroforest). However, in response to global demand, coffee farming has intensified. Farmers grow high-yielding varieties, have all but eliminated shade trees, and amend their crops with large quantities of nitrogen (N) fertilizer (200-250 kg N $ha^{-1}yr^{-1}$; Reynolds-Vargas and Richter 1995; Harmand et al. 2007). Nitrogen is the primary nutrient limiting *Coffea* productivity and growth (Carvajal 1984), and the use of mineral N fertilizers has vastly improved yields. However, isotopic tracer studies indicate that the crop only utilizes 30-40% of the applied N, and as much as 50% of all nitrate (NO_3^-) applied is leached below the crop root zone (Sommer 1978; Salas et al. 2002). This study examines the effects of fertilizer form (mineral or organic) and species on nitrogen and phosphorus (P) leaching in coffee farms.

Nitrate is highly soluble in water and mobile in the soil matrix. Nitrates are the most common contaminant of groundwater, and their presence is often attributed to the heavy use of mineral fertilizers (WHO 1996). Nevertheless, NO_3^--N retention in tropical soils is extremely variable (Reynolds-Vargas et al. 1994). The public health standard limit on NO_3^--N is 10 mg L^{-1}, and high concentrations of nitrate may cause methemoglobinemia or “blue baby syndrome” (WHO 1996). Several studies have shown

that groundwater in Costa Rica's Central Valley may exceed these standard limits (Reynolds-Vargas and Richter 1995), and stable isotope studies show steady increases in groundwater nitrate concentrations in this region over the last 20 years in response to the heavy use of fertilizers and urbanization (Reynolds-Vargas et al. 2006). Although compared to N, phosphorus (P) is leached in smaller quantities, it can still have a large effect on eutrophication in surface waters because its solubility makes it readily available to biota (Tiessen 2008). However, tropical systems, in particular, show little P loss due to the strong mechanisms for P retention in highly weathered soils (Jordan 1982). Nevertheless, heavy P fertilization can lead to higher rates of P loss compared to natural levels, and P inputs into freshwaters can increase algae and aquatic weed growth and lead to anoxic conditions as these organisms decompose (Tiessen 2008). In early efforts to improve water quality, the re-incorporation of shade trees and maintenance of ground cover were suggested as more ecologically sustainable alternatives to intensively managed coffee plantations (Bene et al. 1977; 1987; Foster and Hirata 1988). This study examines how nitrate and phosphate leaching varies among and within agroforests as a function of the key management parameters: fertilizer type and shade tree species.

Babbar and Zak (1995) found that annual N losses were three times higher in unshaded compared to shaded coffee plantations despite extremely high N-input in both farms (300 kg N $ha^{-1}yr^{-1}$), thus confirming that the "tight", efficient nutrient cycles in agroforests prevent loss (Glover and Beer 1986; Imbach et al. 1989). In contrast to unshaded plantations where nutrients located below the coffee root zone are essentially lost to the system, more deeply-rooted shade trees can access water and nutrients stored

beyond the reach of coffee plants (Berendse 1979; Seyfried and Rao 1991; Van Noordwijk et al. 1996; Schroth et al. 2001). For example, N losses (at 200cm) were twice as high in an unshaded coffee plantation than in an adjacent *Eucalyptus deglupta* (timber species) agroforest (regardless of heavy fertilization in both systems; Harmand et al. 2007). Both these studies clearly demonstrate the enhanced nutrient efficiency of agroforests over monocultures. However, no research has examined how nutrient loss may be enhanced or diminished by different management strategies *among* agroforests.

A spectrum of coffee agroforests exist in Costa Rica, ranging from high-input systems in which *Coffea* is cultivated in association with regularly planted leguminous (N-fixing) tree species, to less orderly systems in where *Coffea* is grown under the shade of trees of various species (Moguel and Toledo 1999) and alongside other crops for local sale or household consumption (e.g. sugar cane, beans, pineapple, etc.; Albertin and Nair 2004). Agroforests are distinguished not only by the variety and density of species present, but also by the timing and intensity of management practices (e.g. fertilization and pruning of shade trees). The diversification of these farming techniques in Latin America can be linked, in part, to severe economic forcings in the late 1990s.

International coffee prices dropped to a historic low between 1999 and 2003, leading to the development of niche markets for coffee farmers. In response, specialty coffees certified as Fair Trade®, Bird-Friendly® (shade-grown), and Certified Organic® (ICO 2010; Boyce et al. 1994; Goodman 1999; Lyngbæk et al. 2001) began to emerge. Organic coffee farming excludes the use of synthetic fertilizers and pesticides. Organic farmers rely instead on compost and manure to maintain soil fertility and farm

productivity and on naturally occurring pesticides or integrated pest management to control pests. Many studies in temperate regions (especially in Europe) have shown that organic farming has positive impacts on soil organic matter (SOM; Clark et al. 1999), microbial biomass (Fließbach 2002), soil biological activity (Fließbach et al. 2007), mineralizable N (Clark et al. 1998), cation exchange capacity (Reganold et al. 1993), water-holding capacity (Liebig and Doran 1999), and permeability (Drinkwater et al. 1995). Nevertheless, very few studies evaluating the potential advantages and disadvantages of organic farming have been conducted in tropical regions (but see Payán et al. 2009). In addition, hardly any research (temperate or tropical) has measured leaching from mineral and organically-fertilized farming systems. Estimates are typically based on nutrient budgets (but see Stopes et al. 2002 and Torstensson et al. 2006). Many factors in the soil, not reflected in budgets, may be crucial to the processes regulating N and P leaching (Ulén et al. 2005; Aronsson et al. 2007). Understanding factors regulating leaching losses is critical for the design and implementation of ecologically sustainable agricultural practices and policies in the tropics. Thus, the first goal of this research was to examine how mineral and organic fertilizer management alters soil leachate dynamics in coffee agroforests.

In addition to fertilizer management techniques, nutrient dynamics in agroforests will be modified by the ecological feedbacks among the species present. Different functional types (e.g. N-fixers, herbaceous species, etc.) will alter the physical environment, soil community (Eviner and Chapin 2003), and microclimate through effects on percent shade cover (Vanlauwe et al. 1997) and standing biomass; thus, choice

of shade species should alter nutrient cycling (Mack and D'Antonio 2003). For example, bacterial symbionts in the root nodules of *Erythrina poeppigiana* convert atmospheric N into a bioavailable form, directly enhancing soil N. Other shade species such as *Musa acuminata* (banana) produce fruit that is exported from the farm; this species may deplete the soil of N and cations (especially potassium; Turner et al. 1989). Therefore, some shade species may be considered a nutrient source (*Erythrina*) and others a nutrient sink (*Musa, Coffea*). Therefore, the second goal of this research was to examine species effects on fluctuations in soil nutrient and leachate dynamics, and how these dynamics differ depending on management type.

I predicted that N and P would not differ among mineral and organically-fertilized agroforests (Chapter 2, Figure 6 and 7). However, I expected organically-fertilized soils would have higher soil pH than mineral-fertilized soils (Fließbach et al. 2007). I expected NO_3^--N, NH_4^-N, and PO_4^{3-}-P concentrations to be highest in monoculture farms compared to agroforests at both 15 and 100cm depth; and among agroforests, I expected higher concentrations in mineral-fertilized agroforests. *Erythrina* transfers nutrients to the system via its association with N-fixing bacteria and high quality leaves (Chapter 6), and therefore I expected to observe higher N and P concentrations in soil and leachate collected under this species than under the other either *Coffea* or *Musa*. Finally, I expected temporal patterns in leachate and soil nutrient concentrations would differ as a function of the timing (and release) of nutrients from mineral and organic fertilizers (González-Aguilar et al. 1985).

2.0 Methods

2.1 Study sites

The Central Valley of Costa Rica has a humid tropical climate favorable to agroforestry systems. This study was conducted in three locations near Turrialba, Costa Rica (9°53'N 83°40'W; Figure 1). Farms were located within 3 km of each other (9°52'-9°54'N, 83°41-83°42'W). The altitude of sites ranged from 783 to 1017 meters above sea level. Soils are of volcanic origin and are characterized as Typic Humitropepts (Selvaradjou et al. 2005) with a clay-loam texture.

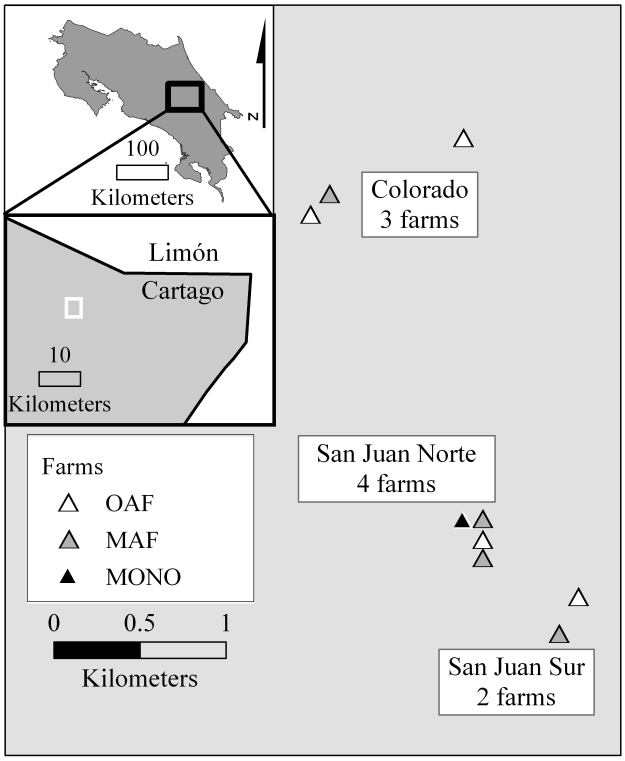


Figure 1: Map of farm locations in Central Valley of Costa Rica. Map of the locations of the three regions where selected coffee farms instrumented in 2008. All farms are located between 783 to 1017 masl. Four farms are mineral-fertilized agroforests (MAF), 4 are organically-fertilized agroforests (OAF), and 1 is an mineral-fertilized monoculture (MONO).

The study region in the Central Valley receives on average 2600mm of rainfall per year and has a mean annual temperature of 22.6°C. Rainfall is seasonal, and the dry season extends from February through May with March being the driest month. The study period (13 months from September 2008-October 2009) was wetter than average, with a cumulative rainfall of 3530mm from first to final collection. In addition, the region experienced an unusually wet December (708mm) and February (544mm) and an extremely dry April (39mm; Figure 2). When soils are at field capacity, monthly evapotranspiration in coffee agroforests is roughly 100mm (3.5mm per day, van Kanten and Vaast 2006). Plants were very likely water-stressed between April and May of 2009. Also at this time, soil solutions were difficult to collect due to the highly negative capillary pressure of remaining soil water.

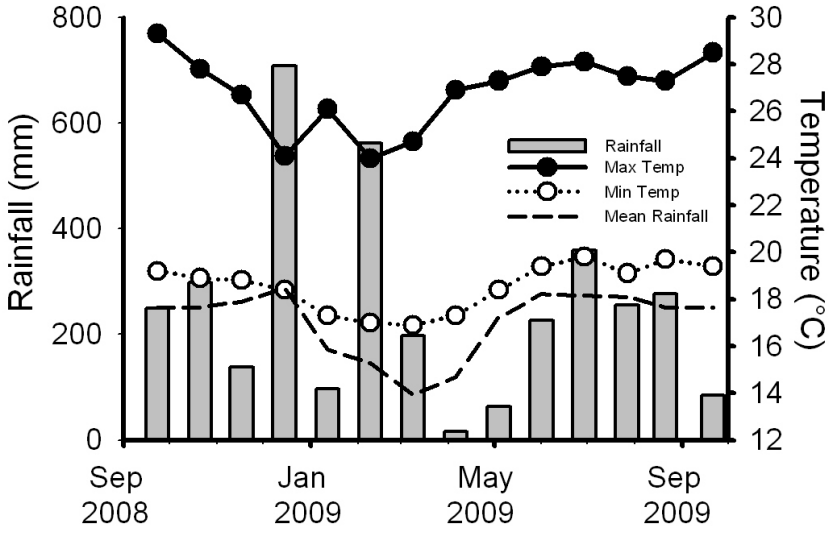


Figure 2: Rainfall and temperature throughout the study period (September 2008-October 2009) in the Central Valley of Costa Rica. Cumulative rainfall and mean daily minimum and maximum temperatures for each 28-day period prior to sample collection. Dashed line represents the mean monthly rainfall from 1942-2009.

After extensive interviews and preliminary sampling (Chapter 2), four organically-fertilized and four mineral-fertilized coffee agroforests were selected for instrumentation (Table 1). In addition, one mineral-fertilized coffee monoculture was selected to serve as a control, testing for nutrient dynamics in the absence of trees (Table 1). In the shaded farms, *Coffea arabica* is cultivated (3300-6600 plants per ha) under a combination of *Erythrina poeppigiana* and *Musa acuminata*. Nitrogen-fixing *Erythrina* represents 60% of the total shade stem (non-coffee) density in mineral-fertilized agroforests and 45% of the total shade stem density in organically-fertilized agroforests, with the rest consisting of banana. On average, mineral-fertilized agroforests received 96kg (± 16) of N and 18kg (± 5) of P ha^{-1} yr^{-1} in the form of mineral fertilizer. Organically-fertilized agroforests received an average of 38kg (± 20) of N and 16kg (± 7) of P ha^{-1} yr^{-1} in the form of manure, compost, and/or fermented coffee husks (called *broza* locally). The mineral-fertilized monoculture received 300 kg N and 33 kg of P ha^{-1} yr^{-1}.

Table 1: Characteristics of coffee farms selected for lysimeter instrumentation in Costa Rica. Farm characteristics, density, size of soil nutrient pools, and nutrient fluxes in four mineral-fertilized agroforests (MAF), four organically-fertilized agroforests (OAF), and one mineral-fertilized monoculture (MONO).

						Density (plants ha^{-1})			Soil (0-80cm) Mg ha^{-1}			Fluxes (kg ha^{-1} yr^{-1})		
Management	**Age (yrs)**	**Prior land use**	**Location**	**Prunings per year**	**Area (ha)**	***Coffea***	***Erythrina***	***Musa***	**N Pool**	**P Pool**	**C Pool**	**Yield**	**Fert N**	**Fert P**
OAF	20	Managed	SJN	2	2	5600	400	200	20	8	254	1800	58	29
OAF	30	Managed	SJS	2	8.5	4900	900	1100	18	10	249	1193	0.5	0.2
OAF	44	Forest	COL	2	2.8	5300	200	700	16	14	135	900	84	27
OAF	20	Forest	COL	2	0.7	3300	700	800	16	9	132	514	10	8
MAF	50	Managed	SJN	3	2	6600	500	200	22	7	274	2160	46	13
MAF	30	Managed	SJN	2	2.1	4800	700	1000	19	11	213	4114	115	32
MAF	20	Forest	SJS	2	2	4000	400	200	24	6	204	2250	121	18
MAF	50	Forest	COL	2	3	5000	600	400	18	8	177	1800	100	8
MONO	15	Managed	SJN	0	14	5000	0	0	20	8	254	4719	300	62

2.2 Potential for loss: Gravity lysimeters (at 15cm)

To examine the effect of management and species on potential nutrient loss, three lysimeter stations were established in each agroforest (instrumentation in the monoculture farm is described below). A pit was excavated to roughly 80cm within 50cm of the bases of (1) *Coffea* beside *Erythrina*, (2) *Coffea* beside *Musa* and (3) *Coffea* plant 5m away from any shade tree (called "distant *Coffea* plant"; Figure 3A and B). Gravity lysimeters are frequently used in tropical systems where rainfall is high (Russell and Ewel 1985; Radulovich and Sollins 1991; Campo et al. 2001). Gravity lysimeters were constructed from 3-inch polyvinyl chloride (PVC) tubes cut on an angle to create an oval opening with semi-axis of 10.16 and 5.08cm. The surface area of the lysimeter exposed to leachate was 40.45cm^2 such that 10mm of leachate should yield 40.45mL. Trenches were dug and lysimeters were installed 15cm below the soil surface and roughly 15cm from the base of the trunk. As the majority of root activity occurs in the topsoil (Schroth 1998), soil solutions collected at 15cm indicate the soluble nutrient concentrations to which the majority of roots are exposed. Two lysimeters were installed under each species in the pair (e.g. coffee and adjacent shade tree), and lysimeters from each species were connected to one 1L volumetric high-density polyethylene (HDPE) collection bottle by PVC tubing. In other words, the two lysimeters under *Erythrina* were connected to the same bottle and the two lysimeters under *Coffea* were connected to another bottle. A wooden box (internal dimensions: 85cm x 10cm x 20cm) was installed in the pit to prevent back-filling, and the HDPE bottles were placed at the bottom. Bottles were treated with three drops of chloroform ($CHCl_3$) to prevent bacterial growth, and all parts

of the lysimeter were acid-washed prior to deployment. Every four weeks, the volume in the bottles was measured and a sub-sample of soil water collected for nutrient analysis. Old bottles were replaced with acid-washed, chloroform-treated bottles. All samples were frozen prior to analysis. Soils (0-10cm) were also collected within 1m of each lysimeter station at the time of leachate collection (n=3 per farm per collection).

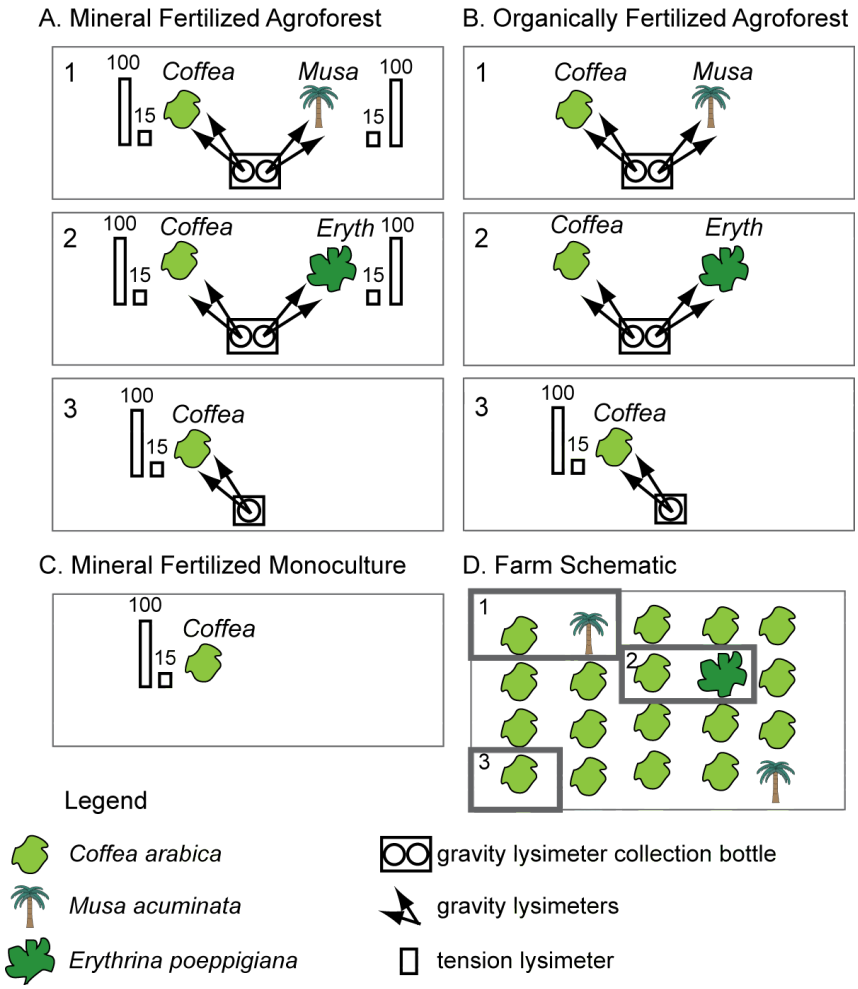


Figure 3: Diagram of lysimeter sampling design in coffee farms in Costa Rica. Lysimeter sampling design in (A) mineral-fertilized agroforests, (B) organically-fertilized agroforests, and (C) mineral-fertilized monocultures. Locations of gravity lysimeters are indicated by black triangles (all at 15cm) and rectangles indicate locations of tension lysimeters (15 and 100cm). In agroforests, a pair of gravity lysimeters were installed at 15cm below *Coffea* and adjacent *Musa*, *Coffea* and adjacent *Erythrina*, and *Coffea* 5m from the nearest shade tree. Only tension lysimeters were installed in the monoculture.

2.3 Actual loss: Tension lysimeters (at 15 and 100cm)

Tension lysimeters use suction to collect water from the soil matrix. Tension lysimeters were constructed by attaching ceramic cups (SoilMoisture Corp., Goleta, CA) to 3-inch PVC tubes and sealing the end with a rubber stopper. In mineral-fertilized agroforests, tension lysimeters were installed at 15cm and 100cm below the soil surface within 30cm of the base of the *Coffea, Erythrina*, *Musa*, and *Coffea* adjacent to each shade species (2 depths per plant x 5 plants; n=10 per farm; Figure3A). I examined soil solutions at 100cm as (1) *Coffea* roots do not typically extend below 50cm (Garriz 1978), (2) 80% of *Erythrina* roots are concentrated in the first 20cm (Chesney 2008), and (3) 70% *Musa* roots are concentrated in the first 40cm (Draye 2002; Chapter 4, Table 2). Due to the low root activity at this depth, I assumed that soil solutions collected at 100cm indicated the soluble nutrient concentrations lost from the system. In organically managed farms, a pair of tension lysimeters (15cm and 100cm) was installed within 30cm of the base of the previously selected distant *Coffea* plant (5m from nearest shade tree; Figure 3B). Lysimeter pairs were installed under *Coffea* plants in an unshaded monoculture farm (Figure 3C).

Tension lysimeters were allowed to equilibrate with their surroundings for one month before collecting the first sample, after which samples were collected every four weeks (starting in September 2008). Lysimeters were filled with distilled water before each sampling period. The day before sample collection, tension lysimeters were purged of any remaining water, and an internal pressure of -0.05 to -0.06 MPa was applied using

a hand-held vacuum pump. The following day, the soil solutions were extracted from the lysimeters. All samples were frozen until analysis.

2.4 Nutrient analysis

Leachate collected from gravity and tension lysimeters was transported to the University of Virginia for nutrient analysis. Samples were filtered through a 40μm filter to remove any debris. Inorganic NO_3^-, NH_4^+, and PO_4^{3-} in leachate were analyzed on a LACHAT QuikChem (LACHAT Instruments Loveland, CO) on filtered samples. Concentrations are reported in mg L^{-1} (where mass values pertain to N or P component of NO_3^-, NH_4^+, PO_4^{3-}).

The same day as collection, field-moist sub-samples of soils were weighed, oven-dried for 24 hours at 105°C, and re-weighed to determine gravimetric soil moisture ($mass_{water}/mass_{soil}$). Remaining soils were air-dried for 3 days in an air-conditioned room, and then passed through a 2mm mesh sieve. Soil pH was determined on air-dried sub-samples in a 2:1 (soil:water) slurry.

The percent weight lost after ignition represents the percent organic carbon in the soil. Bioavailable soil P was determined using a modified Bray-extraction. Three grams of sieved, air-dried soils were shaken for 1 minute in 25mL of a 0.03mol L^{-1} NH_4F and 0.025mol L^{-1} HCl solution (Bray and Kurtz 1945). Extracts were filtered and P concentration was determined colorimetrically using a molybdate blue methodology on an Alpkem Flow Solution IV Autoanalyzer (OI Analytical, College Station, Texas, USA). A portion of soil was also ground to <149μm and dry-combusted on an elemental

analyzer to determine total N and C. All data are reported on an oven-dry mass basis. Nutrient ratios (C:N or N:P) are reported on a molar basis.

2.7 Statistical approach

2.7.1 Effect of management and species on potential nutrient loss

To examine the effect of management and species on potential nutrient loss, I examined leachate concentrations collected from gravity lysimeters located in shallow surface soils (15cm; Figure 3A and B). I performed a two-way repeated measures analysis of variance (RANOVA) with management and species as main effects, time as a random effect, and with individual lysimeters nested within farm to account for variability among instruments. The gravity lysimeter experiment was designed as a 2 x 5 matrix (management x species combination), and therefore the treatment effects of management and species are included in one statistical model. However, in the text I will first address management effects on potential and actual nutrient loss followed by discussion of species effects.

Surface soil properties might also indicate or help explain drivers of nutrient loss. I performed a similar RANOVA (management and species as main effects, time as random effect) on soil nutrient concentrations. In this case, farms were blocked by location in order to account for inherent differences in substrate (even though all farms were located within 3km of one another; Figure 1). To test the ability of soil nutrients to predict leachate concentrations and explain potential drivers of potential loss, I performed univariate regressions with Bonferroni corrections ($=\alpha/6$) on nutrient concentrations (15cm) from soil leachate as a function of available P and total soil N.

2.7.2 Effect of management on actual nutrient loss

To examine the effects of management on leachate nutrient concentrations below the root zone (100cm), I performed a one-way RANOVA with management as the main effect, time as a random effect (uneven sampling design), and with lysimeters nested within farms. Specifically, I examined leachate concentrations collected from tension lysimeters at 100cm depth in organically-fertilized agroforests, mineral-fertilized agroforests, and in the mineral-fertilized monoculture using measurements gathered from under *Coffea* plants only (Figure A3, B3, and C). I calculated the change in nutrient concentration with depth by subtracting concentrations at 100cm from concentrations at 15cm. I examined the effect of management on the change in concentration using a one-way RANOVA with management as the main effect, time as a random effect.

I tested the predictive power of leachate concentrations at 15cm on concentrations at 100cm using univariate regression. Finally I performed multivariate correlations on soil nutrients (10cm), leachate concentrations at 100cm to determine any correlations in shallow soil nutrient content and leaching at depth.

2.7.2 Effect of species on actual nutrient loss

Nutrient concentrations in leachate collected from tension lysimeters were examined for species effects. I performed a two-way RANOVA (with species and depth as main effects, time as random effect, and lysimeters nested within farms) on leachate concentrations collected from tension lysimeters at 15 and 100cm in mineral-fertilized agroforests (Figure 3B). I tested five species and/or species combinations: distant *Coffea* (with nothing other than Coffea within 5m)*, Erythrina, Musa, Coffea* adjacent to

Erythrina (C+E), and Coffea adjacent to Musa (C+M). Again, I examined the percent change in NO_3^--N and PO_4^{3-}-P concentrations as mentioned above, and examined the role of species in reducing concentrations between surface and deep soils. All data were analyzed using JMP 8.0 for Macintosh.

3.0 Results

3.1 Potential nutrient loss is lower in organically-fertilized than in mineral-fertilized agroforests

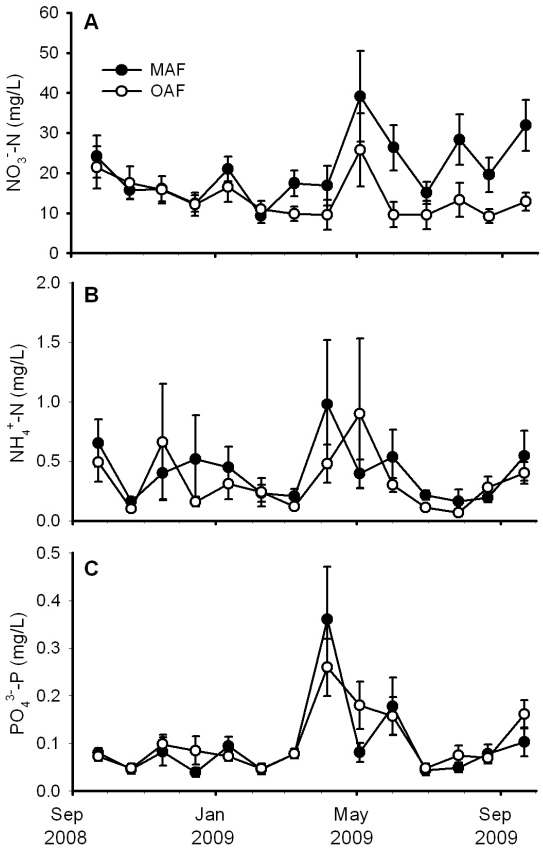


Figure 4: Nutrient concentrations in gravity lysimeter leachate in coffee agroforests in Costa Rica. Concentrations of (A) NO_3^--N, (B) NH_4^+-N, and (C) PO_4^{3-}-P in leachate (15cm; gravity lysimeter) in organically- (OAF) and mineral-fertilized (MAF) agroforests throughout the study period (September 2008-October 2009). Closed circles indicate mineral-fertilized agroforests (n=4), and open circles indicate organically-fertilized agroforests (n=4). Bars represent standard error of the mean.

In surface soils (15cm), NO_3^--N concentrations were higher in leachate collected from mineral- compared to organically-fertilized agroforests ($p=0.014$). I did not observe significant differences in NH_4^+-N or PO_4^{3-}-P concentrations between organically- and mineral-fertilized agroforests (Figure 4B and C). Temporal patterns in NO_3^--N concentrations were remarkably similar in the two management types (e.g. management*time effect n.s.; Figure 4A), with elevated concentrations in both farm types in May (and in mineral-fertilized farms again in August). Ammonium-N and PO_4^{3-}-P also varied significantly through time with 3- and 9-fold increases, respectively, between April and June.

Soil N (to 10cm) and C were significantly higher in mineral- compared to organically-fertilized agroforests ($p<0.0001$ in both cases). Bioavailable soil P was higher in organically- compared to mineral-fertilized agroforests ($p<0.0001$; Figure 5B). Organically-fertilized soils were also more basic than mineral-fertilized soils ($p<0.0001$; Figure 5D), but gravimetric soil moisture was higher in mineral-fertilized agroforests ($p=0.0062$). Temporal patterns in soil N (and C) were similar under the two management schemes, with relatively constant concentrations until June and July, when they declined (Figure 5A). Soil N and C remained low until September when they began to increase. The patterns in bioavailable P through time were very similar between management types (Figure 5B), declining in January and February and again in June and July. Soil moisture varied significantly throughout the year and was lowest in April. Soil pH did not vary significantly throughout the course of the study (time effect n.s.).

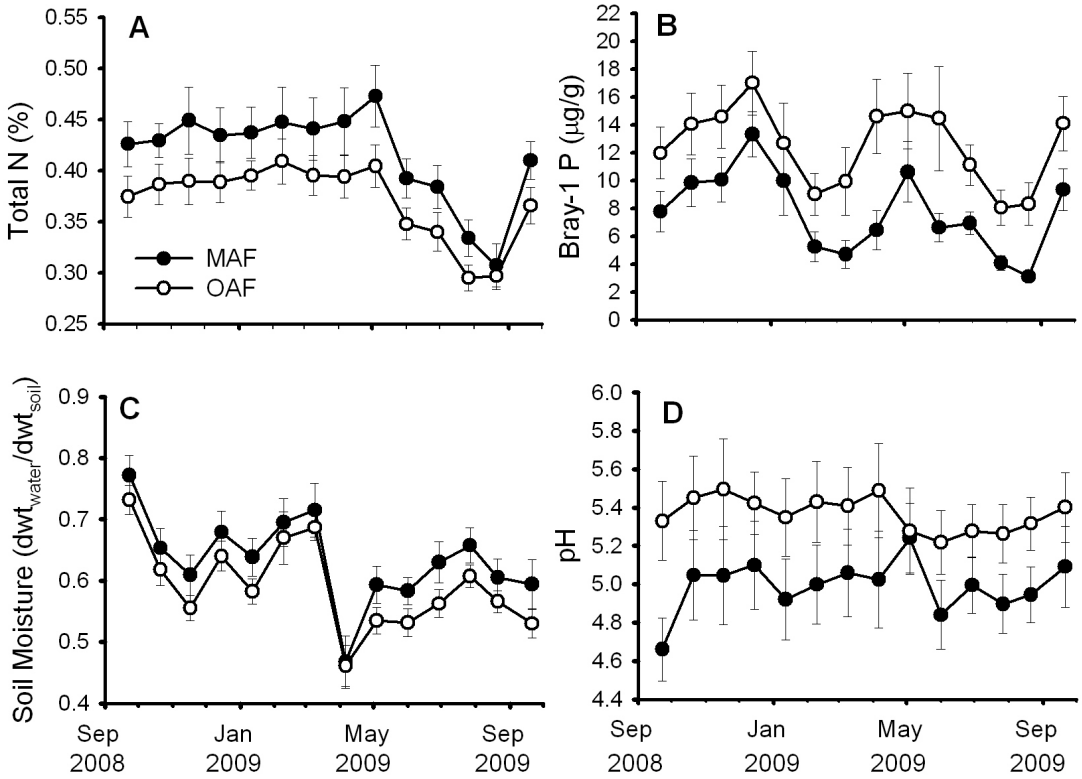


Figure 5: Soil nutrient concentrations in top 10cm in coffee agroforests in Costa Rica. (A) Total N, (B) Bray-1 P, (C) soil moisture, and (D) pH in soils collected from 0-10cm in organically- (OAF) and mineral-fertilized (MAF) coffee agroforests throughout the study period (September 2008-October 2009). Closed circles indicate mineral-fertilized agroforests (n=4), and open circles indicate organically-fertilized agroforests (n=4). Bars represent standard error of the mean.

3.2 Presence of trees, not fertilizer type, determines actual nutrient loss at depth

Below the rooting zone (100cm), leachate NO_3^--N concentrations were more than twice as high in unshaded monocultures compared to coffee agroforests (p=0.0024). The change in NO_3^--N concentration between 15 and 100cm was the largest in unshaded monocultures (43 mg NO_3^--N L^{-1}), with similar changes in concentrations among organically- and mineral-fertilized agroforests (12 and 17 mg NO_3^--N L^{-1}, respectively; p=0.054; Figure 6C). Management did not explain variation in PO_4^{3-}-P or NH_4^+-N at 100cm or the change in PO_4^{3-}-P or NH_4^+-N between shallow and deep soils. Overall,

leachate NO_3^--N concentrations were about three times higher at 15cm than they were at 100cm (p=0.0042; Figure 6).

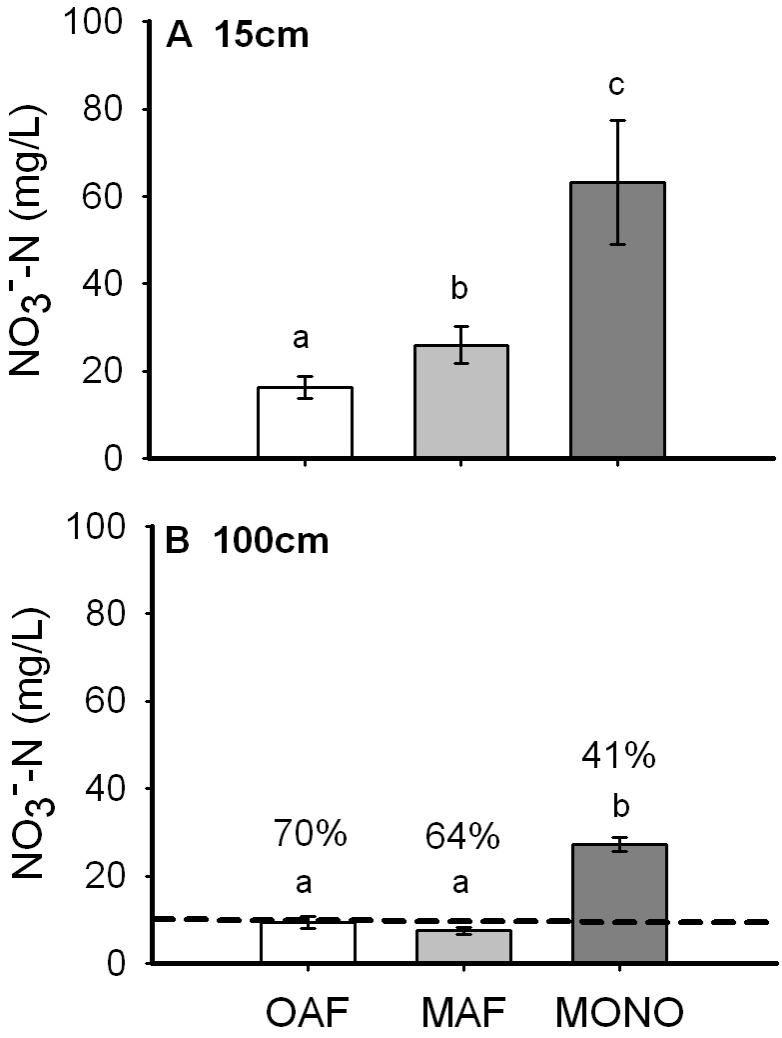


Figure 6: Nitrate concentrations in tension lysimeter leachate in three coffee management systems in Costa Rica. Mean NO_3^--N concentrations at (A) 15 and (B) 100cm depth in leachate from organically-fertilized agroforests (OAF), mineral-fertilized agroforests (MAF), and mineral-fertilized monocultures (MONO) across the study period (n=14 collections). Bars represent the standard error of the mean of three locations per farm across the study period. Differences in nitrate concentrations among the three management types were tested by ANOVA. Values that were significantly different at $p<0.05$ are indicated by different letters. Dashed line represents public health standard limit of 10mg NO_3^--N L^{-1} (WHO 1996).

Nitrate-N concentrations at 15cm explained 44% and 47% of the spatial variability in NO_3^--N concentrations at 100cm in organically- and mineral-fertilized farms, respectively, but only 25% of the spatial variability in unshaded monocultures (Figure 7). Soil N was not related to NO_3^--N or NH_4^+-N concentrations, and bioavailable P was not related to PO_4^{3-}-P concentrations at depth.

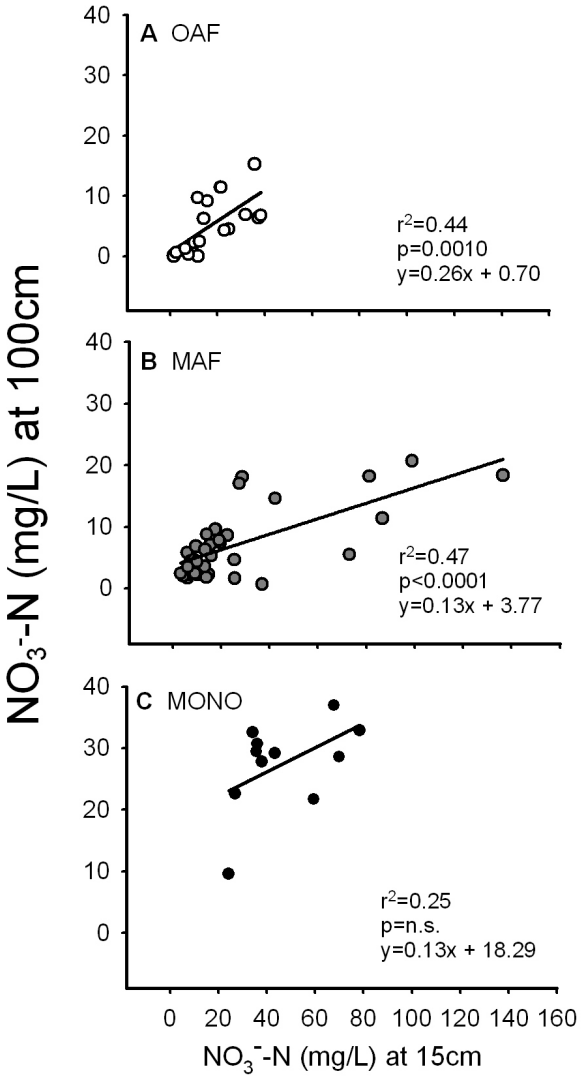


Figure 7: Correlation between nitrate concentrations at 15cm and 100cm in coffee farms in Costa Rica. Mean NO_3^--N concentrations at 100cm as a function of concentrations at 15cm in (A) organically-fertilized agroforests, (B) mineral-fertilized agroforests, and (C) mineral-fertilized monocultures.

3.3 Potential nutrient loss is higher under nitrogen fixers

Nitrate concentrations in leachate collected at 15cm under Erythrina were nearly double those collected under the other species (p=0.0020; Figure 8A). I did not observe significant differences in leachate PO_4^{3-}-P or NH_4^+-N concentrations among species. As expected, soil N and C concentrations were higher (by 17%) near *Erythrina* (N-fixer; p<0.0001, in both cases).

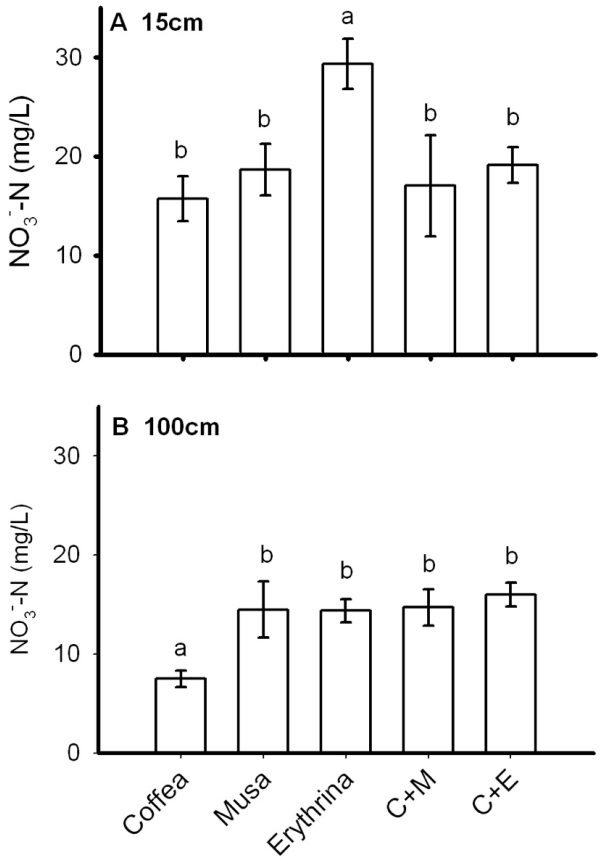


Figure 8: Nitrate concentrations in tension lysimeter leachate under five different species combinations in mineral-fertilized agroforests in Costa Rica. Mean NO_3^--N concentrations in leachate at (A) 15cm and (B) 100cm under *Coffea, Musa,* and *Erythrina* and across study period (n=14 collections). C+M indicates samples taken from under *Coffea* grown adjacent to *Musa*; C+E, samples from under *Coffea* grown adjacent to *Erythrina*. Bars represent standard error of the mean. Differences in nitrate concentrations among the five species combinations were tested by ANOVA. Values that significantly different at p<0.05 are indicated by different letters.

Available P was significantly higher in soils near shade trees than near distant *Coffea* plants ($p<0.0001$; Table 2). Soil pH was higher near *Musa* than near *Erythrina* or solitary *Coffea* plants ($p<0.0001$), and soil moisture was greatest near *Erythrina* ($p<0.0001$; Table 2).

Table 2: Soil characteristics among species (0-10cm) averaged across study period. Differences in nutrient concentrations, soil moisture, and pH among the three species combinations were tested by ANOVA. Values that were significantly different at $p<0.05$ are indicated by different letters.

	Total N (%)	Bray-1 P (µg/g)	Gravimetric Soil Moisture (dwt_{water}/dwt_{soil})	pH
Coffea	0.37 a	8.2 a	28.9 a	5.0 a
Coffea + *Musa*	0.37 a	11.3 b	30.6 a	5.5 b
Coffea + *Erythrina*	0.43 b	10.8 ab	34.8 b	5.0 a

3.4 Species mixtures may have higher nitrate losses at depth

Despite significantly higher NO_3^--N concentrations at 15cm, leachate concentrations at 100cm were not elevated under Erythrina. However, NO_3^--N concentrations were significantly lower (by one half) below Coffea plants surrounded by Coffea compared to Coffea grown adjacent to shade trees (Figure 8B). I observed the smallest drawdown in nitrate concentrations between 15 and 100cm in Musa plants (9 mg NO_3^--N L^{-1}) and the largest under Coffea near Erythrina (25 mg NO_3^--N L^{-1}; $p=0.056$). As for shallow depths, there was no effect of species on PO_4^{3-}-P or NH_4^+-N concentrations in deep soil leachate, or in the differences in concentrations between shallow and deep soils.

4.0 Discussion

4.1 Potential nitrogen loss is lower in organically-fertilized than in mineral-fertilized agroforests

The mineral-fertilized agroforests in this study received two and a half times more N than the organically-fertilized agroforests, a difference similar to that observed among 28 farms in the region (Chapter 2, Figure 4A). Larger N inputs corresponded to higher soil N and leachate NO_3^--N concentrations. As more N accumulates in the deeper soil layers in mineral-fertilized agroforests (Chapter 5, Figure 2), they may have greater potential N loss than organically-fertilized agroforests. In contrast to N inputs, P inputs were very similar among agroforests, and PO_4^{3-}-P concentrations did not differ between mineral and organically-fertilized agroforests. As P is conservatively cycled in highly weathered tropical soils (Jordan 1982; Vitousek 1984), it is not surprising that leachate PO_4^{3-}-P concentrations were low and similar to near by tropical forests (around 0.12mg P L^{-1}; Radulovich and Sollins 1991). In agroforests, it appears that regardless of origin, additional P is either (1) being adsorbed onto clay minerals and effectively removed from the P cycle (Uehara and Gillman 1981) and/or (2) quickly assimilated by plants and microbes. That is, potential P loss is low in both mineral and organically-fertilized agroforests.

Despite similar PO_4^{3-}-P concentrations in leachate, bioavailable P concentrations were higher in organically-fertilized agroforests. This result is somewhat surprising, especially given that Certified Organic® farmers rely more heavily on pruning residues to nourish the crop. These residues tend to be deficient in P, and do not meet crop

demands on an annual basis (Szott and Kass 1993; Palm 1995). However, the addition of organic fertilizers may alter the soil properties and chemical reactions that control P availability. First, the application of manure can result in greater base cation exchange, increasing the pH of soils (Kretzschmar et al. 1991; Oehl et al. 2002), while the application of mineral fertilizers in the form of NH_4-NO_3 acidifies mineral-fertilized soils relative to organically-fertilized systems, consistent with my observations here (Fließbach et al. 2007). In more acidic (mineral-fertilized) soils, P binds more readily to iron and aluminum oxides; as P is adsorbed onto the surfaces of clay minerals such as kaolinite, it is no longer bioavailable (Brady 1974; Uehara and Gillman 1981). Second, P sorption capacity of soils may be reduced when amended with organic residues due to the complexation of fixation sites by organic and humic acids originating from organic residues (Iyamuremye et al. 1996; Reddy et al. 1980). As P sorption capacity decreases, bioavailable P increases (Easterwood and Sartain 1990). Mineral-fertilized farms tended to have larger iron oxide pools (Chapter 5, Table 1) which readily adsorb P and lead to lower concentrations of bioavailable P. Despite more binding sites in mineral-fertilized farms, chemisorbed P (NaOH- and HCl-fractions) tended to be higher in organically-fertilized agroforests (Chapter 5, Table 1; n.s.), suggesting that the application of organic fertilizers may result in a build up of P in binding sites associated with Fe and Al complexes in the clays. This supports the theory that P adsorption capacity may be lower in organically-fertilized soils (Iyamuremye et al. 1996). Third, the relative proportion of nutrients added via fertilizers differs greatly between management types. Mineral fertilizers have an N:P ratio roughly 5:1 while organic fertilizers have a ratio of 2:1. In

general, microbes require 7 units of N for every unit of P (Cleveland and Liptzin 2007), much closer to the stoichiometry of mineral fertilizers. Therefore, the accumulation of P in organically-fertilized soils may also be the result of low availability of N in relation to P. That is, microbes must seek out at least 5 additional units of N for every unit of P to meet their stoichiometric demands and to function (e.g. to metabolize, perform decomposition) leading to lower N and higher P concentrations in soils. Thus, despite the addition of pruning residues in both agroforest types, management alters soil properties and chemistry, with significant consequences for short- and long-term P supply.

4.2 Presence of trees, not fertilizer type, determines actual nutrient loss at depth

The monoculture farm received three times more N per year than the mineral-fertilized agroforests and eight times more N per year than the organically-fertilized agroforests (Table 1). Therefore, it is not surprising that leachate NO_3^--N concentrations below the root zone (100cm) were three times higher in monocultures. All management types showed significantly lower NO_3^--N concentrations at depth. However, only agroforests were able to reduce concentrations below the public health standard limit of 10mg NO_3^--N L^{-1}. Further, at 100cm, NO_3^--N concentrations were the same in organically- and mineral-fertilized agroforests. Clearly the structure of an agroforest was capable of buffering the effects of fertilizer application. That is, regardless of the manner (e.g. N fixation vs. fertilizer application) or form (mineral or organic) of N inputs, shade trees are capable of preventing N loss. In agroforests, NO_3^--N concentrations in soil solutions collected below the root zone were well-predicted (44 and 47% variability explained) by NO_3^--N concentrations in solutions collected in the most active soil layer

(at 15cm; Figure 7). Nitrate-N concentrations in agroforests were more or less 3 times lower at depth, and in monocultures, about 1.3 times lower at depth.

Phosphate-P concentrations were significantly reduced at depth and did not differ among cultivation types. Low leachate PO_4^{3-}-P concentrations reflect conservative cycling in productive systems where P may become limited in the face of repeated heavy inputs of N. I did not find any strong predictors of PO_4^{3-}-P concentrations at depth, but concentrations are too low to endanger water quality or to lead to eutrophication.

4.3 Potential nutrient loss is higher under nitrogen fixers

Although farmers may not understand the underlying chemistry, they are keenly aware of the nutritional benefits imparted by the presence of N-fixers (Albertin and Nair 2004). Enhanced N availability in the soil and higher concentrations in soil solution under *Erythrina* trees (Table 2; also see Payán et al. 2009 for more detailed spatial effects) is both a direct result of N-fixation and an indirect effect of the transfer of high quality leaves during annual prunings. *Erythrina* leaves are high in N and P and decompose quickly, rapidly releasing nutrients to the soil (Chapter 6, Figure 4). Nitrogen fixed by *Erythrina* may elevate surface soil solution concentrations potentially transferring more N to deeper soil layers.

In addition, PO_4^{3-}-P concentrations in soil water tended to be 26% higher under *Erythrina* than *Coffea* plants (p value n.s.). Suggesting *Erythrina* may also play an important role in P cycling in agroforestry systems (Chapter 6, Figure 11), especially when mychorrhizal associations are considered (Danso et al. 1992). For example, bioavailable soil P *was* higher under *Coffea* near shade trees compared to distant *Coffea*

plants. Species diversity increases potential for symbiotic associations (Eriksson 2001). Phosphorus availability may be greater in species mixtures as mychorrizal fungi are capable of liberating P not available to plants through enhanced mineral weathering (Jongmans et al. 1997) and by associating with bacteria that secrete phosphatases or excrete organic acids (Smith et al. 1997). In addition, greater quantities of nutrient-rich litter and pruning residues are produced under species mixtures, which over time may enhance the quality of surrounding soils. For example, initial P-release from decomposing *Coffea* leaves is much higher when mixed with *Musa* and *Erythrina* (Chapter 6, Figure 11).

4.4. Species mixtures have greater nitrate losses at depth

Although I observed elevated NO_3^--N concentrations under *Erythrina* in shallow soils, there was no evidence of enhanced leaching under N-fixers below the root zone (100cm depth). Therefore, biota appear to efficiently utilize N derived directly or indirectly from N-fixing trees in agroforestry systems, and inclusion of N-fixing species will not contribute to groundwater contamination.

Nitrate-N concentrations were lower under *Coffea* plants distant from shade trees. I expected concentrations to be lower under shade trees, as their roots can access nutrients at deeper soil layers. However, increasing diversity does not necessarily guarantee enhanced nutrient retention (Hooper and Vitousek 1998; Bigelow & Ewel *in press*). The shallow, lateral roots of shade trees are quite extensive, and easily span the 5m to the solitary *Coffea* plants studied (Jonsson et al. 1988; Dhyani et al. 1990; Schroth et al. 1995; personal observation). Due to the equilibrating effect of root activity in

surface soils, it is not surprising that I did not observe significant differences in leachate concentrations under *Coffea* plants at 15cm. However, vertical root distributions vary among species, and thus species effects on nutrient loss may be better examined at depth. For example, the low concentrations of NO_3^--N under distant *Coffea* plants (high percent NO_3^--N recovery) reflect the high crop N-demand in microsites where less biological N-fixation occurs and smaller quantities of high-quality litter are concentrated (as a function of distance from shade trees). In sum, there is a great deal of spatial variability in nutrient leaching and soil availability in agroforests. Researchers should take care to spatially standardize sampling protocols in order to accurately compare and contrast nutrient losses from these systems.

4.5 Temporal variation in leachate and soil nutrients have different drivers

Despite spatial correlation between high soil N and leachate NO_3^--N concentrations, their temporal patterns were not correlated. Surface leachate concentrations increased slightly in January following the December-January prune and during the January-February fertilization. However, the highest NO_3^--N concentrations coincided with the May-June fertilizer application, which followed a prolonged dry period (Figure 9A). This suggests that fertilizer N not assimilated by plants or microbes was mobilized and "flushed" out of the surface soils come the first heavy rains (Greenland 1958; Robinson 1960). Phosphate (and ammonium) concentrations increased 10-fold during the driest period (before the May-June fertilizer application; Figure 9A). As phosphate is tightly bound to soil colloids (only a small fraction is mobile), and in a typical month, this soluble fraction is diluted to a concentration roughly 0.08 mg P L^{-1}.

However, in the dry months (<3.5mm per day) soluble P was diluted in less water leading to higher concentrations at the end of the dry period. In contrast to leachate NO_3^--N concentrations, which are 200 times higher and are sensitive to management interventions, PO_4^{3-}-P concentrations in leachate are low and controlled by soil chemistry. However, in both cases, seasonal water regimes play a key role in determining leachate concentrations.

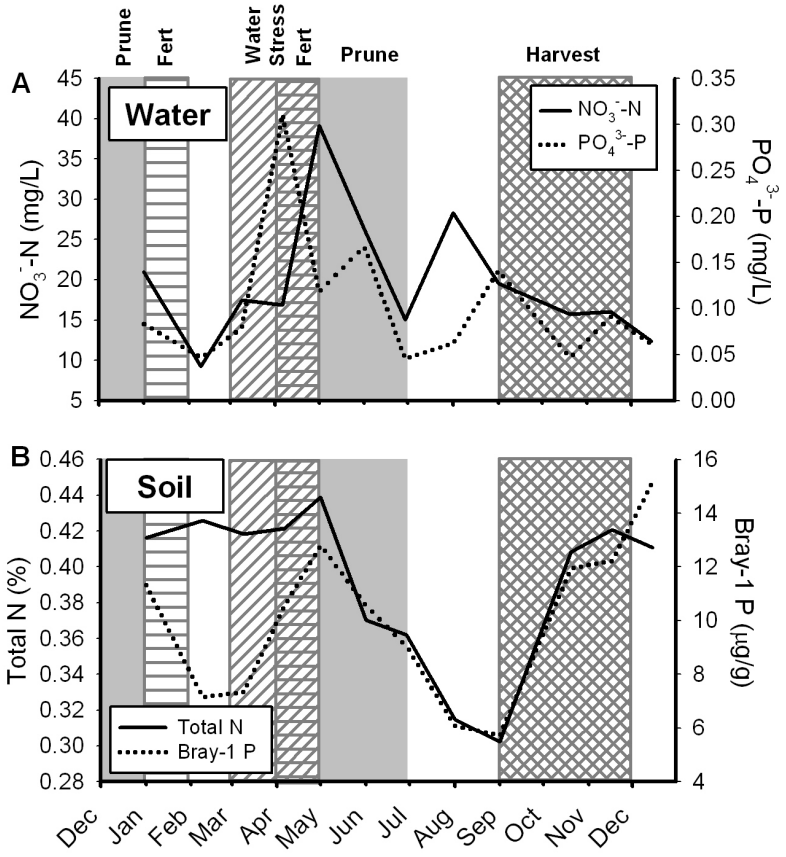


Figure 9: Conceptual diagram of soil and leachate nutrient concentrations in Costa Rican coffee agroforests. Mean (A) NO_3^--N and PO_4^{3-}-P concentrations in leachate and (B) total N and Bray-1 P concentrations in soils across mineral and organically-fertilized agroforests plotted with the timing of management activities. Diagonal lines represent period of water stress <100mm per month (March through mid-May), shaded areas represent pruning periods (December-January and May-July), horizontal lines indicate fertilization events (January-February and May-June), and cross-hatches represent harvest period (September-December with the harvest peak in November).

Temporal variations in soil N more closely tracked plant nutrient demand (Figure 9B). The draw-down of soil N (by 31%) between May and August is likely the result of enhanced Coffea uptake as the crop matures (i.e. berries ripen), which appears to demand high quantities of nutrients. Soil N increased near the end of the harvest (and a reduction in reproductive growth), due to a reduction in nutrient demand by Coffea plants (Figure 9B). Soil P tracked soil N patterns from May though November, suggesting that plant nutrient demand may also drives variation in bioavailable P. However, soil P also declined dramatically between January and March, which is better explained by changes in soil chemistry due to pruning regimes. Therefore, the May-November decline in bioavailable P could be driven by an entirely different mechanism. The pruning of shade trees is typically dictated by Coffea phenology as a means to increase light-availability during critical growth periods (i.e. in December-January to promote flowering and fruit set and in May-July to promote berry ripening; Beer 1988). Several studies indicate the importance of light availability for promoting reproductive growth in coffee plants (Montoya et al. 1961, Castillo and López 1966, Cannell 1975). The decline in bioavailable P following prunings is likely the result of enhanced microbial demand for P during the initial stages of leaf decomposition (Chapter 6). Therefore, while soil N declined as plant nutrient demand increased during the berry-ripening process, similar declines in soil P could be driven by higher microbial nutrient demand during the initial phases of the decomposition process.

5.0 Conclusions

Both farm management and species effected soil and leachate nutrients. Although mineral-fertilized agroforests had higher soil and leachate N in concentrations surface soils than organically-fertilized agroforests, nitrate concentrations at 1 meter were similar among agroforests. This suggests that the presence of trees play an important role in reducing N losses from agroforests. This hypothesis is further supported by the elevated nitrate concentrations in monoculture farms where coffee is grown in the absence of trees. Although bioavailable P concentrations were higher in organically-fertilized farms, there were no effects of fertilizer management on phosphate concentrations. I also observed species-specific effects on leachate nitrate concentrations in both shallow and deep soil water. N-fixing *Erythrina* elevated leachate nitrate concentrations in surface soils, however this trend was not apparent at depth. On the other hand, low leachate concentrations at depth from below *Coffea* plants several meters away from shade trees show that N is in greater demand in microsites where less high quality litter is concentrated.

References

Albertin, A., and P. K. R. Nair. 2004. Farmers' perspectives on the role of shade trees in coffee production systems: an assessment from the Nicoya peninsula, Costa Rica. Human Ecology 32:443-463.

Aronsson, H., G. Torstensson, and L. Bergström. 2007. Leaching and crop uptake of N, P and K from organic and conventional cropping systems on a clay soil. Soil Use and Management 23:71-81.

Babbar, L. I., and D. R. Zak. 1995. Nitrogen loss from coffee agroecosystem in Costa Rica: leaching and denitrification in the presence and absence of shade trees. Journal of Environmental Quality 24:227-233.

Bene, J. G., H. W. Beall, and A. Cote. 1977. Trees, Food and People. IDCR, Ottawa, Canada.

Berendse, F. 1979. Competition between plant populations with different rooting depths I. Theoretical considerations. Oecologia 43:19-26.

Boyce, J., A. Fernández, E. Fürst, and O. Segura. 1994. Café y desarrollo sostenible. Universidad Nacional, San Jose.

Brady, N. C. 1974. The nature and properties of soils. MacMillan Publishing Co. New York.

Bray, R. M., and L. T. Kurtz. 1945. Determination of total organic and available forms of phosphorus in soils. Soil Science 59:39-45.

Campo, J., M. Maas, V. J. Jaramillo, A. Martnez-Yrizar, and J. Sarukhan. 2001. Phosphorus cycling in a Mexican tropical dry forest ecosystem. Biogeochemistry 53:161-179.

Cannell, M. G. R. 1975. Crop physiological aspects of coffee bean yield: a review. Journal of Coffee Research 5:7-20.

Carvajal, J. F.. 1984. Cafeto-cultivo y fertilizacion. Instituto de la potasa, Berna, Suiza.
Castillo, Z. J., and A. R. López. 1966. Nota sobre el efecto de la intensidad de la luz en la floración del cafeto. Cenicafé (Col.) 17:5160.

CEPAL, 2002 CEPAL (Comisión Económica para América Latina y el Caribe), 2002. Centroamérica: El Impacto de la Caída de los Precios del Café en 2001. URL: http://www.fondominkachorlavi.org/cafe/docs/cepal2002.pdf.

Chesney, P. 2008. Nitrogen and fine root length dynamics in a tropical agroforestry system with periodically pruned *Erythrina poeppigiana*. Agroforestry Systems 72:149-159.

Clark, M. S., W. R. Horwath, C. Shennan, K. M. Scow, W. T. Lantni, and H. Ferris. 1999. Nitrogen, weeds and water as yield-limiting factors in conventional, low-input, and organic tomato systems. Agriculture, Ecosystems & Environment 73:257-270.

Clark, K. L., N. M. Nadkarni, D. Schaefer, and H. L. Gholz. 1998. Atmospheric deposition and net retention of ions by the canopy in a tropical montane forest, Monteverde, Costa Rica. Journal of Tropical Ecology 14:27-45.

Cleveland, C. C., and D. Liptzin. 2007. C: N: P stoichiometry in soil: is there a "Redfield ratio" for the microbial biomass? Biogeochemistry 85:235-252.

Danso, S., G. D. Bowen, and N. Sanginga. 1992. Biological nitrogen fixation in trees in agro-ecosystems. Plant and Soil 14:177-196.

Dhyani, S. K., P. Narain, and R. K. Singh. 1990. Studies on root distribution of five multipurpose tree species in Doon Valley, India. Agroforestry Systems 12:149-161.
Draye, X. 2002. Banana roots: Architecture and genetics. Plant Roots: The Hidden Half 3:261-277.

Drinkwater, L. E., D. K. Letourneau, F. Workneh, A. H. C. Van Bruggen, and C. Shennan. 1995. Fundamental differences between conventional and organic tomato agroecosystems in California. Ecological Applications 5:1098-1112.

Easterwood, G. W., and J. B. Sartain. 1990. Clover residue effectiveness in reducing orthophosphate sorption on ferric hydroxide coated soil. Soil Science Society of America journal 54:1345-1350.

Eriksson 2001. Arbuscular mycorrhiza in relation to management history, soil nutrients and plant species diversity. Plant Ecology 155:129-137.

Eviner, V. T., and F. S. Chapin III. 2003. Functional matrix: a conceptual framework for predicting plant effects on ecosystem processes. Annual Review of Ecology and Systematics 34:455-485.

Fließbach, A., H. -R. Oberholzer, L. Gunst, and M. Mader. 2007. Soil organic matter and biological soil quality indicators after 21 years of organic and conventional farming. Agriculture, Ecosystems and Environment 118:273-284.

Foster, S. S. D., and R. C. A. Hirata. 1988. Groundwater pollution risk assessment; a methodology using available data. CEPIS Technical Report, Lima.

Garriz, P. I. 1978. Distribución radicular de tres cultivares de Coffea arabica L. de 24 anõs de edad, en un suelo limo-arcilloso. Agronomía Tropical 9:91-103.

Glover, N., and J. Beer. 1986. Nutrient cycling in two traditional Central American agroforestry systems. Agroforestry Systems 4:77-87.

González-Aguilar, M. A., R. Pacheco-Salazar, and J. A. Briceño-Salazar. 1985. Efecto de la urea y del nitrato de amonio sobre la movilidad de nitrato, potasio, calcio y magnesio en un Dystrandept de Costa Rica. Agronomía Costarricense. 9:241-252.

Goodman, D. 1999. Agro-food studied in the 'Age of Ecology': Nature, corporeality, bio-Politics. Sociologio Ruralis 19:17-38.

Greenland, D. J. 1958. Nitrate fluctuations in tropical soils. The Journal of Agricultural Science 50:82-92.

Harmand, J. M., H. Avila, E. Dambrine, U. Skiba, S. de Miguel, R. V. Renderos, R. Oliver, F. Jimenez, and J. Beer. 2007. Nitrogen dynamics and soil nitrate retention in a *Coffea arabica-Eucalyptus deglupta* agroforestry system in Southern Costa Rica. Biogeochemistry 85:125-139.

Hooper, D. U., and P. M. Vitousek. 1998. Effects of plant composition and diversity on nutrient cycling. Ecological Monographs 68:121-149.

Imbach, A. C., H. W. Fassbender, R. Borel, J. Beer, and A. Bonnemann. 1989. Modelling agroforestry systems of cacao (*Theobroma cacao*) with laurel (*Cordia alliodora*) and poró (*Erythrina poeppigiana*) in Costa Rica IV. Water balances, nutrient inputs and leaching. Agroforestry Systems 8:267-287.

International Coffee Organization. 2010. Annual and monthly averages of coffee process. Retrieved November 31, 2010, from http://www.ico.org/prices/p2.htm.

Iyamuremye, F., R. P. Dick, and J. Baham. 1996. Organic amendments and phosphorus dynamics: I. Phosphorus chemistry and sorption. Soil Science 161:426.

Iyamuremye, F., R. P. Dick, and J. Baham. 1996. Organic amendments and phosphorus dynamics: II. Distribution of soil phosphorus fractions. Soil Science 161:436.

Jongmans, A. G., N. Van Breemen, U. Lundström, P. A. W. Van Hees, R. D. Finlay, M. Srinivasan, T. Unestam, R. Giesler, P. A. Melkerud, and M. Olsson. 1997. Rock-eating fungi. Nature 389:682-683.

Jonsson, K., L. Fidjeland, J. A. Maghembe, and P. Högberg. 1988. The vertical distribution of fine roots of five tree species and maize in Morogoro, Tanzania. Agroforestry systems 6:63-69.

Jordan, C. F. 1982. The nutrient balance of an Amazonian rain-forest. Ecology 63:647-654.

Kretzschmar, R. M., H. Hafner, A. Bationo, and H. Marschner. 1991. Long-and short-term effects of crop residues on aluminum toxicity, phosphorus availability and growth of pearl millet in an acid sandy soil. Plant and soil 136:215-223.

Liebig, M. A., and J. W. Doran. 1999. Impact of organic production practices on soil quality indicators. Journal of Environmental Quality 28:1601-1609.

Lyngbæk, A. E., R. G. Muschler, and F. L. Sinclair. 2001. Productivity and profitability of multi-strata organic versus conventional coffee farms in Costa Rica. Agroforestry Systems 53:205-213.

Mack, M. C., and C. M. D'Antonio. 2003. Exotic grasses alter controls over soil nitrogen dynamics in a Hawaiian woodland. Ecological Applications 13:154-166.

Moguel, P., and V. M. Toledo. 1999. Biodiversity conservation in traditional coffee systems in Mexico. Conservation Biology 15:11-2.

Montoya, L. A., P. G. Sylvain, and R. Umaña. 1961. Effect of light intensity and nitrogen fertilization upon growth differentiation balance in *Coffea arabica* L. Coffee 3:97-104.

Oehl, F., A. Oberson, H. U. Tagmann, J. M. Besson, D. Dubois, P. Mäder, H. R. Roth, and E. Frossard. 2002. Phosphorus budget and phosphorus availability in soils under organic and conventional farming. Nutrient Cycling in Agroecosystems 62:25-35.

Palm, C. A. 1995. Contribution of agroforestry trees to nutrient requirements of intercropped plants. Agroforestry Systems 30:105-124.

Payán, F., D. L. Jones, J. Beer, and J. M. Harmand. 2009. Soil characteristics below Erythrina poeppigiana in organic and conventional Costa Rican coffee plantations. Agroforestry systems 76:81-93.

Radulovich, R., and S. Sollins. 1991. Nitrogen and phosphorus leaching in zero-tension drainage from a humid tropical soil. Biotropica 23:84-47.

Reddy, K. R., M. R. Overcash, R. Khaleel, and P. W. Westerman. 1980. Phosphorus adsorption-desorption characteristics of two soils utilized for disposal of animal wastes. Journal of Environmental Quality 9:86-92.

Reganold, J. P., A. S. Palmer, J. C. Lockhart, and A. N. Macgregor. 1993. Soil quality and financial performance of biodynamic and conventional farms in New Zealand. Science 260:344.

Reynolds-Vargas, J., J. Fraile-Merino, and R. Hirata. 2006. Trends in nitrate concentrations and determination of its origin using stable isotopes (^{18}O and ^{15}N) in groundwater of the western Central Valley, Costa Rica. Ambio 35:229-236.

Reynolds-Vargas, J. S., and D. D. Richter. 1995. Nitrate in groundwaters of the Central Valley, Costa Rica. Environment International 21:71-79.

Reynolds-Vargas, J. S., D. D. Richter, and E. Bornemisza. 1994. Environmental impacts of nitrification and nitrate adsorption in fertilized andisols in the Valle Central of Costa Rica. Soil Science 157:289.

Robinson, J. B. D. 1960. Nitrogen studies in a coffee soil I. Seasonal trends of natural soil nitrate and ammonia in relation to crop growth, soil moisture and rainfall. The Journal of Agricultural Science 55:333-338.

Russell, A. E., and J. J. Ewel. 1985. Leaching from a tropical andept during big storms: a comparison of three methods. Soil Science 139:181-189.

Salas, R., E. Bornemisza, F. Zapata, V. Chaves, and A. Rivera. 2002. Absorción del fertilizante nitrogenado por la planta de café y su influencia sobre la contaminacion de las águas subterraneas. Manejo Integrado de Aguas Subterraneas. EUNED, San José, Costa Rica 89-104.

Schroth, G. 1998. A review of belowground interactions in agroforestry, focusing on mechanisms and management options. Agroforestry systems 43:5-34.

Schroth, G., D. Kolbe, B. Pity, and W. Zech. 1995. Searching for criteria for the selection of efficient tree species for fallow improvement, with special reference to carbon and nitrogen. Nutrient Cycling in Agroecosystems 42:297-314.

Schroth, G., E. Salazar, and J. P. Da Silva. 2001. Soil nitrogen mineralization under tree crops and a legume cover crop in multi-strata agroforestry in Central Amazonia: Spatial and temporal patterns. Experimental Agriculture 37:253-367.

Selvaradjou, S-K., L. Montanarella, O. Spaargaren, and D. Dent. 2005. European Digital Archive of Soil Maps (EuDASM) - Soil Maps of Latin America and Caribbean Islands. Office of the Official Publications of the European Communities, Luxembourg.

Seyfried, M. S., and P. S. C. Rao. 1991. Nutrient leaching from two contrasting cropping systems in the humid tropics. Tropical Agriculture 68:9-18.

Smith, T., H. H. Shugart, and F. I. Woodward. 1997. Plant Functional Types. Cambridge University Press, Cambridge, UK.

Sommer, K.1978. Use of radioisotopes in agriculture. Report to the Government of Costa Rica, Vienna, Austria.

Steppler, H. A. and P. K. R. Nair. 1987. Agroforestry: A Decade of Development. ICRAF, Nairobi, Kenya.

Stopes, C., E. I. Lord, L. Phillips, and L. Woodward. 2002. Nitrate leaching from organic farms and conventional farms following best practice. Soil Use and Management 18:256-263.

Szott, L. T., and D. C. L. Kass. 1993. Fertilizers in agroforestry systems. Agroforestry Systems 23:157-176.

Tiessen, H. 2008. Phosphorus in the global environment. The ecophysiology of plant-phosphorus interactions 1-7.

Torstensson, G., H. Aronsson, and L. Bergstrom. 2006. Nutrient use efficiencies and leaching of organic and conventional cropping systems in Sweden. Agronomy Journal 98:603-615.

Tucker, C. M., H. Eakin, and E. J. Castellanos. 2010. Perceptions of risk and adaptation: Coffee producers, market shocks, and extreme weather in Central America and Mexico. Global Environmental Change 20:23-32.

Turner, D. W., C. Korawis, and A. D. Robson. 1989. Soil analysis and its relationship with leaf analysis and banana yield with special reference to a study at Carnarvon, Western Australia. Fruits 44:193-203.

Uehara G and G. Gillman. 1981. The Mineralogy, Chemistry, and Physics of Tropical Soils with Variable Charge Clays. Westview Press, Boulder, Colorado.

Ulén, B., H. Aronsson, G. Torstensson, and L. Mattsson. 2005. Phosphorus and nitrogen turnover and risk of waterborne phosphorus emissions in crop rotations on a clay soil in southwest Sweden. Soil Use and Management 21:221-230.

van Kanten, R., and P. Vaast. 2006. Transpiration of arabica coffee and associated shade tree species in sub-optimal, low-altitude conditions of Costa Rica. Agroforestry Systems 67:187-202.

Vanlauwe, B., N. Sanginga, and R. Merckx. 1997. Decomposition of four Leucaena and Senna prunings in alley cropping systems under sub-humid tropical conditions: the process and its modifiers. Soil Biology and Biochemistry 29:131-137.

van Noordwijk, M., G. Lawson, A. Soumare, J. J. R. Groot, and K. Hairiah. 1996. Root distribution of trees and crops: competition and/or complementarity. In Tree-Crop Interactions. C.K. Ong and P. Huxley, Editors. CAB International: Wallingford, UK, pp 319-364.

Vedenov, D., J. Houston, and G. Cardenas. 2007. Production efficiency and diversification in Mexican coffee-producing districts. Journal of Agricultural and Applied Economics 39:749.

Vitousek, P. M. 1984. Litterfall, nutrient cycling, and nutrient limitation in tropical forests. Ecology 65:285-293.

WHO. 1996. Guidelines for Drinking Water Quality (2nd ed., Vol. 2). Health Criteria and Other Supporting Information. World Health Organization, Geneva, pp. 313–324.

Chapter 4: Trees minimize nutrient loss from coffee management systems

Abstract

The consequences of vegetation structure on ecosystem processes in agroforests are typically evaluated in contrast to monocultures with little attention paid to how management strategies can differ from farm to farm. This study was conducted on nine coffee farms in the Central Valley of Costa Rica and aimed to determine how management affects nutrient losses and what particular mechanisms prevent nutrient loss. I compared nitrogen (N) and phosphorus (P) losses among coffee agroforests amended with mineral fertilizer (n=4), agroforests amended with organic fertilizer (n=4), and one coffee monoculture amended with mineral fertilizers. Monthly N and P concentrations were measured in soil water collected monthly from tension lysimeters (at 15 and 100cm) between October 2008 and September 2009. A water balance was developed to estimate annual N and P losses from the three coffee systems. Despite differences in the quantity and form of fertilizer inputs, N and P losses at one meter did not differ significantly among agroforests (91±18 kg N ha^{-1} yr^{-1} and 1.4±0.1 kg P ha^{-1} yr^{-1}). However, estimated losses from mineral-fertilized monocultures were nearly four times as great for N and twice as great for P (307 kg N ha^{-1} yr^{-1} and 2.3 kg P ha^{-1} yr^{-1}). These results suggest that trees play a critical role in reducing leaching losses from tropical agroecosystems. Further, N losses (at 100cm) declined significantly with increasing aboveground biomass, and with shade tree biomass in particular. Phosphorus, on the other hand, declined with increasing soil iron (Fe) pools, indicating the important role of both biotic and abiotic processes in reducing nutrient loss.

1.0 Introduction

Northern Latin America is home to seven of the ten countries with the highest deforestation rates in the world (Rice 1999). Once a vast tropical forest, only 43% of this original land cover remains (FAO 2008). Land has been and continues to be cleared for agriculture and pasture, specifically the cultivation of coffee, banana, and sugarcane and the production of cattle (Myster 2004). This research focused specifically on the patterns of nutrient loss from different coffee cultivation systems in Costa Rica.

Coffee (*Coffea arabica*) is native to the Horn of Africa, and was already widely cultivated by the 14th century (Topik 2000). The Dutch brought it to the New World in the 1700s where it thrived in the mid- and high-elevation mountainous regions of tropical America (Rice 1999). Currently, coffee is the top exported commodity in Central America by dollar value ($2.4 billion; FAO 2008), and its cultivation and production supports over 8.5 million people in the region (Vedenov et al. 2007). About 2.7 million hectares of land are devoted to the cultivation of coffee across northern Latin America (FAO 2009), comprising the majority of the agricultural land cover between 500 and 2000 meters above sea level (Perfecto et al. 1996).

Coffee was traditionally cultivated as an understory shrub in an otherwise intact tropical forest (in other words, in an agroforest). However, in response to a fungal pathogen, which swept across Central America in the 1970s, coffee farmers began to "technify" their farms. In technified coffee farms overstory trees are either removed entirely or replaced by regularly-spaced, short (5-8m) service trees. Service species such as *Erythrina poeppigiana* have symbiotic associations with nitrogen (N)-fixing bacteria

which provide a source of nutrients to the system. However, their wood is of extremely low quality, and they offer little in the form of habitat for local species. Further, they are regularly pruned such that for the majority of year, the landscape essentially behaves as an unshaded monoculture. Although some agroforests remain, currently 80-85% of coffee in Central America is cultivated in monoculture or highly technified conditions (FAO 2008).

In recent years, remnant agroforests have received much attention from land managers, international development agencies, and researchers for their potential to improve environmental and economic sustainability (Hassan et al. 2005; UNFCCC 2008). Agroforests store more carbon (Nair et al. 2009), recycle greater quantities of nutrients (Palm 1995; Schroth et al. 2001), maintain higher levels of biodiversity (Perfecto et al. 1996), and have a greater potential to improve regional economic stability by diversifying livelihoods (Leakey et al. 2005; Amekawa et al. 2010) than industrialized agriculture. Nevertheless, in most research, the benefits of agroforests are assessed in comparison to regional extremes such as mature forests on the one hand and heavily fertilized monocultures on the other. Little research has investigated how management within the spectrum of agroforestry can alter ecosystem processes.

In this study, I examine how nutrient loss is affected by management practices. Farmers influence nutrient inputs (fertilization regimes), and therefore different management techniques may have consequences for on-farm nutrient loss. Three coffee management systems are considered: (1) mineral-fertilized agroforests (MAF); (2) organically-fertilized agroforests (OAF); and (3) mineral-fertilized monocultures

(MONO). Among management systems, mineral-fertilized monocultures receive the highest fertilizer amendments (300 kg N and 33 kg of P ha^{-1} yr^{-1}). Mineral-fertilized agroforests receive a third as much N and P as monocultures (average of 96 kg of N and 18 kg of P ha^{-1} yr^{-1}), and organically-fertilized agroforests receive even less (28 kg of N and 16 kg of P ha^{-1} yr^{-1}on average). Among agroforests, shade trees are pruned frequently (2-3 times a year) and aggressively (nearly all branches removed) in mineral-fertilized agroforests. Pruning is less frequent and less intense (fewer branches removed) in organically-fertilized agroforests, thus fewer nutrients are transferred to the soil, and the crop is exposed to less sun.

The observed variation in farm management strategies is likely to have consequences for the coupled cycling of nutrients, water, and energy. This research is the first to examine how different forms of agroforest management can affect nutrient loss. I accomplish this by calculating on-farm nutrient loss through direct measurements of soil water nutrients paired with farm-specific modeled water budgets. I then examine how nutrient loss differs among coffee management systems, and investigate three factors that could potentially regulate nutrient loss: fertilizer inputs, aboveground biomass, and nutrient retention capacity of the soil.

2.0 Materials and Methods

2.1 Study Sites

This study was conducted on four organically-fertilized coffee agroforests (OAF), four mineral-fertilized coffee agroforests (MAF), and one mineral-fertilized monoculture coffee plantation (MONO) located in Costa Rica's Central Valley near Turrialba

(between 9°52'N and 9°54'N and between 83°41'W and 83°42'W). In 2008, farmers were interviewed to acquire information on fertilization regimes (Table 1).

Table 1: Fertilization rates among coffee management systems in Costa Rica. Nitrogen (N) and phosphorus (P) amended as either mineral or organic fertilizer. OAF refers to organically-fertilized agroforests, MAF to mineral-fertilized agroforests, and MONO to mineral-fertilized monocultures.

		Fertilizer ($kg\ ha^{-1}\ yr^{-1}$)	
Management	**Location**	**N**	**P**
OAF	SJN	58	29
OAF	SJS	0.5	0.2
OAF	COL	84	27
OAF	COL	10	8
MAF	SJN	46	13
MAF	SJN	115	32
MAF	SJS	121	18
MAF	COL	100	8
MONO	SJN	300	62

The study region receives an average 2600mm of rainfall per year and has a mean annual temperature of 22.6°C. Rainfall is seasonal, and the dry season typically extends from February through May. The study year (12 months from October 2008-September 2009) was wetter than usual, with a cumulative rainfall of 3257mm. The region experienced an unusually wet December (708mm) and February (544mm) and very dry April (39mm; Chapter 2, Figure 1).

2.2 Nutrient Leaching

Soil solutions were collected every 28 days from October 2008 through September 2009 using tension lysimeters. Each consisted of a porous cup (SoilMoisture Corp., Goleta, CA) attached to a 3-inch PVC tube sealed at the end with a rubber stopper. In mineral- and organically-fertilized agroforests, tension lysimeters were installed at 15

and 100cm depth within 30-50cm from the base of a *Coffea* plant located at least 5m from the nearest shade tree. In Chapter 3, this distance proved sufficient to minimize effects resulting from species interactions. In the unshaded monoculture, two pairs of tension lysimeters were installed under two different *Coffea* plants within 5m of each other (Chapter 3, Figure 3). Below 100cm, root activity is negligible for *Coffea* and shade trees (Garriz 1978; Araya 2005; Lehmann 2003). In this study, nutrients in solution at this depth are defined as lost to the system.

Tension lysimeters were filled with distilled water before each sampling period. The day before sample collection, they were purged of any remaining water, and an internal pressure of -0.05 to -0.06 MPa was applied using a hand-held vacuum pump. The following day, the soil solutions were extracted from the lysimeters. All samples were frozen prior to analysis at the University of Virginia.

Samples were filtered through a 40μm filter to remove any debris and analyzed for inorganic NO_3^- and NH_4^+ on a LACHAT QuikChem (LACHAT Instruments Loveland, CO). A potassium persulfate digestion (also on filtered samples) converted organic P to an inorganic form. Analyzing this solution on a LACHAT yielded total P concentration.

Nitrogen fluxes (also called "losses" when referring to 100cm fluxes) were calculated by adding NO_3^--N and NH_4^+-N concentrations for a given sample period, expressing them on an area basis, and summing over the year. To determine area-based estimates, the amount of drainage water (details below) was multiplied by the soil water N concentration at 15 and 100cm for the same sampling period. Organic N was not

considered for this analysis. Phosphorus fluxes were calculated in the same manner, by multiplying total P concentrations at 15 and 100cm by the drainage water associated with a given sample period. Data were gap-filled where necessary by farm-farm regressions.

2.3 Water balance model

To estimate drainage *D* (in mm), I constructed a water balance.

$$D = P - I - T - OF \quad \text{(Eq. 1)}$$

where *P* is the precipitation, *I* is canopy interception, *T* is transpiration, and *OF* is the water lost during infiltration excess overland flow (Hortonian overland flow).

2.3.1 Precipitation

Each farmer recorded daily rainfall from a graduated rain gauge located in an open area near the farmer's house. Rainfall data for each neighborhood was screened for errors, averaged, and compared to CATIE (Centro Agronómico Tropical de Investigación y Enseñaza) meteorological data. According to this analysis, each farm received comparable rainfall inputs, and on-farm data did not differ systematically from CATIE's data. Therefore, CATIE's meteorological data were used to compute water budgets because of the fine temporal resolution available (necessary for some parts of the model).

2.3.2 Evapotranspiration

Interception was estimated at 20% of precipitation based on basal area of coffee plants and shade trees (Lin and Richards 2007; van Kanten and Vaast 2006). Interception in full sun farms was estimated at 17% of precipitation (Harmand et al. 2007). Transpiration rates were modeled using sap flow data collected from a *Coffea-Erythrina* agroforest in southern Costa Rica (van Kanten and Vaast 2006). Rainfall in that study

was not correlated with transpiration rates and was very similar in quantity to this study (3500mm compared to 3300mm per year; respectively). Transpiration rates were reported in mm day^{-1} for *Coffea* and *Erythrina* growing together, but were originally calculated by scaling individual plant (*Coffea* or *Erythrina*) water consumption (l d^{-1} $plant^{-1}$) by the number of plants (of each species) per hectare. *Coffea* density was reported as 5000 plants ha^{-1} and *Erythrina* as 156 plants ha^{-1}. Therefore, I was able to calculate transpiration rates for individual *Coffea* plants or *Erythrina* trees (mm·ha $plant^{-1}$ d^{-1}). These rates were then scaled by the plant (and tree) density of each study farm (Table 5).

Table 2: Estimated daily transpiration rates for *Coffea* and *Erythrina* in agroforest in southern Costa Rica (based on van Kanten and Vaast 2006). Calculated transpiration rates (mm·ha $plant^{-1}$ d^{-1}) for *Coffea* and *Erythrina* based on plant density in the study coffee agroforest in southern Costa Rica.

	Transpiration rate mm·ha $plant^{-1}$ d^{-1}	
	Coffea (with *Erythrina*)	*Erythrina* (with *Coffea*)
Sep	0.00026	0.0031
Oct	0.000292	0.0022
Nov	0.000334	0.0037
Dec	0.00027	0.0029
Jan	0.00043	0.0045
Feb	0.000466	0.0041
Mar	0.000278	0.0051
Apr	0.000816	0.0008
May	0.000332	0.0031
Jun	0.00036	0.0037
Jul	0.0005	0.0039
Jul	0.0005	0.0039
Aug	0.000292	0.0042
Sep	0.00026	0.0031

The sum of *Coffea* and *Erythrina* transpiration rates resulted in an estimate of the combined daily transpiration rate for the shaded farms (Eq. 2).

$$T_{100} = [T_{Cof}(\text{mm·ha plant}^{-1}\,\text{d}^{-1})*\text{plants ha}^{-1} + T_{Eryth}(\text{mm·ha plant}^{-1}\,\text{d}^{-1})*\text{trees ha}^{-1}]*28\text{d} \quad \text{(Eq. 2)}$$

However, this model tended to underestimate transpiration rates in coffee monocultures as the monoculture *Coffea* plants in the study were not fully mature. Siles (et al.) 2010 estimated that transpiration rates in mature *Coffea* grown in monoculture were only reduced by 10% compared to an agroforest (4722 *Coffea* plants ha^{-1}; 278 trees ha^{-1}). Therefore, I modeled a farm with this plant and tree density and reduced monthly transpiration rates by 10% to estimate monoculture transpiration rates (Harmand et al. 2007).

Finally, when soils are at field capacity, monthly evapotranspiration in coffee agroforests is roughly 100mm (3.5mm per day, van Kanten and Vaast 2006). Rainfall depths prior to April and May 2009 collection were 17 and 63mm, respectively. Thus, plants were likely water-stressed at these times, and drainage was estimated as zero for both months in all farms.

Based on the literature, I estimated that 50% of the root mass lay between 0 and 15cm depth (Table 3). Thus, I assumed that one half of the transpiration comes from water consumption in the first 15cm (Eq. 3). I assumed all transpired water originated between 0-100cm.

$$T_{15} = T_{100}*0.5 \quad \text{(Eq. 3)}$$

Table 3: Fraction of roots in *Coffea* and shade species in the top 15cm of soil as reported in the literature. These data were used to estimate that 50% of roots are found in the top 15cm, and were used to correct for transpiration rates at this depth.

Species	Fraction of total root mass above 15cm depth	Reference
Coffea arabica	0.4	Lehmann et al 2003; Garriz 1978
Erythrina poeppigiana	0.7	Chesney 2008
Musa acuminata	0.4	Araya 2005
Coffea-Erythrina-Musa	0.5	Average of *Coffea*, *Erythrina, and Musa*

Bare soil evaporation was insignificant in the shaded plantations due to the high density of trees. However, bare soil evaporation could potentially contribute to water loss in the coffee monoculture. In a monoculture coffee plantation of similar density, Marin et al. (2005) derived a relationship between total evapotranspiration ($ET=I+T$) and crop transpiration (T),

$$ET=T/0.76 \quad \text{(Eq. 4)}$$

I assumed that during periods between rainfall events,

$$E_{bare}=ET-T \quad \text{(Eq. 5)}$$

For each month, I estimated bare soil evaporation rates and subtracted this quantity from the total amount draining out of the coffee monoculture farm.

3.3.3 Overland Flow

To characterize the variability in hydraulic properties within each farm, infiltration was measured using a Decagon mini disk infiltrometer (-3.0cm suction; Decagon Devices, Pullman, USA). Three measurements were taken within 50cm of each of the tension lysimeter pairs (on dry days following rainy days; November-December 2010). The data from the infiltration measurements were used to plot the cumulative

infiltration curves for each sampling point and determine the saturated hydraulic conductivity (K_{sat}; Table 4). I compared bihourly rainfall data (CATIE meteorological data; corrected for interception) to the computed K_{sat} (mm $120min^{-1}$) for each farm to determine those periods when rainfall exceeded infiltration rates (Hortonian overland flow). Water volume estimated from times when rainfall exceeded infiltration rates was subtracted from the final value of drainage water.

Table 4: Saturated hydraulic conductivity (K_{sat}) in study farms in Costa Rica. Infiltration rates were calculated using a mini-disk infiltrometer, and then values converted to K_{sat}. Bi-hourly rainfall intensity was compared to K_{sat} to determine infiltration excess overland flow (in mm). OAF refers to organically-fertilized agroforests, MAF to mineral-fertilized agroforests, and MONO to mineral-fertilized monocultures.

Management	Location	K_{sat} (mm/hr)
OAF	San Juan Norte	8.7
OAF	San Juan Sur	13.8
OAF	Colorado	12.9
OAF	Colorado	5.8
MAF	San Juan Norte	4.1
MAF	San Juan Norte	13.7
MAF	San Juan Sur	20.7
MAF	Colorado	8.7
MONO	San Juan Norte	7.3

2.5 Factors regulating nutrient loss from coffee farms

While the addition of fertilizers may enhance nutrient loss, biotic and abiotic factors may act to prevent nutrient loss from agroecosystems. I calculated (1) aboveground shade tree biomass and aboveground total biomass (shade trees plus crop plants) as a potential biotic factor and (2) total soil iron (Fe) pools as a potential abiotic (chemical) factor preventing on-farm nutrient loss. In June-August of 2008, $100m^2$ plots were established in each farm. I randomly selected 10 coffee stems and recorded their

diameter at 15cm as well as height. Similarly, diameter at breast height (dbh) was measured for all shade trees (*Erythrina* and *Musa*) in the plot. Individuals of all species were counted, and shade tree and coffee density were scaled up to a hectare. Aboveground biomass was estimated using allometric equations derived for *Coffea* (Segura et al. 2006; Chapter 2, Appendix A.1); *Erythrina* (Frank and Eduardo 2003; Chapter 2, Appendix A.2), and *Musa* (Hairiah et al. 2001; Chapter 2, Appendix A.4; Table 5). Pruning affects tree biomass, and the equation used to calculate *Erythrina* biomass included a term for the level of pruning (Chapter 2, Appendix A.2). Pruning intensity was categorized into two levels: *partial* (all but 3-5 of the tallest branches removed); and *total* (all branches removed; Table 5).

Table 5: Shade tree pruning intensity and aboveground biomass on study farms in Costa Rica. Pruning intensity refers to the partial (3-5 of the tallest branches removed) or total (all branches removed) removal of *Erythrina* branches. Data on pruning techniques were gathered from farmer interviews. Aboveground biomass was calculated by pairing field measurement with allometric equations. OAF refers to organically-fertilized agroforests, MAF to mineral-fertilized agroforests, and MONO to mineral-fertilized monocultures.

			Density (plants ha^{-1})		**Aboveground biomass (Mg ha^{-1})**		
Mgmt	**Location**	**Pruning intensity**	***Coffea***	***Eryth***	***Coffea***	**Shade**	**Total**
OAF	San Juan Norte	partial	5600	400	5.0	148.8	153.8
OAF	San Juan Sur	partial	4900	900	20.1	206.3	226.5
OAF	Colorado	partial	5300	200	14.7	128.0	142.7
OAF	Colorado	partial	3300	700	9.8	361.2	371.0
MAF	San Juan Norte	partial	6600	500	24.8	406.0	430.7
MAF	San Juan Norte	total	4800	700	19.6	282.5	302.0
MAF	San Juan Sur	total	4000	400	20.1	87.1	107.2
MAF	Colorado	total	5000	600	13.6	135.4	148.9
MONO	San Juan Norte	-	5000	0	14.5	0.0	14.5

In June-July of 2008, soil pits were dug in agroforests to 80cm within 50cm of coffee plants, and soil samples were taken at six depths (Chapter 5). Bulk density was measured using a steel canister. Non-crystalline plus crystalline (total) Fe concentrations were estimated by a 16-hour dithionite-citrate-bicarbonate (DCB) extraction (Darke and Walbridge 1994). Resulting solutions were analyzed on an atomic absorption spectrometer for Fe. Using Fe concentrations and bulk density, I calculated the size of the DCB-extractable Fe pool (in kg ha^{-1}) for each farm. The owner of the coffee monoculture did not agree to soil-pit excavation, and therefore, the coffee monoculture was excluded from analysis of the effect of soil Fe on nutrient loss.

Table 6: DCB-extractable Fe pools in soils (0-80cm) of study farms in Costa Rica. DCB-extractable Fe concentrations were scaled up to Mg ha^{-1} using soil bulk density (see Chapter 5). OAF refers to organically-fertilized agroforests, MAF to mineral-fertilized agroforests, and MONO to mineral-fertilized monocultures.

Management	Location	DCB-extractable Fe (Mg ha^{-1})
OAF	San Juan Norte	537
OAF	San Juan Sur	620
OAF	Colorado	3912
OAF	Colorado	448
MAF	San Juan Norte	487
MAF	San Juan Norte	540
MAF	San Juan Sur	439
MAF	Colorado	536
MONO	San Juan Norte	537

2.6 Statistical Approach

Farm-level annual nutrient fluxes (at 15cm and 100cm) were analyzed using a one-way ANOVA with management system (organically-fertilized agroforest, mineral-fertilized agroforest, or mineral-fertilized monoculture) as the main effect.

Further, N and P losses could be affected by the quantity of N and P added as fertilizer. Therefore, univariate regression was performed to test the relationship between N and P loss at 100cm from agroforests and fertilizer N and P inputs on all farms (n=9). Univariate regression was also performed to test the relationship between N and P loss at 100cm and total aboveground biomass (shade tree plus coffee plants; n=9), shade tree aboveground biomass (agroforests only, n=8), and the size of the iron-oxide pool (agroforests only, n=8). All data were analyzed using JMP 8.0 for Macintosh.

3.0 Results

3.1 Nutrient losses differ among management systems

Among the components of the water balances, transpiration rates tended to differ the most between monocultures and agroforests, with smaller estimated transpiration flux in the monoculture (Table 7). Overland flow also tended to be higher in monocultures than in agroforests.

Table 7: Components (mm) of the water balance in three coffee management systems from October 2008 to September 2009 in Costa Rica. Components of the water balances in mineral-fertilized monocultures (MONO), mineral-fertilized agroforests (MAF), and organically-fertilized agroforests (OAF). Water balance components in agroforests represent the mean across farms (MAF n=4; OAF n=4).

Water Flux	MONO		MAF		OAF	
	(mm)	(% of rainfall)	(mm)	(% of rainfall)	(mm)	(% of rainfall)
Rainfall	3257	100	3257	100	3257	100
Throughfall	2703	83	2606	80	2606	80
Interception (I)	554	17	651	20	651	20
Transpiration (T_{plant})	961	30	1425	47	1380	42
Bare soil Evap (E_{soil})	293	9	0	0	0	0
AET(I+ T_{plant} +E_{soil})	1808	56	2076	64	2031	62
Runoff	149	5	126	4	102	3
Water Drainage	1300	40	1055	32	1124	35

Nitrogen fluxes at 15 and 100cm were significantly higher (by about 72%) in the mineral-fertilized monoculture compared to the coffee agroforests (Figure 1). However, N fluxes did not differ significantly among mineral- and organically-fertilized agroforests. Nitrogen losses were reduced at depth in all farms, and were 70%, 76%, and 57% lower at 100cm than at 15cm in mineral-fertilized monocultures, mineral-fertilized agroforests, and organically-fertilized agroforests, respectively.

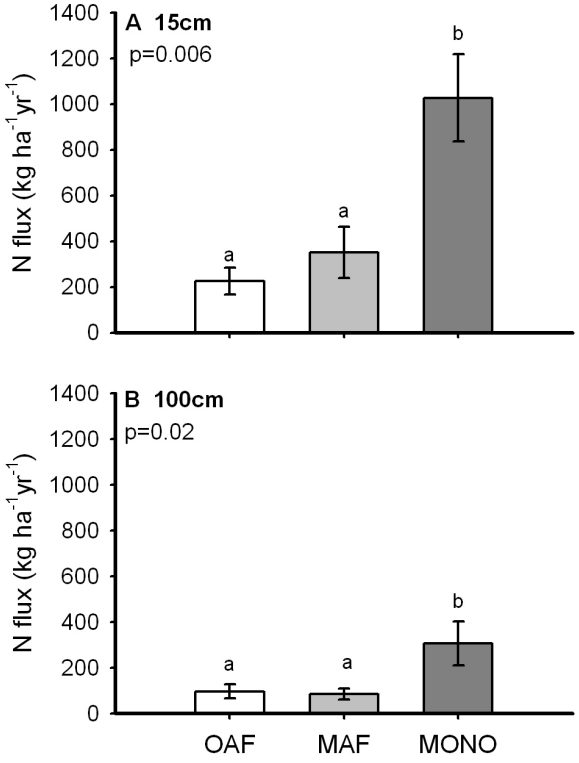


Figure 1: Annual nitrogen fluxes at 15 and 100cm in three coffee management systems in Costa Rica. Nitrogen flux at (A) 15cm and (B) 100cm in organically-fertilized agroforests (OAF), mineral-fertilized agroforests (MAF), and the mineral-fertilized monoculture (MONO). Nitrogen concentrations as measured in leachate from tension lysimeters plus estimations of drainage calculated from a water balance model were used to calculate N losses. Monthly losses were summed over the course of the study year. Differences in N losses were tested by ANOVA. Values that were significantly different at $p<0.05$ are indicated by different letters. Error bars represent the standard error of the annual mean.

As for N, P fluxes at 15cm and 100cm were significantly higher (by 50% and 62%) in the mineral-fertilized monoculture compared to coffee agroforests (Figure 2), but did not differ significantly between farms that received mineral fertilizer and those that received organic fertilizer. Phosphorus losses were reduced at depth in similar proportions, and were 35%, 18%, and 21% lower at 100cm than 15cm in mineral-fertilized monoculture, mineral-fertilized agroforests and organically-fertilized agroforests, respectively.

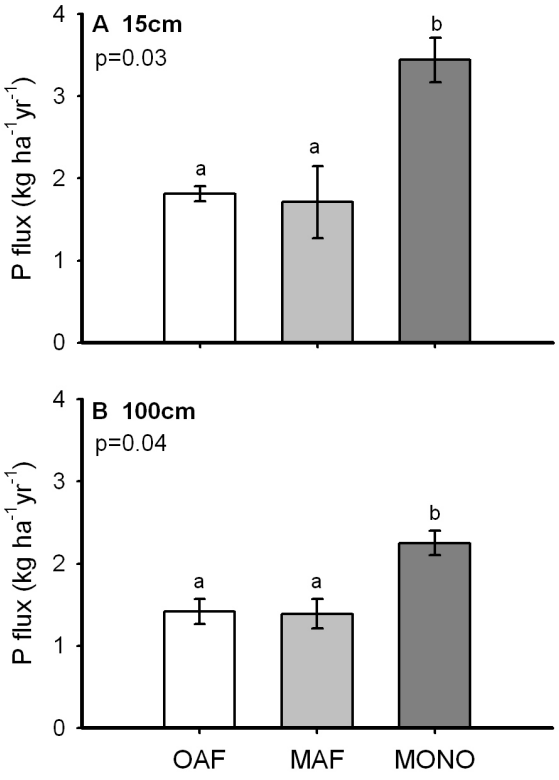


Figure 2: Annual phosphorus fluxes at 15 and 100cm in three coffee management systems in Costa Rica. Phosphorus fluxes at (A) 15cm and (B) 100cm in organically-fertilized agroforests (OAF), mineral-fertilized agroforests (MAF), and mineral-fertilized monoculture (MONO). Phosphorus concentrations were measured in leachate from tension lysimeters and estimations of drainage calculated from a water balance were used to calculate P loss. Monthly losses were summed over the course of the study year. Differences in P losses were tested using ANOVA. Values that were significantly different at $p<0.05$ are indicated by different letters. Error bars represent the standard error of the annual mean.

3.2 Factors regulating nutrient loss from coffee farms

Nitrogen and P losses (at 100cm) were not significantly correlated to fertilizer N and P inputs in agroforests alone. However, when regressions included data from monocultures, N losses increased significantly with fertilizer N additions (r^2=0.73, p=0.003; Figure 3A). Phosphorus losses also increased with fertilizer P additions when monocultures were included in the regression, although the relationship was only marginally significant (r^2=0.43, p=0.06; Figure 3B).

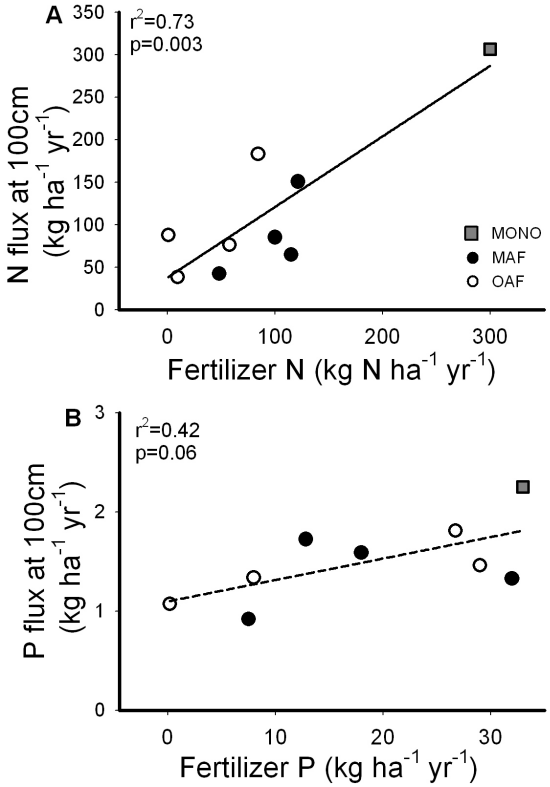


Figure 3: Correlation between fertilizer inputs and nutrient losses in coffee management systems in Costa Rica. (A) annual N loss as a function of annual N fertilizer input and (B) annual P loss as a function of annual P fertilizer input. Regressions performed on data from all farms (n=9). Solid squares represent mineral-fertilized monocultures (MONO), solid circles represent mineral-fertilized agroforests (MAF), and open circles represent organically-fertilized agroforests (OAF). Solid line indicates regression is significant at $p<0.05$ and dashed line indicates regression is not significant at $p<0.05$.

N losses (at 100cm) decreased exponentially with increasing total (coffee plus shade tree) aboveground biomass (r^2=0.87, p=0.0002; Figure 4A) while P loss showed no relationship with aboveground biomass (Figure 4B). Among agroforests, N losses at 100cm declined significantly with increasing shade tree biomass (r^2=0.62, p=0.02; Figure 4C). Phosphorus loss decreased significantly with increasing DCB-extractable iron oxide pools (r^2=0.54, p=0.039; Figure 4F) while N loss did not (Figure 4E).

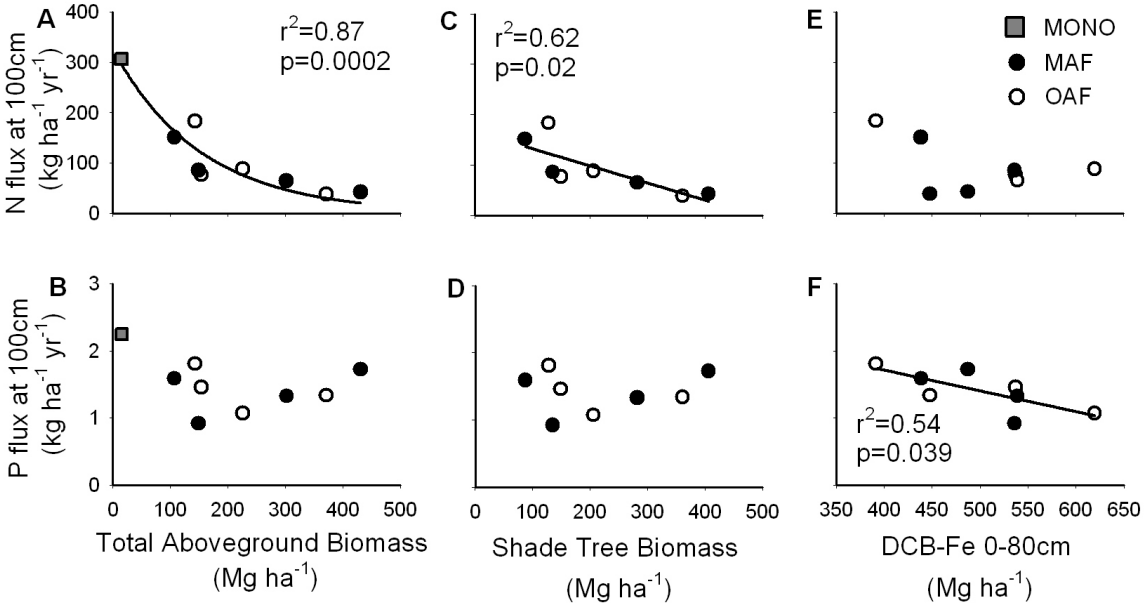


Figure 4: **Correlations between factors regulating nutrient losses in coffee management systems in Costa Rica.** Annual N (A) and P (B) loss (at 100cm) a function of aboveground biomass, annual N (C) and (D) P loss (at 100cm) as a function of total DCB-extractable iron pool (0-80cm), annual N (E) and P (F) loss (at 100cm) a function of aboveground shade tree biomass. Solid squares represent mineral-fertilized monocultures (MONO), solid circles represent mineral-fertilized agroforests (MAF), and open circles represent organically-fertilized agroforests (OAF). Regressions A and B were performed across all farms (n=9). Regressions C-F were only performed on agroforests (n=8).

4.0 Discussion

4.1 Nutrient losses differ among management systems

Based on modeled nutrient balances (Chapter 2; Figure 6), I expected mineral-fertilized agroforests to lose more N than organically-fertilized agroforests. However, despite the fact that mineral fertilizers are more readily bioavailable, and are added in greater quantities (Chapter 3, Table 1), there were no significant differences in N fluxes at 15 and 100cm between agroforests amended with organic fertilizer and those amended with mineral fertilizer. Deep N losses (at 100cm) were 76% lower than surface losses (at 15cm) in mineral-fertilized agroforests compared to organically-fertilized agroforests where deep N losses were only 57% lower than surface N losses. Thus, mineral fertilizer appears to be effectively captured than organic fertilizer as it is quickly assimilated by plants and microbes. However, in both fertilizer management types, the presence of trees on farms appears to draw N losses down to a similar level. In the absence of trees, N losses at 100cm were three times higher (Figure 1). This is not surprising as mineral-fertilized monocultures received three times as much N as mineral-fertilized agroforests and eight times as much N as organically-fertilized agroforests (Table 1). Further, nitrate concentrations were three times higher in mineral-fertilized monocultures than in coffee agroforests (Chapter 3, Figure 6), suggesting a greater potential for loss.

As one might expect, N losses (at 100cm) increased with fertilizer inputs (Figure 3A). However, this relationship was driven by the high N inputs and losses in monocultures. Despite variation in the quantity and form of nutrients added, fertilizer N inputs alone could not explain N losses in agroforests. Neither was it simply a matter of

the presence or absence of trees (as past studies have identified, see Babbar and Zak 1995). Rather, it appears that losses decline steadily as biomass (and therefore, tree density) increases in farms.

Phosphorus tends to be very conservatively cycled in tropical systems, and typically annual P losses are very low (Jordan 1982; Bruijnzeel 1991; Campo et al. 2001; Markewitz et al. 2004). Therefore, it is not surprising that annual P fluxes at 15cm and 100cm were not significantly different between mineral- and organically-fertilized agroforests. As for N, P losses were significantly higher in mineral-fertilized monocultures compared to agroforests, suggesting that the presence of trees may help reduce P losses as well. As for N, P losses tended to increase with increasing fertilizer P inputs (Figure 3B), suggesting that P losses may be enhanced in systems with large annual P additions. However, P losses (at 100cm) from all farms were within the expected range of losses from intact tropical systems (Radulovich and Sollins 1991) suggesting a large capacity of these tropical soils to retain P.

4.2 Mechanisms preventing N and P loss

The most consistent predictor of N loss across farms was the size of the aboveground biomass pool (Figure 4A). This study suggests that the capacity of farms to retain nutrients will be enhanced on farms with higher plant density as well as greater structural diversity where resource partitioning (e.g. via rooting depth) allows for the co-existence of multiple plant functional types. Among agroforests, N losses declined with increasing shade tree biomass (Figure 4C) due to both direct and indirect mechanisms. First, as tree density increases, so does plant N demand, which provides a sink for added

N (direct mechanism). Second, as tree density increases, greater quantities of litterfall and pruning residues are transferred to the soil. As these tissues decompose, they contribute to the soil organic matter pool, which immobilizes N and prevents loss (indirect mechanism; Chapter 5).

Agroforesters should not increase the density of shade trees on their farms simply to mitigate nutrient losses, as yields also decline with increasing shade tree density (Beer et al. 1998). However, the strong relationship between biomass and the reduction of N loss is evidence of a valuable ecosystem service provided by shade trees. Further, biomass is easily quantifiable from simple field measurements, and therefore may serve as a simple and efficient metric for evaluating potential N loss from agroecosystems. For example, such a metric, coupled with a model of the relationship between shade and yield could be used to design management systems that mitigate N losses while still remaining economically viable.

Among agroforests, P losses declined with increasing DCB-extractable Fe oxide pools in the soil (Figure 4F) as more P can be held on positively-charged clay surfaces. Unfortunately, in contrast to tree biomass which is easily estimated from tree density, soil Fe content may not be a useful metric for estimating P losses from coffee farms since the quantification of soil Fe pools is expensive and time-consuming. Further research should investigate if Fe pools are correlated with more basic soil properties such as soil texture (e.g. percent clay) or pH, which might serve as a proxy for Fe pools, and thus serve as a metric for evaluating potential P loss from agroecosystems.

5.0 Conclusions

Despite a wide range in the forms (NH_4-NO_3 to chicken manure) and quantities (1 to 122 kg of N ha^{-1} yr^{-1}) of added nutrients, N and P losses were not significantly different among agroforests amended with mineral fertilizer and those amended with organic fertilizer. Rather, the greatest differences in nutrient losses were between coffee farms with trees and coffee farms without trees. Mineral-fertilized monocultures lost three times as much N as agroforests indicating the crucial role that trees play in mitigating the negative effects of fertilizer application. Further, N losses (at depth) declined with increasing aboveground biomass, reinforcing the role of shade trees—92% of aboveground biomass— in mitigating N losses. On the other hand, soil chemical processes, rather than trees, seem to play a major role in the mitigation of P losses.

References

Amekawa, Y., H. Sseguya, S. Onzere, and I. Carranza. 2010. Delineating the multifunctional role of agroecological practices: Toward sustainable livelihoods for smallholder farmers in developing countries. Journal of Sustainable Agriculture 34:202-228.

Araya, M. 2005. Stratification and spatial distribution of the banana (Musa AAA, Cavendish subgroup, cvs' Valery'and Grande naine') root system. Banana Root System: Towards a Better Understanding for its Productive Management 83-103.

Babbar, L. I., and D. R. Zak. 1995. Nitrogen loss from coffee agroecosystem in Costa Rica: leaching and denitrification in the presence and absence of shade trees. Journal of Environmental Quality 24:227-233.

Beer, J., R. Muschler, D. Kass, and E. Somarriba. 1998. Shade management in coffee and cacao plantations. Agroforestry Systems 38:139-164.

Bruijnzeel, L. A. 1991. Nutrient input output budgets of tropical forest ecosystems- a review. Journal of Tropical Ecology 7:1-24.

Campo, J., M. Maas, V. J. Jaramillo, A. Martnez-Yrizar, and J. Sarukhan. 2001. Phosphorus cycling in a Mexican tropical dry forest ecosystem. Biogeochemistry 53:161-179.

Darke, A. K., and M. R. Walbridge. 1994. Estimating noncrystalline and crystalline aluminum and iron by selective dissolution in a riparian forest soil. Communications in Soil Science and Plant Analysis 25:2089-2101.

Food and Agriculture Organization of the United Nations (FAO). 2008. FAO Statistical Databases < http://www.fao.org/corp/statistics/en/>

Food and Agriculture Organization of the United Nations (FAO). 2009. FAO Statistical Databases < http://www.fao.org/corp/statistics/en/>

Frank, B., and S. Eduardo. 2003. Biomass dynamics of Erythrina lanceolata as influenced by shoot-pruning intensity in Costa Rica. Agroforestry Systems 57:17-26.

Garriz, P. I. 1978. Distribución radicular de tres cultivares de Coffea arabica L. de 24 anõs de edad, en un suelo limo-arcilloso. Agronomía Tropical 9:91-103.

Hairiah, K., S. M. Sitompul, M. van Noordwijk, and C. Palm. 2001. ASB Lecture Note 4B. International Center for Research in Agroforestry (ICRAF), Bogor.

Harmand, J. M., H. Avila, E. Dambrine, U. Skiba, S. de Miguel, R. V. Renderos, R. Oliver, F. Jimenez, and J. Beer. 2007. Nitrogen dynamics and soil nitrate retention in a *Coffea arabica-Eucalyptus deglupta* agroforestry system in Southern Costa Rica. Biogeochemistry 85:125-139.

Hassan, R., R. Scholes, and N. Ash. 2005. Ecosystems and human well-being: current state and trends, Volume 1. Findings of the Conditions and Trends Working Group of the Millennium Ecosystem Assessment Island Press: Washington.

Leakey, R. R. B., Z. Tchoundjeu, K. Schreckenberg, S. E. Shackleton, and C. M. Shackleton. 2005. Agroforestry tree products (AFTPs): targeting poverty reduction and enhanced livelihoods. International Journal of Agricultural Sustainability 3:1-23.

Lehmann, J. 2003. Subsoil root activity in tree-based cropping systems. Plant and Soil 255:319-331.

Lin, B. B., and P. L. Richards. 2007. Soil random roughness and depression storage on coffee farms of varying shade levels. Agricultural Water Management 92:194-204.

Markewitz, D., E. Davidson, P. Moutinho, and D. Nepstad. 2004. Nutrient loss and redistribution after forest clearing on a highly weathered soil in Amazonia. Ecological Applications 14:S177-S199.

Myster, R. W. 2004. Post-agricultural invasion, establishment, and growth of neotropical trees. The Botanical Review 70:381-402.

Nair, P. K., V. D. Nair, B. M. Kumar, and S. G. Haile. 2009. Soil carbon sequestration in tropical agroforestry systems: a feasibility appraisal. Environmental Science & Policy 12:1099-1111.

Palm, C. A. 1995. Contribution of agroforestry trees to nutrient requirements of intercropped plants. Agroforestry Systems 30:105-124.

Perfecto, I., R. A. Rice, R. Greenberg, and M. E. VanderVoort. 1996. Shade coffee: A disappearing refuge for biodiversity. BioScience 46:598-608.

Radulovich, R., and S. Sollins. 1991. Nitrogen and phosphorus leaching in zero-tension drainage from a humid tropical soil. Biotropica 23:84-47.

Rice, R. A. 1999. A place unbecoming: the coffee farm of northern Latin America. Geographical Review 89:554-579.

Schroth, G., E. Salazar, and J. P. Da Silva. 2001. Soil nitrogen mineralization under tree crops and a legume cover crop in multi-strata agroforestry in Central Amamzonia: Spatial and temporal patterns. Experimental Agriculture 37:253-367.

Segura, M., M. Kanninen, and D. Suarez. 2006. Allometric models for estimating aboveground biomass of shade trees and coffee bushes grown together. Agroforestry Systems 68:143-150.

Siles, P., P. Vaast, E. Dreyer, and J-M Harmand. 2010. Rainfall partitioning into throughfall, stemflow and interception loss in a coffee (*Coffea arabica L.)* monoculture compared to an agroforestry system with *Inga densiflora*. Journal of Hydrology 395:39-48.

Topik, S. C. 2000. III.4. Coffee. In K. F. Kiple, and K. C. Ornelas., editors. The Cambridge world history of food, New York, NY.

UNFCCC (United Nations Framework Convention on Climate Change). 2008. Challenges and opportunities for mitigation in the agricultural sector. Technical Paper, UNFCCC, Bonn, DE. (Document FCCC/TP/2008/8).

van Kanten, R., and P. Vaast. 2006. Transpiration of arabica coffee and associated shade tree species in sub-optimal, low-altitude conditions of Costa Rica. Agroforestry Systems 67:187-202.

Vedenov, D., J. Houston, and G. Cardenas. 2007. Production Efficiency and Diversification in Mexican Coffee-Producing Districts. Journal of Agricultural and Applied Economics 39:749.

Chapter 5: The effects of organic and mineral fertilizers on nutrient retention capacity and nutrient pools in soils of coffee agroforests

Abstract

The widespread application of fertilizers has led to dramatic increases in crop productivity, but their overuse has created a suite of environmental and public health problems, primarily related to nutrient loading in surface and groundwaters. The cultivation of crops in the presence of trees (agroforestry) is a form of agriculture that is more effective at retaining nutrients and preventing against loss than unshaded monoculture farming. This research investigated how measures of soil nutrient retention capacity and total nutrient pools varied (1) between agroforests receiving organic or mineral fertilizers and (2) among tree species. There were no effects of fertilizer management on nutrient pools in surface and mid soils (0-50cm). Rather, carbon (C) and nitrogen (N) pools were strongly, positively correlated with oxalate-extractable aluminum (Al; an indicator of inherent nutrient retention capacity). However, differences in nutrient pools were apparent at depth (50-80cm) where mineral-fertilized agroforests had larger N pools. However, proportionally large C pools appear to be capable of immobilizing N and preventing against loss. Cation exchange capacity (CEC) and soil phosphorus (P) pools tended to be higher in organically-fertilized agroforests. Nutrient pools and retention capacity did not differ among individual tree species, however trees clearly play a crucial role in maintaining the retentive capacity of soils. For example, the presence of trees appears to maintain soil C pools which in turn can store a large proportion of other nutrients (N and P) and prevent loss.

1.0 Introduction

The heavy use of fertilizers in agriculture can have severe consequences for both environmental and human health. Fertilizer runoff leads to the eutrophication of streams, lakes, and estuaries (Schindler 1977; Howarth 1988; Nixon et al. 1996), and leaching into groundwater supplies can elevate nitrate concentrations to toxic levels (Sharpley et al. 1987). In efforts to reduce the downstream effects of fertilizers, trees are often planted in riparian zones or among crops (agroforest). Alternative agricultural practices, such as organic farming, reduce reliance on agrochemicals and augment ecological processes that promote plant productivity while maintaining soil fertility and conserving water resources (Pimentel et al. 2005). This research investigates how nutrient retention capacity and soil nutrient pools in agroforests are modified by the type of fertilizer applied (mineral or organic).

Nutrient retention capacity is the ability of a soil to retain added nutrients against loss. Nutrients are held in the soil on exchange sites of clay minerals, organic matter, and the clay-humus complex. In general, nutrient retention (and by extension, soil fertility) in soils is controlled by texture, organic carbon, cation exchange capacity (CEC), pH, and the presence of iron (Fe) and aluminum (Al) oxides (Arshad and Coen 1992; Seybold et al. 1997). Soil texture affects nutrient retention by altering available exchange sites in clay minerals and by its control over soil permeability. Soil CEC (and pH) determines the capacity to hold positively charged ions (cations) on negatively charged sites at the surface of the soil minerals and organic matter. Soil organic matter (SOM) has a large surface area and therefore has a high density of exchange sites. The negative charge of

these exchange sites allows them to bind with cations. Microbial biomass also immobilizes nitrogen (N), preventing N losses. Iron and aluminum oxides play an important role in the retention of dissolved inorganic phosphorus (P). In particular non-crystalline (or amorphous) Al and Fe have large surface areas and are often associated with higher P sorption capacity in tropical soils (Agbenin 2003). Although soil fertility is controlled by factors that promote nutrient retention, it can also be empirically determined by measuring the size of soil nutrient pools. While industrialized agriculture can diminish soil nutrient pools and alter soil properties that retain nutrients in the system (Cassman 1999; Sanchez et al. 2003), agroforestry can support nutrient retention and promote soil fertility by preventing erosion and soil compaction (Lal 1989; Young 1989; Young 1986), maintaining SOM (Chander et al. 1998; Lal 2004), and reducing nutrient leaching (Babbar and Zak 1995; Harmand et al. 2007; Chapter 3, Figure 6). In this research, I describe how fertilizer management alters nutrient retention capacity and soil nutrient pools in coffee agroforests.

The coffee (*Coffea arabica*) industry supports over 8.5 million people across Central America, with 4 million people relying directly on farm yields for their livelihoods (Tucker et al. 2010). During the coffee crisis of the late 1990s, niche markets for specialty coffees began to emerge (ICO 2010; Boyce et al. 1994; Goodman 1999). Certification programs such as Rainforest Alliance® and Smithsonian Migratory Bird Center's Bird Friendly Coffee® support more sustainable forms of agriculture and provide many small farmers with price premiums for shade-grown coffee (Lyngbæk et al. 2001). Nevertheless, in efforts to improve yields, many agroforesters may continue to

apply large quantities of inorganic nitrogen (N) fertilizers (Szott and Kass 1993). In contrast, Certified Organic® farmers rely on animal manure, compost, and worm castings to provide nutrients to their crop. As organic inputs confer less N, organic coffee yields are typically 20% lower than in mineral-fertilized managed coffee plantations (Lyngbæk et al. 2001). While both mineral and organically-fertilized agroforests maintain the same ecosystem structure (e.g. canopy trees), different input and output rates over many years will likely alter soil nutrient retention capacity and the size of nutrient pools.

In tree-less farms (e.g. rotational crops of tomato, beans, corn, melon, etc.), organic management can lead to higher soil water holding capacity (Liebig and Doran 1999; Wells et al. 2000), pH (Fließbach et al. 2007), CEC (Haynes and Naidu 1998), and greater aggregate stability (Gerhardt 1997; Mader et al. 2002) compared to mineral-fertilized farms. After 22 years of cultivation, organically-fertilized soils accumulated nitrogen (N) over initial conditions, and contained more soil carbon (C) than mineral-fertilized farms (Pimentel et al. 2005). Organically-fertilized farms may also have higher soil microbial biomass (Clark et al. 1998; Melero et al. 2006), higher basal respiration (Fließbach et al. 2007), and root length colonized by mycorrhizae (Mader et al. 2002) than mineral-fertilized farms. The aforementioned research focuses on agricultural systems of minimal structural diversity (e.g. monocultures or polycultures of similar height). To my knowledge, this is the first research to investigate the effects of organic and mineral fertilizers on soils when the crop is grown the presence of trees. I expect the inclusion of trees to modify the effect of fertilizer management on nutrient retention capacity and soil fertility. Further, different canopy species may have different effects on

soil retention and pools. Thus, my secondary goal was to determine the effect of different species on their immediate soil environment and if their effects differ depending on management type.

Different plant functional types (e.g. N-fixers, herbaceous species, evergreen, deciduous, etc.) can alter the physical environment and soil community (Eviner and Chapin 2003). For example, bacterial symbionts in the root nodules of *Erythrina poeppigiana* convert atmospheric N into a bioavailable form, directly enhancing soil N availability, but also acidifying soils (Haynes 1983). The effects of species on patterns and rates of decomposition and nutrient release are discussed in detail in Chapter 6. Clearly species-specific variation in the residence time of nutrients in the litter pool has consequences for nutrient retention. Variation in nutrient allocation is also likely to play a role. *Musa* plants, often interplanted with coffee, produce fruit that is high in N and cations (especially potassium; Turner et al. 1989). Repeated harvest of the fruits deplete these soil nutrient pools. Thus, some shade species may be considered a net nutrient source (*Erythrina*) and others a net nutrient sink (*Musa, Coffea*). I expect nutrient pools to be smaller under "sink species" and larger under "source species". The primary crop itself, *Coffea,* can remove as much as 68 and 27 kg of N ha^{-1} yr^{-1} in mineral and organically-fertilized agroforests, respectively (Appendix Chapter 3, Figure 5). Therefore, soil nutrient pools below *Coffea* may be the lower than the other species.

2.0 Methods

2.1 Study sites

The Central Valley of Costa Rica has a humid tropical climate favorable to coffee agroforestry. Specifically, this study was conducted in four locations near Turrialba, Costa Rica (9°53'N 83°40'W; Chapter 3, Figure 1). Coffee agroforests are located within 3km of one another between 9°52'N and 9°54'N and between 83°41'W and 83°42'W. The altitude of sites ranged from 783 to 1017 meters above sea level. The study region in the Central Valley receives on average 2600mm of rainfall per year with a mean annual temperature of 22.6°C. Rainfall is seasonal, and the dry season extends from February through May with March being the driest month. In region received 2670mm of rainfall in the year prior to sampling, with a mean temperature of 22.3°C.

Soils are of volcanic origin and are characterized as a Typic Humitropept (Selvaradjou et al. 2005) with a clay-loam texture. After extensive interviews and initial sampling on 28 farms (Chapter 2), four organically managed and four mineral-fertilized coffee agroforests were selected for additional sampling. In these agroforests, *Coffea arabica* is cultivated under the shade of *Erythrina poeppigiana* and *Musa acuminata* (Chapter 3, Table 1). On average, mineral-fertilized agroforests received 96kg of N and 18kg of P ha^{-1} yr^{-1} in the form of mineral fertilizer (Appendix Chapter 3, Figure 4). Organically-fertilized agroforests received an average of 38kg of N and 16kg of P ha^{-1} yr^{-1} in the form of manure, compost, and fermented coffee husks (*broza*; Chapter 4, Figure 4).

2.2 Soil sampling and initial processing

Early in the wet season (June) of 2008, soil pits to 80cm were excavated in three locations in each agroforest (Chapter 3, Figure 3). The pits were located within 50cm of the base(s) of (1) *Erythrina* and *Coffea*, (2) *Musa* and *Coffea*, and (3) *Coffea* at least 5m radius from the nearest shade tree (distant *Coffea* plant). Soil samples were taken at six depths along the soil profile: 0-5cm; 5-10cm; 10-15cm; 15-30cm; 30-50cm; and 50-80cm using a $100cm^3$ metal cylinder (one sample per depth between 0-15cm; two samples per depth between 15-80cm). Once the cylinder was full of soil (but not overflowing), samples were bagged to retain moisture and immediately returned to the lab for processing.

2.3 Soil properties and nutrient analysis

Upon returning to the lab at Centro Agronómico de Investigación y Enseñaza (CATIE), the entire soil sample was weighed wet and a subsample was weighed wet, dried for 24 hours at 105°C, and re-weighed to determine moisture content. This wet-to-dry conversion factor was applied to the total sample wet weight, and the product was divided by the total volume of the core to determine bulk density (each core $100cm^3$). The volume and mass of roots and rocks in samples was determined and subtracted from the bulk density value. The remaining soil was air-dried and sieved (2mm mesh) prior to transport back to the University of Virginia.

Sub-samples of the processed soils were then ground to 149μm and dry-combusted on an elemental analyzer (Carlo Erba, Model NA 2500; Milan, Italy) to

determine total N and C. Soil texture analysis was performed by Brookside Laboratories, Inc. (New Knoxville, OH) on composite samples from 0-15cm.

I used a barium chloride ($BaCl_2$) extraction to determine cation exchange capacity as agricultural soils are regularly amended with calcium, and studies have shown $BaCl_2$ is a slightly more efficient replacer of exchangeable calcium, magnesium, and sodium than ammonium acetate in these soils (Gillman 1979). In this procedure, soils are extracted with 0.1M $BaCl_2$, followed by three extractions with 0.0002M $BaCl_2$. Finally, soils are extracted with 0.005M magnesium sulfide ($MgSO_4$), and resulting solutions decanted and analyzed for magnesium on an atomic absorption spectrometer (AAnalyist 100; Perkin Elmer; Connecticut, USA).

Non-crystalline (amorphous) and crystalline Al and Fe were estimated by the method of Darke and Walbridge (1994). A sequential 0.2M acid ammonium oxalate (pH 3.0) extraction followed by 0.1M NaOH extraction using air-dried soils was used to estimate non-crystalline Al and Fe (Al_o and Fe_o) and crystalline Al (Al_c), respectively. Non-crystalline plus crystalline (total) Fe was estimated by a separate 16-hour dithionite-citrate-bicarbonate (DCB) extraction (Fe_T). Crystalline Fe (Fe_c) was estimated as the difference between Fe_T and Fe_o. Resulting solutions were analyzed on an atomic absorption spectrometer for Al and Fe.

Soil subsamples were digested using a modified Kjeldhal protocol, and total P in resulting solutions was determined colorimetrically using the molybdate blue methodology on an Alpkem Flow Solution IV Autoanalyzer (OI Analytical, College Station, Texas, USA). Sodium hydroxide extracts (Al and Fe sorbed P) were also

analyzed for P on an Alpkem. I refer to this pool as NaOH-P, but as more labile pools were not sequentially extracted (e.g. resin P, bicarbonate-extractable P; Tiessen and Moir 1983), the NaOH fraction in this study represents both the plant-available P pool as well as the less labile P associated with non-crystalline Fe and Al phosphates (Cross and Schlesinger 1995). The bioavailable P pool (resin + bicarbonate) is likely to be 42% of the pool I extracted (Szott and Melendez 2001).

Finally, primary calcium-associated P was determined by extracting soils with 1M HCl; extracts were analyzed for P on an Alpkem. HCl and NaOH concentrations were statistically identical suggesting that the majority of P is bound to iron and aluminum oxides.

Soil nutrient pools were calculated for each depth using bulk density and profile thickness to scale up nutrient concentrations (N, P, C) pools (Mg ha^{-1}). Soil C:N:P ratios were computed on a mass basis (using total nutrient pools). Compared to nutrient concentrations, soil pools are a more realistic representation of the environment to which plants are exposed. Therefore I focus my discussion on the effects of management and species on nutrient pools. I examined soil nutrient pools (and properties) at three depths: (1) "surface soils" comprising the most active root layer (0-15cm); "mid soils" comprising the rest of primary rooting zone (15-50cm); and "deep soils" extending below the rooting zone (50-80cm). Nutrient pools were summed and properties averaged for each of these depths. All data are reported on a dry mass basis.

2.4 Statistical Approach

For each of the three depth profiles, I performed a two-way analysis of variance (ANOVA) on soil pools and properties with management and species as main effects. In order to account for inherent soil variability, the concentration of oxalate-extractable Al (Ox-Al) was used as an indicator of inherent soil fertility as (a) it is not likely affected by management practices and (b) differed among locations ($p=0.003$). The soil depths examined were of variable thickness (0-5cm compared to 30-50cm). Therefore, I examined the vertical distribution of nutrient pools on a relative basis at each soil depth (Jobbágy and Jackson 2001). First, I calculated nutrient pools on a volumetric basis using bulk density (e.g. Mg N m^{-3}). Then I estimated the relative pool size in 5cm of soil (Eq. 1).

$$\text{Relative pool (Mg ha}^{-1}\text{)} = \text{conc. (mg g}^{-1}\text{)} * \text{bulk density (g cm}^{-3}\text{)} * 5\text{cm} * 10^{-1} \quad \text{(Eq. 1)}$$

Measures of nutrient retention capacity were averaged at each section of the soil profile. All statistical analyses were performed using JMP 8.0 for Macintosh (2009, SAS Institute, Inc.).

3.0 Results

3.1 Fertilizer management affects nutrient pools at depth

Fertilizer management did not drive patterns in nutrient pools in surface soils (0-15cm) or mid soils (15-50cm). Rather, across management types, oxalate-extractable Al was the strongest predictor of soil C and N pools in both surface ($r^2=0.57$ and $r^2=0.40$, respectively) and mid soils ($r^2=0.48$ and $r^2=0.62$, respectively; Figure 1).

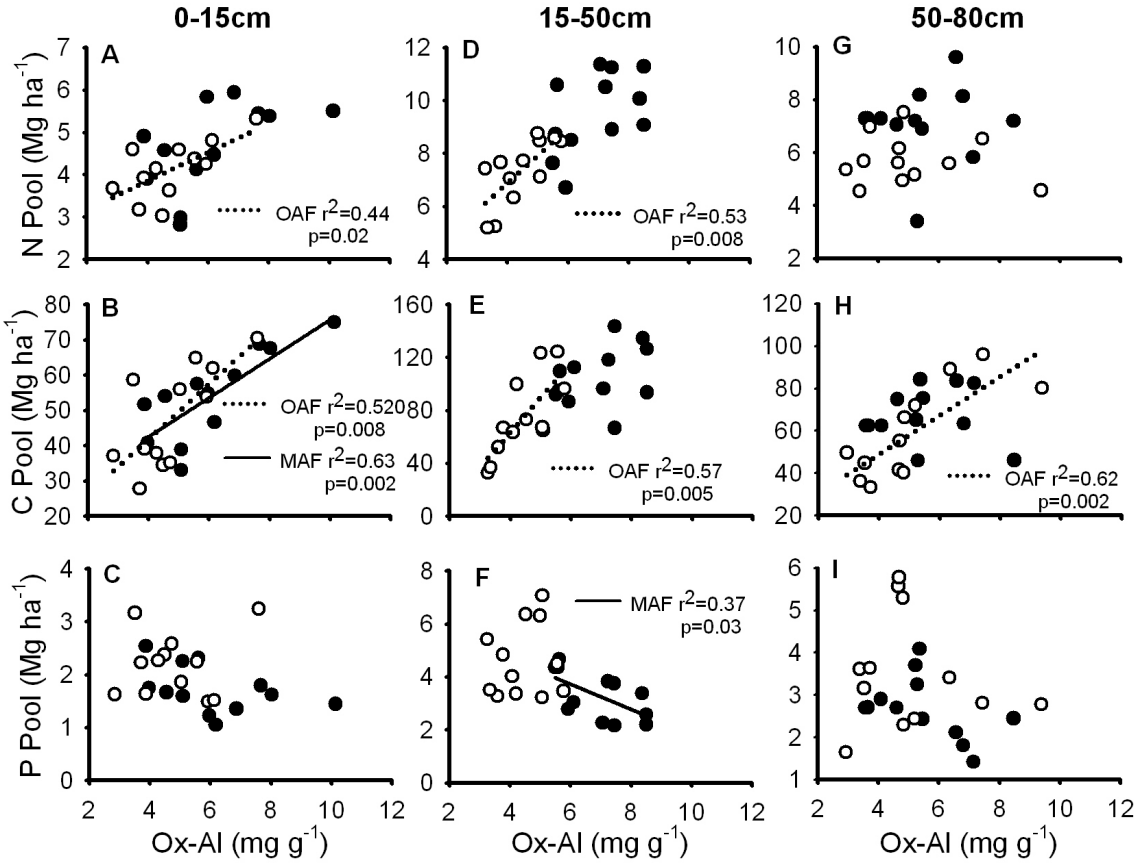


Figure 1: Soil N, C, and P pools as a function of oxalate-extractable Al at three depths between 0 and 80cm in coffee agroforests in Costa Rica. Nitrogen, carbon, and phosphorus pools as a function of oxalate-extractable Al at (A-C) 0-15cm, (D-F) 15-50cm, and (G-I) 50-80cm depth. Closed circles represent mineral-fertilized agroforests (MAF; n=4) and open circles represent organically-fertilized agroforests (OAF; n=4). Solid (mineral-fertilized) and dotted (organically-fertilized) linear lines of best fit are presented.

However, the effects of fertilizer management emerged when I examined deep soils (50-80cm). Soil N pools were 24% larger in mineral-fertilized (7.11±0.42 Mg N ha^{-1}) than in organically-fertilized agroforests (5.72±0.27 Mg N ha^{-1}; p=0.0219; Figure 2A). Phosphorus pools, on the other hand, tended to be larger in organically- (3.52±0.39 Mg P ha^{-1}) compared to mineral-fertilized agroforests (2.67±0.22 Mg P ha^{-1}; p=0.09; Figure 2C). There were no effects of fertilizer management on soil C pools at depth as they were still strongly correlated with oxalate-extractable Al (r^2=0.62; p=0.002; Figure 1H).

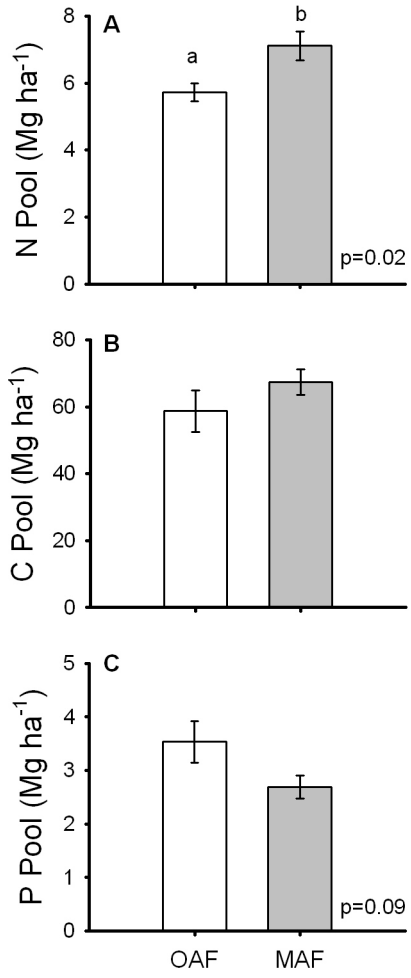


Figure 2: Soil N, C, and P pools at depth (50-80cm) in coffee agroforests in Costa Rica. Total soil (A) nitrogen, (B) carbon, and (C) phosphorus pools between 50 and 80cm depth in mineral-fertilized (n=4) and organically-fertilized (n=4) agroforests. Error bars represent standard error of the mean. Differences in C, N, and P pools among the two management types were tested by ANOVA. Values that were significantly different at $p<0.05$ are indicated by different letters.

3.2 Fertilizer management does not affect nutrient retention capacity in soils

There were no significant effects of fertilizer management on nutrient retention capacity in soils at any depth. Soil texture did not differ significantly among organically- and mineral-fertilized agroforests, but CEC tended to be higher in organically-fertilized agroforests (Table 1).

Table 1: Soil nutrient pools and properties in coffee agroforests in Costa Rica. Mean soil properties and cumulative pools by fertilizer management in surface (0-15cm), mid (15-50cm), and deep (50-80cm) soils. Differences in soil properties among the four locations were tested by ANOVA. Values that were significantly different at $p<0.05$ are indicated by different letters.

	MAF	**OAF**
0-15cm		
N (Mg ha^{-1})	4.65	4.12
P (Mg ha^{-1})	1.72	2.19
C (Mg ha^{-1})	54.02	48.09
C:N (molar)	14	13
NaOH-P (Mg ha^{-1})	0.23	0.23
HCl-P (Mg ha^{-1})	0.19	0.39
CEC (cmol+ kg^{-1})	3.63	3.92
Ox-Al (mg g^{-1})	6.09	4.81
NaOH-Al (mg g^{-1})	7.39	5.77
Ox-Fe (mg g^{-1})	7.59	8.44
DCB-Fe (mg g^{-1})	69.19	61.56
Bulk Density (g cm^{-3})	0.82	0.87
15-50cm		
N (Mg ha^{-1})	9.55	7.33
P (Mg ha^{-1})	3.28	4.61
C (Mg ha^{-1})	106.09	75.05
C:N (molar)	13	12
NaOH-P (Mg ha^{-1})	0.41	0.51
HCl-P (Mg ha^{-1})	0.18	0.53
CEC (cmol+ kg^{-1})	2.97	3.59
Ox-Al (mg g^{-1})	6.95	4.44
NaOH-Al (mg g^{-1})	9.09	5.74
Ox-Fe (mg g^{-1})	9.21	7.29
DCB-Fe (mg g^{-1})	69.83	64.81
Bulk Density (g cm^{-3})	0.93	1.04
50-80cm		
N (Mg ha^{-1})	7.11 a	5.72 b
P (Mg ha^{-1})	2.69	3.53
C (Mg ha^{-1})	67.38	58.73
C:N (molar)	11	12
NaOH-P (Mg ha^{-1})	0.33	0.36
HCl-P (Mg ha^{-1})	0.17	0.38
CEC (cmol+ kg^{-1})	3.05	3.30
Ox-Al (mg g^{-1})	5.52	5.08
NaOH-Al (mg g^{-1})	7.95	7.52
Ox-Fe (mg g^{-1})	7.56	7.11
DCB-Fe (mg g^{-1})	62.78	62.76
Bulk Density (g cm^{-3})	0.96	1.06

Nevertheless, across management types, CEC declined significantly with increasing NaOH-extractable Al (sum of crystalline and non-crystalline Al; r^2=0.41; Figure 3).

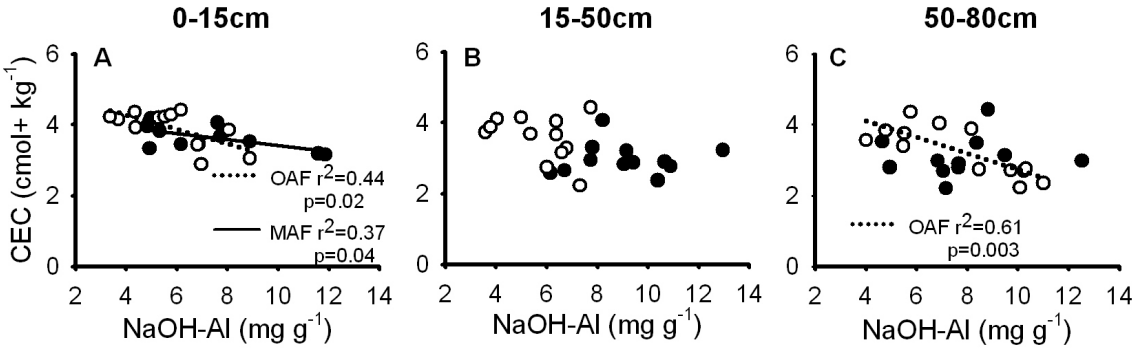


Figure 3: Correlation between cation exchange capacity and NaOH-extractable Al at three depths between 0 and 80cm in coffee agroforests in Costa Rica. Soil CEC as a function of NaOH-extractable Al at (A) 0-15cm, (B) 15-50cm, and (C) 50-80cm depth. Closed circles represent mineral-fertilized agroforests (MAF; n=4) and open circles represent organically-fertilized agroforests (OAF; n=4). Solid (mineral-fertilized) and dotted (organically-fertilized) linear lines of best fit are presented.

Percent silt and sand differed significantly by location with the lowest silt content in San Juan Sur (p=0.03) and the lowest sand content in Colorado (p=0.02; Table 2). Unlike Al, Fe fractions did not differ among locations, and were not significantly related to any measures of soil nutrient retention or nutrient pools.

Table 2: Soil properties (0-10cm) at the three sampling locations in Costa Rica. Differences in soil properties among the three locations were tested by ANOVA. Values that were significantly different at p<0.05 are indicated by different letters.

	Colorado	San Juan Norte	San Juan Sur
Silt (%)	48.1 a	45.7 ab	38.8 b
Clay (%)	38.5	25.7	34.7
Sand (%)	13.4 a	28.6 b	26.5 ab
pH	5.07	4.71	5.63
CEC (cmol+ kg^{-1})	4.08	3.80	3.70
Ox-Al (mg g^{-1})	4.15 b	6.70 a	5.65 ab
Ox-Fe (mg g^{-1})	63.52	67.32	70.57
Bulk Density (g cm^{-3})	0.92 a	0.81 b	0.80 ab

3.3 The effect of depth

Bulk density increased significantly with depth ($p<0.0001$; Figure 4). On the other hand, the quantity of C, N, and P stored was relatively high in surface soils and declined with depth (Figure 5). While the patterns of C were similar among mineral and organically-fertilized agroforests, the vertical distribution of soil N and P differed among management types. For example, at depth, soil N and P pools continued to decline in organically-fertilized agroforests while they remained the same size in mineral-fertilized agroforests (Figure 5A and C).

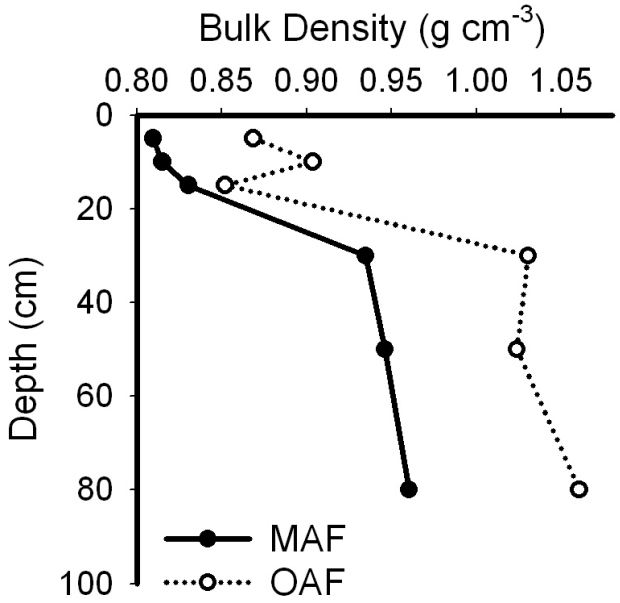


Figure 4: Vertical patterns of mean bulk density in coffee agroforests in Costa Rica. Solid lines indicate mineral-fertilized agroforests (MAF; n=4) and dotted lines indicate organically-fertilized agroforests (OAF; n=4).

While the vertical profiles of oxalate-extractable Fe were relatively similar among management types, oxalate-extractable Al was very different in mineral-fertilized and organically-fertilized agroforests (Figure 6A). In surface soils (0-15cm) oxalate-extractable Al concentrations declined in organically-fertilized and increased in mineral-

fertilized agroforests. Oxalate-extractable Al remained relatively consistent in organically-fertilized soils between 15 and 80cm, but declined sharply in mineral-fertilized soils. Finally, the vertical patterns of CEC also differed among management types with higher and more consistent CEC in organically- compared to mineral-fertilized agroforests (significant only at 15-30cm p=0.034). Mineral-fertilized agroforests farms showed a much steeper decline in CEC with depth than organically-fertilized agroforests (Figure 6C).

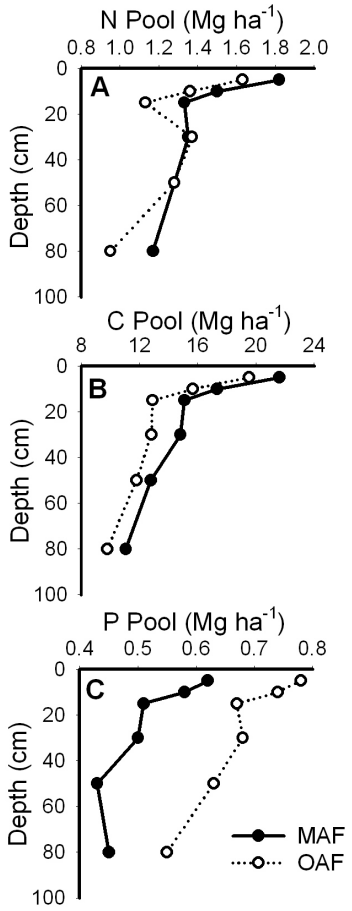


Figure 5: Vertical patterns in N, C, and P pools in coffee agroforests in Costa Rica. Vertical distribution of relative (A) nitrogen, (B) carbon and (C) phosphorus pools in mineral and organically-fertilized agroforests. Each point represents a mean pool in 5cm of soil at that depth. Solid lines indicate mineral-fertilized agroforests (MAF; n=4) and dotted lines indicate organically-fertilized agroforests (OAF; n=4).

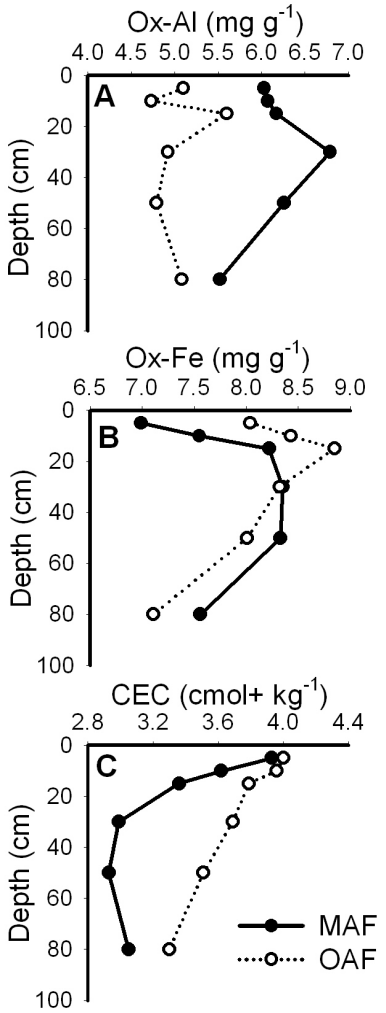


Figure 6: Vertical patterns in oxalate-extractable Al and Fe and cation exchange capacity in coffee agroforests in Costa Rica. Vertical patterns of (A) oxalate-extractable Al, (B) oxalate-extractable Fe, and (C) CEC among mineral (MAF; n=4) and organically-fertilized (OAF; n=4) coffee agroforests. Each point represents the average concentration at each soil depth. Solid lines indicate mineral-fertilized agroforests and dotted lines indicate organically-fertilized agroforests.

4.0 Discussion

4.1 No effect of fertilizer management in surface and mid soils

Surprisingly, there were no significant effects of fertilizer management on soil nutrient pools in the surface or mid soils. Mineral fertilizers have a higher N content than organic fertilizers (18% compared to 3%) and are added in larger quantities (96 compared

to 38 kg N ha^{-1} yr^{-1}; Chapter 4, Figure 5). Soil N (and C) pools did tend to be larger in mineral-fertilized agroforests. However, in surface and mid soils, these trends were best explained by inherent soil properties rather than the effects of management.

Both mineral and organically-fertilized agroforests receive large quantities of organic C through litterfall and pruning residues. For example, organic farmers apply 2.5 Mg of organic material in the form of fertilizers on average, but pruning residues can transfer up to 14 Mg of organic material to the forest floor each year depending on the number of prunings (Beer et al. 1998). Therefore it is not surprising that I saw no effect of management on soil C pools. Rather, soil C in the surface and mid soils was strongly (positively) correlated with oxalate-extractable Al concentration, which is not likely affected by management techniques.

4.2 The effect of inherent nutrient retention capacity in surface and mid soils

Aluminum can serve as a bridging cation for organic substances to clay minerals, which can alter soil properties (Vance et al. 1996). Therefore as oxalate-extractable Al content increased in these soils, so did the number of positively charged binding sites. Soil organic matter is negatively charged due the presence of carboxyl and phenol groups. It also comprises a large portion of the C pool explaining the strong correlation between C and Al. Nitrogen can be bound to either labile OM (short residence time; Vitousek 1984; Davidson et al. 1990) or recalcitrant fractions of SOM (long residence time; Clark 1977). Therefore, the positive relationship between oxalate-extractable Al and soil N pools was likely due to nitrogen's association with carbon. Finally, the (negative) relationship between CEC and NaOH-extractable Al simply reflects the

replacement of base cations (e.g. Ca^{2+}, Mg^{2+}, and K^{+}) with acidic cations (e.g. Al^{3+}; Figure 3).

In mid soils, where the relationship between C pools and oxalate-extractable Al was the strongest, I observed a significant decline in P pools with Al in mineral-fertilized agroforests. This was likely due to the competitive sorption of organic compounds (e.g. humic and fulvic acids) with phosphate on exchange sites, which can prevent P from binding onto clay surfaces (Lopez-Hernandez and Rodriguez 1986; Sibanda and Young 1986; Kafkafi et al. 1988). Further, soils are more acidic in mineral- compared to organically-fertilized agroforests (5.0 compared to 5.4; Chapter 3, Figure 5D) as the additions of mineral fertilizers (e.g. NH_4-NO_3) tend to acidify soils relative to organically-fertilized systems (Clark et al. 1998; Drinkwater et al. 1995; Fließbach et al. 2007). Increased negative charge on the soil surface can decrease the point of zero charge and reduce P sorption and maintain more P in solution (Bowden et al. 1980; Lopez-Hernandez and Rodriguez 1986). As total soil P pools tended to be lower in mineral-fertilized agroforests (Figure 2C) much of this desorbed P is likely being utilized by plants. This idea is further supported by the fact that aboveground P pools were larger in mineral (300 kg P ha^{-1}) compared to organically-fertilized agroforests (160 kg P ha^{-1}; Chapter 4, Figure 6) despite similar quantities of aboveground biomass.

4.3 Accumulation of nutrients in deep soils

The effects of fertilizer management were more pronounced below the rooting zone where soil properties have the greatest influence nutrient leaching into groundwater. Applied nutrients (after gaseous losses) have two fates: they can (1) accumulate in soil

and/or biomass pools or (2) leach to deeper soil layers, below the rooting zone and potentially out of the system. Nearly twice as much N and P is stored in plant biomass in mineral-fertilized agroforests, suggesting that much of the applied N is being assimilated by biota. Further, larger quantities of N and P are leaching at 15cm compared to 100cm (Chapter 3, Figure 4 and 5). Therefore, plants are assimilating a portion of the applied N and P in surface soils with the excess leached to deeper soil layers.

At depth, organically- and mineral-fertilized agroforests have similar annual losses of N (97 and 86 kg N ha^{-1} yr^{-1}; Chapter 4, Figure 1B). Further, the majority of root activity occurs in the surface soils, and nutrient uptake by plants is diminished at depth. Therefore, applied nutrients that have leached to deeper soil layers are chemically bound to the soil and stored. As mineral fertilizers confer larger quantities of nutrients than organic fertilizers, mineral-fertilized agroforests have larger N pools at depth. Nitrogen immobilization rates tend to increase with increasing C content (Barrett and Burke 2000). Overall soil C:N ratios were the same in organically- and mineral-fertilized agroforests (13:1), suggesting that although N inputs are higher in mineral-fertilized agroforests, the proportionally large C pool has the capacity to immobilize the additional N. Phosphorus losses were identical among organically- and mineral-fertilized agroforests at depth (1.4 kg P ha^{-1} yr^{-1}; Chapter 4, Figure 2B). Although P pools tended to be larger in organically-fertilized agroforests at depth, the high P sorption capacity and the presence of iron oxides in highly weathered, tropical soils is likely responsible for retaining P and preventing loss regardless of pool size (Appendix Chapter 3, Figure 3B). Further, higher soil bulk density at depth (Figure 4) increases the number of binding sites ions, and

provides a mechanism for both N and P retention. Finally, increased anion adsorption has been proposed as a secondary mechanism for nitrate and phosphate retention in deep soils as pH tends to increase at depth in these soils (Harmand et al. 2007).

Nutrient leaching and plant cycling drive the distribution of soil nutrients with depth. For example, while nutrient leaching tends to lead to an accumulation of nutrients at depth (below the rooting zone), plant cycling has the opposite effect: drawing deep nutrients up to the surface leading to accumulation in surface soils (Jobbágy and Jackson 2001). Although there was no effect of individual canopy species on nutrient pools or retention capacity, the effect of biology was apparent as I examined the vertical distribution of nutrients. In all cases, soil nutrient pools declined with depth (with sharp declines in the first 15cm) indicating the important role plants play in removing nutrients from deeper soil layers and preventing nutrient loss (Figure 5).

5.0 Conclusions

The cultivation of permanent crops such as shade-grown coffee, which are not subject to total annual biomass removal or tilling, prevent erosion and soil compaction compared to annual crops. This research asked the question, once a farmer makes the decision to maintain an agroforest, are there species- or fertilizer-specific effects on soil nutrient retention capacity or soil nutrient pools? Overall, I observed very few differences between mineral and organically-fertilized agroforests, and no effect of species. Clearly, the inclusion of trees mediates the negative effects of heavy mineral fertilizer applications through their active role in nutrient cycling. For example, the addition of high quantities of litter and pruning residues helps maintain high soil C pools,

and the storage of nutrients in aboveground biomass prevents nutrient loss. Therefore, the structure of an agroforests can preserve inherent soil properties and structures that retain nutrients and prevent loss.

References

Agbenin, J. O. 2003. Extractable iron and aluminum effects on phosphate sorption in a savanna alfisol. Soil Science Society of America Journal 67:589.

Arshad, M.A. and G.M. Coen. 1992. Characterization of soil quality: Physical and chemical criteria. American Journal of Alternative Agriculture 7:25-31.

Babbar, L. I., and D. R. Zak. 1995. Nitrogen loss from coffee agroecosystem in Costa Rica: leaching and denitrification in the presence and absence of shade trees. Journal of Environmental Quality 24:227-233.

Barrett, J. E., and I. C. Burke. 2000. Potential nitrogen immobilization in grassland soils across a soil organic matter gradient. Soil Biology and Biochemistry 32:1707-1716.

Barrow, N. J. 1984. Modelling the effects of pH on phosphate sorption by soils. European Journal of Soil Science 35:283-297.

Beer, J., R. Muschler, D. Kass, and E. Somarriba. 1998. Shade management in coffee and cacao plantations. Agroforestry Systems 38:139-164.

Bowden, J. W., A. M. Posner, J. P. Quirk, and B. K. G. Theng. 1980. Adsorption and charging phenomena in variable charge soils. Soils with variable charge. 147-166.

Boyce, J., A. Fernández, E. Fürst, and O. Segura. 1994. Café y desarrollo sostenible. Universidad Nacional, San Jose.

Cassman, K. G. 1999. Ecological intensification of cereal production systems: Yield potential, soil quality, and precision agriculture. Proceedings of the National Academy of Sciences of the United States of America 96:5952.

Chander, K., S. Goyal, D. P. Nandal, and K. K. Kapoor. 1998. Soil organic matter, microbial biomass and enzyme activities in a tropical agroforestry system. Biology and Fertility of Soils 27:168-172.

Clark, F. E. 1977. Internal cycling of nitrogen in shortgrass prairie. Ecology 58:1322-1333.

Clark, K. L., N. M. Nadkarni, D. Schaefer, and H. L. Gholz. 1998. Atmospheric deposition and net retention of ions by the canopy in a tropical montane forest, Monteverde, Costa Rica. Journal of Tropical Ecology 14:27-45.

Cross, A. F., and W. H. Schlesinger. 1995. A literature review and evaluation of the Hedley fractionation: Applications to the biogeochemical cycle of soil phosphorus in natural ecosystems. Geoderma 64:197-214.

Darke, A. K., and M. R. Walbridge. 1994. Estimating noncrystalline and crystalline aluminum and iron by selective dissolution in a riparian forest soil. Communications in Soil Science and Plant Analysis 25:2089-2101.

Davidson, E. A., J. M. Stark, and M. K. Firestone. 1990. Microbial production and consumption of nitrate in an annual grassland. Ecology 71:1968-1975.

Drinkwater, L. E., D. K. Letourneau, F. Workneh, A. H. C. Van Bruggen, and C. Shennan. 1995. Fundamental differences between conventional and organic tomato agroecosystems in California. Ecological Applications 5:1098-1112.

Eviner, V. T., and F. S. Chapin, III. 2003. Functional matrix: a conceptual framework for predicting plant effects on ecosystem processes. Annual Review of Ecology and Systematics 34:455-485.

Fließbach, A., H. -R. Oberholzer, L. Gunst, and M. Mader. 2007. Soil organic matter and biological soil quality indicators after 21 years of organic and conventional farming. Agriculture, Ecosystems and Environment 118:273-284.

Gerhardt, R. A. 1997. A comparative analysis of the effects of organic and conventional farming systems on soil structure. Biological Agriculture and Horticulture: United Kingdom.

Gillman, G. P. 1979. A proposed method for the measurement of exchange properties of highly weathered soils. Australian Journal of Soil Research 17:129-139.

Goodman, D. 1999. Agro-food studied in the 'Age of Ecology': Nature, corporeality, bio-Politics. Sociologio Ruralis 19:17-38.

Harmand, J. M., H. Avila, E. Dambrine, U. Skiba, S. de Miguel, R. V. Renderos, R. Oliver, F. Jimenez, and J. Beer. 2007. Nitrogen dynamics and soil nitrate retention in a *Coffea arabica-Eucalyptus deglupta* agroforestry system in Southern Costa Rica. Biogeochemistry 85:125-139.

Haynes, R. J. 1983. Soil acidification induced by leguminous crops. Grass and forage science 38:1-11.

Haynes, R. J., and R. Naidu. 1998. Influence of lime, fertilizer and manure applications on soil organic matter content and soil physical conditions: a review. Nutrient Cycling in Agroecosystems 51:123-137.

Howarth, R. W. 1988. Nutrient limitation of net primary production in marine ecosystems. Annual Review of Ecology and Systematics 19:89-110.

International Coffee Organization. 2010. Annual and monthly averages of coffee process. Retrieved November 31, 2010, from http://www.ico.org/prices/p2.htm.

Jobbágy, E. G., and R. B. Jackson. 2001. The distribution of soil nutrients with depth: global patterns and the imprint of plants. Biogeochemistry 53:51-77.

Kafkafi, U., B. Bar-Yosef, R. Rosenberg, and G. Sposito. 1988. Phosphorus adsorption by kaolinite and montmorillonite: II. Organic anion competition. Soil Science Society of America Journal 52:1585-1589.

Lal, R. 1989. Agroforestry systems and soil surface management of a tropical alfisol: II. water runoff, soil erosion, and nutrient loss. Agroforestry Systems 8:97-111.

Lal, R. 2004. Soil carbon sequestration impacts on global climate change and food security. Science 304:1623.

Liebig, M. A., and J. W. Doran. 1999. Impact of organic production practices on soil quality indicators. Journal of Environmental Quality 28:1601-1609.

Lopez-Hernandez, D. S., and G. Rodriguez. 1986. Competitive Adsorption of Phosphate with Malate and Oxalate by Tropical Soils1. Soil Science Society of America Journal 50:1460.

Lotter, D. W. 2003. Organic agriculture. Journal of Sustainable Development 21:59-128.

Lyngbæk, A. E., R. G. Muschler, and F. L. Sinclair. 2001. Productivity and profitability of multi-strata organic versus conventional coffee farms in Costa Rica. Agroforestry Systems 53:205-213.

Mader, p., A. Fließbach , D. Dubois, L. Gunst, P. Fried, and U. Niggli. 2002. Soil fertility and biodiversity in organic farming. Science 269:1694-1697.

Melero, S., J. C. R. Porras, J. F. Herencia, and E. Madejon. 2006. Chemical and biochemical properties in a silty loam soil under conventional and organic management. Soil and Tillage Research 90:162-170.

Nixon, S. W., J. W. Ammerman, L. P. Atkinson, V. M. Berounsky, G. Billen, W. C. Boicourt, W. R. Boynton, T. M. Church, D. M. Ditoro, and R. Elmgren. 1996. The fate of nitrogen and phosphorus at the land-sea margin of the North Atlantic Ocean. Biogeochemistry 35:141-180.

Pimentel, D., P. Hepperly, J. Hanson, D. Douds, and R. Seidel. 2005. Environmental, energetic, and economic comparisons of organic and conventional farming systems. Bioscience 55:573-582.

Sanchez, P. A., C. A. Palm, and S. W. Buol. 2003. Fertility capacity soil classification: a tool to help assess soil quality in the tropics. Geoderma 114:157-185.

Schindler, D. W. 1977. Evolution of phosphorus limitation in lakes. Science 195:260-262.

Selvaradjou, S-K., L. Montanarella, O. Spaargaren, and D. Dent. 2005. European Digital Archive of Soil Maps (EuDASM) - Soil Maps of Latin America and Caribbean Islands. Office of the Official Publications of the European Communities, Luxembourg.

Seybold, C. A., M. J. Mausbach, D. L. Karlen, and H. H. Rogers. 1997. Quantification of soil quality. Soil processes and the carbon cycle 387-404.

Sharpley, A. N., S. J. Smith, and J. W. Naney. 1987. Environmental impact of agricultural nitrogen and phosphorus use. Journal of Agricultural and Food Chemistry 35:812-817.

Sibanda, H. M., and S. D. Young. 1986. Competitive adsorption of humus acids and phosphate on goethite, gibbsite and two tropical soils. European Journal of Soil Science 37:197-204.

Szott, L. T., and D. C. L. Kass. 1993. Fertilizers in agroforestry systems. Agroforestry Systems 23:157-176.

Szott, L. T., and G. Melendez. 2001. Phosphorus availability under annual cropping, alley cropping, and multistrata agroforestry systems. Agroforestry Systems 53:125-132.

Tiessen, H., and J. O. Moir. 1983. Characterization of available P by sequential extraction. In M. R. Carter. Editor. Soil Sampling and Methods of Analysis, Ann Arbor.

Tucker, C. M., H. Eakin, and E. J. Castellanos. 2010. Perceptions of risk and adaptation: Coffee producers, market shocks, and extreme weather in Central America and Mexico. Global Environmental Change 20:23-32.

Turner, D. W., C. Korawis, and A. D. Robson. 1989. Soil analysis and its relationship with leaf analysis and banana yield with special reference to a study at Carnarvon, Western Australia. Fruits 44:193-203.

Vance, G. F., F. J. Stevenson, and F. J. Sikora. 1996. Environmental chemistry of aluminum-organic complexes. In The Environmental Chemistry of Aluminum, G. Sposito, Editor. CRC Press, Florida. p169-220.

Vitousek, P. M. 1984. Litterfall, nutrient cycling, and nutrient limitation in tropical forests. Ecology 65:285-298.

Wells, A. T., K. Y. Chan, and P. S. Cornish. 2000. Comparison of conventional and alternative vegetable farming systems on the properties of a yellow earth in New South Wales. Agriculture, ecosystems & environment 80:47-60.

Young, A. 1986. Effects of trees on soils. *In* R.T. Prinsley and M.J. Swift, Editors. Amelioration of soils by trees: A review of current concepts and practices. London: Commonwealth Science Council.

Young, A. 1989. Agroforestry for soil conservation. CAB International Wallingford, UK.

Chapter 6: Canopy and leaf composition drive patterns of nutrient release from pruning residues in a coffee agroforest

Abstract

In a coffee-agroforest, the crop is cultivated under the shade of canopy trees. These trees are periodically pruned to promote flowering and fruiting as well as to make nutrients stored in shade-tree biomass available to plants. I investigated the effect of canopy composition and substrate quality on decomposition rates and patterns of nutrient release from pruning residues in a coffee agroforest located in Costa Rica's Central Valley. Initial phosphorus (P) release was enhanced under a canopy comprised solely of nitrogen (N)-fixing, *Erythrina poeppigiana* compared to a mixed canopy of *Erythrina-Musa acuminata* (banana). Although initial rates of mass loss and N release were the same between the two canopy types, after one year of decomposition, a higher proportion of initial N had been released under the mixed canopy. Decomposition rates and patterns of nutrient release were not predicted by initial substrate quality. However, mass remaining of species mixtures was well-predicted by mean mass of their component species. This study identifies pruning regimes that may regulate N and P release during crucial growth periods, and suggests that in some cases less is more. For example, during the onset of rapid fruit growth, a two-species mixture may release more P than three-species mixture. However by the time of the harvest, the two- and three-species mixtures have released roughly the same amount of N and P. These nutrients do not always follow the same pattern, as N release is maximized in single-species substrates, while P release is enhanced in species mixtures. Thus, this study indicates the importance of (a) maintaining a diverse canopy and (b) timing of prunings to match crop nutrient demand.

1.0 Introduction

The presence of trees in cropping systems (agroforests) can enhance soil fertility both through the recycling of nutrients in litterfall and pruning residues and via N-fixation of leguminous species. In an agroforest, the decomposition of pruning residues will constrain the release of nutrients to the crop and may also inhibit the delivery of nutrients applied as mineral fertilizer. Understanding the timing and rate of nutrient release from pruning residues will help inform management strategies which optimize on-farm nutrient use. Unlike unmanaged systems where decomposition occurs primarily on senesced leaf tissues, the periodic pruning of shade trees transfers large quantities of green leaf tissues to the forest floor. Little is known about how the high nutrient content of these leaves and their mixtures will alter decomposition rates and patterns of nutrient release. Conducted in a *Coffea arabica* (coffee) agroforest, this research examines the effects of (a) canopy composition and (b) leaf composition on decomposition rates and nutrient release.

Shade trees add structural and functional diversity to the agroecosystem, and by extension, affect nutrient cycling. Structurally, plant species may alter decomposition rates by altering the physical environment or the soil community (Eviner and Chapin 2003). In addition, canopy architecture fundamentally alters the microclimate through effects on the percent shade cover (Vanlauwe et al. 1997), which in turn can alter biogeochemistry (Mack and D'Antonio 2003).

Functionally, trees can drive changes in the chemical environment. For example, *Erythrina poeppigiana* in combination with its microbial symbionts, fixes atmospheric-N,

contributing N directly to the soil. Furthermore, the decomposition of its high-quality leaves indirectly contributes to soil N availability. The leaves of other shade species such as *Musa acuminata* (banana) contain less N and more lignin (Sun 1998, Oliveira et al. 2006) and may break down more slowly. In addition, bananas are cultivated as a secondary crop and, therefore, may compete with the coffee crop for nutrients. *Erythrina* trees, on the other hand, are grown solely for their ability to supplement the system with nutrients. Thus, some trees may be a nutrient source (*Erythrina*) in the system, and others may be a nutrient sink (*Musa*).

The periodic pruning of shade trees provides a major source of nutrients to the crop (Schroth et al. 1992) and helps maintain soil fertility (Youkhana and Idol 2009). With the exception of P, nutrient transfers from litterfall and pruning residues are roughly equal to the flux of nutrients exported annually in the crop. Therefore, in theory, they can supply sufficient nutrients to meet plant demand (Szott and Kass 1993, Palm 1995). However, incubation studies show that the crop only assimilates about 20% of the N released from pruning residues or litter (Palm 1995). The remaining 40-80% of the N from pruning residues and litter may be incorporated into soil organic matter (SOM; Haggar et al. 1993) or lost via leaching. Due to the slow nutrient release from SOM, organic inputs may have more enduring positive effects on soil fertility than synthetic fertilizers. Therefore the quality of pruning residues and litter may constrain both the initial uptake of nutrients by the crop as well as long term soil fertility and SOM formation.

Previous research in agroforests has shown that elevated tissue quality (Palm and Sanchez 1990; McGrath et al. 2000; Jaramillo-Botero et al. 2008; Lamers et al. 2010), precipitation (Teklay 2007; Das and Das 2010), microbial activity (Kurzatkowski et al. 2004), and soil nutrient availability (Gnankambary et al. 2008) can enhance rates of mass loss and nutrient release. All of the previously mentioned studies were conducted on single leaf species, with very few examining decomposition rates in mixtures of leaves commonly found in agroforests (see Youkhana and Idol 2009 and Gnankambary et al. 2008). In theory, nutrients can transfer among leaf types such that diversity enhances decomposition rates (Hättenschwiler et al. 2005). Transfers can occur either through leaching or via fungal hyphae and microbes, and may reduce nutrient limitation on decomposition rates such that high-quality litter with fast decomposition rates may increase decomposition rates in lower-quality litter (Seastedt 1984; Wardle et al. 1997). To my knowledge, no previous studies have examined how leaf mixtures of commonly cultivated shade trees may facilitate or impede nutrient release from decomposing litter. In addition, throughout the decomposition literature, the effect of canopy composition (e.g. effect of different functional types in the canopy) on decomposition rates is not well described (but see Hobbie et al. 2006).

Agroforests are inherently diverse ecosystems in which nutrient and light management is critical to crop productivity. Information regarding the timing of nutrient release from leaves and how these rates are driven by leaf quality and canopy species can help inform management practices that will improve both soil fertility and productivity. Here I examine decomposition rates and patterns of nutrient release under different

canopy types (single-*Erythrina* vs. mixed-*Musa*-*Erythrina* canopies) and in different leaf mixtures of *Coffea* and three canopy trees commonly cultivated in the Central Valley of Costa Rica. I predicted that the single-canopy will facilitate rapid initial rates of mass loss and nutrient release (and ultimately more complete loss) as the presence of high quality *Erythrina* leaves on the forest floor will promote nutrient transfer among leaf materials. Due to high tissue quality in *Erythrina* and *Cordia* leaves, I predicted rapid initial mass loss and nutrient release from these substrates and from mixtures containing either of these two species. *Coffea* and *Musa* leaves are of lower quality, and therefore I expected to observe slower rates and less complete nutrient release from these substrates and their mixtures.

2.0 Methods

2.1 Study Site

The Central Valley of Costa Rica has a humid tropical climate favorable to agroforestry systems. The study region in the Central Valley receives on average 2600mm of rainfall per year with a mean annual temperature of 22.6°C. Rainfall is seasonal; the dry season extends from February through May with March being the driest month. The study period (July 2008-July 2009) was exceptionally wet, with a cumulative rainfall of 3530mm from initial deployment to the final collection. In addition, the region experienced an unusually wet December (708mm) and February (544mm) and an extremely dry April (39mm). The mean monthly maximum and minimum temperatures during the dry season were 35.1°C and 23.8°C (respectively; Figure 1). Soils are of volcanic origin and are characterized as a Typic Humitropept (Selvaradjou et al. 2005)

with a clay-loam texture. *Coffea arabica* (coffee) is typically cultivated in combination with shade trees such as *Erythrina poeppigiana*, *Musa acuminata*, *Cordia alliodora*, as well as a variety of fruit trees.

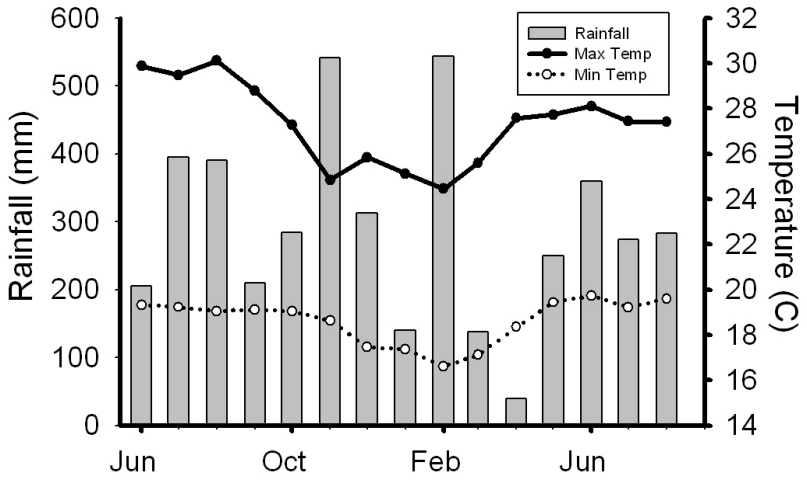


Figure 1: Monthly rainfall and temperature from July 2008-July 2009 in Central Valley of Costa Rica. Rainfall and temperature across the study period. Solid line indicates mean daily maximum temperature and dotted line indicated mean daily minimum temperature.

2.2 Field Experiment

The decomposition experiment was conducted in a coffee agroforest (9°53'N 83°41'W) near Turrialba, Costa Rica. The farm is located 1000masl and is divided into two distinct but adjacent sections. In the eastern part of the farm coffee is cultivated solely under the shade of *Erythrina* (single-canopy). In the western section, both *Erythrina* and *Musa* (mixed-canopy) comprise the canopy. The sections had similar shade tree density with an average of 700 trees ha^{-1}. In the mixed-canopy, *Musa* comprised roughly 30% of shade trees.

Fresh leaves of *Erythrina, Musa,* and *Cordia* were collected from the crowns of randomly selected shade trees during June of 2008. *Cordia* leaves were harvested from a

neighboring farm, as they were not present in the focal farm. Fully expanded *Coffea* leaves were collected at the same time. Leaves for individual species were composited and mixed before air-drying in a greenhouse for 4 days. Sub-samples of individual species and mixtures were taken to determine oven-dry weight (65°C) and initial nutrient concentrations.

To test the effect of canopy type (single and mixed) and leaf species (*Coffea*, *Erythrina*, *Cordia*, and *Musa* and their two-species and three-species combinations) on decomposition rates and patterns of nutrient release, I established a 2 x 7 factorial randomized block design with four replicates and seven repeated measurements over the course of one year (total of 392 bags). To further test the effect of leaf species and leaf species combinations, I conducted a "Common Garden" experiment (randomized block) under the mixed canopy. I examined decomposition rates and patterns of nutrient release in *Coffea*, *Erythrina*, *Cordia*, and *Musa* leaves and their 11 possible combinations (Table 1). Five grams total of each individual species and their mixtures were placed in 15cm x 15cm nylon bags (2mm mesh; which is wide enough to allow access to most decomposer macro-fauna such as termites; see Yamashita and Takeda 1998). The decomposition bags were distributed under the trunks of either *Erythrina* in the single-canopy or *Musa* in the mixed-canopy in order to ensure that the bags were exposed to *Musa* leaves. Litterbags were placed on top of the mineral soil, and recovered at 2, 4, 8, 19, 33, 41, and 52 weeks. Four replicates per treatment were collected each time. After collection, bags were air-dried for 4 days. The residues were then removed, cleaned of visible roots, soil, and fungi, oven-dried at 65°C for 48 hours, and weighed. Samples were ground in a 20-

mesh Wiley mill (Thomas Scientific, Inc., Swedesboro, NJ, USA). Subsamples (0.5g) were ashed in a muffle furnace at 500°C for 4 hours to determine inorganic content. All data are reported on an ash-free, dry-weight basis.

Table 1: Initial measures of leaf tissue quality, fitted decomposition rates (observed and expected), and short- and long-term nitrogen and phosphorus release among leaves and leaf-mixtures used in decomposition experiment. Nutrient ratios were calculated on a molar basis.

Species	mg P g^{-1}	%N	%C	C:N	C:P	*k*-value		% P Released		%N Released	
						Obs	Exp	4 wk	52 wk	4 wk	52 wk
Coffea arabica	2.0	3.6	47.2	15	616	2.0		36	95	34	94
Musa acuminata	2.1	3.3	46.4	17	563	1.9		57	88	57	88
Erythrina poeppigiana	3.6	5.7	47.9	10	345	2.0		50	89	54	90
Cordia alliodora	3.7	4.9	46.8	11	331	1.7		36	90	55	88
Coffea-Erythrina	2.6	4.4	48.7	13	489	3.3	2.0	68	99	35	98
Coffea-Musa	1.8	3.6	47.0	15	692	2.0	2.0	36	88	43	87
Cordia-Coffea	2.4	4.4	47.3	12	510	1.8	1.8	63	88	52	88
Musa-Erythrina	2.7	4.6	48.1	12	457	2.1	2.0	42	88	49	89
Musa-Cordia	2.6	4.3	46.1	12	463	1.3	1.8	74	84	50	78
Cordia-Erythrina	2.9	4.9	47.0	11	421	1.1	1.8	52	81	41	67
Coffea-Musa-Erythrina	2.3	4.3	47.6	13	535	1.9	2.0	64	98	39	98
Coffea-Musa-Cordia	2.1	4.0	46.8	14	586	1.8	1.9	67	88	52	89
Coffea-Cordia-Erythrina	2.5	4.8	47.1	11	484	1.7	1.9	51	80	22	84
Musa-Cordia-Erythrina	2.6	4.5	47.3	12	474	2.2	1.9	60	87	44	85
Coffea-Musa-Cordia-Erythrina	2.2	4.2	46.8	13	541	1.9	1.9	14	83	28	86

Ground samples were dry-combusted on an elemental analyzer (Carlo Erba, Model NA 2500; Milan, Italy) to determine total C and N. Total P was determined using a modified Kjeldahl protocol and the resulting solutions were analyzed on an Alpkem Flow Solution IV Autoanalyzer (OI Analytical; College Station, Texas, USA).

2.3 Soil Nutrients

Beginning in September of 2008, soil samples (0-10cm) were collected monthly from each canopy type (Chapter 3, Methods). Soils were air-dried and sieved (2mm mesh) prior to nutrient analysis. Bioavailable soil P was determined using a modified Bray-extraction. Three grams of sieved, air-dried soils were shaken for 1 minute in 25mL of a 0.03mol L^{-1} NH_4F and 0.025mol L^{-1} HCL solution (Bray and Kurtz 1945). Extracts were filtered and P concentration was determined colorimetrically using a molybdate blue methodology on an Alpkem Flow Solution IV Autoanalyzer (OI Analytical, College Station, Texas, USA). A portion of soil was also ground to 149µm and dry-combusted on an elemental analyzer to determine total N and C (Carlo Erba, Model NA 2500; Milan, Italy). Subsamples were oven-dried at 105°C for 24 hours to determine gravimetric soil moisture. All data are reported on a dry mass basis.

2.4 Statistical Approach

I used two-way analysis of variance (ANOVA) to examine the effect of time and canopy type on soil C, N, P, and gravimetric soil moisture (Appendix Table 1). Single-exponential decomposition models were calculated for each canopy type based on mass remaining at each collection (Eq. 1; n=4 for each substrate at each time interval).

$$X=100e^{-kt} \quad \text{(Eq. 1)}$$

Where X is the proportion of initial mass remaining at time *t*, *k* is the decomposition rate constant, and *t* is the time in weeks. The model was fitted with the restriction that at time=0 all of initial litter was present (X_0=100; Wieder and Lang 1982), and that *k*-values could not be <0 (Wieder and Wright 1995). Decomposition rates were also calculated for each canopy type based on mass remaining at each collection.

Percent of the original N or P remaining was calculated by dividing the nutrient content of the substrate at each collection by the original nutrient content of that substrate and multiplying by 100. Nutrient content at each collection was calculated by multiplying the nutrient concentration by the mass remaining (ash-free weight). Nutrient content was then reported as a proportion of the initial leaf content, with values greater than 1 corresponding to an accumulation of nutrients through microbial immobilization (atmospheric deposition is negligible at the scale of one sample, see Imbach et al. 1989).

The effects of canopy composition on decomposition rates and patterns of nutrient release were determined by ANOVA with mass, N, and P remaining (in milligrams) as dependent variables and week, canopy type, and substrate tested as main effects (Appendix Table 2). However, I discuss these results in terms of percent of original mass, N, or P remaining. Substrate effects in the "Common Garden" were compared using a two-way ANOVA with mass, N, P as dependent variables and time and leaf substrate as main effects (Appendix Table 3). Significant differences among treatments were determined using *post-hoc* Tukey-Kramer multiple comparison tests. I analyzed the relationship between measurements of substrate quality (N and P concentrations, C:N, C:P, and N:P) and decomposition rates (*k*) using Kendall's rank correlation coefficient.

To determine whether combining leaf species enhanced decomposition rates, I calculated predicted decomposition rates of multi-species mixtures as the mean decomposition rate of monocultures of each component species (k_i) (Eq. 2; n=11).

$$\text{Predicted decomposition rate } (k) = \frac{1}{n} \cdot \sum_{i=1}^{n} k_i \qquad \text{(Eq. 2)}$$

A matched pairs analysis was performed to determine if decomposition rates differed significantly from predicted values, and the ability to predict decomposition rates based on component species was determined by linear regression. Modifying Eq. 2, I calculated predicted mass remaining of multi-species mixtures as the mean mass remaining of monocultures of each component species for every collection. I performed a matched pairs analysis to determine if the observed values different from the predicted values, and linear regressions to determine if the slope differed significantly from 1. Finally, I performed multivariate correlations on mass remaining (g) between leaf mixtures and each monoculture substrate to determine which species was driving mass loss (each substrate n=28).

Data were natural log-transformed where necessary to meet the assumptions of the models. All data were analyzed using JMP 8.0 for Macintosh.

3.0 Results

Soil moisture soil C concentrations, N concentrations (but not C:N), and soil moisture were significantly higher in the single-canopy compared to the mixed-canopy (Table 2; $p<0.0001$ in all 3 cases and F-value=91.73, 80.85, and 62.14, respectively). Bioavailable P, on the other hand did not differ between canopy types. Bray P and soil

moisture varied significantly through time (Figure 2; p=0.014 *F*-value=3.60 and p<0.0001 and *F*-value=10.94, respectively).

Table 2: Selected chemical properties of the surface soil (0-10cm) from two different canopy types in a coffee agroforest in Costa Rica. Soil samples were measured monthly; values not connected by the same letter are significantly different.

	Single Canopy	**Mixed Canopy**
Total N (%)	0.63 a	0.37 b
Total C (%)	8.56 a	4.98 b
C:N (molar)	16	16
Bray1-P (μg/g)	3.54	4.12
Soil moisture (wt_{water}/dwt_{soil})	0.51 a	0.33 b

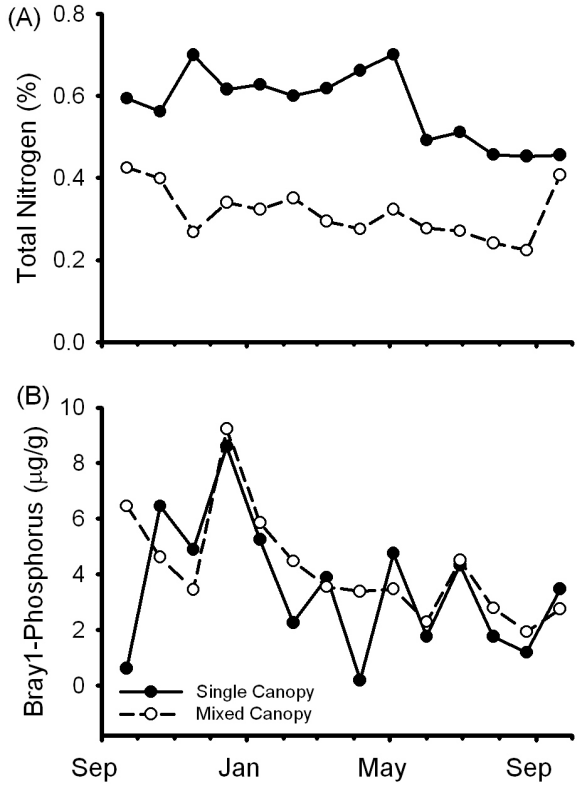


Figure 2: Soil N and P patterns through time by canopy type in a coffee agroforest in Costa Rica. (A) total soil N and (B) Bray-extractable P through time by canopy type. Solid lines indicate single-species (*Erythrina*) and dashed lines indicate mixed (*Erythrina-Musa*) canopy.

3.1 Canopy composition drives patterns in mass loss and nutrient release

Although initial and final decomposition rates were similar between the two canopy types, mass loss was greater under the mixed-canopy after five months of decomposition (week 19; Tukey $p<0.0001$, canopy*week F-value=9.27; Figure 3A). Phosphorus became available more rapidly under the single-canopy (weeks 2 and 4; Tukey $p<0.0001$, and $p=0.0020$ respectively; ANOVA $p<0.0001$, F-value=38.98; Figure 3C). Initial N release was rapid in both canopy types, but as for mass loss and P release, I observed a period of greater N release in the mixed-canopy after 5 months of decomposition (Tukey $p=0.0021$, canopy*week F-value=4.80; Figure 3B).

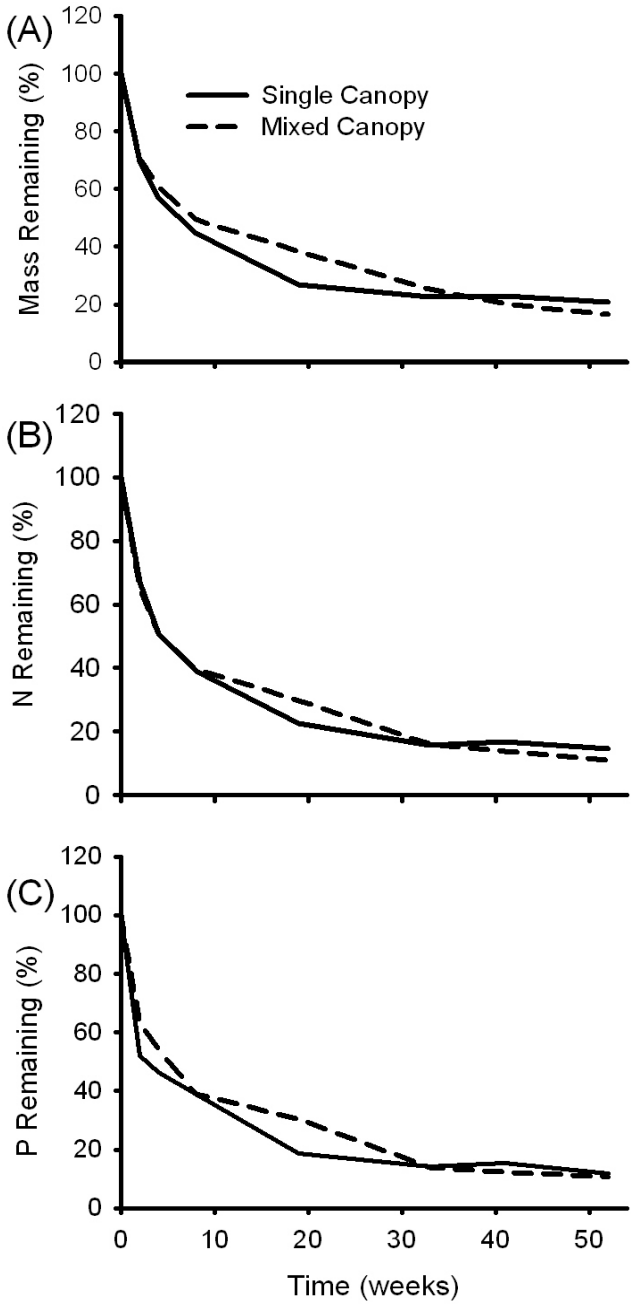


Figure 3: **Mass, N, and P remaining in leaves by canopy type over a year of decomposition in a coffee agroforest in Costa Rica.** Percent original (A) mass, (B) nitrogen, and (C) phosphorus remaining by canopy type. Solid lines indicate single-species (*Erythrina*) and dashed lines indicate mixed (*Erythrina-Musa*) canopy.

3.2 Leaf composition drives patterns in mass loss and nutrient release

Decomposition rates were rapid initially with nearly 60% mass lost in the first 4 weeks, and 80% of the mass lost on average over the one-year study period (Figure 4A). During the initial stages of decomposition (at 4 wks), mass loss was rapid in *Erythrina* leaves (53% remaining) while *Cordia* leaves showed slower (and linear; r^2=0.95; 72% remaining) mass loss. *Coffea* and *Musa* leaves had similar, intermediate rates of mass loss with an average of 66% of the original mass remaining at 4 weeks. After these initial stages, decomposition proceeded slowly in comparison to tropical lowland forests in Costa Rica, with higher temperatures and greater rainfall, where complete mass loss can be achieved in 4-6 months (Gessel et al. 1980; Wood et al. 2009). However, some substrates, including individual *Coffea* and *Cordia* leaves, as well as their combination, *Coffea-Cordia*, showed near complete decomposition within a year.

Although mass loss differed significantly among leaf type ($p<0.0001$, *F*-value=5.60), decomposition rates could not be predicted based on initial substrate quality (N and P concentrations, C:N or C:P), but was predicted by initial %C ($p=0.002$; r^2=0.53). Overall, decomposition rates (*k*-values) and mass remaining for leaf mixtures were predicted by the rates of individual species ($p<0.0001$ in both cases, respectively; Figure 5L). Some mixtures showed faster (*Coffea-Erythrina;* Figure 5A) and others slower (*Cordia-Erythrina* and *Cordia-Musa;* Figure 5E and F) mass loss than was predicted based on the mean loss in individual species. In general, mass, N, and P remaining in mixtures were best predicted by mass, N, and P remaining in *Erythrina* leaves.

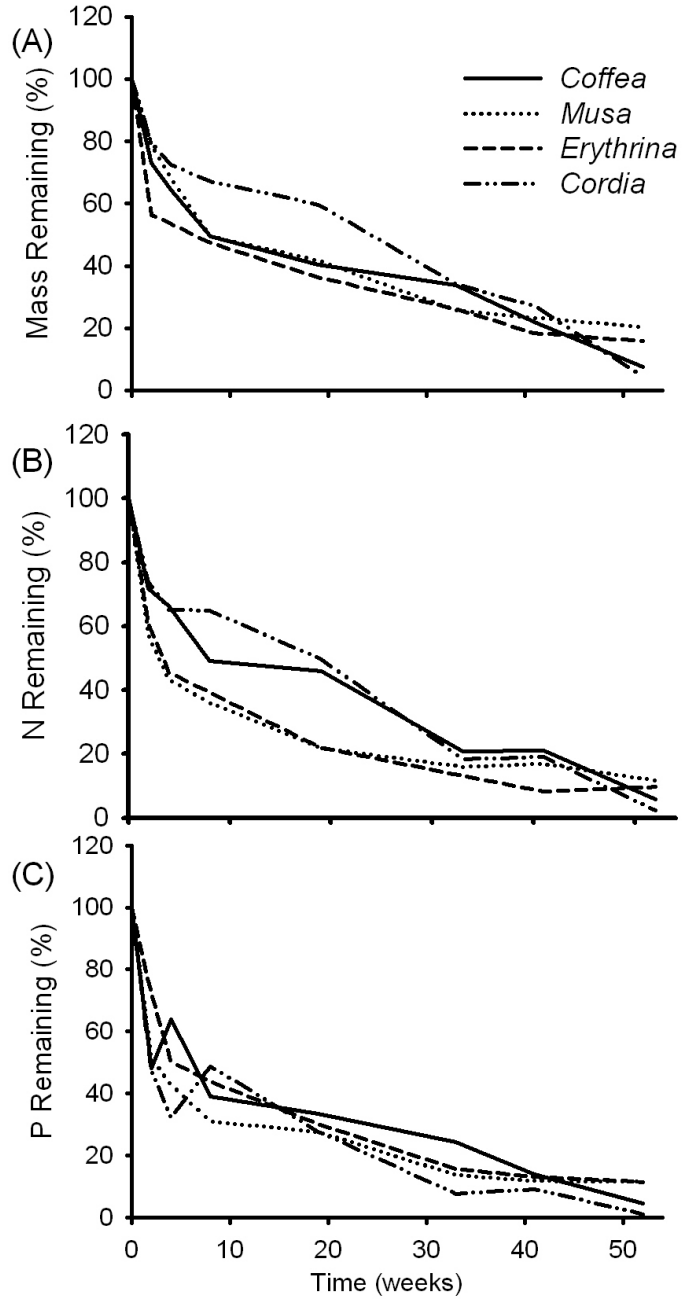


Figure 4: Mass, N, and P remaining by leaf type over a year of decomposition in a coffee agroforest in Costa Rica. Percent original (A) mass, (B) nitrogen, and (C) phosphorus remaining in the leaf tissues of *Coffea*, *Cordia*, *Erythrina* and *Musa*. Solid lines indicate *Coffea*, dotted lines indicate *Musa*, dashed lines indicate *Erythrina*, and dash-dot lines indicate *Cordia.*

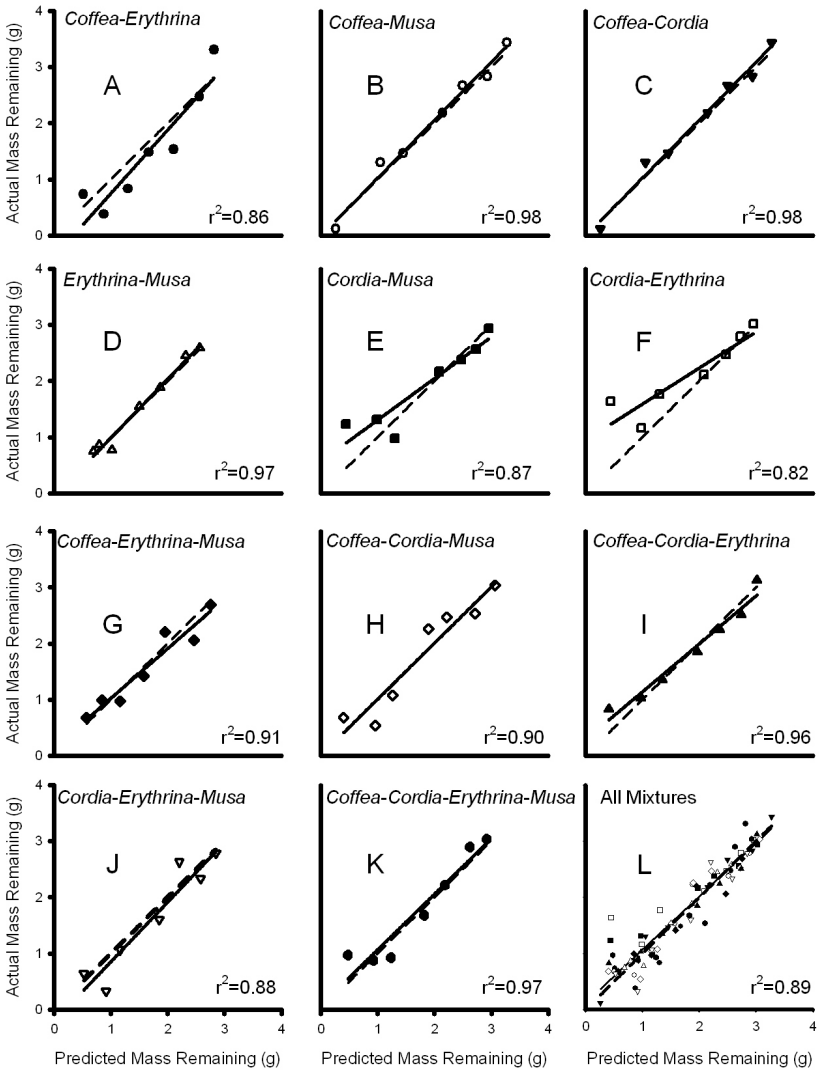


Figure 5: Actual versus predicted mass remaining among leaf mixtures over a year of decomposition in a coffee agroforest in Costa Rica. Predicted values derived from the mean of all component species. Solid represents line of best-fit, and dashed represents 1:1 line. All regressions are significant at $p<0.01$.

Patterns of N release differed widely among leaf materials (Figs 6-7; ANOVA $p<0.0001$, F-value=8.26), and the high quality leaves (*Erythrina* and *Cordia*) did not lose N at the same rate. In the short-term (first 4 weeks), I observed rapid N release from *Erythrina* leaves, which was not improved by combining with other species, but which was equal to a mixture of *Erythrina* and *Musa* leaves (Figure 6D). Over the same period, less N was released from *Cordia* leaves, but N release was enhanced when *Cordia* was

combined with other species, particularly *Musa* leaves (Figure 7D and E). In the long term (after a year of decomposition), I observed near identical N release from individual *Erythrina* leaves and its combinations, with the exception of *Erythrina* and *Cordia* mixtures where N release was inhibited (Figure 8D). Over the same time period, I observed almost complete N release from individual *Cordia* leaves, which was not improved by mixing *Cordia* with any other combination of species, but was equal to mixtures of *Cordia* and *Coffea* leaves (Figure 9D). Again, the four-species mixture failed to release as much N as mixtures with fewer species, and rather, was inhibited in the four-species mixtures (Figures 6 and 7).

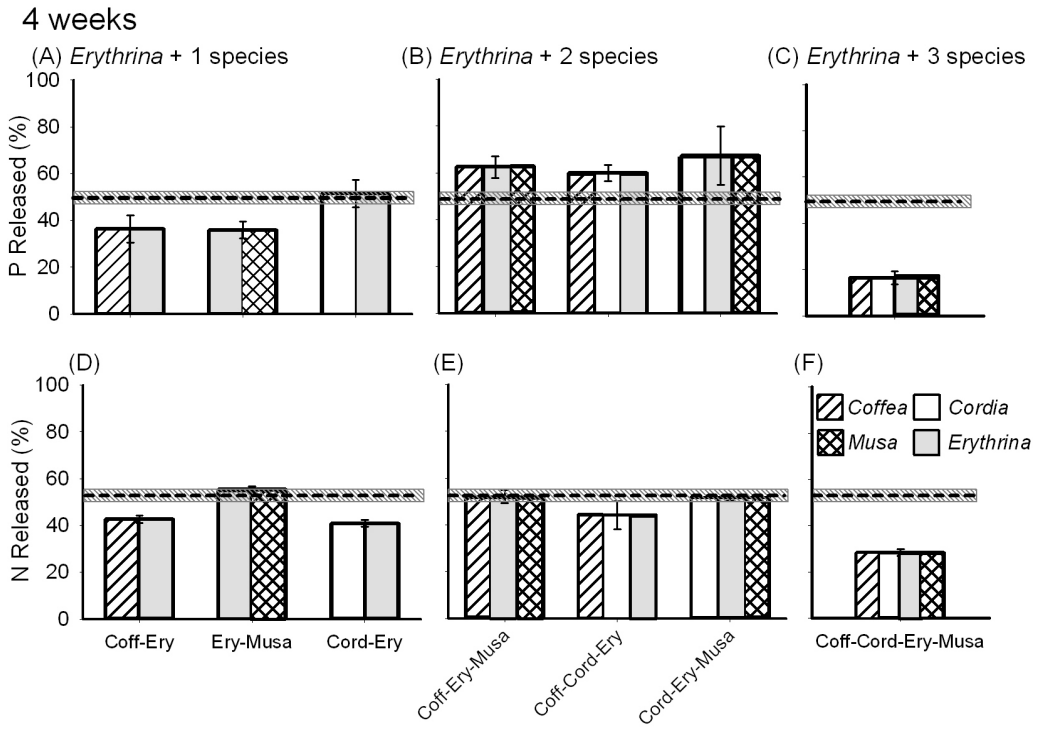


Figure 6: P and N release after 4 weeks of decomposition in *Erythrina* and *Erythrina*-mixtures in a coffee agroforest in Costa Rica. Phosphorus and nitrogen release after 4 weeks of decomposition in (A, D) *Erythrina* +1 species, (B, E) *Erythrina* +2 species and (C, F) *Erythrina* + 3 species leaf-mixtures. Error bars represent standard error if the mean, and dashed lines indicate nutrient release from *Erythrina* leaves alone; shaded areas represent standard error of the mean.

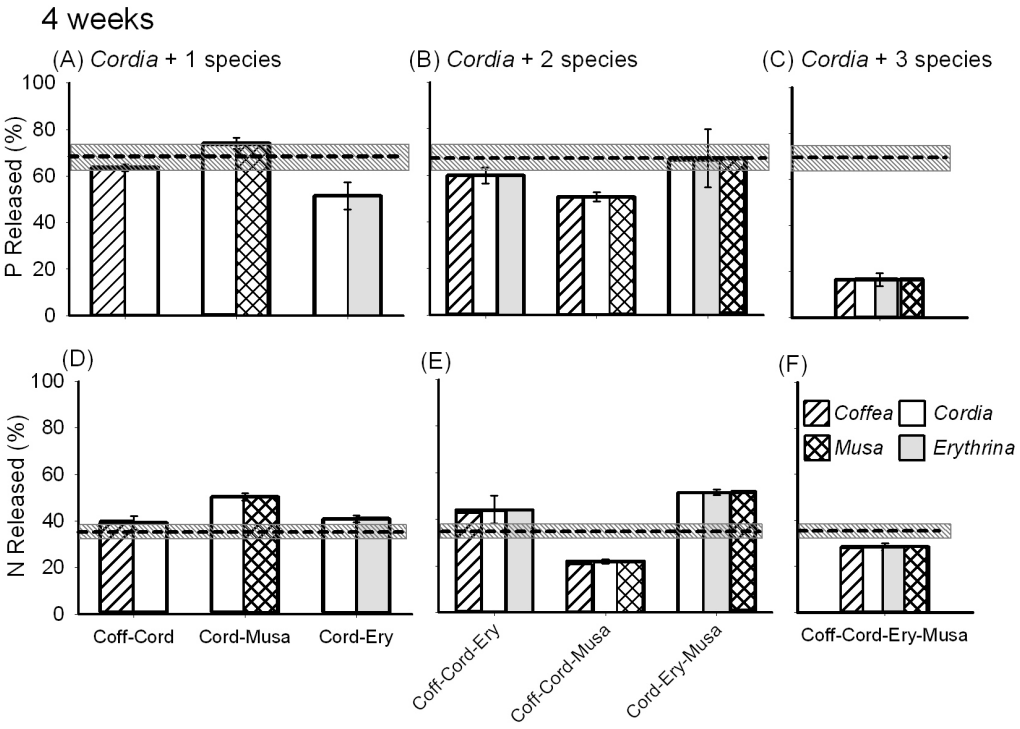


Figure 7: P and N release after 4 weeks of decomposition in *Cordia* and *Cordia*-mixtures in a coffee agroforest in Costa Rica. Phosphorus and nitrogen release after 4 weeks of decomposition in (A, D) *Cordia* +1 species, (B, E) *Cordia* +2 species and (C, F) *Cordia* + 3 species leaf-mixtures. Error bars represent standard error if the mean, and dashed lines indicate nutrient release from *Cordia* leaves alone; shaded areas represent standard error of the mean.

Initial phosphorus concentration was highest in *Cordia* and *Erythrina* leaves and lowest in *Coffea* leaves and the *Coffea*-*Musa* mixture. Although C:P ratios could not explain patterns of phosphorus release throughout the study, the patterns differed significantly among substrates (ANOVA p=0.003, *F*-value=2.46). Rapid P release was observed in all leaf materials with over half of P mineralized in the first 4 weeks (Figure 6 and 7). Phosphorus release was more synchronized among leaf materials than was N release. Unlike N where *Cordia* leaves inhibited initial mineralization, P release from *Cordia* leaves was rapid and was not improved in mixtures (although release was equally rapid in the *Cordia* and *Musa* mixture, Figure 7A). On the other hand, P release from

Erythrina leaves was enhanced in mixtures, especially when combined with two other species (Figure 6B). As for N, P release was significantly retarded in the four-species mixture.

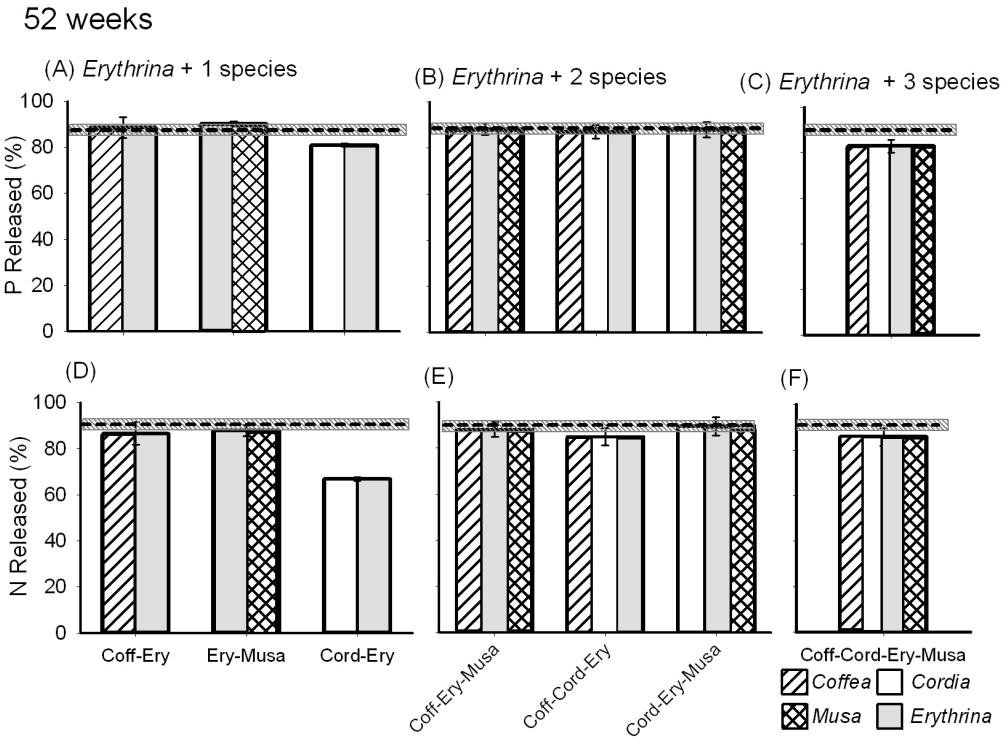


Figure 8: P and N release after 52 weeks of decomposition in *Erythrina* and *Erythrina*-mixtures in a coffee agroforest in Costa Rica. Phosphorus and nitrogen release after one year of decomposition in (A, D) *Erythrina* +1 species, (B, E) *Erythrina* +2 species and (C, F) *Erythrina* + 3 species leaf-mixtures. Error bars represent standard error if the mean and dashed lines indicate nutrient release from *Erythrina* leaves alone; shaded areas represent standard error of the mean.

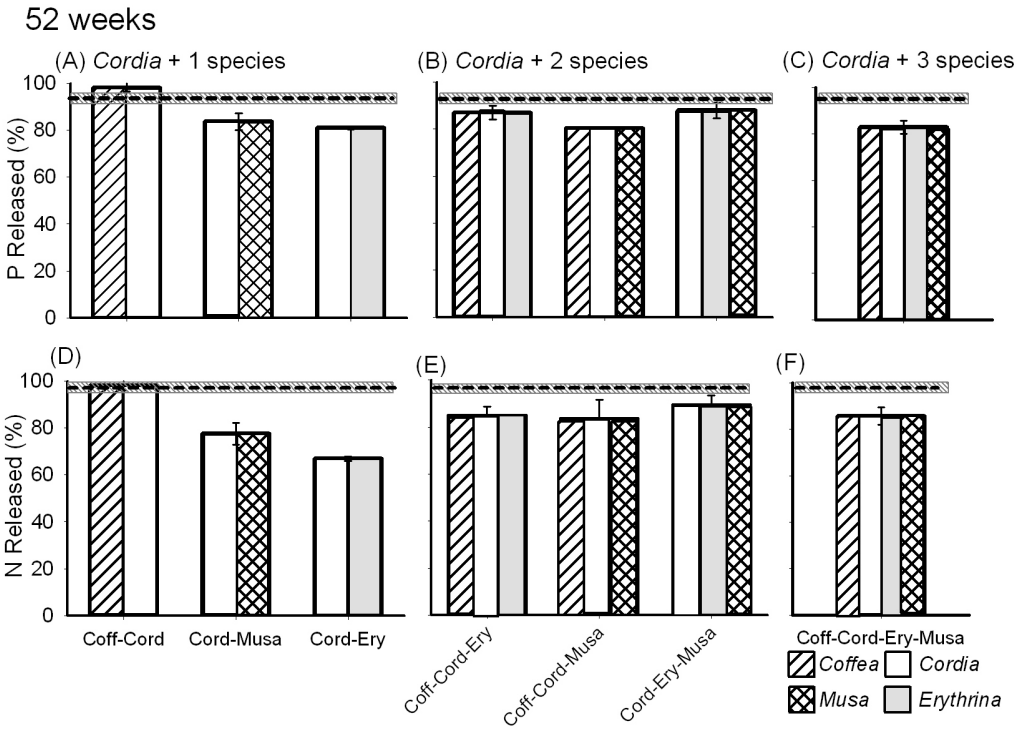


Figure 9: P and N release after 52 weeks of decomposition in *Cordia* and *Cordia*-mixtures in a coffee agroforest in Costa Rica. Phosphorus and nitrogen release after one year of decomposition in (A, D) *Cordia* +1 species, (B, E) *Cordia*+2 species and (C, F) *Cordia* + 3 species leaf-mixtures. Error bars represent standard error of the mean, and dashed lines indicate nutrient release from laurel leaves alone; shaded areas represent standard error of the mean.

In the long term (12 months), *Erythrina* and its mixtures showed roughly the same amount of P release (Figure 8). As for N, I observed almost complete P release from *Cordia* leaves, which was not improved by mixing with any other species, with the exception of *Coffea* leaves (Figure 9A). Overall, phosphorus remaining was well-predicted by the mean P remaining of component species ($p<0.0001$, $r^2=0.84$), and actual remaining P was not significantly different from predicted P remaining (t-test).

4.0 Discussion

4.1 Canopy composition drives patterns of mass loss and nutrient release

Initially, mass loss and nitrogen release were nearly identical under the single- and mixed-canopies, however, their patterns diverged after five months of decomposition with greater change under the mixed-canopy. Therefore, the maintenance of a diverse canopy may not affect immediate N release, but over the medium term, more nitrogen may be made available. Over the course of the study, total soil N (to 10 cm) was consistently higher under the single- compared to the mixed-canopy. This suggests that increasing the density of N-fixing *Erythrina* trees enhances soil N. This N may either (a) accumulate in soils/plant biomass or (b) leach to deeper soil layers. Leaching experiments under these species indicate that higher soil N under *Erythrina* trees corresponds with higher nitrate concentrations under these species compared to *Musa* (Chapter 3, Figure 8A).

Unlike patterns of mass loss and N release, initial P release from leaves was faster under the single-canopy, with a convergence in release between the two canopies over time. Bioavailable soil P varied significantly through time, but did not appear to be accumulating in the surface soils of either canopy type (Figure 2B). Thus, bioavailable P is likely quickly assimilated by plants or adsorbed onto clay surfaces and converted into a more recalcitrant form.

4.2 Leaf composition drives patterns of mass loss and nutrient release

Leaf mixtures had different rates of mass loss and patterns of nutrient release. Although these patterns were not predicted by initial leaf quality, decomposition rates

were strongly related to the percent N and P released after 52 weeks of decomposition. Not surprisingly, those tissues with faster decomposition rates also showed the greatest amount of N and P released by the end of the study. Although *Erythrina* leaves decomposed very rapidly (as is expected of high-nutrient plant tissues), *Cordia* leaves presented a slow, linear pattern of mass loss inconsistent with its high tissue quality. Nevertheless, after one year, *Erythrina* retained 15% and *Cordia* leaves only 4% of their original mass. Linear mass loss from *Cordia* leaves was also observed in an experimental forest in Panama, which receives similar amounts of rainfall, but has less fertile soils (Scherer-Lorenzen et al. 2007). On the other hand, mass loss in *Erythrina* leaves occurred twice as fast in the Upper Amazon Basin of Perú (comparable rainfall, but more fertile soils; Palm and Sanchez 1991).

The difference between *Cordia* and *Erythrina* may be due to higher lignin content in *Cordia* (Table 6). Thus, the rate of *Cordia* mass loss may be constrained by the rate at which microbes biodegrade lignin, while *Erythrina* mass loss is constrained by soil nutrient availability. Patterns of nutrient release also differed between *Erythrina* and *Cordia* leaves. In the short-term, individual *Erythrina* leaves released more N while individual *Cordia* leaves released more P. However, combining *Erythrina* with two species facilitated P release (Fig 6B) while combining *Cordia* with one or two other species enhanced N release (Figure 7D and E). Together, these results suggest that P tends to be immobilized in *Erythrina* leaves and N tends to be immobilized in *Cordia* leaves and that certain combinations of leaves can balance substrate stoichiometry to promote nutrient release.

Table 3: **Leaf or litter lignin concentrations in agroforestry trees.** Literature values of species-specific lignin concentrations.

Species	Lignin (%)	Substrate	Reference
Musa acuminata	30	leaves	Oliveira et al. 2006
Cordia alliodora	19	litter	Scherer-Lorenzen et al. 2007
Erythrina poeppigiana	9.7	leaves	Palm and Sanchez 1991

Overall, decomposition rates (*k*-values) were well predicted by the mean rates of their component species (Table 1). However, decomposition rates were faster in *Coffea-Erythrina* mixtures and slower in *Cordia-Erythrina* and *Cordia-Musa* mixtures compared to mean rate of their component species. The predicted mass and nutrients remaining were very similar to the actual mass and nutrients remaining for each time period (Figure 5). Thus, increasing leaf diversity will not necessarily enhance mass loss or nutrient release, but specific mixtures of species can either enhance or impede decomposition rates. In agroforests, where farmers control canopy composition and manage pruning regimes, it may be possible to optimize patterns of nutrient release by managing canopy composition (with impacts on both the environment and substrate for decomposition) and timing of pruning.

4.3 Effective Shade Coffee Management

Although N is typically considered to be the primary nutrient limiting productivity in agricultural systems, tropical ecosystems are, in general, P-limited compared to temperate systems (Vitousek 1984). It has been suggested that while the addition of leaves through pruning residues and natural litterfall have the potential to meet crop N and K demands, P demands cannot be met by biomass transfer as P is already in deficit in many of these systems (Palm and Sanchez 1991; Salazar et al. 1993;

(Palm 1995). The variety of patterns of N release I observed among leaf tissues may reflect a heterogeneous degree of N-limitation with an agroforest, dependent on microsite feedbacks between plants and soils. A similarity in patterns of P release among species may be indicative of general P-limitation. Periods of P-uptake by the decomposing material (such as *Coffea* and *Cordia*) suggest that P availability may constrain decomposition rates at certain periods (Figure 4C). However, none of the substrates, individual species or mixtures, displayed net nutrient immobilization (an increase in nutrients *over* the amount initially present).

Nevertheless, nutrient release was inhibited in the four-species mixture contrary to my expectation that this mixture would mimic nutrient release in component species. An equal amount (in grams) of each species was represented in the mixtures, a combination not often mirrored in agroforests as pruning typically overwhelms the system with one or two leaf mixtures. Inhibited nutrient release in the four-species mixture may suggest a management strategy of staggered pruning of select species. Furthermore, this result suggests that the theory of increased nutrient transfer with increased leaf diversity may not apply to agroforests. Humans manipulate C, N and P inputs to such a degree that simply increasing leaf diversity (to enhance nutrient transfer) will not meet the altered stoichiometric demands. This is not to say that nutrient release cannot be manipulated through pruning techniques, as this study clearly shows that certain combinations out-perform others. However, inhibited nutrient release in the four-species mixture does suggest that overwhelming the system with fresh leaves of all species present will not necessarily promote rapid nutrient release.

Erythrina trees can contribute on average 50% more organic material through litterfall than *Cordia* (Beer et al. 1990), and they are pruned regularly (2-3 times a year). Pruning is typically dictated by *Coffea* phenology as a means to increase light-availability during crucial growth periods (i.e. in January-February to promote flowering and fruit set and in June-July to promote berry ripening; Beer 1988). Many studies have shown the importance of light availability for promoting reproductive growth in coffee plants (Montoya et al. 1961; Castillo 1966; Cannell 1975).

On the other hand, pruning causes complete mortality for the bacterial symbionts in *Erythrina* root nodules, bringing nitrogen-fixation to a halt for about 10 weeks following pruning (Nygren and Ramirez 1995). Fortunately, large quantities of nutrients are added to the system following shade-tree pruning, but it is unclear whether this pulse of nutrients is synchronized with the period of greatest nutrient-demand by the crop species. Flowers begin to open in January, and the fruit set initiates in February and March. At this time coffee plants are pruned selectively to remove old and unhealthy branches, and *Erythrina* is pruned aggressively (all branches removed) to provide light, which promotes coffee fruit development (Maria Luisa Araya Fuentes *personal communication*). Initially (first 4-8 weeks) fruits grow slowly, followed by a period of rapid growth and an increase in volume at 10-11 weeks (mid-April; Arcila-Pulgarin et al. 2002). On the focal-farm, *Erythrina* is pruned again five months later (early July-only low-lying branches removed) to promote fruit ripening. *Coffea* fruits take 7-8 months to mature, and are harvested in this region between September and late November.

Nutrients may be in greater demand when fruits begin to grow rapidly (16 weeks after the first pruning), at which point, nutrient release should be greater under the *Erythrina* single-species canopy. However, in equatorial regions, there is often an overlapping of reproductive stages in *Coffea* such that flowers, green fruits, and mature fruits may coexist on a single plant. The overlap leads to competition for assimilates between organs and between vegetative and reproductive stages (Trojer 1968; Cannell 1985; Arcila-Pulgarín 1990). This suggests that there may not be a single period of great nutrient demand, but several throughout the course of fruit development. Nutrient release may differ initially between canopy types, but based on my data, the proportion of N and P released from the first pruning should be very similar under the two canopy types by the time of *Coffea* harvest (Figure 10). Thus, if *Coffea* plants require nutrients at a constant rate throughout the year, perhaps shade canopy type may not constrain nutrient availability. However, if *Coffea* plants have periods of strong demand that correspond with the enhanced photosynthetic activity brought on by increased light availability (immediately following pruning), the composition of the canopy may improve or inhibit their ability to turn more light into more fruit.

In addition to the effects of canopy composition are the effects of substrate on patterns of nutrient release, which farmers can manipulate by pruning management. In general, P release from all leaf mixtures was facilitated by the addition of *Cordia* leaves, suggesting that moderate pruning of *Cordia* trees should accompany any other major pruning event if P is considered in deficit. However, if *Cordia* is not present, as is the case on the focal farm, more complex strategies may be necessary. On a typical

schedule, both *Erythrina* and *Coffea* are pruned between December and January. Given the results of the study, initial N and P release from *Coffea-Erythrina* mixtures (4 weeks) can be facilitated by the addition of *Musa* leaves (Figure 6). However, 10 weeks following fruit set (period of rapid fruit growth), both two- and three-species mixtures released similar amounts of N and P (Figure 11). In July, when *Erythrina* is pruned a second time, these results suggest that the addition of *Musa* leaves would substantially improve P, but not N release come the harvest (Figure 11).

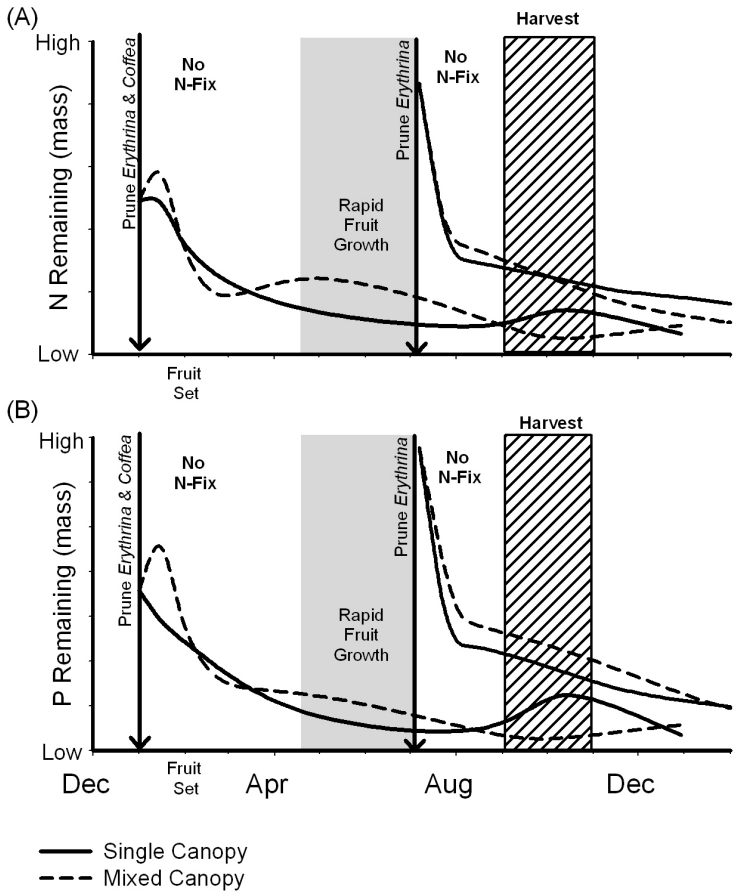


Figure 10: Conceptual model of N and P release from pruned leaves by canopy type in a coffee agroforest. Simulated remaining (A) nitrogen and (B) phosphorus in *Coffea-Erythrina* (January) and *Erythrina* (July) under single and mixed canopy. Shaded areas represent the initiation of rapid fruit growth 10-weeks following fruit set. For ten weeks following *Erythrina* pruning, shade trees are not fixing nitrogen.

In sum, N release is often optimized in single-species substrates while mixing species facilitates P release. Pruning regimes can clearly alter the timing of N and P availability. In the future it will be important to identify (1) the periods of greatest plant nutrient demand (e.g. is it more important that nutrients are available in large quantities for short periods of time or available in moderate quantities during the entire process of fruit maturation), and (2) which nutrient is most likely to limit plant productivity (as different pruning regimes optimize N than optimize P release).

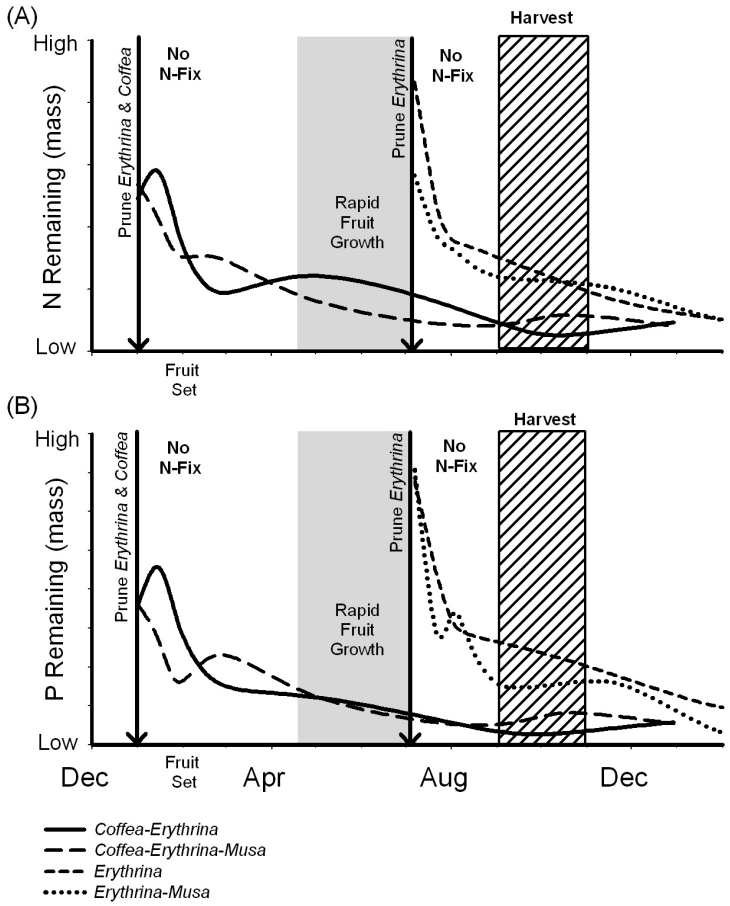


Figure 11: Conceptual model of N and P release from pruned leaves by leaf type in a coffee agroforest. Simulated remaining (A) nitrogen and (B) phosphorus in *Coffea-Erythrina* and *Coffea-Erythrina-Musa* mixtures following the January pruning, and in individual *Erythrina* leaves and in the *Erythrina-Musa* mixture following the July pruning. . Shaded areas represent the initiation of rapid fruit growth 10-weeks following fruit set. For ten weeks following *Erythrina* pruning, shade trees are not fixing nitrogen.

5.0 Conclusions

Patterns of nutrient release differ under the plant canopies typically found in Northeastern Costa Rica and among the different mixtures of leaves resulting from canopy diversity. Thus, farmers can accelerate or decelerate the release of N and P in anticipation of the coffee crop's nutrient-demand through simple modifications to their management practices. Indeed, periods of greatest nutrient-demand may coincide with periods of greatest light-demand, as pruning promotes coffee flowering and fruiting. Specifically, simultaneous pruning of *Erythrina* and *Coffea* in January can optimize P availability during the period of rapid fruit growth. However, in tropical regions it is common to observe the coincidence of flowers and fruits of different maturities in the same farm and even on the same plant. Thus, it is also important to consider the quantity of nutrients released from pruning residues from fruit set to harvest. Nutrient release from late-season (July) pruning residues will differ more widely: N availability is enhanced in *Erythrina* only leaf residues, but P availability is enhanced in *Erythrina-Musa* mixtures.

Nutrients do not seem to be accumulating in the surface soils over the course of one growing season. Therefore, if plants are pruned during periods of low nutrient-demand, excess nutrients may be subject to loss through runoff or leaching. Although *Coffea* agroforests utilize nutrients far more efficiently than unshaded monocultures, this study shows that canopy composition may control the timing of nutrient release. Attention should be paid to how canopy composition and pruning regimes may mediate patterns of nutrient *loss* as well. Finally, *Coffea* phenology is closely linked to seasonal

and environmental cues (DaMatta 2004), and global climate change may affect these cues. In the future farmers may need to adjust the timing of pruning, which will clearly have consequences for on-farm nutrient cycling.

References

Arcila-Pulgarin, J., L. Buhr, H. Bleiholder, H. Hack, U. Meier, and H. Wicke. 2002. Application of the extended BBCH scale for the description of the growth stages of coffee (Coffea spp.). Annals of Applied Biology 141:19-27.

Arcila-Pulgarín, J. 1990. Productividad potencial del cafeto en Colombia. 50 Años de Cenicafé 1938-1988. Conferencias Conmemorativas.

Beer, J. 1988. Litter production and nutrient cycling in coffee (*Caffea arabica*) or cacao (*Theobroma cacao*) plantations with shade trees. Agroforestry Systems 7:103-114.

Beer, J., A. Bonnemann, W. Chavez, H. W. Fassbender, A. C. Imbach, and I. Martel. 1990. Modelling agroforestry systems of cacao (*Theobroma cacao*) with laurel (*Cordia alliodora*) or poró (*Erythrina poeppigiana*) in Costa Rica. Agroforestry systems 12:229-249.

Bray, R. M., and L. T. Kurtz. 1945. Determination of total organic and available forms of phosphorus in soils. Soil Science 59:39-45.

Cannell, M. G. R. 1975. Crop physiological aspects of coffee bean yield: a review. Journal of Coffee Research 5:

Cannell, M. G. R. 1985. Physiology of the coffee crop. Coffee: botany, biochemistry and production of beans and beverage 109-134.

Castillo, Z. J. and A. R. López. Nota sobre el efecto de la intensidad de la luz en la floración del cafeto. Cenicafé 17:5160.

DaMatta, F. M. 2004. Ecophysiological constraints on the production of shaded and unshaded coffee: a review. Field Crop Research 86:99-114.

Das, T., and A. K. Das. 2010. Litter production and decomposition in the forested areas of traditional homegardens: a case study from Barak Valley, Assam, northeast India. Agroforestry systems 79:157-170.

Eviner, V. T., and F. S. III. Chapin. 2003. Functional matrix: a conceptual framework for predicting plant effects on ecosystem processes. Annual Review of Ecology and Systematics 34:455-485.

Gessel, S. P., D. W. Cole, D. W. Johnson, and J. Turner. 1980. The nutrient cycles of two Costa Rican forest. Progress in Ecology 3:23-44.

Gnankambary, Z., J. Bayala, A. Malmer, G. Nyberg, and V. Hien. 2008. Decomposition and nutrient release from mixed plant litters of contrasting quality in an agroforestry parkland in the south-Sudanese zone of West Africa. Nutrient Cycling in Agroecosystems 82:1-13.

Haggar, J. P., E. J. V. Tanner, J. W. Beer, and D. C. L. Kass. 1993. Nitrogen Dynamics of tropical agroforestry and annual cropping systems. Soil Biology and Biochemistry 25:1363-1378.

Hättenschwiler, S., A. V. Tiunov, and S. Scheu. 2005. Biodiversity and litter decomposition in terrestrial ecosystems. Annual Review of Ecology, Evolution, and Systematics 191-218.

Hobbie, S. E., P. B. Reich, J. Oleskyn, M. Ogdahl, R. Zytkowiak, C. Hale, and P. Karolewski. 2006. Tree species effects on decomposition and forest floor dynamics in a common garden. Ecology 87:2288-2297.

Imbach, A. C., H. W. Fassbender, R. Borel, J. Beer, and A. Bonnemann. 1989. Modelling agroforestry systems of cacao (*Theobroma cacao*) with laurel (*Cordia alliodora*) and poro (*Erythrina poeppigiana*) in Costa Rica IV. Water balances, nutrient inputs and leaching. Agroforestry Systems 8:267-287.

Jaramillo-Botero, C., R. H. S. Santos, M. P. Fardim, T. M. Pontes, and F. Sarmiento. 2008. Litter production and potential nutrient input of native tree species in an agroforestry system at Zona da Mata, MG, Brazil. Revista Árvore 32:869-877.

Kurzatkowski, D., C. Martius, H. Hofer, M. Garcia, B. Forster, L. Beck, and P. Vlek. 2004. Litter decomposition, microbial biomass and activity of soil organisms in three agroforestry systems in central Amazonia. Nutrient Cycling in Agroecosystems 69:257-267.

Lamers, J. P. A., C. Martius, A. Khamzina, M. Matkarimova, D. Djumaeva, and R. Eshchanov. 2010. Green foliage decomposition in tree plantations on degraded, irrigated croplands in Uzbekistan, Central Asia. Nutrient Cycling in Agroecosystems 1-12.

Mack, M. C., and C. M. D'Antonio. 2003. Exotic grasses alter controls over soil nitrogen dynamics in a Hawaiian woodland. Ecological Applications 13:154-166.

McGrath, D. A., N. B. Comerford, and M. L. Duryea. 2000. Litter dynamics and monthly fluctuations in soil phosphorus availability in an Amazonian agroforesty. Forest Ecology and Management 131:161-181.

Montoya, L. A., P. G. Sylvain, and R. Umaña. 1961. Effect of light intensity and nitrogen fertilization upon growth differentiation balance in Coffea arabica L. Coffee 3:97-104.

Nygren, P., and C. Ramirez. 1995. Production and turn-over of N_2 fixing nodules in relation to foliage development in periodically pruned Erythrina poeppigiana (Leguminosae) trees. Forest Ecology and Management 73:59-73.

Oliveira, L., D. V. Evtuguin, N. Cordeiro, A. J. D. Silvestre, A. M. S. Silva, and I. C. Torres. 2006. Structural Characterization of Lignin from Leaf Sheaths of "Dwarf Cavendish" Banana Plant. Journal of Agriculture and Food Chemistry 54:2598-2605.

Palm, C. A., and P. A. Sanchez. 1990. Decomposition and nutrient release patterns from the leaves of three tropical legumes. Biotropica 22:330-338.

Palm, C. A., and P. A. Sanchez. 1991. Nitrogen release from the leaves of some tropical legumes as affected by their lignin and polyphenolic contents. Soil Biology and Biochemistry 23:83-88.

Palm, C. A. 1995. Contribution of agroforestry trees to nutrient requirements of intercropped plants. Agroforestry Systems 30:105-124.

Salazar, A., L. T. Szott, and C. A. Palm. 1993. Crop-tree interactions in alley cropping systems on alluvial soils of the Upper Amazon Basin. Agroforestry systems 22:67-82.

Scherer-Lorenzen, M., L. Bonilla, and C. Potvin. 2007. Tree species richness affects litter production and decomposition rates in a tropical biodiversity experiment. Oikos 116:2108-2124.

Schroth, G., W. Zech, and G. Heimann. 1992. Mulch decomposition under agroforestry conditions in a sub-humid tropical savanna processes and influence of perennial plants. Plant and Soil 147:1-11.

Seastedt, T. R. 1984. The role of microarthropods in decomposition and mineralization processes. Annual Review of Entomology 29:25-46.

Selvaradjou, S-K., L. Montanarella, O. Spaargaren, and D. Dent. 2005. European Digital Archive of Soil Maps (EuDASM) - Soil Maps of Latin America and Caribbean Islands. Office of the Official Publications of the European Communities, Luxembourg.

Sun, R. C. 1998. Fractionation and characterization of polysaccharides from abaca fibre. Carbohydrate Polymers 37:351-359.

Szott, L. T., and D. C. L. Kass. 1993. Fertilizers in agroforestry systems. Agroforestry Systems 23:157-176.

Teklay, T. 2007. Decomposition and nutrient release from pruning residues of two indigenous agroforestry species during the wet and dry seasons. Nutrient Cycling in Agroecosystems 77:115-126.

Trojer, H. 1968. The phenological equator for coffee planting in Colombia. Agroclimatological Methods, Proceedings of the Reading Symposium, Paris (France).
Vanlauwe, B., N. Sanginga, and R. Merckx. 1997. Decomposition of four *Leucaena* and *Senna* prunings in alley cropping systems under sub-humid tropical conditions: the process and its modifiers. Soil Biology and Biochemistry 29:131-137.

Vitousek, P. M. 1984. Litterfall, nutrient cycling, and nutrient limitation in tropical forests. Ecology 65:285-298.

Wardle, D. A., K. I. Bonner, and K. S. Nicholson. 1997. Biodiversity and plant litter: experimental evidence which does not support the view that enhanced species richness improves ecosystem function. Oikos 79:247-258.

Wieder, R. K., and G. E. Lang. 1982. A critique of the analytical methods used in examining decomposition data obtained from litter bags. Ecology 63:1636-1642.

Wieder, R. K., and S. J. Wright. 1995. Tropical forest litter dynamics and dry season irrigation on Barro Colorado Island, Panama. Ecology 76:1971-1979.

Wood, T. E., D. Lawrence, D. A. Clark, and R. L. Chazdon. 2009. Rain forest nutrient cycling and productivity in response to large-scale litter manipulation. Ecology 90:109-121.

Yamashita, T., and H. Takeda. 1998. Decomposition and nutrient dynamics of leaf litter in litter bags of two mesh sizes set in two dipterocarp forest sites in peninsular Malaysia. Pedobiologia 42:11-21.

Youkhana, A., and T. Idol. 2009. Tree pruning mulch increases soil C and N in a shaded coffee agroecosystem in Hawaii. Soil Biology and Biochemistry 41:2527-2534.

Appendix Chapter 6

Appendix Table 1: Two-way ANOVA of soil Bray P, total N, and C in canopy tree experiment. Results of a two-way ANOVA for soil Bray P, total N and C with canopy and date as independent variables (d.f. = degrees of freedom).

Source	d.f.	F-value	p-value
Bray P (μg/g)			
Canopy	1	1.50	0.2417
Date	13	3.60	0.014
Total N (%)			
Canopy	1	91.73	<.0001
Date	13	1.27	0.3382
Total C (%)			
Canopy	1	80.85	<.0001
Date	13	1.10	0.4366
Soil Moisture (wt_{water}/dwt_{soil})			
Canopy	1	62.14	<.0001
Date	13	10.94	<.0001

Appendix Table 2: Three-way ANOVA of mass, N, and P remaining in canopy tree experiment. Results of a three-way ANOVA for leaf mass, N and P remaining (by weight) with canopy, species, and week as independent variables (d.f. = degrees of freedom).

Source	d.f.	F-value	p-value
Mass Remaining (g)			
Canopy	1	0.41	0.5231
Species	6	5.42	<.0001
Week	6	491.49	<.0001
Canopy*Species	6	1.55	0.1619
Canopy*Week	6	9.27	<.0001
Canopy*Week*Species	36	1.54	0.0289
Week*Species	36	4.34	<.0001
P Remaining (mg)			
Canopy	1	38.98	<.0001
Species	6	40.31	<.0001
Week	6	369.31	<.0001
Canopy*Species	6	6.91	<.0001
Canopy*Week	6	11.13	<.0001
Canopy*Week*Species	36	2.87	<.0001
Week*Species	36	6.87	<.0001
N Remaining (mg)			
Canopy	1	0.76	0.3845
Species	6	19.13	<.0001
Week	6	588.46	<.0001
Canopy*Species	6	3.31	0.0037
Canopy*Week	6	4.80	0.0001
Canopy*Week*Species	36	2.55	<.0001
Week*Species	36	5.94	<.0001

Appendix Table 3: Two-way ANOVA of mass, N, and P remaining in common garden experiment. Results of a two-way ANOVA for leaf mass, N and P remaining (by weight) with species and week as independent variables (d.f. = degrees of freedom).

Source	DF	F Ratio	Prob > F
Mass Rem			
Species	14	5.60	<.0001
Week	6	494.80	<.0001
Species*Week	84	3.06	<.0001
P Remaining (mg)			
Species	14	2.46	0.0027
Week	6	344.66	<.0001
Species*Week	84	4.32	<.0001
N Remaining (mg)			
Species	14	8.26	<.0001
Week	6	431.67	<.0001
Species*Week	84	3.80	<.0001