LINKING LAND USE – LAND COVER CHANGE AND ECOSYSTEM FUNCTION IN TROPICAL LOWLAND WATERSHEDS OF BELIZE, CENTRAL AMERICA

By

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This dissertation is dedicated to my wife, Ellie Harrison-Buck for all of her support during this research;

and

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By

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Chair: Mark Brenner
Major: Interdisciplinary Ecology

Ecosystems in tropical regions are expected to experience substantial impacts related to human land use and land cover change in the coming decades. Freshwater ecosystems will likely experience a disproportionate impact relative to other tropical ecosystems. My dissertation research takes an interdisciplinary approach to examine interactions between humans and their environment in lowland tropical watersheds of Belize, Central America.

I present results from a rapid assessment technique (Chapter 2) designed to estimate the intensity of impacts on freshwater ecosystems stemming from riparian zone land use practices. The technique provides an accessible, low-tech method for identifying river reaches experiencing high degrees of impact. Chapter 3 also examines riparian zone land use practices and focuses on the use of agrochemicals by small-scale Q'eqchi' Maya farmers. A high percentage of farmers in the Temash River watershed of Belize are cultivating corn within riparian zones and are using herbicides without adequate training. The continued use of herbicides in these riparian zone farms represents an emerging threat to both human and ecosystem health in the Temash River.

Chapters 4 and 5 look at slash-and-burn agriculture (locally referred to as *milpa*) in the Temash River watershed and its impact on soil nutrient dynamics (Chapter 4) and the loss of nutrients from terrestrial landscapes to streams (Chapter 5). In general, soil nutrients in the upper 10 cm are not significantly impacted by milpa. Plant-available nitrogen (as NO_3^--N) declines significantly in milpa fields and recovery to re-disturbance concentrations requires approximately 15 years, suggesting that 15 years is an ideal fallow period for milpa farmers in the Temash. In-stream nutrients in watersheds dominated by milpa are generally P-limited, as reflected in TN:TP concentrations. A seasonal flux of nutrients is exported from catchments at the onset of the rainy season but estimates for annual export of nutrients are similar to or less than other tropical watersheds.

CHAPTER 1
BACKGROUND

Human – Environment Interactions in Tropical Watersheds

The humid tropics are expected to experience significant declines in ecosystem services and functions due to land use and land cover change (LULCC) (Millennium Ecosystem Assessment 2005) and these impacts will likely not be distributed evenly across tropical biomes. Tropical rivers and their riparian zones serve as centers for rural livelihoods and high deforestation rates coupled with population growth are expected to have a disproportionately large impact on tropical freshwater ecosystems (Sala et al. 2000). In tropical regions, both rural and urban people alike depend on freshwater resources for drinking and utility water, irrigation, protein via fishing, transportation and recreation. During the next several decades, it is expected that nearly 90% of the Earth's population growth will occur in tropical countries (McClain 2002). As populations increase, greater dependence on freshwater resources will follow, along with the potential for greater impact on these resources.

Ecologists have long understood the importance of connectivity between streams and rivers and their catchments (Likens and Bormann 1974, Hynes 1975, Weins 2002) and understanding the influence of land use practices on aquatic ecosystem structure and function often depends on the spatial scale of the analysis. Regional conditions such as geomorphology, soils, topography, and land use – land cover change can influence nutrient supply, sediment delivery, hydrology, and channel characteristics. Local riparian conditions such as vegetation cover can exert control over in-stream habitat structure and organic matter input (Allan et al. 1997, Ankers et al. 2003). In some cases the most explanatory variable for variations in aquatic ecosystem structure

may simply be proximity to human infrastructure (Scheuerell and Schindler 2004). Legacy effects from past land uses in the catchment may also be an important factor in defining the observed distribution of organisms within a catchment (Harding et al. 1998). In addition, the annual hydrologic regime can have a significant effect and may overshadow the apparent control of physical/geologic characteristics in the catchment on dynamics such as nutrient supply and sediment delivery (Johnson et al. 1997) (Table 1-1).

Although extensive literature exists linking human land use practices and aquatic ecosystem structure and function in temperate watersheds, studies incorporating both terrestrial and aquatic ecosystems in tropical regions are exceedingly rare (Allan et al. 1997, McClain 2002). Previous studies of nutrient dynamics in tropical watersheds suggest that nitrogen (N) load in rivers is slightly higher than in temperate systems (Downing et al. 1999, Filoso et al. 2003). Phosphorus (P) content in tropical streams is positively correlated with soil nutrient concentrations (Biggs et al. 2002) and can influence decomposition and N mineralization rates in tropical ecosystems (Hobbie and Vitousek 2000).

Overall, land use impacts on the biogeochemistry of tropical streams are not well understood nor quantified (Biggs et al. 2004). In a large, tropical watershed (~12,400 km^2) with high human population densities, total N export was correlated with anthropogenic N inputs (Filoso et al. 2003). In watersheds converted to pasture, streams were N-limited and had lower NO_3^- concentrations than forested streams of similar size (Neill et al. 2001). In a small watershed (0.25 km^2) 80% deforested by slash-and-burn, high concentrations of TN and TP were observed in overland flow

(Williams and Melack 1997), yet no significant increases in NO_3^-, NH_4^+, or PO_4^{3-} were observed in streams when compared with a forested catchment (Williams and Melack 1997). Biggs et al. (2004) observed a stream nutrient response to deforestation in watersheds (ranging in size from 2 to 300 km^2) that were 66-75% cleared, suggesting that stream nutrient concentrations are resistant to land cover conversion below this threshold.

Belize, Central America provides a unique opportunity to test hypotheses related to impacts of LULCC on aquatic systems. A substantial portion (~35%) of the country is held in some sort of natural or cultural protected area (Meerman and Wilson 2005)), and the severity of ecosystem stress may be relatively low by regional standards. The rivers of Belize provide a linkage between terrestrial ecosystems, wetlands and estuaries of the coastal zone, and the longest barrier reef in the Western Hemisphere, the Mesoamerican Reef (MAR; or the Mesoamerican Barrier Reef System, MBRS). Increased awareness of this connectivity has added to the value of a watershed-scale perspective for conservation initiatives in Belize (Nunny et al. 2001, Bailey et al. 2007).

Aquatic Resources of Belize

A comprehensive water quality survey of Belize identified sixteen major watersheds and many small coastal tidal creek watersheds (Lee et al. 1995). These watersheds vary in size, traverse a variety of geologies and soil types, and also drain diverse terrestrial ecosystems and human-dominated landscapes (Lee et al. 1995, Esselman and Boles 2001). The Maya Mountain Massif (MMM) forms the headwaters for 12 of the 16 major watersheds (Table 1-2), from the Belize River in central Belize to the Moho River in southern Belize. Most of these rivers originate as high-gradient, relatively low-pH streams within granite and metamorphic rock catchments, traverse

limestone dominated landscapes, build up alluvial plains and wetlands, and ultimately discharge into the inner channel or shelf lagoon that separates the coast from the barrier reef. The Rio Hondo (forming the northern border of Belize) and New River, originate in karst hills, drain the low relief limestone landscape of northern Belize and discharge into Chetumal Bay, which in turn discharges into the inner channel. The headwaters of the Temash River begin in Guatemala and flow eastward across the southern Toledo district. The Sarstoon River (demarcating the southern border of Belize) originates in Guatemala, within the mountain range and foothills of the Sierra de Santa Cruz (Esselman and Boles 2001, Esselman et al. 2006).

In addition to lotic (flowing) waters, Belize has an abundance of lagoons and wetlands. Many of the freshwater wetlands occur in northern Belize and contain large complexes of lentic (still) water habitats including swamp forests, herbaceous marshes, and open water areas (Esselman and Boles 2001). Of the many freshwater wetlands in Belize, two have been internationally recognized as RAMSAR sites including Crooked Tree Lagoon and a *Sphagnum* bog within the boundaries of the Sarstoon-Temash National Park. A few karst, sink-hole lakes are found in the country, with Five Blues Lake in the Cayo District being the most notable. Although recognized as a significant component of the freshwater resources of Belize, groundwater resources remain almost completely unstudied (ERMA 2007).

Research on freshwater biodiversity and the ecology of aquatic organisms is limited in Belize. Significant works on aquatic macroinvertebrates have been conducted in the Belize River (Gonzalez 1980) and the Sibun River (Boles 1998). Both studies provided descriptive information on the community of aquatic macroinvertebrates in

these two watersheds and serve as important reference documents for future monitoring efforts. The ecology and distribution of tropical disease-carrying mosquitoes has also received significant research attention. Mosquitoes of the genus *Anopheles* (known malaria vectors) were observed to have a strong habitat preference within 1 km of rivers (Roberts et al. 1996), and larvae were significantly associated with certain riverine habitats (Manguin et al. 1996). Seasonal variation in the distribution of these mosquitoes also illustrates the role of increased rainfall in expanding the breeding habitat of these organisms (Roberts et al. 2002).

The freshwater fishes of Belize have been studied in greatest detail of any freshwater organism. *Freshwater Fishes of the Continental Waters of Belize* (Greenfield and Thomerson 1997) is the definitive work on freshwater fishes, and includes taxonomic keys and descriptions of each of Belize's 118 species. The importance of longitudinal connectivity and geology to native fish fauna has been documented in the Monkey River watershed (Esselman et al. 2006), whereas a country-wide survey of freshwater fishes was recently completed that (1) documents national patterns in freshwater fish diversity, (2) predicts the eventual range of invasive tilapias as they continue to expand their range in Belize, and (3) identifies a network of freshwater fish conservation areas that, if protected, would conserve the most fish biodiversity with the least amount of investment (Esselman 2009).

Statement of Objectives

Following this introduction to Belize and its freshwater resources, I present four studies that examine impacts of LULCC on aquatic systems at various scales ranging from the riparian zone to the entire catchment. Chapter 2 addresses LULCC impacts at the scale of the riparian zone and presents results from a study utilizing a rapid impact

assessment technique to estimate potential stresses on aquatic ecosystems stemming from riparian land uses. Chapter 3 takes a closer look at riparian zone land use and focuses on farming practices of the Q'eqchí Maya of southern Belize and their utilization of riparian zones for corn cultivation during the dry season. This chapter compares and contrasts traditional slash-and-burn agriculture with the slash-and-mulch agriculture that is practiced within riparian zones of the Temash River watershed and also discusses impacts related to an increased use of pesticides within these slash-and-mulch fields. Chapter 4 examines LULCC at the watershed scale in the Temash and focuses on soil nutrient dynamics across a chronosequence of active and abandoned agricultural fields, pasture, small-holder cacao plantations, and mature forest. Chapter 5 examines the spatial and temporal variability of in-stream nutrients in four small watersheds within the Temash River watershed. Land use within these catchments is primarily slash-and-burn and the discussion focuses on the relationship between nutrients, land-use practices, and abiotic variables such as soil characteristics and basement geology.

Table 1-1. Principal mechanisms by which land use influences stream ecosystems (modified from Allan 2004)

Environmental Stressor	Effects
Sedimentation	↑ turbidity, scouring, abrasion ↓ primary production, depth heterogeneity of stream
Nutrient enrichment	↑ autotrophic production, ↑ favorable conditions for filamentous algae, ↑ litter breakdown rates ↓ dissolved oxygen; species shift from sensitive to tolerant species
Contaminant pollution	↑ heavy metals and toxic organic substances resulting in deformities and mortality ↓ growth and reproduction rates and survival among fishes
Hydrologic alteration	Alters runoff-evapotranspiration balance ↑ vulnerability to erosion; ↑ potential for transport of nutrients, sediments and contaminants
Riparian clearing	↑ light penetration and water temperature ↓ bank stability, litter and woody debris inputs and retention of nutrients and sediments Alters quantity and quality of organic matter

Table 1-2. Sixteen major watersheds of Belize, drainage areas and underlying geologies. Data from Lee et al. 1995.

Major Watershed	Area (km^2)	Geology
Rio Hondo	15,075.5	Limestone rocks and variable soils; alkaline geochemistry
Belize River	9,434.2*	Mix of limestone, igneous and metamorphic rocks
Sarstoon River	2,217.5	Limestone and tertiary coastal sediments
New River	1864.0	Limestone rocks and variable soils; alkaline geochemistry
Monkey River*	1275.4*	Igneous rocks, metasediments, volcanic rocks, and limestone
Moho River	1,188.5*	Limestone
Sibun River	967.8*	Acidic igneous rocks, metasediments, and limestone
Rio Grande	718.5*	Limestone
Manatee River	484.0*	Metasediments and limestone
Temash River*	474.6	Limestone and tertiary coastal sediments
Sittee River	451.2*	Metasediments and tertiary coastal sediments
Deep River	347.9*	Limestone
North Stann Creek	281.4*	Ancient igneous rocks and metasediments; some limestone
South Stann Creek	258.0*	Ancient igneous rocks and metasediments
Golden Stream	204.1*	Limestone
Mullins River	156.9*	Metasediments and limestone

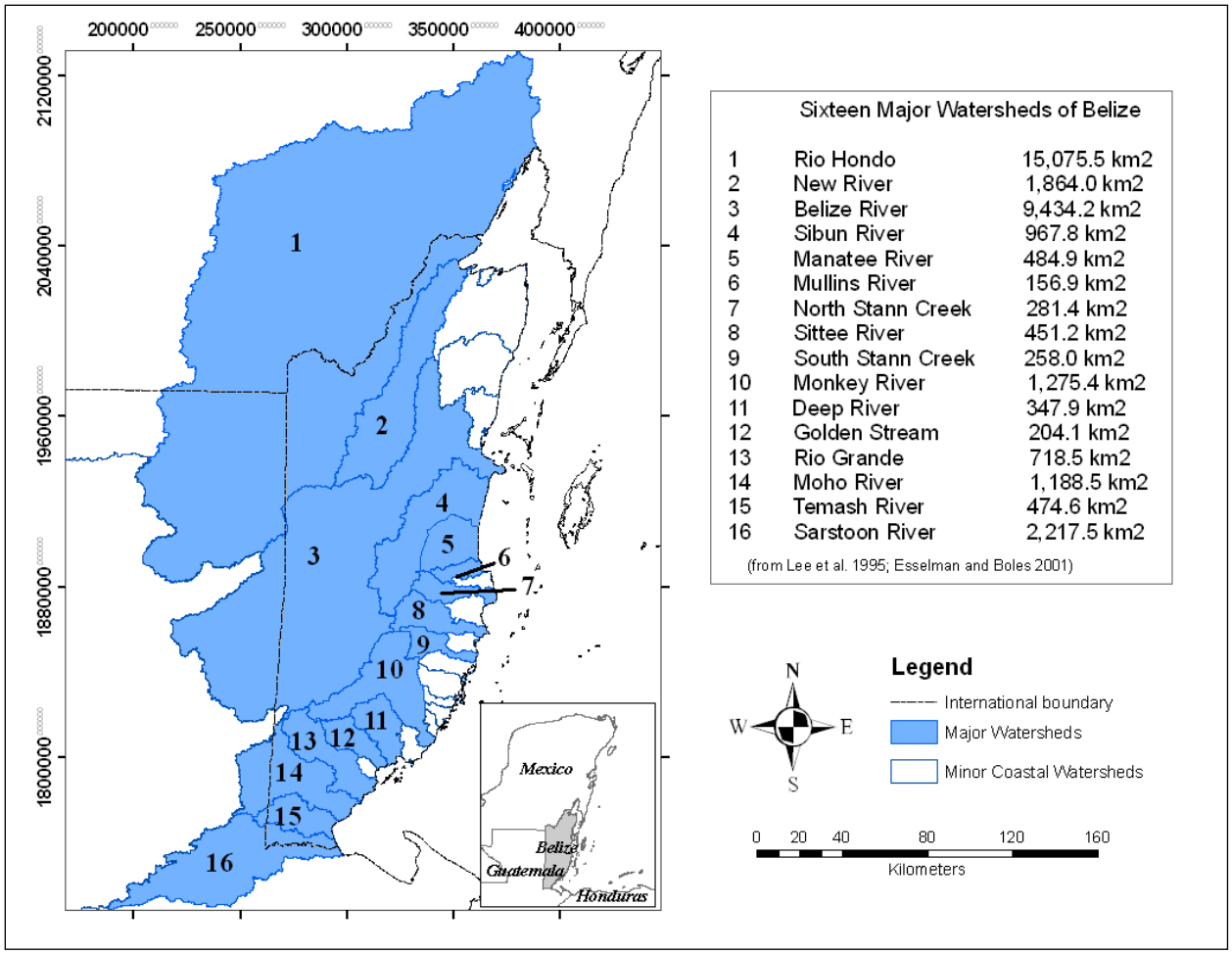

Figure 1-1. The sixteen major watersheds of Belize

CHAPTER 2
MONITORING LAND USE CHANGES ALONG RIPARIAN CORRIDORS IN LOWLAND TROPICAL WATERSHEDS: APPLICATION OF HUMAN IMPACT MAPPING AND ESTIMATION OF LOCAL STRESS INTENSITY (ELSI)

Introduction

Riparian forests are central to river conservation activities because of their unique ecosystem structure and function within the landscape. Riparian forests maintain bank stability and serve as buffers against excess sediment and nutrient transport across the terrestrial – aquatic interface. Vegetation in riparian zones regulates in-stream temperatures and provides organic matter that is essential for aquatic food webs. Riparian forests also exert control on the hydrologic balance in watersheds by influencing runoff, subsurface water storage, and evapotranspiration. In addition, riparian forests contain biological communities that are adapted to disturbance regimes associated with hydrologic events. They are also key components for maintaining biological connectivity across the landscape. Together with aesthetic and cultural values related to water quality, reduced flood damage, and recreation, riparian forests are important target habitats in a conservation portfolio (Naiman and Decamps 1997, Naiman et al. 2005, Sabo et al. 2005).

Conservation of riparian forests, their adjacent freshwater resources, and combined terrestrial and aquatic biodiversity, requires an integrated, whole-catchment approach to management that incorporates a science-based strategy for conservation and a plan for the resource needs of local stakeholders (Pringle et al. 1993, Saunders et al. 2002). This is particularly important in tropical regions where rivers and riparian zones have long served as sites for rural livelihoods. In the humid tropics, the deforestation rate is ~5.8 million ha/yr (Achard et al. 2002), and land use and land cover

change has a disproportionately large effect on tropical freshwater ecosystems and the goods and services they provide (Sala et al. 2000, Millennium Ecosystem Assessment 2005).

Impacts on riparian forests occur across multiple spatial and temporal scales. Patterns of land use and land cover change at the catchment scale influence nutrient supply, sediment delivery, hydrology and geomorphic characteristics. Local riparian conditions such as vegetation cover can exert control over in-stream habitat structure and organic matter input. In addition to spatial controls on freshwater ecosystems, the legacy of past land uses also influences freshwater ecosystems (Allan et al. 1997, Harding et al. 1998, Allan 2004).

The scale of analysis for determining the impact of land use and land cover change on riparian zones and their associated freshwater resources is largely dependent on the available technology and the time allotted for analysis (Johnson and Gage 1997). In developing countries, technological capability and the transfer of appropriate methodologies between developed and developing countries is key for the success of freshwater monitoring programs (Resh 2007). In Belize, Central America, aquatic scientists, in collaboration with local natural resource managers and conservationists, have developed a monitoring technique for human impact mapping that estimates the severity and irreversibility of impacts on freshwater systems stemming from riparian zone land use (Esselman 2001). The Expected Local Stress Intensity (ELSI) Index has been utilized in numerous freshwater settings in the country (Esselman and Boles 2001, Lee 2002, TIDE 2003, Karper and Boles 2004, Esselman and Buck 2007, Requena 2008). Human impact mapping and ELSI represent an

inexpensive, and “low-tech” method for natural resource managers to collect and communicate spatially-explicit data on riparian zone land uses and estimate the impacts of these land uses on freshwater resources.

Here I examine changes in riparian zone land use and the ELSI Index from the Monkey River watershed (MRW), Belize, between 2000 and 2007. This paper addresses three primary concerns for the conservation of freshwater resources in the MRW: (1) What are the primary land uses within the riparian zone of the MRW responsible for stresses in riparian and aquatic ecosystems? (2) Where, within the watershed, do these land uses occur?, and (3) How severe are the stresses on riparian and aquatic ecosystems? A clear understanding of the drivers of ecological stress within riparian zones, together with knowledge of where land use change is occurring in the MRW, can enable effective management strategies that address riparian zone and freshwater conservation at the river reach, sub-catchment, and catchment scale.

Study Site

The Monkey River is located on the southeastern flank of the Maya Mountains, in southern Belize (Figure 2-1). The Monkey River watershed consists of three main branches (Bladen, Trio and Swasey) that join in the coastal plain and enter the Caribbean Sea as a 6th-order river. The Monkey River is the fourth-largest watershed in Belize (1275 km^2). The headwaters of all three branches drain mountainous, primarily undisturbed tropical broadleaf forest. The middle reaches flow through human-dominated landscapes that include commercial banana cultivation, pasture, gravel mining, and subsistence agriculture. Human settlements are also concentrated within these middle reaches (Figure 2-1). The lower reaches, below the confluence of the Bladen and Swasey Branches, are largely undeveloped.

The Monkey River is the largest of the six watersheds (Rio Grande, Middle River, Golden Stream, Deep River, Payne's Creek, and Monkey River) that together form the Maya Mountain Marine Corridor (MMMC). The MMMC is a landscape-scale conservation initiative that connects headwater regions of the Maya Mountains to the coastal waters and coral reefs in the Gulf of Honduras. The MMMC is a matrix of national protected areas, extractive reserves, and human-dominated landscapes that includes an expansive wetland and estuary network and a coastal embayment with ~130 cayes and a diverse fishery.

The MMMC is a collaborative conservation initiative directed by the Toledo Institute for Development and the Environment (TIDE), the Ya'axché Conservation Trust, and The Nature Conservancy (TNC)-Belize Program. A Conservation Action Plan was recently developed for the MMMC and freshwater systems were identified as a primary conservation target for the corridor (Salas and Meerman 2007). Within this conservation target, riparian zone connectivity was identified as a key ecological attribute. In the MMMC, riparian zone connectivity includes longitudinal connectivity from headwaters to the sea as well as lateral connectivity from the river's edge across the riparian zone and associated flood plain (Buck et al. 2008).

Methods

Human impact mapping was conducted in the MRW in 2000, 2002, and 2007 (Esselman 2001, TIDE 2003, Esselman and Buck 2007). Detailed descriptions of methods used in human impact mapping and the ELSI Index are presented elsewhere (Esselman 2001). Briefly, the methods include 7 primary steps: (1) Observations of human impacts within the riparian zone are made along the main river channel, from a kayak or canoe. Each impact is marked using a global positioning system (GPS), and

each impact is classified as one of ten “sources of stress” (Table 2-1). (2) The sources of stress are then linked to 6 primary stresses (Table 2-1). (3) The contribution of each source of stress to the primary stresses are ranked using a combination of scientific literature review, best professional assessment, and pre-determined ranking tables developed by TNC (TNC 2000, Esselman 2001, Esselman and Buck 2007). (4) Using a Geographic Information System (GIS), the river is segmented into 1-km reaches in an upstream direction beginning from the river mouth. Segments are converted to polygons that include a 100-m buffer around the river channel. These segmented polygons become the basis of analysis for analyzing stresses across the riparian corridor. (5) Using the data collected with the GPS, the ranked stress scores (*step 3*) are summed for each river segment. (6) Spatially-explicit maps, detailing expected stress intensity, are developed for each primary stress. For ease of communication, categorical break points were established by ranking each stress score and dividing the rankings into quartiles. Each quartile then represents one of four levels of stress (Low, Medium, High, Very High). (7) Rankings for each stress are then summed to provide a cumulative stress score for each river reach, thus providing an overall expected stress intensity.

Human impact maps and ELSI scores for the MRW from 2000 and 2007 were analyzed to identify river reaches where changes in stress intensity had occurred. In the case of the Trio Branch of the Monkey River, impact data from 2002 (TIDE 2003) were utilized because Trio Branch was not included in the original 2000 human impact mapping. The stress and source rankings from 2000/02 and 2007 are presented in Table 2-2. All rankings were equal across both study years except with respect to

drainage ditches associated with industrial banana plantations within the MRW. As of 2002, the Belize Banana Growers Association has been working to improve drainage ditch design through financial assistance from the European Union's Special Framework of Assistance to Traditional Suppliers of Bananas (GOB 2002). Stress rank scores for 2000/02 were subtracted from stress rank scores from 2007. These data reveal the direction of land use change – a positive value suggests increased impact between 2000 and 2007; a negative value suggests reduced impact between 2000 and 2007. Observations of sources of stress were summarized for each branch of the MRW to further isolate areas of land use change and the drivers of change within each branch.

Results

Change in overall distribution of primary stress types within the MRW between 2000 and 2007 was minimal (Figure 2-2, Figure 2-3). The dominant stress in the MRW was sedimentation. Sources of stress contributing to flow alteration were minimal. The number of observed sources of stress within the riparian zone of the MRW increased from 138 observations in 2000 to 635 observations in 2007 (Figure 2-4). Riparian zones with no buffer or thin buffer (<10m) increased across all branches of the Monkey River. In addition, observations of cattle grazing within the riparian zone increased between 2000 and 2007 in each of the three main branches, with the greatest increase observed in the Trio Branch (0 observations in 2000; 44 observations in 2007) (Figure 2.4). Changes in the ELSI Index from 2000 to 2007 illustrate the spatial distribution of stress within the MRW and how it has changed (Figure 2-5)

Discussion

Human impact mapping within the MRW identified river segments that require conservation action to alleviate further stresses on the aquatic resources of the

watershed. The Trio Branch experienced the most rapid human-induced changes. The expansion of cattle within Trio has severely impacted the middle and lower reaches of this tributary (Esselman and Buck 2007). Cattle contribute to multiple stresses including sedimentation, nutrient enrichment, and habitat alteration. Many of the new pasture areas along the Trio Branch use barbed wire fences that are stretched across the main channel. During flood events, riparian trees cleared during pasture development create large snags where barbed wire crosses the channel, impeding the natural flow of the river (Esselman and Buck 2007).

Small-scale agricultural activity also impacts the riparian zone of the MRW. Slash-and-burn agriculture, primarily for corn cultivation, is a common practice in both the Bladen and Swasey Branches and contributes to the loss of riparian buffers (Esselman and Buck 2007). Human activities associated with slash-and-burn (e.g. irrigation, pesticide use) also contribute to the impact of this land-use practice on aquatic ecosystems in the MRW.

In-stream gravel mining, largely in the Swasey Branch, is expected to change the natural flow regime and alter habitats. In addition, commercial banana farming in the Swasey and Bladen Branches represents a continued stress on riparian zones through reduced buffer width. Although the re-engineering of drainage ditches has reduced the amount of run-off from banana plantations reaching the main channel, irrigation for banana production may affect the hydrologic regime of the MRW. It is estimated that between 2.5 and 3.5 X 10^6 gallons of water per day are pumped from the Monkey River (primarily Swasey and Bladen Branches) for banana irrigation (GUARD 2007). In addition, Monkey River Village (Figure 2-1), located at the mouth of the Monkey River,

is experiencing rapid erosion that threatens villagers' livelihoods and the future of this village (GUARD 2007). Although erosion is a natural process that occurs along the coastal margin, reduced transport and delivery of material from the middle reaches of the MRW to the river's mouth contributes to the impacts observed in Monkey River Village. The combined impacts of channel alteration resulting from gravel mining and water abstraction contribute to reducing the amount of material transported to the river's mouth (GUARD 2007).

As with any rapid assessment technique, ELSI has several limitations. First, impact mapping provides an imperfect assessment of conditions and relies on assumptions regarding terrestrial-aquatic interactions. These assumptions come from expert opinions and publications on temperate zone ecosystems, and require quantitative validation within tropical contexts. Second, ELSI provides an estimate of the intensity of stressors in each river reach, and does not consider propagated, cumulative downstream effects in the river network. This is evident in the ELSI percent distribution of stressors related to flow alteration and the resulting change detection map (Table 2-2, Figure 2-4). Although flow alterations are few relative to other observed stress types, the downstream effect of these impacts (e.g., in-stream gravel mining, damming and pump houses for irrigation) may cause the most severe impact on the integrity of the Monkey River (GUARD 2007). In addition, the calculation of ELSI scores assumes that stress ranks are additive and the interaction between stress types is linear. Finally, ELSI, because it draws on the location of visible activities in the riparian corridor, cannot detect unseen threats (e.g., agrochemical pulses), and has no way to identify legacy effects of past land use on freshwater systems. It is recommended that

ELSI serve as a first step that should be incorporated into a multi-scalar monitoring program that includes catchment-scale monitoring, and detection of non-visible threats such as toxics and heavy metals.

ELSI provides an accurate assessment of the location and severity of stresses. Assessing these threats to riparian and freshwater ecosystem structure and function, however, is not enough. Conservation-minded organizations must follow up on the information gathered to close the management gap between data collection and decision-making. In Belize, a legal framework exists for conservation of riparian zones. The Crown Lands Rule (Subsidiary Laws of Belize Vol. IV, 14) requires that 66 feet from the high water mark be left in un-cleared vegetation along all water frontage in rural lands (Boles et al. 2008). In addition, the National Lands Act (No. 6 of 1992) requires that a 66-foot-wide strip of land adjacent to streams, rivers or open water be left in its natural state unless the Ministry of Natural Resources and Environment gives approval for it to be used in a specific manner (Boles et al. 2008). Although these laws provide a means for riparian forest conservation, successful conservation of freshwater resources in Belize will require the combined efforts of the science, policy and education communities (Buck et al. 2008). Science—including rapid assessment approaches like ELSI, backed up by more focused assessment of specific problems—can be used to inform conservation strategies and define conservation priorities. Ultimately, however, political entities, educational institutions, NGOs and private industries need to develop integrated management structures that facilitate inter-party coordination on water-related issues.

Conclusions

Effective conservation of aquatic resources within the Maya Mountain Marine Corridor (MMMC) and the greater Port Honduras watershed requires an integrated management strategy that incorporates human activities at the terrestrial-aquatic interface. ELSI is a rapidly-deployed, low-cost approach to quickly gather information that can yield spatially- and temporally-detailed information about human activities that contribute to ecosystem stress. Resulting maps communicate results in a simple, compelling way and are very useful in outreach activities and as primary information sources for decision-making. When applied repeatedly through time, ELSI can detect trends in human uses of the riparian corridor. It is important that the impacts identified by ELSI be substantiated with follow-up studies to assess quantitatively the greatest environmental threats. Complementary approaches to ELSI (e.g., biotic indices of environmental quality) should also be used to identify invisible threats and legacy effects that are not detected by human impact mapping. Applications of ELSI in Belize demonstrate that the method can be applied across many contexts, suggesting it is an appropriate approach for use outside of Belize.

Table 2-1. Stress-source relationships and the scientific literature used to justify these relationships. When no scientific literature was available, personal observation was used to justify relationships (adapted from Esselman 2001)

Stress	Sources of Stress*	References
Sedimentation	No riparian buffer (NB)	Wood and Armitage 1997; Osborne and Kovacic 1993; Lowrance et al. 1997
	Drainage ditches (DD)	Usher and Pulver 1994
	In-stream gravel mining (GRV)	Brown et al. 1998, Sandecki 1989
	Channelization (CHN)	Brookes 1986
	Grazing (GRZ)	Metzeling et al. 1995; Owens et al. 1996
	Road access (RD)	Cline et al. 1982; Metzeling et al. 1995
	Thin riparian buffer (TB)	Wood and Armitage 1997
Nutrient Loading	Drainage ditches (DD)	Usher and Pulver 1994
	No riparian buffer (NB)	Osborne and Kovacic 1993; Peterjohn and Correll 1984; Lowrance et al. 1984
	Community use (CU)	Quddus 1980
	Grazing (GRZ)	Line et al. 2000
	Thin riparian buffer (TB)	Lowrance et al. 1997
Toxins/Contaminants	Drainage ditches (DD)	Usher and Pulver 1994
	No riparian buffer (NB)	Usher and Pulver 1994; Lowrance et al. 1997; Nearly et al. 1993
	Thin riparian buffer (TB)	Lowrance et al. 1997; Nearly et al. 1993
Altered Flow Regime	Drainage ditches (DD)	Poff et al. 1997
	Water pumping (PH)	Poff et al. 1997
	In-stream gravel mining (GRV)	Mas-Pla et al. 1999
	In-stream dam (DAM)	
Thermal Alteration	No riparian buffer (NB)	Osborne and Kovacic 1993; Gregory et al. 1991
	Drainage ditches (DD)	
	In-stream dam (DAM)	
Direct Habitat Alteration	No riparian buffer NB)	Gregory et al. 1991; Harmon et al. 1986
	In-stream gravel mining (GRV)	Brown et al. 1998; Sandecki 1989; Kondolf 1997
	Channelization (CHN)	Brookes 1986
	Water pumping (PH)	

* Codes for each source of stress are in parenthesis

Table 2-2 Stress-source rankings used for human impact mapping scoring criteria and the developing of the ELSI Index. The contribution (contrib) and estimated irreversibility (Irrevers) of each source of stress is combined to create the qualitative source rank of low (L), medium (M), high (H), and very high (VH). These are converted to numberical values on a 1-10 scale to create the rank score.

Stress	Sources / (Source Code)	Contrib.	Irrevers.	Source Rank	Rank Score
Sedimentation					
	No riparian buffer (NB)	VH	H	VH	10
	Thin riparian buffer (TB)	L	H	M	5
	Drainage Ditch (DD)††	H	H	H / M	7.5 / 5
	In-stream gravel mining (GRV)	H	H	H	7.5
	Channelization (CHN)	H	H	H	7.5
	Cattle grazing (GRZ)	VH	H	VH	10
	Road access (RD)	M	H	H	7.5
Nutrient Loading					
	Drainage ditch (DD)	VH	M	H	7.5
	No riparian buffer (NB)	H	H	H	7.5
	Community Use (CU)	M	M	M	5
	Cattle grazing (GRZ)	H	M	M	5
	Thin riparian buffer (TB)	L	M	L	2.5
Toxins / Contaminants	Drainage ditch (DD)	V	H	V	10
	No riparian buffer (NB)	H	H	H	7.5
	In-stream gravel mining (GRV)	L	H	M	5
	Thin riparian buffer (TB)	M	H	M	5
	Pump house (PH)	L	H	M	5
	Road access (RD)	L	H	M	5
Altered Flow Regime	Drainage ditch (DD)	L	M	L	2.5
	Pump house (PH)	M	M	M	5
	In-stream gravel mining (GRV)	L	M	L	2.5
Thermal Alteration	No riparian buffer (NB)	H	H	H	7.5
	Drainage ditch (DD)	L	M	M	5
	Thin riparian buffer (TB)	L	M	M	5
Direct Habitat Alteration	No riparian buffer (NB)	M	H	M	5
	In-stream gravel mining (GRV)	H	M	M	5
	Channelization (CHN)	H	H	H	7.5
	Pump house (PH)	L	M	L	2.5
	Cattle grazing (GRZ)	M	M	M	5
	Sandbag dam (DAM)	L	L	L	2.5
	Thin riparian buffer (TB)	L	M	L	2.5

†† The Drainage Ditch Source Rank between 2000/02 and 2007 decrease from high to medium because of changes to drainage ditch design implemented by the Belize Banana Growers Assoc.

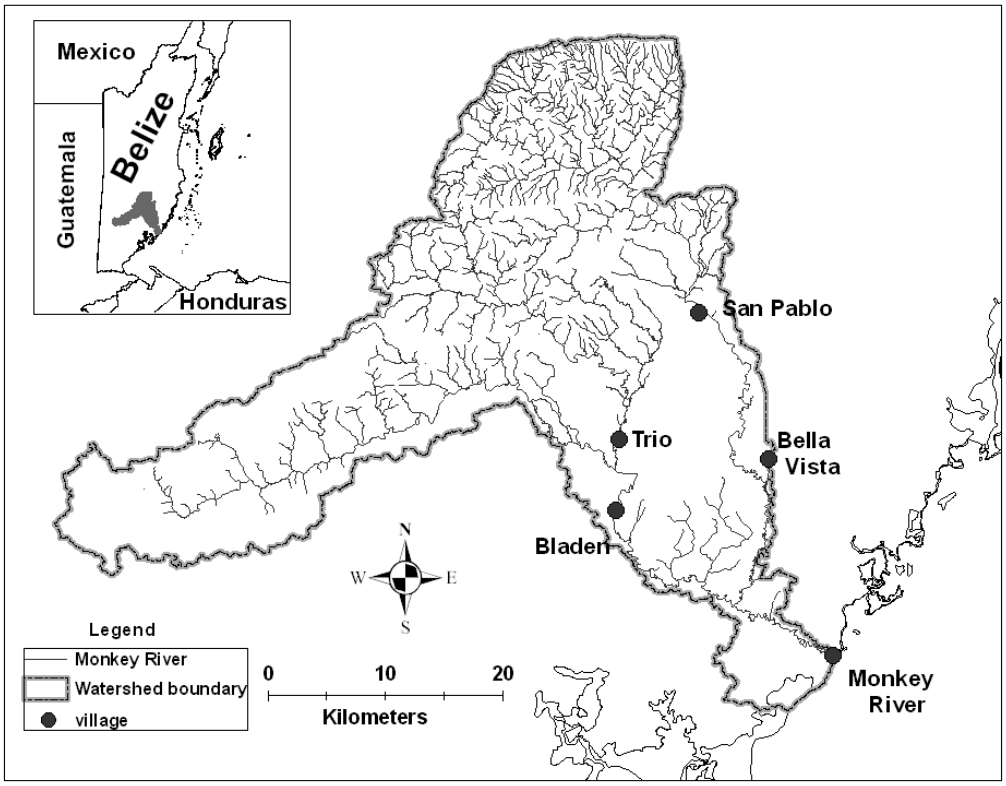

Figure 2-1. The Monkey River Watershed and the primary villages located within the catchment.

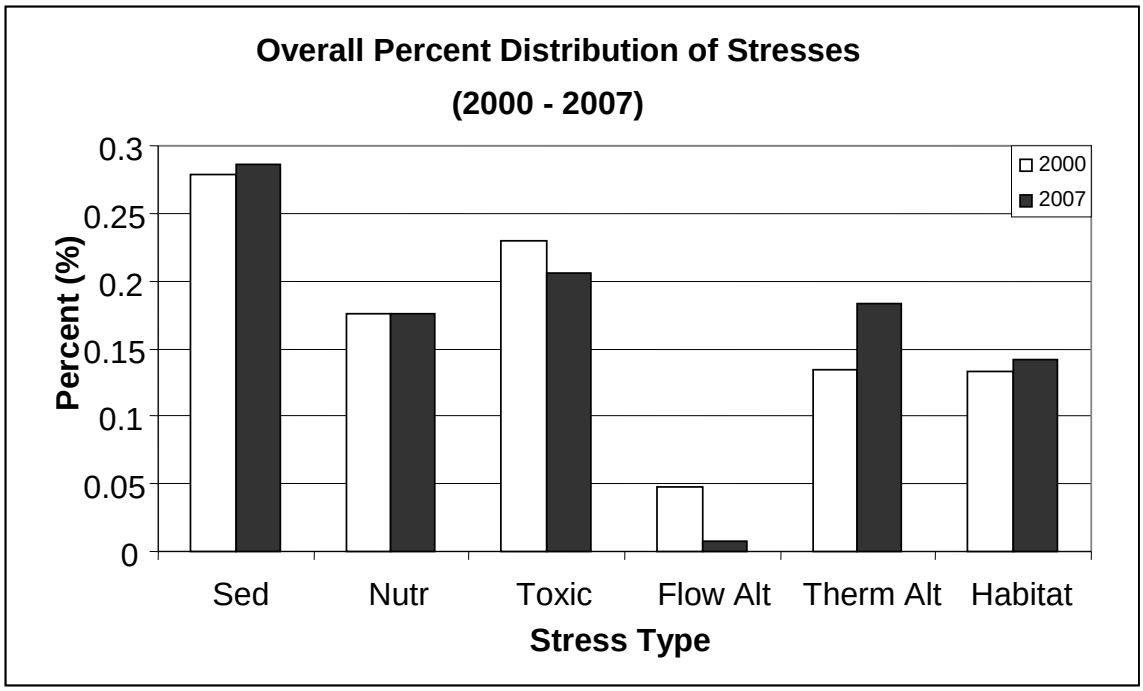

Figure 2-2. Percent distribution of stresses observed from human impact mapping in the MRW in 2000 and 2007.

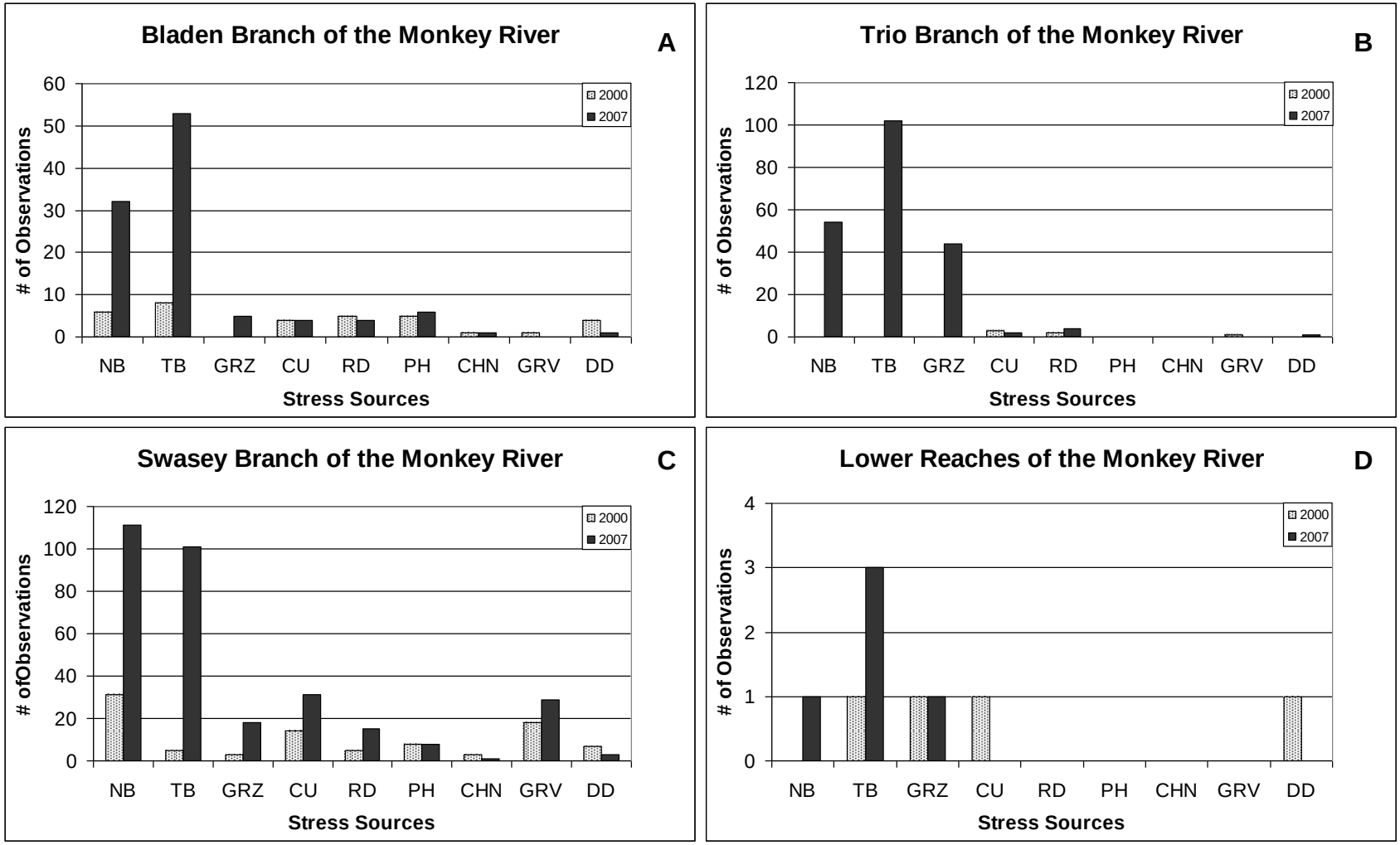

Figure 2-3. The observed sources of stress in each of the three branches of the Monkey River (Bladen Branch = A; Trio Branch = B; Swasey Branch = C) as well as the main stem of the Monkey River (D) below the confluence of the Bladen and Swasey branches. Very few observations were made along the Monkey River (D). (Note the differing 'y' axes). (NB=no bank; TB=thin bank; GRZ=grazing; CU=community use; RD=road; PH=pump house; CHN=channelization; GRV=gravel mining; DD=drainage ditch).

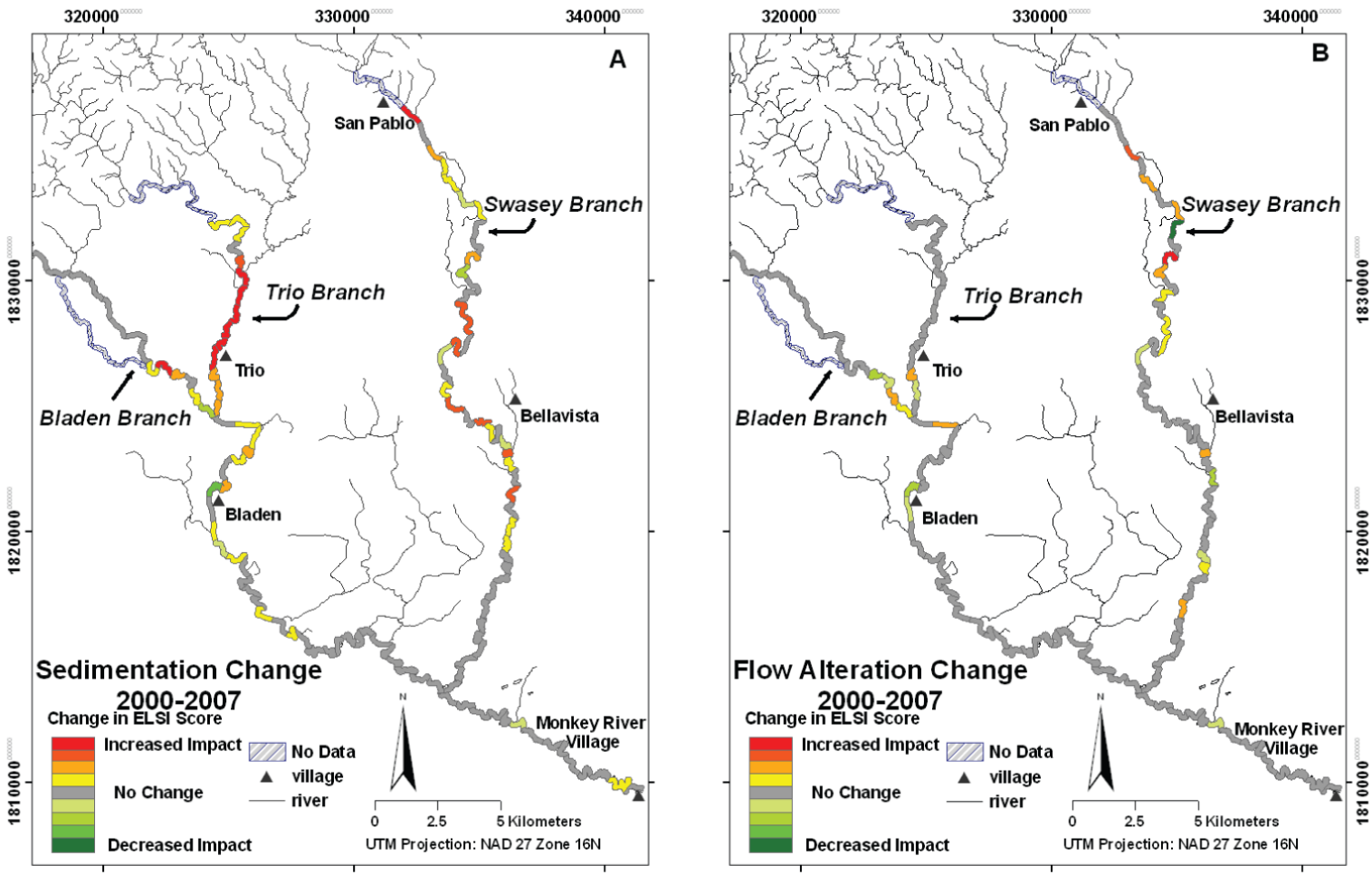

Figure 2-4. Change in impacts and estimations of stress intensity for sedimentation (A) and flow alteration (B) for the MRW from 2000 – 2007. Similar maps were created for all primary stresses. While ELSI does not accurately account for the upstream-downstream communication of impacts within the riparian zone corridor, sedimentation and flow alteration are changing dynamics related to material transport in the MRW and severely impacting downstream habitats and human livelihoods (e.g., Monkey River Village).

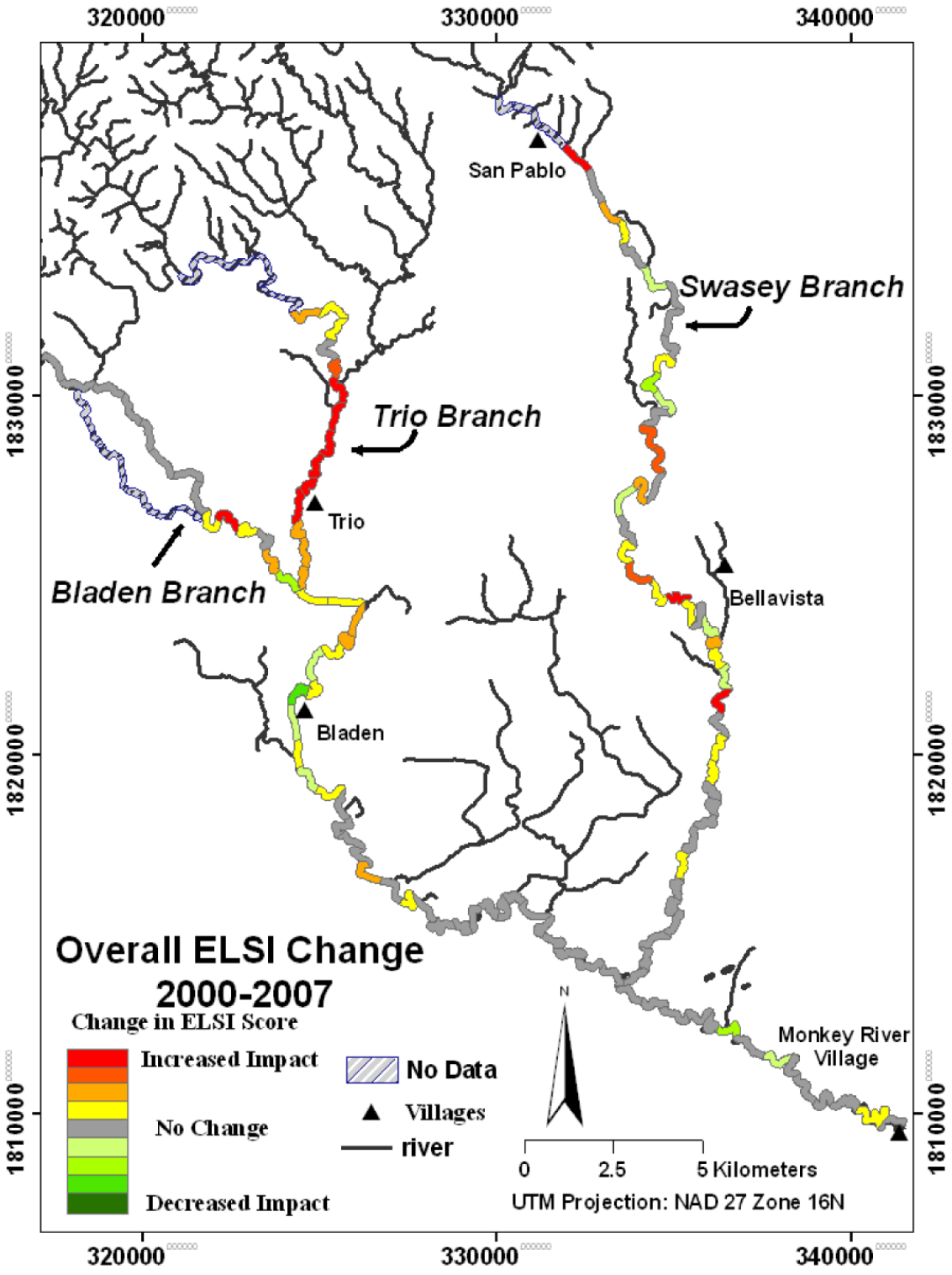

Figure 2-5. Change in the Overall ELSI Index for the MRW from 2000-2007.

CHAPTER 3
LAND USE PRACTICES, PESTICIDE APPLICATIONS, AND FRESHWATER RESOURCE CONSERVATION IN A LOWLAND TROPICAL WATERSHED

Introduction

Pesticide use in Central America is driven by large-scale agricultural production, particularly for export crops sugar cane, coffee, bananas, and other fruits. This agricultural sector accounts for ~85% of pesticides used in the region (Galvao et al. 2004). Importation of pesticides in Central America nearly tripled between 1992 and 2001, totaling more than 46,000 tons in 2001 (Galvao et al. 2004). In Belize, pesticide importation increased 36% between 1992 and 2000, from 952 to 1,296 tons. Lower pesticide importation in 2001 (831 tons: (Galvao et al. 2004) was followed by several years of high importation. Pesticide imports in 2002, excluding chemicals associated with wood preservatives, totaled 1,311 tons. Import totals to Belize for 2003, 2004 and 2005 were 1,339, 1,235, and 1,290 tons, respectively (BPCB).

Similar to the economies of other Central American countries, Belize's economy is dominated by large-scale agricultural production. Agricultural exports in 2005 and 2006 totaled 85% and 74% of all exports, respectively, with sugar cane, papaya, citrus and banana accounting for approximately half of that production (BMAF 2006). In addition to this large-scale agricultural production, small-scale agriculturalists provide the domestic Belize market with a substantial amount of food products. Belize has been self-sufficient in rice production for approximately 10 years (BMAF 2006). Historically, Belize's rice production has been undertaken by small-scale farmers. For many years, rural Q'eqchí and Mopan Maya farmers in southern Belize produced between one third and one half of the rice consumed in the country (Wilk 1997). Today they produce approximately 15% (BMAF 2006). The majority of the country's cassava (*Manihot*

esculenta) production occurs in southern Belize. In addition, cacao (*Theobroma cacao*) production by Q'eqchí and Mopan Maya farmers has expanded rapidly in the region and is supplying a growing international chocolate market (Levasseur and Olivier 2000, Emch 2003).

The trend towards market integration among rural farmers has been observed across Central America (Humphries 1993, Wilk 1997, Emch 2003, Keys 2004, Hamilton and Fischer 2005, Keys and Chowdhury 2006). Integration with larger markets and associated strategies for intensification, have been accompanied by increased introduction of pesticides into rural communities. Pesticide introduction has been associated with an increase in public and environmental health concerns (Humphries 1993, Whitaker 1993, Popper et al. 1996, Azaroff and Neas 1999, Shriar 2001, Blanco et al. 2005). Education efforts in rural communities can increase awareness about safe handling of pesticides (Popper et al. 1996), but low literacy and education levels, inadequate protective clothing, and poorly functioning equipment preclude the safe use of pesticides, thereby impacting rural farmers and their households negatively (Popper et al. 1996, Azaroff and Neas 1999, Blanco et al. 2005).

Although public health concerns regarding pesticide use by rural, small-scale farmers in Central America have received some attention in the literature (Azaroff and Neas 1999, Blanco et al. 2005), little attention has been given to small-scale pesticide use and its impacts on the natural resource base on which these rural communities rely. Rural communities throughout tropical regions tend to aggregate near freshwater resources that are used for drinking, washing, livestock, irrigation, and utilities, as well as sources of dietary protein (e.g., snails, fish, turtles) (Wilk 1985, Sala et al. 2000,

McClain and Cossio 2003, McClain 2008). Pesticide use, particularly proximate to freshwater resources, and the cleaning of equipment such as backpack sprayers and pesticide containers in rivers and streams represents a threat to freshwater resources (Castillo et al. 1997).

This study presents survey data on pesticide use among rural, Q'eqchí Maya farmers of southern Belize. The survey focuses on riparian zone land use and the use of pesticides during corn cultivation in riparian zones. I identify the most commonly used pesticides and discuss potential threats to freshwater resource conservation in a lowland tropical watershed.

Study Area

Study Villages and Watershed

Fieldwork was conducted in three Q'eqchí Maya villages within the Temash River watershed (Figure 3-1). Crique Sarco, on the banks of the Temash River, was founded about 1912 (Grandia 2004) and is one of the oldest villages in the watershed. Lucky Strike lies near Tomagas Creek, a tributary of the Temash River. Lucky Strike, the youngest village in the watershed, was settled in 2003 by an extended family that came from nearby Crique Sarco. Sunday Wood is approximately 25 years old. It was founded by farmers from the nearby village of San Lucas (Grandia 2004). The village is near Sunday Wood Creek, where several small, 1st-order tributary creeks drain the surrounding landscape.

The headwaters of the bi-national Temash River begin in Guatemala and flow eastward across the southern Toledo District of Belize towards the Caribbean Sea and Gulf of Honduras (Figure 3-1). Approximately 400 km^2 of the watershed's 475 km^2 are in Belize. Soils are derived largely from Cretaceous-age limestone with patches of

dolomite and siliceous limestone (Wright et al. 1959). Mean annual rainfall is ~3800 mm with a dry season from December to April and a wet season from May to November. Mean monthly temperatures range from 18.7°C (January) to 32.1°C (May). Vegetation cover in the Temash River watershed is dominated by tropical evergreen lowland forests (Meerman et al. 2003b).

The Sarstoon Temash National Park (STNP) is situated within the lower reaches of the Temash River (Figure 3-1) and the study villages are all located within the buffer zone of the national park. The STNP is a 40,000-acre community-co-managed national park. The Sarstoon-Temash Institute for Indigenous Management (SATIIM), a local non-governmental organization, currently co-manages the STNP in collaboration with the Belize Forest Department. SATIIM works with five communities adjacent to the park (including two of the three study villages, Crique Sarco and Sunday Wood). SATIIM's management plan for the STNP (Herrera 2004) was officially approved by the government of Belize in June 2005. Many of the ecosystems within the STNP depend on seasonal hydrology that leaves large areas of the park inundated for much of the year. The conservation and management of riparian zones within the Sarstoon and Temash Rivers is a central component of the recently ratified management plan (Herrera 2004).

Riparian Zone Land Use within the Temash River Watershed

Livelihood strategies among the Q'eqchí Maya of southern Belize revolve around the cultivation of corn. Corn cultivation commonly occurs twice annually including a slash-and-burn crop (*milpa*) that is grown during the rainy season (June – November) and a slash-and-mulch crop (*matambre*) that is grown during the dry season (January – May). Riparian zones and other low-lying areas are preferred locations for planting

matambre (*saq'ecuaj* in Q'eqchí). Matambre fields within riparian zones capitalize on ecosystem services related to soil moisture retention and soil nutrient replenishment from occasional deposition of nutrient-rich sediments during flooding. In addition, green mulch minimizes competition from weeds and protects the soil from compaction (Wilk 1997). These characteristics allow Q'eqchí farmers to reuse the same riparian field repeatedly, and when fields are left fallow, matambre fallow periods are much shorter than wet season milpa fallow periods (Wilk 1985, 1997). Previous research suggests that matambre is considered a less productive corn cropping method by Q'eqchí farmers (Wilk 1997), but is recognized as an integral component of the agricultural cycle. Contemporary matambre agriculture reflects long-term cultural practices of the Q'eqchí that minimize risks associated with their agricultural livelihoods (Wilk 1985).

Methods

I conducted a series of semi-structured and structured interviews in three Q'eqchí Maya villages in the Temash River watershed. Semi-structured interviews with seven village elders were conducted to better understand the history of matambre, general agricultural practices, and governance of natural resources, with particular attention to riparian zones and associated freshwater ecosystems. Structured interviews were conducted with heads of households in the three villages. Ninety interviews were conducted out of a possible 103 households (87%). Structured interviews were conducted between April and June 2008 and focused on practices associated with matambre, perceptions about water quality, use of pesticides in matambre fields, and the potential for conservation in the Temash River watershed. Responses from structured interviews were summarized using SPSS software, release 10.0.0 (SPSS, Inc.).

Results

Households and Matambre

In Crique Sarco village, 43 of 52 households (83%) were interviewed. Nine households either refused to participate or the head of the household was working outside the village at the time the survey was conducted (Table 3-1). In Lucky Strike village, eight of nine households (89%) were surveyed with one head of household working outside the village at the time the survey was conducted (Table 3-1). In Sunday Wood village, 39 of 42 households (93%) were interviewed, with three households refusing to participate (Table 3-1).

In Crique Sarco, 29 of the 43 interviewed households reported planting matambre in the dry season of 2008. Sixty-two percent of those households selected fields located within riparian zones, while 38% used areas classified as upland. Mean matambre field size in Crique Sarco was 1.03 hectares (ha). When asked to compare matambre and milpa field productivity, approximately half of the farmers in Crique Sarco (48%) believed that matambre provided them with more corn (estimated as bags of corn per hectare) than the wet season milpa (Table 3-2).

All households (100%) interviewed in Lucky Strike reported planting matambre fields in 2008, with 14% of those fields located within riparian zones. The reported mean field size of matambre in Lucky Strike was 1.45 ha. Matambre provided a higher yield of corn than milpa for all households surveyed in Lucky Strike (Table 3-2).

In Sunday Wood, 32 of the 39 households surveyed (82%) reported planting matambre in 2008. Sixty-two percent of these households reported planting their matambre in riparian zones, with an additional 19% planting in low-lying areas away from a creek or river. Mean matambre field size in Sunday Wood was 0.96 ha. Seventy

percent of the respondents in Sunday Wood stated that matambre provided more corn for their household than the wet season milpa (Table 3-2).

Water Use and Water Quality

In Crique Sarco, four households (<10%) reported collecting drinking water from either a creek or spring, whereas all respondents in Lucky Strike and Sunday Wood said they used either a creek or spring as their primary source for drinking water. All households in Crique Sarco reported utilizing the public water system that was installed in 2006, with 32% of the households augmenting with rainwater collection. None of the households in Lucky Strike reported utilizing rainwater and only 10% of households (n=4) reported collecting rainwater in Sunday Wood.

When asked about the water quality in the Temash River, respondents in Crique Sarco gave mixed responses, with 65% saying it was good. Twenty-five percent of households in Lucky Strike and 19% in Sunday Wood thought the Temash River water was clean enough to drink. When asked about the water quality in tributary creeks draining lands near their villages, responses differed. Fifty-six percent in Crique Sarco and 63% in Lucky Strike felt that water in the tributary creeks was clean, whereas only 8% of the households in Sunday Wood felt so. In Crique Sarco, respondents considered pasture to have the largest impact on water quality, followed by domestic use (e.g., washing, bathing), and waste deposited directly into the river. Households in Lucky Strike also considered pastures to be the primary stressor on water quality. Sunday Wood farmers reported that animals, particularly pigs, and domestic uses impacted water quality in Sunday Wood Creek and its tributaries.

Matambre Field Management

Matambre farmers who use riparian zones reported leaving a forested buffer strip between their fields and adjacent freshwater. In Crique Sarco, 69% of farmers reported leaving a buffer, while 75% of farmers in Lucky Strike and 89% of farmers in Sunday Wood reported leaving a buffer strip. The mean width of buffer strips reported by matambre farmers across all villages is 2.4 m. All matambre farmers reported weeding their fields at least once following planting. Eighty-four percent of these households used herbicides, typically applied four weeks after planting (Table 3-3). The most common herbicide used was Gramoxone®, followed by 2,4-D and Round-Up® (Table 3-4). Other agrochemicals used included Folidol and Tamaron. Of the farmers who reported using agrochemicals, 37% said they received training from the Belize Pesticide Control Board (BPCB).

Discussion

Matambre and Riparian Conservation

Riparian forests and their associated floodplains provide an important buffer for freshwater ecosystems against human impacts stemming from land use practices and land cover change (Naiman and Decamps 1997, Allan 2004). In Belize, a legal framework exists for the conservation of riparian zones. The *National Lands Act of Belize* (No. 6 of 1992) requires that a 66-foot-wide strip of land adjacent to streams, rivers or open water be left in its natural state (Boles et al. 2008). Field observations and interviews with village elders confirm that matambre fields are generally cleared to the edge of rivers or streams. Results from the structured household interviews contradict this claim. In part due to community outreach efforts by SATIIM, matambre farmers in the Temash are aware of the conservation community's perceptions of

matambre and its impact on riparian zones. The apparent disconnect between interview responses and matambre practices may reflect a “deference effect,” in which the respondent wishes to tell the interviewer what he/she perceives to be the desired response (Bernard 2006). SATTIM has engaged the local communities in a discussion about matambre through its outreach and education efforts, including a detailed mapping exercise of all matambre fields located within the STNP boundary. Likewise, the STNP management plan states that matambre within riparian zones is a direct threat to the management of the national park and its biodiversity (Herrera 2004). Despite these efforts, this conservation issue remains at odds with the land use choices made by the Q’eqchí farmers in the Temash.

Matambre, Pesticides and Public Health

Of greater concern, from both an environmental and human health perspective, is the use of agrochemicals in matambre fields. Prior research on matambre was limited to studies conducted 20 years ago among Q’eqchí farmers in the neighboring Moho River watershed (Wilk 1985, 1997). Although it was noted that Q’eqchí rice farmers commonly used 2,4-D for weed control, Wilk (1997) made no mention of its use within matambre fields. Similarly, Grandia (2004) documented the preference of Q’eqchí milpa farmers in the Temash to use machetes for weeding instead of agrochemicals because of both potential health risks and possible loss of wild foods that grow within agricultural fields. Results from farmer interviews in this study point to increasing use of agrochemicals among Q’eqchí farmers in the Temash watershed (Table 3-3, Table 3-4).

Public health concerns related to the improper use and disposal of agrochemicals in developing countries has been well documented (McConnell and Hruska 1993, Murray 1994, van Wendel de Joode et al. 1996, Escobichon 2001, Wesseling et al.

2001a, Wesseling et al. 2001b, Wesseling et al. 2001c, Murray et al. 2002, Wesseling et al. 2005). Official surveys of regional hospitals, rural health posts and doctors from across Central America, conducted in coordination with the PanAmerican Health Organization (PAHO) between 1992 and 2001, documented 43,368 acute intoxications, with an estimated 4,323 deaths resulting from exposure to pesticides (Galvao et al. 2004). No information on pesticide-related illnesses or deaths are available for Belize and these values for Central America likely represent a minimum, as the majority of pesticide-related illnesses go unreported in the region. Estimates as high as 400,000 poisonings per year have been suggested for Central America (Murray et al. 2002).

The agrochemical used most by matambre farmers in the Temash River watershed is paraquat (sold as Gramoxone®). Paraquat (1,1'dimethyl-4,4'-bipyridinium dichloride) is one of the highest selling pesticides in the world (Wesseling et al. 2001c). Paraquat, sold under many of its trade names, is listed as a restricted use pesticide by Belize's Pesticide Control Board (BPCB 2008). In Central America, farmers and plantation workers are most often exposed to paraquat via contact with the skin as a result of poorly functioning application equipment (e.g., backpack sprayers) or chemical spills. These occupational exposures cause chemical burns and lesions, most often on the back, wrists, and/or groin area (van Wendel de Joode et al. 1996, Penagos 2002). Paraquat is one of the primary pesticides associated with acute poisonings worldwide (Wesseling et al. 2005), and is the most common pesticide associated with fatal and suicidal poisonings in Central America (Wesseling et al. 2001a).

The second most commonly used pesticide in the Temash River watershed is the herbicide 2,4-D (2,4-Dichlorophenoxyacetic acid). Extensive research has been

conducted in the developed world on 2,4-D and potential links to non-Hodgkins lymphoma (Hardell 1979, Zahm et al. 1990, Cantor et al. 1992). Although mechanisms of causality have been debated in the literature (Garabrant and Philbert 2002), studies have repeatedly identified farmers and agricultural workers as an at-risk population for non-Hodgkins lymphoma. Pesticide use, particularly use of 2,4-D, has been identified as a primary cause (Dreiher and Kordysh 2006). Exposure to 2,4-D has also been linked with increased risk of gastric cancer (Ekström et al. 1999, Mills and Yang 2007) although this has been refuted by the pesticide industry (Burns et al. 2007).

Results from the household survey show that only 37% of matambre farmers using pesticides have received training in safe handling of pesticides from the BPCB. In order to purchase pesticides, farmers are required to present proof of training by the BPCB. The Belize Pesticide Control Board struggles to cover rural areas of Belize, particularly the southern part of the country, where only two staff members are currently charged with pesticide training and monitoring. Farmers without official training likely obtain pesticides either from other farmers within their village that have the necessary proof of training or through the unregulated sale and trade of items with traveling sales people from neighboring Guatemala.

Environmental Fate of Agrochemicals in the Temash

Ecotoxicology studies and the assessment of environmental impacts of pesticides have historically been conducted in temperate regions. There are insufficient data on almost all pesticides - DDT being a notable exception (Kammerbauer and Moncada 1998) - at higher temperature and higher humidity in tropical environments (Castillo et al. 1997, Kammerbauer and Moncada 1998, Marques et al. 2007). The research and regulatory capacity in most Central American countries to monitor pesticides in the

environment is generally inadequate, although Costa Rica and Nicaragua have extensive experience that could be applied to other areas in the region (Wesseling et al. 2001a). Studies in North America on the fate and transport of paraquat and 2,4-D suggest that they pose a threat to the environmental health of the watershed.

Paraquat has a long half-life in soils, ranging from 16 months to 13 years (Rao and Davidson 1980). Paraquat and its breakdown products can adsorb onto soil particles and be easily transported to freshwater environments during runoff. Exposure to paraquat can directly alter the structure and function of aquatic and semi-aquatic environments. Paraquat can cause fatalities in the zooplankton and benthic communities of lakes (Gagneten and Marchese 2003) and streams (Burnet 1972), resulting in prolonged alteration to community structure. It is considered slightly to moderately toxic to many species of fish and damages lung, liver and kidney tissue (Eisler 1990, Parma de Croux et al. 1999).

Although 2,4-D decays rapidly in the environment (half-life of 7 – 10 days), it is on the U.S. Environmental Protection Agency's (US EPA) list of organic chemicals that are water contaminants (USEPA 2008) and has been documented in groundwater in several states in the US and Canada (Howard 1989, Fitzgerald et al. 2001). Although its toxic effect on many trophic levels in aquatic environments is considered slight or negligible (Table 3-4), it has been documented to impact feeding behavior and can cause mortality in at least one fish species (NRCC 1978). Post-mortality studies following a massive die-off of corals in the Gulf of Chiriqui, Panama, documented high 2,4-D residues on coral heads (Glynn et al. 1984, Castillo et al. 1997). The source of pesticide was upstream rice farms (Castillo et al. 1997). This should be of particular

concern for Belize, where productive mangroves and sea grass beds along the coastal shelf give way to the longest barrier reef in the western hemisphere, the Mesoamerican Barrier Reef System (MBRS). The MBRS and the associated inner channel serve as rich nursery, breeding and feeding grounds, supporting a diverse and productive fishery. Current assessments of the coral reefs along the Belizean coast suggest the system is heavily impacted by tourism, coastal development and agricultural practices in the watersheds that drain into the MBRS . Organochlorines and heavy metals have been documented in top predators across multiple aquatic habitats in Belize (Wu et al. 2000, Rainwater et al. 2002, Rainwater et al. 2007, Evers 2008) suggesting that agriculture practices may be responsible for contaminating freshwater, coastal and marine habitats. Bioaccumulation of these and other toxins should be investigated in Belize and the greater MBRS region.

Conclusions

Riparian buffer strips can greatly reduce the amount of pesticides being transported across the terrestrial-aquatic boundary (Asmussen et al. 1977, Lowrance et al. 1997a). Movement of herbicides in surface runoff typically occurs during a short period of time after application and the condition of the riparian zone at the time of application is an important factor in limiting pesticide runoff (Lowrance et al. 1997b). The infiltration of pesticides is strongly influenced by soil properties, particularly soil organic matter and pH (Reddy and Gambrell 1987), and the presence of pesticides and their degradation products in shallow groundwater depends on these surface soil properties and the time lag required for infiltration (Lowrance et al. 1997a, Puckett and Hughes 2005). In riparian zones of southern Belize where matambre farmers are applying herbicides annually, there is great potential for combined runoff and shallow

groundwater infiltration. Research is needed on the movement and transport of pesticides in the lowland tropical watersheds that are home to these rural farming communities.

Conservation of riparian zones within the Temash River watershed requires a focus on their combined social and ecological function. Riparian zones have a central role in the livelihood strategies and the cultural tradition of matambre for Q'eqchí farmers in the Temash watershed and the rest of southern Belize (Wilk 1985, 1997). In addition to their ability to reduce the transport of contaminants across the terrestrial-aquatic boundary, riparian zones provide numerous other ecosystem functions related to biodiversity conservation (Allan 2004, Naiman et al. 2005).

Future directions for conservation of riparian zones should focus on: 1) education and outreach related to the importance of riparian zones in maintaining ecosystem function; 2) human and environment health concerns associated with agrochemical use; and 3) determination of optimum riparian buffer widths that will allow matambre farmers to cultivate within riparian zones while minimizing adverse effects from riparian forest removal and pesticide application. Land use activities within the riparian zone of the Temash River highlight the importance of reconciling the perceived differences between conservation goals and the land use practices and resource needs of Maya communities in southern Belize.

Table 3-1. Number of households surveyed and the mean household size (% of total households in village; ± = standard deviation)

	Crique Sarco	Lucky Strike	Sunday Wood	Overall
households surveyed	43 (83%)	8 (89%)	39 (93%)	90 (87%)
mean household size	5 (± 2.4)	4.6 (± 1.4)	5.4 (± 2.1)	5.1 (±2.2)

Table 3-2. Summary of *matambre* characteristics for each village (± = stdev)

	Crique Sarco	Lucky Strike	Sunday Wood	Overall
households practicing *matambre*	29 (67.4%)	7 (87.5%)	32 (82.1%)	68 (75.6%)
% of households with higher yield from *matambre*	48%	100%	70%	64%
% of *matambre* farms within riparian zone	62%	14%	62%	57%
matambre in upland area	38%	86%	19%	34%
matambre in lowland area	- - -	- - -	19%	8%
mean travel time to field (minutes walking)	35 (± 22)	12 (± 8)	52 (± 17)	41 (±22)
mean size of field (hectares)	1.03 (± 0.74)	1.45 (± 0.32)	0.96 (± 0.39)	1.04 (± 0.57)

Table 3-3. Agrochemical use among *matambre* farmers

	Crique Sarco	Lucky Strike	Sunday Wood	Overall
matambre farmers using herbicides	21 (n = 29)	7 (n = 7)	29 (n = 32)	57 (n = 68)
frequency of herbicide application (% of farmers applying herbicide)	once = 95% twice = 5%	once = 72% twice = 28%	once = 68% twice = 32%	once = 79% twice = 21%
matambre farmers receiving agrochemical training from Pesticide Control Board	6	1	18	25

Table 3-4. Common agrochemicals of *matambre* farmers and their toxicity to aquatic organisms

		Acute Toxicity to Aquatic Organisms*				
	# of farmers	Amphibians	Insects	Crustaceans	Molluscks	Fish
Paraquat (sold as Gramoxone®)	30 (53%)	Slight	Not Acute	Moderate	Slight	Moderate to High
2,4-D	18 (32%)	Slight	Slight	Not Acute	Not Acute	Moderate to High
Other‡	9 (15%)	n/a	n/a	n/a	n/a	n/a

*Toxicity summarized from (Kegley et al. 2008)

‡ Other pesticides used include Glyphosate (RoundUp®), Folidol, and Tamaron

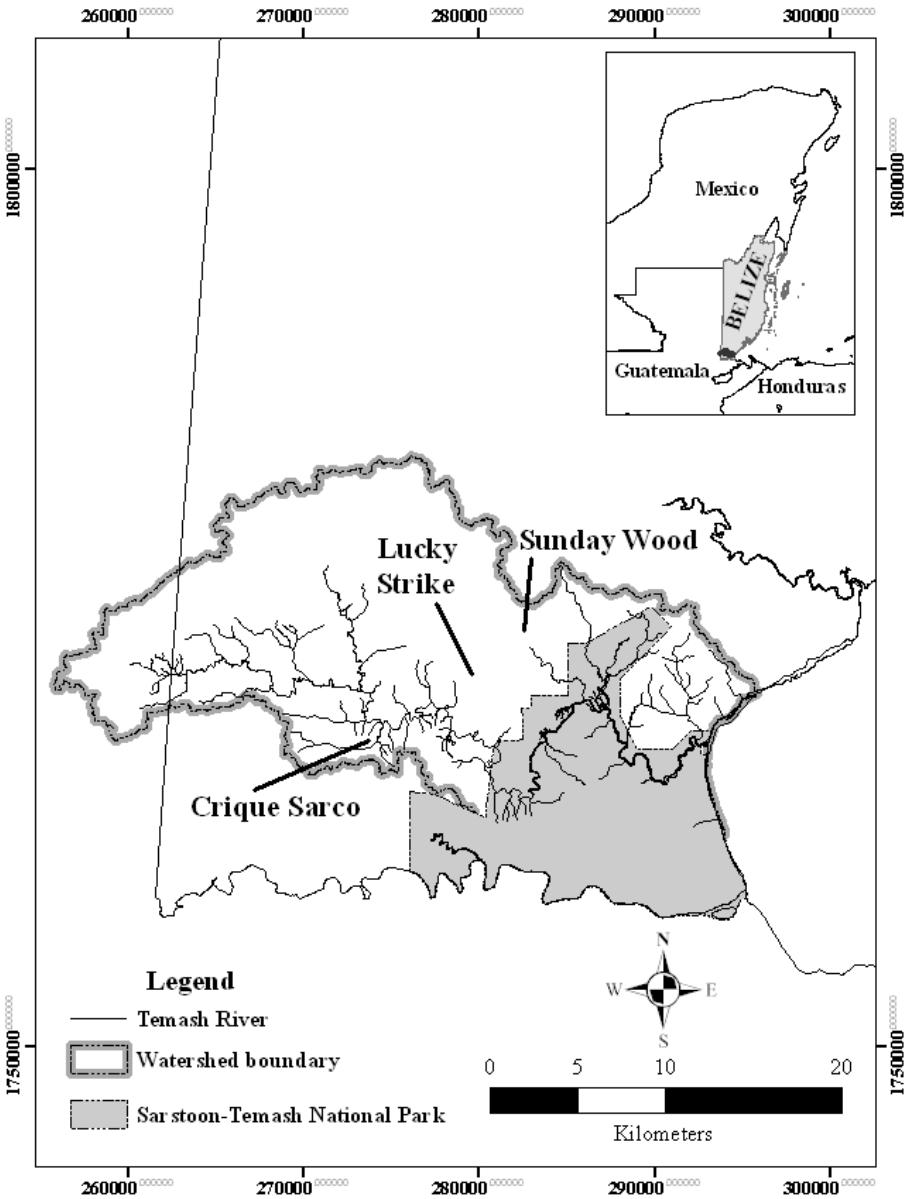

Figure 3-1. Map of the Temash River watershed including the three study villages. The lower reaches include the Sarstoon-Temash National Park.

CHAPTER 4
SOIL NUTRIENT DYNAMICS, ORGANIC MATTER TURNOVER AND LAND USE WITHIN AN ANTHROPOGENIC LANDSCAPE DOMINATED BY SHIFTING CULTIVATION

Introduction

The Maya Forest region of northern Mesoamerica includes much of southern Mexico (the states of Chiapas, Campeche, Yucatán, and Quitana Roo), the northern Guatemalan department of the Petén, and Belize (Figure 4-1). The region contains a wealth of natural and cultural resources and includes the most extensive tract of continuous tropical forest in the Americas, outside of Amazônia. The primary livelihood strategy in rural areas of the Maya Forest region is shifting cultivation, locally referred to as *milpa*. The practice consists of clearing forests by slash-and-burn, planting and cultivation of corn and other food crops, followed by fallowing before the area is re-cleared and cultivated again. *Milpa* has been practiced by indigenous people across Mesoamerica for millennia (Turner et al. 2001) and the contemporary expression of *milpa* agriculture in the Maya Forest region represents a "multiple-use livelihood strategy" (Toledo et al. 2003) with corn (*Zea mays*) being the staple crop, but farmers also actively engage in other land use activities including the cultivation of cash crops and pasture that serve to integrate rural farmers into local and regional markets (Humphries 1993, Keys 2004, Keys and Chowdhury 2006).

Significant research has been conducted on the impacts of *milpa* within the dry tropical forest of the Maya Forest region, particularly in the southern Yucatán peninsular region (SYPR). *Milpa* in the SYPR predominantly occurs within "*ejidos*" or community-owned lands where *ejido* members practice *milpa* and often engage in the extraction of timber and non-timber forest products as part of the land management (Turner et al.

2001, Vance and Geoghegan 2002, Bray et al. 2004). Scholars consider *milpa* to be a dominant driver of land cover change in the SYPR, resulting in large areas of secondary forest in varying stages of succession (Turner et al. 2001, Chowdhury 2006). Above-ground biomass within these secondary forests increases rapidly during succession while a return to values similar to mature forest stands requires between 55-95 years (Read and Lawrence 2003). A recent study also suggests that repeated cycles of clearing and burning can ultimately reduce the amount of carbon stored in the above-ground biomass of secondary forests by 64% (Eaton and Lawrence 2009). However, in soils subjected to repeated cycles of clearing, burning, cultivation, and abandonment, the size of the carbon stock (i.e., g-C/m^2) remains relatively stable (Eaton and Lawrence 2009). Hughes et al. (1999) noted a similar pattern in the soils of secondary forests of the Los Tuxtlas region, northwest of the Yucatán, where *milpa* is also the dominant land use. Variability in the spatial distribution of soil nutrients in the SYPR is less influenced by land use than by tree-related patterns of soil nutrients (Døckersmith et al. 1999) or variations in precipitation and topography across the peninsula (Lawrence and Foster 2002).

Elsewhere in the Maya Forest region, the Lacandon Maya of southern Chiapas actively manage secondary forests to promote certain species with greater leaf-litter production that aids in the re-accumulation of soil organic matter following shifting cultivation (Levy-Tacher and Golicher 2004). Several of these tree species have also been shown to limit harmful soil nematodes, effectively improving soil conditions during fallow (Diemont et al. 2006). The system of *milpa* as practiced by the Lacandon Maya

maintains local biodiversity and soil ecology within a traditional subsistence livelihood strategy (Nations and Nigh 1980, Diemont and Martin 2009).

Early studies of the impact of *milpa* on soils in the Petén region of the Maya Forest suggest that burning above-ground biomass results in increases in some major exchangeable cations, whereas organic matter, nitrogen, and phosphorus all decrease after burning (Cowgill 1962). During and directly after cultivation, soil nutrients declined relative to neighboring forest soils and this decline was most pronounced in fields subject to successive years of cultivation (Cowgill 1962). Following *milpa* abandonment, early forest succession occurs rapidly in the Petén (Ferguson et al. 2003). However, episodes of human migration, dating back to the late 19th century and continuing through the late 20th century, have resulted in significant changes in land use patterns in Petén. Approximately half of Petén forests have been cleared within the past 40 years as a result of rapid colonization, and *milpa* now competes with pasture development as the primary land use (Sader et al. 1997, Carr 2004, 2008). In addition, immigrants to Petén who practice *milpa*, whether Mestizo/Ladino or Q'eqchí Maya from other regions of Guatemala, are rapidly clearing forest, do not share the same degree of ecological knowledge about the forests of Petén as resident Itzá Maya farmers, and do not follow some of the soil conservation practices of resident Itzá Maya farmers (Atran 1993, Atran et al. 1999).

This study was conducted in the southeastern lowlands of the Maya Forest region, in southern Belize. The study area is home to Q'eqchí Maya that immigrated into the area in the late 19th century from Guatemala, have no legal tenure to the land, but maintain customary land use practices that revolve around *milpa* agriculture and also

include organic cacao (*Theobroma cacao*) agroforestry and pasture. I characterize soils within this anthropogenic landscape using a synchronic sampling design and present data on soils collected across a chronosequence of land covers including active milpa fields, fallowed fields of varying successional stages, mature forests, cacao orchards, and pasture. The objectives of the research were to: (i) describe soil nutrients within the anthropogenic landscape of Q'eqchí Maya of southern Belize; (ii) evaluate the impact of *milpa*, cacao agroforestry and pasture relative to mature forests by estimating the time needed for soils to recover to pre-disturbance conditions; and (iii) consider the sustainability of these different land uses within the context of the Maya Forest.

Study Area

The study was conducted in the Temash River watershed of southern Belize that is home to several rural Q'eqchí Maya communities (Figure 4-1). Whereas prehistoric records of Q'eqchí in Belize are unknown, the first historic records suggest they arrived in southern Belize from the Alta Verapaz of Guatemala during the 1890s as part of a large-scale labor migration to satisfy the demands of a growing cacao, coffee and banana plantation economy (Wilk 1997, Grandia 2004). Although the plantation economy declined during the 1910s, the Q'eqchí Maya persisted and adopted a diversified production strategy that included food staples and products for external markets. The primary land use of today's Q'eqchí Maya of southern Belize is *milpa* agriculture. Detailed descriptions of this land use practice for the Maya communities of Southern Belize can be found elsewhere (Wilk 1997, Grandia 2004). In addition to *milpa* agriculture for corn and other food crops, the Q'eqchí, along with nearby Mopan Maya communities, have become suppliers to a rapidly expanding international market for organically grown cacao (Levasseur and Olivier 2000, Emch 2003). The Q'eqchi'

Maya also raise livestock and pasture is a rapidly expanding land use in the Temash and other parts of southern Belize.

Mean annual rainfall in southern Belize is ~3800 mm, but is highly seasonal with most precipitation occurring between May and October. Mean monthly temperatures range from 18.7°C (January) to 32.1°C (May). The natural vegetation cover in the watershed is dominated by tropical evergreen lowland forests (Meerman et al. 2003a) and is within the tropical moist forest life zone (Holdridge 1971).

The soils in the Temash River watershed fall within the Temash sub-suite of the Toledo Suite, as originally described by Wright et al. (1959), and generally correspond with gleyic and haplic acrisols of the Food and Agriculture Organization's soil classification system (Baillie et al. 1993). The soils are derived from interbedded calcareous sandstones and mudstones (Wright et al. 1959) and are acidic, with thick clay horizons that result in imperfect drainage (Baillie et al. 1993, Holland et al. 2003).

Methods

Soil Sampling and Land Use History

Soil samples were collected from plots (N = 74) representing seven different land use/land cover categories (Table 4-1). Five soil cores of the uppermost 10 cm (0 – 10 cm) were collected from each plot using an 'X' shaped sampling scheme. In each plot, one core was collected at the approximate central point and four additional cores were collected at the end of 10-m long transects radiating outward from the central point in the shape of an 'X'. The five cores were composited in the field and stored in labeled Whirl-pak® bags. Bulk density for each plot was estimated using the cylinder method (USDA 2004). Samples for nutrient and stable isotope ($\delta^{13}C$) analysis were air dried and sieved to retain the < 2 mm fraction.

The land use history of each soil sampling site was determined through interviews with farmers from each of the five villages. In active *milpa* fields, farmers were asked how many consecutive years the fields had been cultivated and to describe the land cover prior to clearing (i.e., type and age of land cover). In successional stages of regrowth (SS1 – SS3), the age of the fallowed field (locally referred to as *huamil*), and the land cover prior to the original clearing was recorded. In addition, descriptions of the dominant vegetation within each successional stage plot were recorded, including species present, canopy height, canopy closure, and diameter at breast height of canopy and emergent trees. The age classes approximate previously described age classes of successional vegetation as defined by the Q'eqchi' (Wilk 1981). Similar information was recorded for cacao orchards and pastures (Table 4-1). Plant samples of the dominant vegetation types within each land cover were also collected for isotope analysis.

Laboratory Analysis

Soil samples from each plot were homogenized and split for individual analyses. Percent organic matter was determined by loss on ignition (LOI) following Nelson and Sommers (1996) and USDA (2004). Samples for macro- and micronutrient analysis were sent to Waters Agricultural Labs, Inc., and analyzed using methods described in (Gavlak et al. 1994). Soil pH was determined in a 1:1, soil: water solution. Soil NO_3^- was determined by colorimetry following cadmium reduction. All other macro- and micronutrient analyses were conducted using a Mehlich I extraction and measured by inductively coupled plasma on a Thermo ICAP-AES 6500 and reported as parts per million (ppm). Percent C and N were determined by flash combustion on a Carlo-Erba NA 1500 CNS elemental analyzer. Values were converted to g/m^2 using the bulk

density data. Bulk stable carbon ($\delta^{13}C$) and isotopic analyses were measured simultaneously using a VG Prism Series II isotope ratio mass spectrometer with a triple trap preparation device linked to the elemental analyzer.

Estimation of SOM Turnover

In landscapes with undisturbed C3 forest vegetation, active *milpas*, pastures, and fields that have been fallowed for different lengths of time, the $\delta^{13}C$ of SOM in surface soils can be used to determine the relative contribution of C3 vs. C4 vegetation to the soil. Isotopic values of the 'new' SOM at time *t* ($\delta^{13}C_t$) and of all end-members must be evaluated, including C3 vegetation ($\delta^{13}C_{3\text{-veg}}$), C3-derived SOM ($\delta^{13}C_{3\text{-soil}}$), C4 vegetation ($\delta^{13}C_{4\text{-veg}}$), and C4-derived SOM ($\delta^{13}C_{4\text{-soil}}$). In situations where a distinct SOM-$\delta^{13}C$ of soil derived solely from C4 vegetation is not available, the $\delta^{13}C$ of the C4 vegetation ($\delta^{13}C_{4\text{-veg}}$) can be used (Wiesenberg et al. 2004). From these values, it is possible to estimate the percent (*x*) of carbon coming from the 'new' C4 plant using the Equation 4-1 (Balesdent et al. 1987):

$$\delta^{13}C_t = (\delta^{13}C_{4\text{-veg}} - \delta^{13}C_{3\text{-soil}}) * x/100 + \delta^{13}C_{3\text{-soil}} \quad (4\text{-}1)$$

Decomposition of SOM is assumed to follow first-order kinetics (Balesdent and Mariotti 1996) and can be calculated using Equation 4-2 (Wiesenberg et al. 2004):

$$k = -\ln(C3_t/C3_{t0}) / (t - t0) \quad (4\text{-}2)$$

Where *k* is the decay constant, $C3_t$ equals the percentage of C3-carbon remaining in the soil after t years since clearing of original C3 vegetation, and $C3_{t0}$ is the percentage of C3-carbon in the soils at the time of clearing (assumed to equal 100). The mean residence time (MRT) equals the average time soil C resides in the SOM reservoir under steady state conditions (Derrien and Amelung 2011). MRT can be calculated using Equation 4-3 (Wiesenberg et al. 2004):

$$MRT = 1/k \quad (4\text{-}3)$$

Statistical Analyses

The non-parametric Mann-Whitney U statistic (p-value ≤ 0.05) was used to test for significant differences between mature forest soils and soils from all other land cover classes. When ties in the rank scores for soil classes were present, the distribution estimate was corrected for ties, including a correction for continuity (Hintze 2001). The z-score and corresponding probability value are provided for these cases.

Correlations between soil properties were determined using Pearson's product-moment. Correlations where |r| ≥ 0.70 were considered to be strong, following a post-hoc bonferroni adjustment for multiple tests (adjusted P-value = 0.00067).

Results

Soil Physical and Chemical Characteristics

Percent organic matter (%OM) varied little across land use/land cover classes. *Milpa* fields had the lowest %OM whereas cacao orchards had the highest %OM, although no anthropogenic soil classes were significantly different from forest soils (Table 4-2). Similar patterns for carbon (g-C/m^2) and nitrogen (g-N/m^2) were also observed (Table 4-2). Median values for the C:N ratio for pasture soils were higher than the C:N ratio of forest soils (U = 18, P = 0.027) (Table 4-2, Figure 4-2). Cation exchange capacity (CEC) for all soils ranged from 14.2 to 17.3 meq/100g and soils across all classes were acidic, with pH values between 5.0 and 5.7 (Table 4-2). Aluminum (Al) concentrations in SS1 soils were lower than forest soils (U = 45, z = 2.2181, P = 0.027).

Soil nitrate (NO_3^-N) concentrations from *milpa* (U = 17, P = 0.012), SS1 (U = 45, P = 0.027), SS2 (U = 23.5, z = 2.457, P = 0.014) and pasture (U = 8, P = 0.001) were

significantly lower than forest soil NO_3^-N (Table 4-3, Figure 4-2). Phosphorus (P) concentrations were low across all soil classes. Potassium (K) concentrations were elevated in *milpa* (U=8, z = -3.1161, P = 0.002), SS1 (U = 39, z = -2.489, P = 0.013), cacao (U = 0, P = 0.0014) and pasture (U = 9, P = 0.002) soils relative to forest soils (Table 4-3, Fig 4-2). Magnesium (Mg) concentrations in *milpa* soils were also elevated relative to forest soils (U = 19, P = 0.02) (Table 4-3, Figure 4-2). No significant differences were observed in calcium (Ca) concentrations across any soil classes. Sulfur (S) concentrations were lower than forest soils within *milpa* (U = 21, z = 2.1787, P = 0.029), cacao (U = 2.5, z = -2.3503, P = 0.019) and pasture (U = 19, z = 2.1411, P = 0.032) (Table 4-3, Figure 4-2). No significant differences were detected between the soils of anthropogenic land covers and forest soils for any of the micronutrients measured (Table 4-4).

Strong positive correlations (r > 0.85) were observed between %OM, g-N/m^2, and g-C/m^2 and these three soil characteristics were also correlated with CEC (r > 0.7) (Table 4-5). Ca was also correlated with %OM, g-N/m^2, and CEC (r > 0.7), with a weaker correlation between Ca and g-C/m^2 (r = 0.65). In addition, Boron (B) was positively correlated (r > 0.70) with g-N/m^2, Ca, and pH, and had a weaker positive correlation with %OM (r = 0.63), g-C/m^2 (r = 0.66), and CEC (r = 0.69). Boron also had a weak negative correlation with copper (Cu) (r = -0.67). Cu was positively correlated (r > 0.70) with Iron (Fe) and both Cu and Fe were negatively correlated with pH (r = -0.72 and r = -0.68, respectively). Cu was also negatively correlated with Ca (r = -0.67). Aluminum (Al) had a weak negative correlation with CEC and a very weak positive correlation with Fe (Table 4-5).

Stable Carbon Isotopes in Vegetation and Soils and the MRT of SOM

Pasture grass (*B. humidicola*) and corn (*Z. mays*) have $\delta^{13}C$ values of C4 vegetation (-13.13‰ and -12.84‰, respectively; Table 4-6). The pasture soils have $\delta^{13}C$ values typical of SOM derived from C4 vegetation whereas *milpa* soils reflect the isotopic signature of C3-derived SOM (Table 4-5). All other land cover classes have soils with $\delta^{13}C$ values typical of SOM derived from its dominant C3 vegetation cover (Table 4-6). Within pasture soils, the relative contribution of C4 pasture grass to SOM ranged from 2.4% in a 1-year old pasture to 24% in a 28-year old pasture (Figure 4-2). Estimates for the MRT of SOM within pastures ranged from 14 yrs to 102 yrs (Figure 4-3).

Discussion

Soil Nutrient Dynamics in Milpa Versus Forest Soils

The success of *milpa* agriculture is largely dependent on nutrient availability in cultivated soils (Sanchez 1982, Kleinman et al. 1996) and the nutrient-rich ash that remains following burning is a commonly cited mechanism for increasing soil fertility within *milpa* fields (Nye and Greenland 1960). The burning of above-ground biomass releases a pulse of plant-available nutrients and the resulting ash increases soil pH and cation exchange capacity (CEC) in tropical soils (Nye and Greenland 1960, Ewel et al. 1981, Tiessen et al. 1992, Giardina et al. 2000, McGrath et al. 2001, Arunachalam 2002). The pH and CEC of *milpa* soils in the Temash watershed did increase slightly relative to forest soils (Table 4-2), but this increase was only significant at P=0.32 and P=0.24, respectively. Wright et al. (1959) also noted little difference in pH between forested and cultivated soils in the Temash.

Although ash can increase pH and CEC in tropical soils, nutrients - particularly N, P and organic C – can also be lost during and following the burn (Ewel et al. 1981, Tiessen et al. 1992, Giardina et al. 2000, Sommer et al. 2004). In Temash soils, total-N and NO_3^- were reduced in *milpa* fields relative to forests and NO_3^- remained significantly lower than forest soils through the first two phases of succession (Table 4-3, Figure 4-2). The disruption of the N cycle by felling and burning tropical forests greatly reduces the amount of N uptake by vegetation, and increases in NO_3^- in soil solution have been observed following slash-and-burn (Uhl and Jordan 1984, Hölscher et al. 1997, Williams and Melack 1997). The accumulation and immobilization of N in early successional weedy biomass helps to slow the loss of mineralized N via leaching (Lambert and Arnason 1986, Brubacher et al. 1989), but the fact that NO_3^- in *milpa* soils does not return to values similar to forest soils until > 15 years of fallow (SS3) is also a reflection of high N demand by early successional species (Ewel 1986).

Available P content in Temash soils was relatively unchanged through cultivation and 2nd succession and was never significantly different from forest soils (Table 4-3). Soil P is derived from the physical and chemical weathering of geologic parent material, and in highly weathered tropical soils, any biologically available P is readily adsorbed onto clays and other inorganic constituents, becoming biologically unavailable (Vitousek and Sanford 1986, Tiessen et al. 1992, Lawrence and Schlesinger 2001). Although not measured in this study, approximately half of the P from biomass burning reaches the soil surface (Giardina et al. 2000), and the organic-P fraction in soils is often enriched following deforestation (Farella et al. 2007). In the acid soils of the Temash, Al and Fe likely play an important role in reducing the amount of available P and lower

concentrations of Al and Fe during early succession may aid in the maintenance of the limited amounts of P in these soils (Kleinman et al. 1996).

The carbon (C) pool most heavily impacted by shifting cultivation is above-ground biomass whereas soil organic matter is the most stable (Ewel et al. 1981, Kotto-Same et al. 1997). In a moist tropical region of Costa Rica, Ewel (1981) estimated that approximately 31% of C stored in above-ground biomass was lost due to burning. In the Amazon, estimates for the loss of C following slash-and-burn can be greater than 50% of the total above-ground pool (Kauffman et al. 1995, Hughes et al. 1999), and as much as 75% of the initial C can be lost when shifting cultivation is carried out on the same plot for 3-4 years (Uhl 1987). Many other studies have also noted that land use intensity – including successive years of cultivation and repeated cycles of slash-and-burn - is a critical factor when determining the impact of slash-and-burn on above-ground C pools that can ultimately decrease the potential of second-growth forests to sequester C (Hughes et al. 1999, Zarin et al. 2005, Eaton and Lawrence 2009).

How these land use practices impact soil C are not entirely clear. In two of the above-mentioned studies in which C pools in above-ground biomass were impacted by the intensity of prior land use, C pools in soil remained stable (Hughes et al. 1999, Eaton and Lawrence 2009). Zarin et al. (2005) did not directly measure soil C. In other studies of slash-and-burn soils, the soil C content either changed little or increased after felling and burning (Nye and Greenland 1964, Seubert et al. 1977, Kotto-Same et al. 1997). In the Temash milpa soils, %OM and total C showed small but insignificant declines relative to forest soils (Table 4-2). However, some studies have documented declines in C following slash-and-burn. In her early study of milpa soils in the Petén,

Cowgill (1962) documented decreases in %OM following the burn and Kotto-Same et al. (1997) mention several other studies where OM-C declined following burning. Salcedo et al. (1997) measured a 17% decrease in organic C following five years of cultivation on an Oxisol in northeastern Brazil.

Organic matter is commonly volatilized at temperatures between 200°C and 315°C (Lide 2004). During an experimental burn of slash produced from an 8-9- year-old 2nd growth forest in Costa Rica, Ewel (1981) recorded temperatures in excess of 400°C within the burning slash 1-2 cm above the soil surface, but temperatures at the soil surface averaged ~ 200°C and dropped to ~100°C at 1 cm soil depth and <38°C at 3 cm depth. The C:N ratio decreased within the upper 3 cm during the slash-and-burn phase at this site, but overall C storage was high in the soils (Ewel et al. 1981). In a mature (>100-yr-old) dry forest site of central Mexico, surface temperatures exceeded 500°C and declined to 100°C at 3 cm soil depth (Giardina et al. 2000). Total C at this site decreased by ~20% in the upper 2 cm following the slash burning but was unchanged from 2-5 cm soil depth (Giardina et al. 2000). The high soil surface temperatures at this site also altered soil N and P pools. Total N decreased in the upper 2 cm but was offset by an increase in plant-available N within the 0-10 cm horizon, with a similar decrease in organic P and increase in plant-available P also occurring (Giardina et al. 2000). Variations in the maximum soil surface temperature during slash burning, and consequently the impact of burning on soil nutrient pools, are likely related to the age and stature of the cleared forest, the length of the dry season, and soil moisture (Giovannini et al. 1990, Giardina et al. 2000, Knicker 2007).

Cacao Agroforestry and Pasture

In tropical regions, agricultural practices that best mimic the structure and function of natural communities are considered more sustainable (Ewel 1986). Cacao agroforestry attempts to mimic the structure of a tropical forest by utilizing canopy trees for shade, with cacao trees occupying the under story. Cacao is cultivated across multiple regions of the tropics both in large-scale plantations and in smallholder orchards. Cacao cultivation by smallholder Maya farmers in southern Belize has expanded rapidly in recent years in an effort to meet an increasing international market demand for organic cacao beans (Emch 2003). The Toledo Cacao Growers Association (TCGA) was developed in the late 1980s to help facilitate the sale of cacao grown by farmers in southern Belize. Membership in the TCGA has grown from 70 farmers in 1987 to more than 1000 members from 80 villages in 2008 (TCGA, unpublished data).

Cacao orchards in the Temash were planted in 2003-2004 through assistance with TCGA and a local non-governmental organization, the Sarstoon-Temash Institute for Indigenous Management (SATIIM). Cacao soils in the Temash have maintained forest soil characteristics (Tables 4-2, 4-3, 4-4) with K actually being significantly higher than forest soils (Table 4-3).

Pasture expansion has long been considered a driving force behind Central American deforestation, historically driven by external market demands for beef (Myers 1981). Government policies in Guatemala provided large tracts of land at low prices to land speculators in the Petén, resulting in rapid pasture expansion in that region (Schwartz 1990). Although policies have shifted slightly, pasture continues to expand, driven partly by new infrastructure development and by the need for land owners to

establish clear land tenure (Kaimowitz 1995). Similar patterns are observed in Belize as well (Chomitz and Gray 1996). The Belize Ministry of Agriculture conducts annual surveys of cattle pasture and, from 2005 – 2007, the number of cattle in the southern Toledo District increased by 47%, from 3515 to 5163, while pasture (both improved and natural) increased by 42%, from 6181 acres to 8789 acres (Ministry of Agriculture, unpublished data). During this same time period, the number of cattle in the Temash River watershed (within the Toledo District) increased by 110% from 194 head of cattle to 407, while pasture increased by 14% from 776 acres to 883 acres. Although this only accounts for three years of data, this may represent a general trend towards pasture expansion into the more remote areas of southern Belize like the Temash River watershed.

Carbon content in pasture soils in the Temash is not significantly different from the C content of forest soils. Several studies have noted the initial conservation of soil nutrient stocks following forest conversion to pasture (Buschbacher et al. 1988, Neill et al. 1997, Desjardins et al. 2004). Factors controlling the impact on soils of forest conversion to pasture include land use history and pasture management (Buschbacher et al. 1988, Neill et al. 1997). Stabilization of organic C by Al-organic matter complexes can also reduce the amount of carbon loss during conversion (Veldkamp 1994). Pastures do increase soil compaction and reduce root penetration, water infiltration and gas exchange (McGrath et al. 2001), and the low nitrate values in the Temash pasture soils (Table 4-2, Figure 4-2) may be related to compaction. Pastures in the Temash are frequently managed with fire and the increase in K in pasture soils (Table 4-2, Figure 4-2) is likely related to the repeated burning of above-ground biomass.

Stable Isotopes and MRT of SOM in Temash Soils

When considering long-term impacts of land use and land cover change on soils, the mean residence time (MRT) of the carbon in soils is also important and may represent a more accurate measure of impacts from land cover change on soil structure and function than simply a measure of the amount of carbon per square meter in the soil (Trumbore et al. 1995, Balesdent and Mariotti 1996, Six and Jastrow 2002). The $\delta^{13}C$ of milpa soils did not differ significantly from forest soils, thus precluding calculation of MRT using the above-mentioned equations. The $\delta^{13}C$ of corn is isotopically heavier than that of either forest plants or forest soils, but it is no apparent in the $\delta^{13}C$ of the SOM of Temash milpa soils. A primary reason for this is that litter fall from corn fields is only about 10% of forest litter production (Awiti et al. 2008). In addition, milpa sites in the Temash are rarely used for more than two consecutive years, effectively limiting the time for carbon from C4 plants to accumulate within the leaf litter and topsoil. Studies that have documented SOM changes under corn cultivation include sites where corn has been under cultivation for > 20 years (Cayet and Lightfouse 2001, Wiesenberg et al. 2004). Awiti et al. (2008) estimate that maize-derived (C4) carbon would become the dominant source of carbon in soils after 38 years of continuous cultivation.

Pasture soils in the Temash reflect the shift in carbon input from C3 forests to C4 pasture grass. There is a strong positive correlation between the age of the pasture and the amount of C4 carbon in the soil (Table 4-6 Figures 4-3, 4-4). This pattern has been observed elsewhere in the tropics as well (Trumbore et al. 1995, Desjardins et al. 2004) and factors influencing the rate of incorporation of C4-carbon into soils include soil type (clayey versus sandy), soil particle size, climate, type of pasture grass, and pasture management practices (Trumbore et al. 1995, Desjardins et al. 2004). The

MRT of Temash pasture soils is also related to pasture age - the oldest pasture (28 years) has an MRT of ~ 100 years. The estimates for MRT are conservative however, and may in fact be much longer. The model for determining the percent contribution of C3 vs C4 carbon assumes that inputs from C3 carbon sources stop after conversion to pasture. Tropical forest soils have active root zones down below 1 m in the soil and carbon cycling on the order of 100-1000 years can occur at depth (Nepstad et al. 1994, Trumbore et al. 1995). Replacing deep-rooted trees with shallow-rooted pasture grasses disrupts this soil carbon cycling at depth (Trumbore et al. 1995).

Across the tropics, the potential for pasture to serve as a carbon sink varies widely and accurate estimates must also account for the loss of C stored in above-ground biomass (Desjardins et al. 1994, Fearnside and Barbosa 1998). In the Temash soils, MRT of carbon is generally greater than estimates for MRT in tropical soils (~25 yrs), suggesting that in certain circumstances pasture can serve as a carbon sink. However, no estimate for above-ground biomass for Temash forests has been calculated that would allow for a more accurate assessment of the impact of forest conversion to pasture on soils.

Sustainability of Milpa in the Maya Forest

The persistence of milpa as a land use strategy in the Maya Forest region suggests that it is an effective means of adapting to the natural environment that capitalizes on ecosystem services without exhausting them (Alcorn and Toledo 1998, Berkes and Folke 2002, Toledo et al. 2003). In the Temash River watershed, data on soil NO_3^- suggest that soil recovery following slash-and-burn may take upwards of 15 years to return to pre-disturbance conditions. This same timeframe for recovery of soil has been observed in soils used by smallholder farmers in the Brazilian Amazon as well

(Farella et al. 2007). Uhl and Jordan (1984) noted that soil nutrients recovered within five years of fallow in the Venezuelan Amazon, whereas numerous others have documented little if any change in major soil nutrients following slash-and-burn (Uhl 1987, Hughes et al. 1999, Eaton and Lawrence 2009). An important factor in determining the ability of forests to recover following slash-and-burn is land-use intensity, particularly the number of consecutive years under cultivation (Uhl 1987, Zarin et al. 2005). Moran et al. (2000), however noted that relative to other land uses successional forests from slash-and-burn have higher overall above-ground biomass and higher dominance of canopy biomass than forests succeeding pasture or abandoned mechanized agriculture.

An important distinction that should be made when assessing the sustainability of slash-and-burn agriculture in the tropics is the relationship of the farmer to the area he is farming. Myers (1993) noted that the impact of "shifted" versus "shifting" cultivators can be significantly different and is often tied up in larger national and regional development agendas that are not easily recognized at smaller scales (Lambin et al. 2001, Vance and Geoghegan 2002). In Petén, Guatemala, land development policies both inside and outside the region created a group of farmers unfamiliar with the natural environment of the lowland forests (Atran et al. 1999, Carr 2002). These "shifted" cultivators have been a primary driver of deforestation in the Petén. The examples of the Lacandon Maya (Diemont et al. 2006) and the ejido system of the SYPR (Bray et al. 2004) suggest that Maya farmers can develop an agricultural system that is sustainable, and soil data from the Temash support this idea as well. A major challenge facing small-scale agriculturalists in the Maya Forest region and throughout the tropics is

related to market integration and the introduction of new crops that can negatively impact traditional agricultural practices (Humphries 1993, Keys 2004). Cash crops, such as cacao, grown within an agroforestry system can have minimal impacts on local biodiversity and ecosystem function (Steffan-Dewenter et al. 2007), but smallholders would benefit from development assistance that provides training in labor-saving techniques including intensive shade-tree management, arboriculture, and green manure practices (McGrath et al. 2000, Rosenberg and Marcotte 2005).

Table 4-1. Dominant land cover classes in the Temash River watershed. Soils were collected from each land cover class (sample size in parentheses).

Land Cover Class	Q'eqchi' Name*	Land Cover Age	Dominant Vegetation
milpa (n=11)	*c'at c'al*	< 1yr	*Zea mays*
SS1 (n=21)	*coc' pim*	1-3 yrs	Herbaceous species - *Heliconia* sp., *Solanum* sp., *Neurolaena lobata*, *Ipomoea violacea* Broadleaf species - *Cecropia* sp., *Vismia ferruginea*
SS2 (n=14)	*coc' che ru* or *coc' al c'al*	4-15 yrs	Broadleaf species - *Cecropia* sp., *Vismia ferruginea* Melastomatacea, *Acacia cornigera*, *Atalea cohune*, *Bursera simaruba*
SS3 (n=5)	*ninki al c'al*	16-30 yrs	Broadleaf species - *Atalea cohune*, *Bursera simaruba* *Vochysia hondurensis*, *Terminalia amazonia* Understory species - *Sabal mauritiiformis*, *Chamaedorea* spp.
mature forest (n=9)	*nink li q'uiche'* or *q'uiche'*	> 30 yrs	Broadleaf species - *Atalea cohune*, *Vochysia hondurensis*, *Terminalia amazonia*, *Bursera simaruba*, *Pithecellobium arboreum* Understory species - *Sabal mauritiiformis*, *Chamaedorea* spp.
cacao (n=4)	*kakaw*	~ 5-7 yrs	Broadleaf canopy - *Atalea cohune*, *Vismia ferruginea* *Inga* sp. Understory species - *Theobroma cacao*
pasture (n=10)	n.d.	~ 2-28 yrs	*Brachiaria humidicola*

* Q'eqchi' names from Wilk (1981) and/or Grandia (2004)

Table 4-2. Soil properties (mean ± standard error) of the seven primary land use/land cover classes in the Temash River Watershed. The median value is shown in parentheses*.

	Land Use Categories						
	Milpa *(n = 11)*	SS1 *(n = 21)*	SS2 *(n = 14)*	SS3 *(n = 5)*	Forest *(n = 9)*	Cacao Orchard *(n = 4)*	Pasture *(n = 10)*
OM (%LOI)	10.2 ± 0.6 (9.5)	12.4 ± 0.7 (12.3)	11.9 ± 0.9 (12.9)	12.0 ± 1.5 (12.5)	12.4 ± 1.7 (10.4)	13.8 ± 0.4 (14.0)	11.4 ± 0.9 (11.9)
C (g-C/m2)	3805 ± 143 (3673)	4450.3 ± 311 (4259)	4498 ± 417 (4240)	3219 ± 294 (3332)	4526 ± 743 (3521)	5012 ± 320 (4937)	4440 ± 314 (4555)
N (g-N/m2)	293 ± 16 (290)	365 ± 31 (331)	367 ± 38 (365)	261 ± 38 (250)	392 ± 75 (302)	411 ± 37 (404)	332 ± 25 (352)
C:N	13.1 ± 0.3 (12.9)	12.5 ± 0.3 (12.8)	12.5 ± 0.3 (12.3)	12.8 ± 0.9 (12.3)	12.1 ± 0.5 (12.6)	12.3 ± 0.3 (12.3)	13.5 ± 0.3 (13.5)*
CEC (meq/100g)	14.7 ± 0.6 (14.0)	17.3 ± 0.8 (16.9)	16.5 ± 1.6 (17.1)	15.4 ± 2.1 (13.6)	14.2 ± 2.0 (12.6)	16.1 ± 0.4 (16.0)	14.2 ± 0.9 (14.4)
pH (in H_2O)	5.3 ± 0.1 (5.2)	5.7 ± 0.1 (5.7)	5.5 ± 0.2 (5.3)	5.3 ± 0.2 (5.2)	5.0 ± 0.3 (4.9)	5.1 ± 0.1 (5.0)	5.6 ± 0.1 (5.6)
Al (ppm)	144.0 ± 12.0 (156.5)	137.1 ± 19.3 (110.8)*	171.7 ± 35.1 (112.3)	155.8 ± 28.9 (149.5)	243.5 ± 44.7 (205.0)	179.6 ± 33.6 (156.8)	169.9 ± 15.2 (179.5)

* median values with asterisk (*) indicate significant difference from the median forest soil value (Mann Whitney U test statistic, $P<0.05$).

Table 4-3. Macronutrients concentrations (mean ± standard error) of the seven primary land use/land cover classes in the Temash Watershed. The median value is shown in parentheses*.

	Land Use Categories						
	Milpa *(n = 11)*	SS1 *(n = 21)*	SS2 *(n = 14)*	SS3 *(n = 5)*	Forest *(n = 9)*	Cacao *(n = 4)*	Pasture *(n = 10)*
Macronutrients							
NO3-N (ppm)	46 ± 9 (60)*	65 ± 11 (43)*	57 ± 7 (49)*	73 ± 14 (72)	105 ± 18 (91)	129 ± 21 (118)	27 ± 10 (15)*
P (ppm)	3 ± 0.3 (2.7)	2.0 ± 0.1 (1.8)	2.1 ± 0.2 (1.9)	2.4 ± 0.5 (1.9)	2.2 ± 0.3 (1.8)	1.9 ± 0.0 (1.9)	1.8 ± 0.0 (1.8)
K (ppm)	91 ± 15 (84)*	59 ± 5 (57)*	45 ± 4 (41)	47 ± 4 (49)	39 ± 3 (40)	79 ± 8 (75)*	86 ± 10 (89)*
Mg (ppm)	283 ± 19 (281)*	193 ± 20 (147)	170 ± 22 (177)	194 ± 26 (187)	226 ± 49 (183)	253 ± 22 (266)	192 ± 20 (207)
Ca (ppm)	1359 ± 156 (1116)	2223 ± 143 (2143)	2094 ± 290 (2110)	1845 ± 414 (1713)	1488 ± 352 (1613)	1707 ± 146 (1804)	1516 ± 112 (1525)
S (ppm)	3.5 ± 0.4 (4.0)*	7.3 ± 1.3 (4.5)	9.7 ± 2.2 (7.0)	3.9 ± 1.3 (2.5)	7.9 ± 2.0 (5.0)	2.8 ± 0.5 (2.8)*	3.6 ± 0.3 (4.0)*

* median concentrations with asterisk (*) indicate significant difference relative to the median forest soil value (Mann Whitney U test statistic, $P < 0.05$).

Table 4-4. Micronutrient concentrations (mean ± standard error) of the seven primary land use/land cover classes in the Temash Watershed. The median value is shown in parentheses*.

	Land Use Categories						
	Milpa *(n = 11)*	SS1 *(n = 21)*	SS2 *(n = 14)*	SS3 *(n = 5)*	Forest *(n = 9)*	Cacao *(n = 4)*	Pasture *(n = 10)*
Micronutrients							
B (ppm)	0.3 ± 0.0 (0.3)	0.5 ± 0.0 (0.5)	0.4 ± 0.0 (0.3)	0.5 ± 0.2 (0.4)	0.5 ± 0.1 (0.5)	0.3 ± 0.0 (0.3)	0.3 ± 0.0 (0.3)
Zn (ppm)	1.0 ± 0.0 (1.0)	1.1 ± 0.1 (0.9)	1.0 ± 0.1 (0.8)	1.2 ± 0.3 (0.9)	0.9 ± 0.1 (0.8)	0.9 ± 0.1 (0.9)	0.9 ± 0.1 (0.7)
Mn (ppm)	40.4 ± 11.2 (34.0)	26.6 ± 3.1 (26.0)	32.6 ± 5.1 (28.5)	29.6 ± 3.0 (28.0)	31.9 ± 7.4 (30.5)	38.0 ± 7.4 (38.3)	36.9 ± 3.7 (34.0)
Fe (ppm)	12.5 ± 1.5 (11.9)	6.1 ± 0.8 (5.7)	12.2 ± 2.8 (9.4)	11.0 ± 4.5 (6.4)	16.3 ± 6.3 (8.4)	9.5 ± 3.1 (7.7)	12.8 ± 1.5 (13.6)
Cu (ppm)	0.4 ± 0.0 (0.4)	0.3 ± 0.0 (0.2)	0.3 ± 0.0 (0.4)	0.3 ± 0.0 (0.3)	0.4 ± 0.1 (0.2)	0.3 ± 0.0 (0.3)	0.4 ± 0.0 (0.3)

* median concentrations with asterisk (*) indicate significant difference relative to the median forest soil value (Mann Whitney U test statistic, $P < 0.05$).

Table 4-5. Pearson's product moment correlation matrix for soil characteristics from the Temash Watershed†

	%OM	g-N/m2	g-C/m2	C:N	CEC	Ca	S	B	Fe	Cu	Al	pH
%OM	1.00											
g-N/m2	0.85*	1.00										
g-C/m2	0.86*	0.96*	1.00									
C:N	-0.44*	-0.62*	-0.41	1.00								
CEC	0.82*	0.75*	0.71*	-0.52*	1.00							
Ca	0.73*	0.70*	0.65*	-0.54*	0.93*	1.00						
S	0.2	0.31	0.28	-0.23	0.30	0.44*	1.00					
B	0.63*	0.73*	0.66*	-0.49*	0.69*	0.73*	0.35	1.00				
Fe	-0.43*	-0.44*	-0.43*	0.29	-0.54*	-0.65*	-0.18	-0.56*	1.00			
Cu	-0.46*	-0.48*	-0.49*	0.26	-0.57*	-0.69*	-0.33	-0.67*	0.74*	1.00		
Al	-0.40	-0.33	-0.32	0.27	-0.65*	-0.64	-0.15	-0.35*	0.47*	0.35	1.00	
pH	0.39	0.49*	0.47*	-0.25	0.63*	0.75*	0.42	0.71*	-0.68*	-0.72*	-0.42	1.00

* $P < 0.01$ (with post-hoc bonferroni adjustment for multiple comparisons)

† only characteristics that had a significant correlation are shown. All others were insignificant at $P < 0.01$ and $P < 0.05$

Table 4-6. Stable carbon ($\delta^{13}C$ vs. PDB) isotopic concentrations (mean ± standard error) of soil organic matter for the seven primary land cover classes in the Temash Watershed. The median value is shown in parentheses*.

Land Cover Category	Soil $\delta^{13}C$	Dominant Vegetation $\delta^{13}C$
Milpa	-27.71 ± 0.11 (-27.63)	-12.84 (*Z. mays*)
SS1	-27.79 ± 0.10 (-27.77)	-28.27 (*Heliconia* sp.)
SS2	- 27.67 ± 0.12 (-27.81)	-32.16 (*V. ferruginea*)
SS3	- 28.03 ± 0.12 (-28.07)	-30.29 (*A. cohune*)
Forest	- 27.99 ± 0.11 (-27.90)	-30.29 (*A. cohune*)
Cacao Orchard	-28.14 ± 0.10 (-28.09)	n.d. (*T. cacao*)
Pasture	- 25.28 ± 0.36 (-25.22)*	-13.13 (*B. humidicola*)

* median values with asterisk (*) are significantly different from forest soil (Mann Whitney U test statistic, $P < 0.05$)

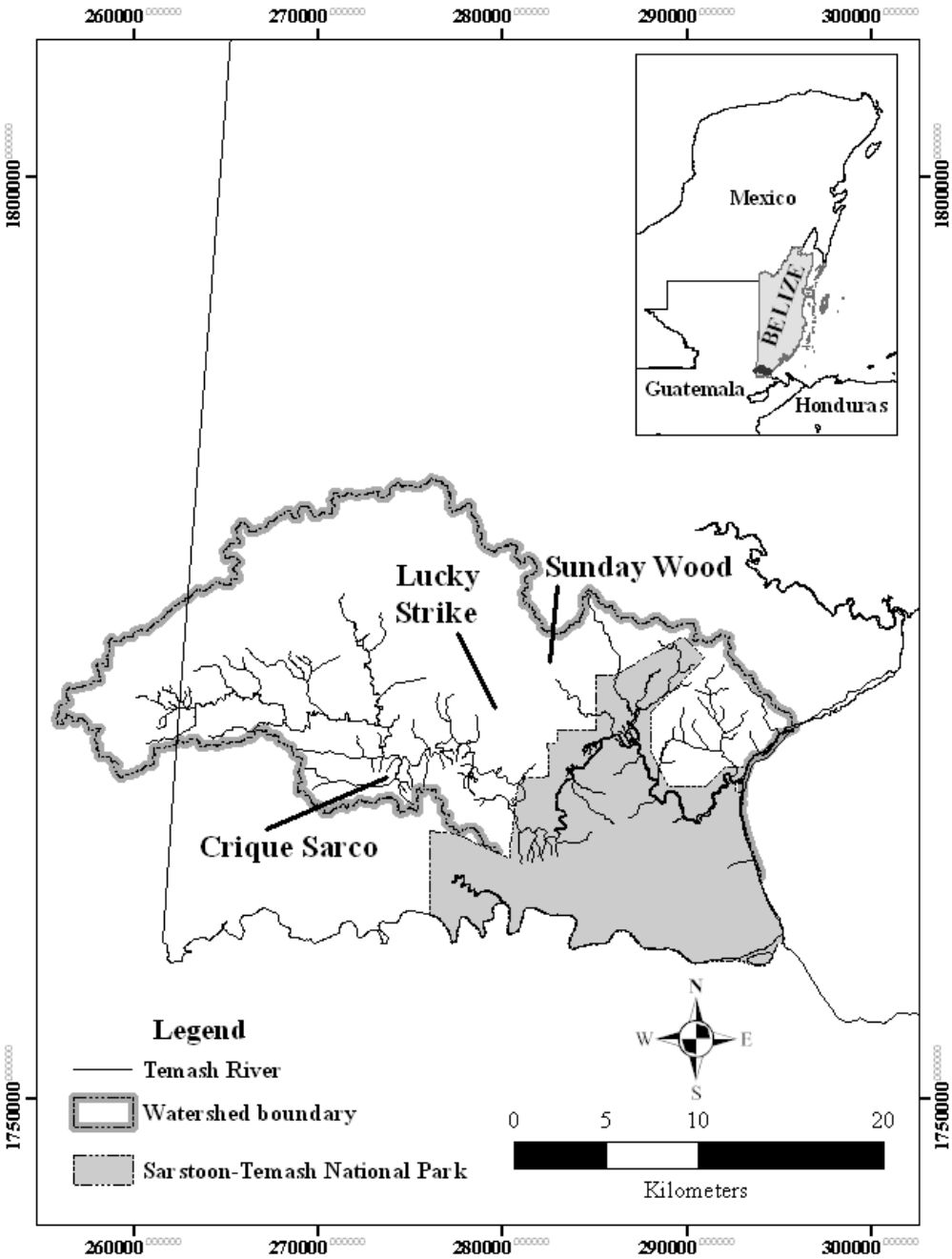

Figure 4-1. Map of study area.

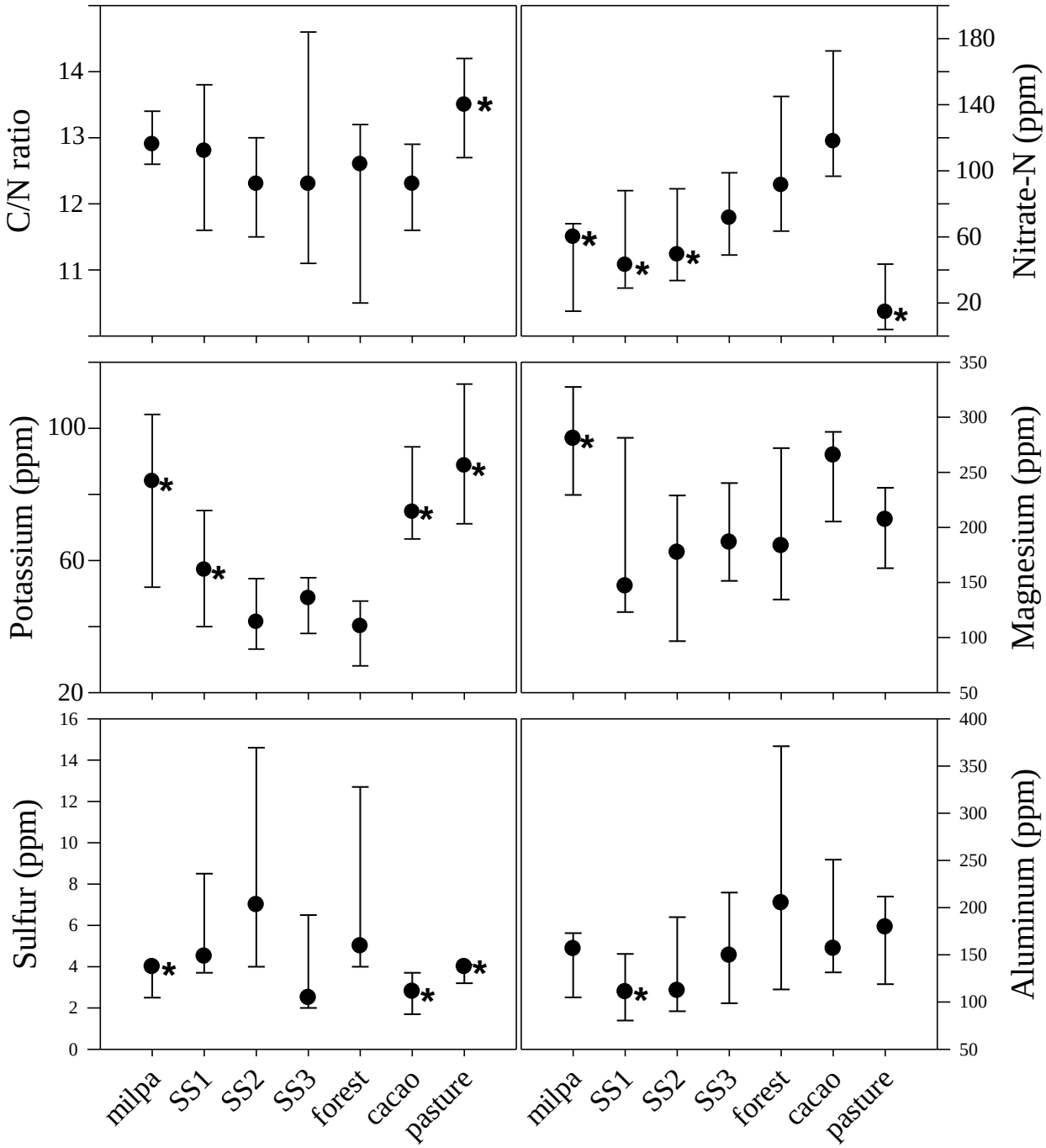

Figure 4-2. Plot of select soil characteristics that include at least one land cover class that is significantly different from forest soils (see Tables 2 and 3 for details; asterisks (*) denote soils that are significantly different from forest soils, Mann Whitney U statistic, $P < 0.05$). (SS1 = 1st succesional stage; SS2 = 2nd successional stage; SS3 = 3rd successional stage)

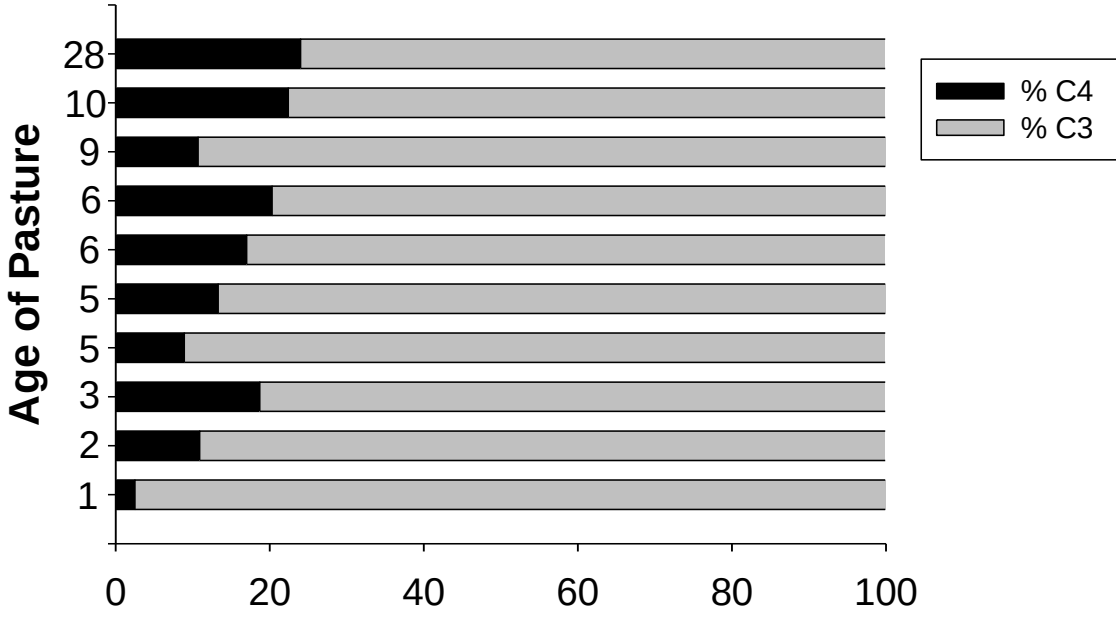

Figure 4-3. Percentages of carbon derived from C3-forest and from C4-pasture.

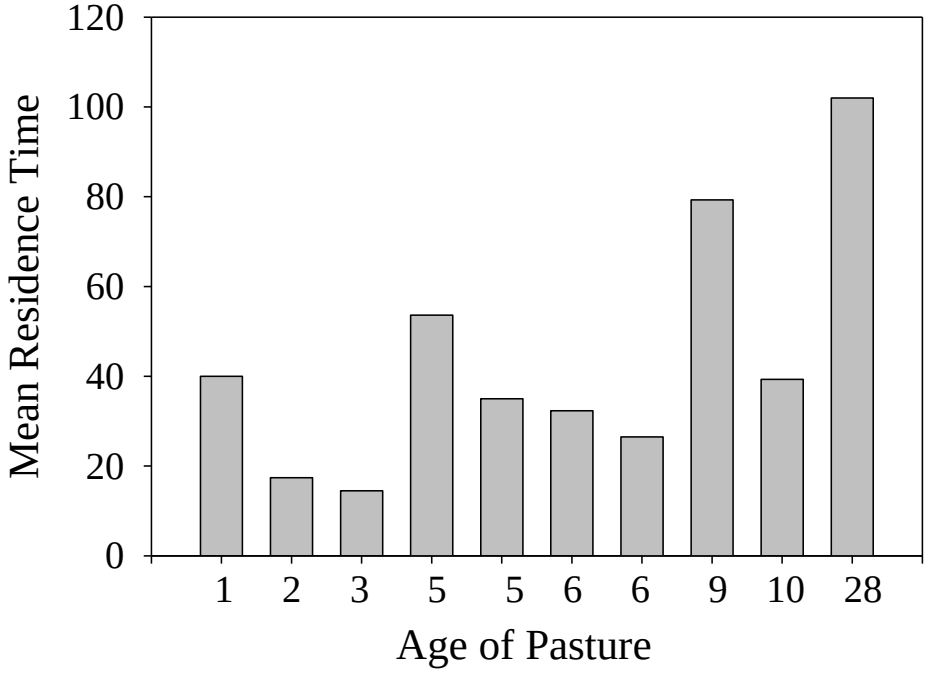

Figure 4-4. The Mean Residence Time (MRT) of carbon in SOM of pasture

CHAPTER 5
TEMPORAL AND SPATIAL VARIABILITY OF DISSOLVED NUTRIENTS IN A SMALL TROPICAL WATERSHED DOMINATED BY SHIFTING CULTIVATION

Introduction

Nutrient concentrations transported in rivers and streams can provide insight into interactions with, and characteristics of the adjacent terrestrial landscape. The transfer of nutrients across the terrestrial-aquatic boundary and the flux of nutrients through stream and river networks depends on a combination of biotic and abiotic factors including catchment and riparian zone vegetation, hydrology, geology (slope and soils), and local/regional climatic patterns. Human land use and land cover conversion, in particular conversion for agricultural purposes, is considered one of the primary drivers of rapid change in nutrient concentrations resulting in the impairment of freshwater and near-shore coastal environments (Carpenter et al. 1998, Caraco and Cole 1999, Downing et al. 1999, Arbuckle and Downing 2001).

In humid tropical watersheds, human-induced change to nutrient export occurs in a context that, in some cases, already experiences nutrient export rates higher than those of temperate watersheds with similar runoff rates (Downing et al. 1999, Lewis et al. 1999). Humid tropical forest ecosystems have high rates of nitrogen (N) fixation and are generally not N-limited (Vitousek and Sanford 1986, Lewis et al. 1995) leading to weak N retention in tropical catchments. However, many aquatic ecosystems in the tropics are considered N-limited because organic-N, rather than plant-available N forms, constitutes a large proportion of the total nitrogen export (Downing et al. 1999, Neill et al. 2001). Phosphorus (P) concentrations in tropical stream networks are dependent on terrestrial sources and reflect the weathering of local catchment geology (Meybeck 1982) and are also positively correlated with soil nutrient concentrations (Biggs et al.

2002). In-stream P can also be strongly influenced by interactions with groundwater and geothermal springs (Pringle and Triska 1991).

Understanding large tropical river responses to catchment-level changes is difficult because of heterogeneity of disturbance at larger spatial scales (McDowell and Asbury 1994). It is generally considered that small catchments are more sensitive to changes in land use, in part because of their small surface-area to channel-length ratio (Thomas et al. 2004), and because they display higher rates of nutrient transformation relative to larger rivers (Peterson et al. 2001). Land cover conversion from forest to agricultural land in small tropical watersheds results in increasing stream discharge because of reduced evapotranspiration (ET) and increased overland flow (Bruijnzeel 2004, Moraes et al. 2006). However, identifying changes in dissolved in-stream nutrient concentrations as a consequence of land cover conversion in small tropical watersheds is understudied and consequently not well understood (Biggs et al. 2004).

Future projections of land use and land cover changes associated with population growth suggest that disproportionately large impacts will be observed in tropical freshwater ecosystems (Sala et al. 2000, Dudgeon et al. 2006, McClain 2008). Although it has often been argued that small-scale farmers are the most dominant agents in tropical forest clearing (Myers 1991, Houghton 1994), large-scale global market processes are considered the ultimate drivers of land use and land cover change in the tropics (Lambin et al. 2001, Geist and Lambin 2002, Filoso et al. 2006). However, more than 250 million people in the tropics practice slash-and-burn agriculture (Attiwill 1994), a land-use strategy that includes clearing forest patches, burning the fallen biomass, planting and cultivating corn and other food crops, followed by fallowing

before the area is re-cleared and cultivated again. In the Maya Forest of northern Central America, slash-and-burn, locally referred to as *milpa*, has been the region's most pervasive land-use strategy for millennia (Turner et al. 2001). Although an extensive literature exists on interactions between *milpa* and wildlife (Gomez-Montes and Bayly 2010), soil nutrients (Diekmann et al. 2007) and forest structure (Døckersmith et al. 1999, Lawrence and Foster 2002), there have been no studies relating *milpa* agriculture and contemporary aquatic ecosystems in northern Central America[1], and there is only one study investigating slash-and-burn agriculture and its impact on stream nutrients from Amazonia (Williams and Melack 1997; Williams et al. 1997).

In this study, I investigate spatial and temporal differences of in-stream nutrient concentrations in four watersheds of varying sizes where the dominant land use is *milpa* agriculture. The objectives for this study are to (i) characterize discharge patterns in relation to precipitation and watershed characteristics for each study catchment, (ii) examine differences in seasonal and longitudinal in-stream nutrient concentrations and nutrient fluxes within each study catchment, (iii) determine relationships between seasonal and annualized flow weighted mean nutrient concentrations and watershed characteristics, and (iv) compare nutrient export coefficients of the study watersheds of the Temash River to other tropical and temperate watersheds.

[1] An extensive literature exists on the use of sediment records from lakes and fluvial geomorphology in the Maya Forest region to better understand ancient human-environment interactions. See (Brenner et al. 2002) for a review.

Methods

Study Site

The headwaters of the binational Temash River begin in Guatemala and flow eastward across the southern Toledo District of Belize towards the Caribbean Sea and Gulf of Honduras (Figure 4.1). Approximately 400 km^2 of the watershed's 460 km^2 are located in Belize. The Temash River is one of 16 major watersheds of Belize and is one of only two of these major watersheds to have never been instrumented with a stream gauge station or other hydrologic monitoring equipment (Lee et al. 1995). Soils are derived largely from Cretaceous limestone with patches of dolomite and siliceous limestone (Wright et al. 1959). Vegetation cover in the Temash River watershed is dominated by tropical evergreen lowland forests (Meerman et al. 2003b). In addition, the lower reaches of the watershed have large stands of red mangrove (*Rhizophora mangle*) as well as the only documented *Sphagnum* bogs in lowland Central America (Meerman et al. 2003b). The lower reaches of the Temash River are within the Sarstoon Temash National Park, a 41,000-acre community co-managed park. The Temash watershed is home to several Q'eqchi' Maya villages. The dominant land use of the local Q'eqchi' is *milpa* agriculture, though pasture and cacao (*Theobroma cacao*) agroforestry are expanding in the region (Levasseur and Olivier 2000, Emch 2003).

Watershed Selection, Delineation, and Characterization

The four study watersheds are 2nd order streams. Yax Cal creek (YXL) and Crique Sarco creek (CRS) drain into the middle reaches of the Temash River. Sunday Wood creek (SWD) and Conejo creek (CON) drain into the lower reaches of the Temash River and a large *R. mangle* estuary. Upstream sampling sites (e.g., YXL01, CRS01, SWD01, CON01) were selected during the dry season and as such represent the

farthest upstream reach that retained water during the dry season. Downstream sites in YXL and CRS (YXL05 and CRS05) are close to the confluence with the Temash River. Downstream sites in SWD and CON (e.g., SWD04 and CON04) were selected to include the nearest village within the catchment areas.

A geographic information system (ArcView and ArcInfo, ESRI, Redlands, CA) was used to delineate watershed boundaries and the catchment areas for each sampling point. Land cover percentages within each watershed were derived from the 1:250,000 Belize Ecosystems Map (Meerman and Sabido 2005). The distributions of soil types within each watershed were derived from a digitized version of the 1:250,000 map of the soils of Belize (Wright et al. 1959).

Annual Rainfall Pattern

Rain gauges (Tru-Chek®, graduated 1 – 150 mm) were installed in three communities within the watershed (Crique Sarco, Lucky Strike, and Conejo), beginning in February 2007 (Figure 5.1). Rain gauges were positioned in the open, at least 5 meters from any structure. Total rainfall was monitored daily and recorded. The transition between the dry and rainy seasons in the Temash Watershed was determined using methods described in Marengo et al. (2001). A five-day running mean was calculated for the total rainfall record from each station. The onset (and end) of the rainy season must satisfy three criteria: (1) mean daily precipitation is more (less) than 4.0 mm; (2) six out of eight of the preceding (subsequent) days must have mean precipitation less (more) than 3.5 mm; (3) six out of eight subsequent (preceding) days must have mean precipitation more (less) than 4.5 mm (Marengo et al. 2001).

Sample Collection and Laboratory Analysis

Water samples were collected using a modified synoptic sampling design (Grayson et al. 1997) to achieve both spatial coverage and longitudinal sampling within each of the study watersheds. Samples were collected twice per month, except during October, between April 2007 and June 2008. Longitudinal samples within each catchment were collected on the same day and individual catchments were sampled within 1-3 days of one another. Water samples for dissolved nutrients were filtered in the field through 0.7-µm glass fiber filters and collected in 60-mL HDPE bottles. Samples for dissolved inorganic N (DIN) analyses were preserved with ~20 µL of H_2SO_4 to bring sample pH to <2. Samples for total soluble N (TSN) and P (TSP) and soluble reactive phosphorus (SRP) were not treated with acid. Whole (unfiltered) water samples for total nitrogen (TN) and total phosphorus (TP) were collected during three dry season months and four rainy season months. Samples were placed on ice in the field and stored frozen (<3 months) until analysis.

Concentrations of dissolved and total N and P were analyzed on a Technicon Autoanalyzer II with a single-channel colorimeter using standard colorimetric techniques. Dissolved inorganic nitrogen (NO_3^- + NO_2^-) was measured by cadmium reduction and SRP concentrations were analyzed using the ascorbic acid-ammonium molybdate method. Total dissolved (and whole-water) P was measured on filtered (and unfiltered) samples by ascorbic acid-ammonium molybdate colorimetry following acidic persulfate digestion. Total dissolved and whole-water N was measured by cadmium reduction, after basic persulfate digestion on filtered and unfiltered samples, respectively (Davis and Simmons 1979, APHA 1989).

Statistical Analysis

Seasonal differences (dry vs. rainy) in discharge were determined using the non-parametric Mann-Whitney U statistic (p-value ≤ 0.05). Correlations between discharge and antecedent rainfall (1-day, 7-day, 14-day, and 28-day prior) were determined using Pearson's product moment. Relationships between discharge and watershed area in each study watershed were determined using linear regression.

Seasonal differences (dry vs. rainy) in nutrient concentrations at the upstream and downstream sampling sites were assessed using t-tests (p-value ≤ 0.05). For cases when the nutrient concentrations were not normally distributed, the non-parametric Mann-Whitney U statistic (p-value ≤ 0.05) was used to test for significant differences between seasons. Within-stream differences between the upstream and downstream sampling sites was also assessed for the dry season and rainy season using the same test parameters.

The calculation of flow-weighted mean concentrations (FWMC) does not provide an error term. However, the variance of a weighted average can be estimated using Equation 5-1:

$$Var(\overline{\bar{x}_w}) = \frac{\sigma^2 \sum_{i=1}^{n} w_i^2}{(\sum_{i=1}^{n} w_i)^2} \tag{5-1}$$

The standard deviation of x (σ) is derived from the in-stream nutrient data and *w* is the weighting factor, in this case the flow (Q). This estimate of variance for the weighted mean nutrient concentrations is used to calculate a standard deviation of the FWMC ($\sigma = \sqrt{Var(\bar{x})}$) for statistical comparisons of FWMC from the four study watersheds.

Discharge, Nutrient Fluxes and Flow-Weighted Mean Nutrient Concentrations

Instantaneous discharge (m^3 s^{-1}) at each sampling site was estimated during each sampling event using the float method (Gore 2006). Mean surface flow velocity (m s^{-1}) was measured using a surface float and adjusted (multiplied by 0.85) to estimate mean velocity. Between two and four floats were used to estimate mean velocity. The mean velocity was then multiplied by the cross sectional area of the stream channel (m^2) at the time of sampling to calculate instantaneous discharge rate (m^3 s^{-1}) (Gore 2006). Instantaneous nutrient fluxes were calculated for each sampling event by multiplying the instantaneous discharge (m^3 s^{-1}) by the nutrient concentration (mg L^{-1} = g m^{-3}) to obtain g s^{-1}, and ultimately expressed as load (kg day^{-1}). Instantaneous discharge rates and instantaneous nutrient fluxes were assumed to represent the average discharge and flux for each sampling interval (~15 days or 30 days in months when only a single sample was collected). Annual nutrient fluxes (kg ha^{-1} yr^{-1}) from each study watershed were then calculated by multiplying instantaneous fluxes (kg day^{-1}) by the number of days in each sampling interval, summing the resulting nutrient load (kg) over the year, and dividing by watershed area (ha). Seasonal (dry and wet) and annual flow-weighted mean nutrient concentrations (mg/L), calculated by dividing the sum of the seasonal (or annual) nutrient loads by the total seasonal (or annual) discharge rate, were used to examine water quality patterns independent of discharge.

Results

Precipitation, Watershed Characteristics, and Seasonal Discharges

The rainy season began on June 4 and the dry season started on December 19 in 2007 (Figure 5-2). The 2007-2008 dry season lasted until May 20, 2008. Total annual precipitation (rainy season 2007 to onset of rainy season 2008) at each of the three

stations was >3600 mm, and >80% of the total annual rainfall fell during the rainy season. Although several large rainfall events occurred during Dec 2007 – Feb 2008, these events were preceded and followed by extended periods of rain-free days (Figure 5-2), thus none of this time period qualified as “rainy season” under the established rainfall criteria. At the Conejo rain gauge station, there was a brief period from November 4 – 18, 2007 that satisfied the criteria for onset of the dry season. This period, however, was preceded and followed by a period of 18 days when the 5-day moving average precipitation did not fall below 4 mm, and consequently was not used to mark the onset of the dry season.

The four study watersheds differ in size, geology, soils, and to a degree, vegetation cover (Table 5-1). Crique Sarco creek (CRS) has a mean stream channel slope of 6.6% and the basement geology is predominantly clastic material. Soils are young (cambisols) and shallow (leptosols). Yax Cal creek (YXL) is the smallest of the 4 study catchments and has a mean stream channel slope of 2.9%. The bedrock of YXL is entirely clastic sedimentary rock and soils are almost entirely young cambisols. Sunday Wood creek (SWD) is the largest of the four study catchments and has a mean channel slope of 5.4% and its bedrock includes alluvial, clastic, and limestone deposits. The soils are predominantly fluvial in origin. Conejo creek (CON) has a mean channel slope of 2.9%. Conejo creek bedrock is predominantly clastic sedimentary rock with some alluvial deposits. Soils are young and of fluvial origin. Land cover in the four watersheds is dominated by agricultural lands. Yax Cal and Crique Sarco creeks have the highest overall percent land cover as agriculture (77% and 75%, respectively). Agricultural lands in Sunday Wood and Conejo creeks compromise approximately 60%

of the total land cover (Table 5-1). The only other dominant land cover in the study catchments includes combined semi-deciduous and evergreen tropical forest.

Discharge varied across watersheds and between sites within watersheds. Mean dry season instantaneous discharge from each watershed ranged from 0.015 ± 0.04 m^3 s^{-1} (YXL) to 0.31 ± 0.24 m^3 s^{-1} (CRS). Wet season instantaneous discharge was greatest in CRS (0.66 ± 0.83 m^3 s^{-1}). Dry season and wet season discharge rates were significantly different across all sampling sites with the exception of CRS05, YXL01 and YXL05 (Table 5-2). Absolute discharge rates generally increased from upstream sites to downstream sites within each watershed (Table 5-2, Figure 5-3).

Mean instantaneous discharge rates were correlated with antecedent rainfall, although the strength of the correlation with different periods of antecedent rainfall (e.g., 1 day, 7-days, 14-days, and 28-days prior) varied across each watershed. Discharge rates in CRS had a strong correlation (r=0.73) with 1-day prior rainfall. The correlation between discharge rate and 1-day prior rainfall was strongest relative to other correlation coefficients for SWD, but the correlation was weak (r=0.52). YXL discharge was strongly correlated with 14-day prior rainfall (r=0.68) while CON discharge was correlated with 28-day prior rainfall (r=0.62) (Table 5-3). Mean seasonal discharge rates were also significantly related to watershed area (Figure 5-4). To remove the effect of watershed area, discharge rates were normalized by watershed area (Figure 5-3). When normalized for watershed area, the general trend towards increasing discharge rates downstream was maintained, with the exception of the most upstream sites in YXL and SWD (Figure 5-3).

Seasonal and Longitudinal Variation of In-stream Nutrient Concentrations

Mean nutrient concentrations for each sampling site within each study watershed are shown in Table 5-4 (N species) and Table 5-5 (P species). In the upper reaches of CRS (CRS01), TN concentrations during the dry season were higher than in the wet season (U=0.0, P=0.024). Seasonal differences in DIN, TSN, SRP, TSP, and TP were not statistically significant (P>0.05) (Table 5-4, 5-5). In the lower reaches of CRS (CRS05), TSN concentrations were significantly higher during the dry season (U=18.5, P=0.020). During the dry season, nutrient concentrations at upstream and downstream sites were not statistically different in CRS. During the rainy season, SRP and TSP concentrations at CRS05 were greater than at CRS01 (t = -2.824, P=0.010 (SRP); U=16.5, P=0.005(TSP)). No other differences between upstream and downstream wet season nutrient concentrations in CRS were observed.

In YXL, no seasonal differences in N-species were observed at the upstream sampling site (YXL01). However, dry season concentrations of SRP (t = 4.969, P<0.001), TSP (t = 3.669, P=0.001) and TP (t = 4.687, P<0.001) were all higher than wet season concentrations at YXL01. These seasonal differences were not observed at the downstream site, with only dry season TSN concentrations being higher than wet season TSN concentrations at YXL05 (U=28, P=0.038). During the dry season, DIN and TSN concentrations were greater at YXL05 than at YXL01 (U=37, P=0.046; U=29, P=0.014, respectively) while the opposite occurred with P nutrients. Dry season concentrations at YXL01 of SRP (U=12, P<0.001), TSP (U=12, P<0.001) and TP (U=7, P=0.026) were all greater than at YXL05. No differences in nutrient concentrations between the upstream and downstream site were observed during the rainy season.

Nutrient concentrations in SWD varied little between the seasons. At SWD01, dry season TN was greater than wet season TN (U=1.0, P=0.048). No other seasonal differences in nutrient concentrations were observed at the upstream or downstream sites. In addition, no longitudinal differences in nutrient concentrations were observed during the wet or dry season (Table 5-4, 5-5). In CON, dry season TN concentrations were greater than wet season concentrations at the downstream site (U=0.0, P=0.017). Differences between upstream and downstream nutrient concentrations were only observed during the wet season with TP being greater at the downstream site (t = -3.50, P=0.025). No other longitudinal differences in nutrient concentrations were observed (Table 5-4, 5-5).

The contribution of DIN to the TSN pool was highly variable across seasons and across study watersheds. DIN comprised approximately 58% of both the dry season and rainy season TSN pool at both upstream and downstream sites. TSN comprised approximately 60% of the TN pool on average during the dry season and contributed more than 85% during the wet season. In YXL the average contribution of DIN to the TSN pool was between 64% and 88%. The TN pool in YXL averaged more than 60% TSN during the dry season and during the rainy season included more than 90% TSN. In SWD, DIN comprised approximate 50% of the TSN pool during the dry season and more than 75% during the rainy season, with a similar pattern of DIN:TSN observed in CON. During the rainy season in both SWD and CON, DIN contributed more than 70% to the TSN pool on average. TSN comprised approximately 90% of the TN pool during the rainy season in SWD and averaged between 73% and 100% in CON.

In CRS, SRP comprised between 17% and 46% of TSP, on average, during the dry season. During the wet season, SRP averaged between 30% and 44% of TSP in CRS. TSP contributed a larger portion to the TP pool in CRS, with SRP:TSP averaging between 50% and 90%. During the dry season in YXL, SRP contributed more than 85% to the TSP pool at the upstream site (YXL01). Lower ratios of SRP:TSP were observed downstream. Dry season average contributions of TSP to the TN pool were similar to SRP:TSP while rainy season TSP comprised more than 90% of the TP pool. In SWD, SRP averaged between 30% and 53% of the TSP pool, regardless of season. The TSP contribution to the TP pool averaged between 38% and 63% during the dry season and between 75% and 97% during the rainy season. Dry season and wet season SRP:TSP ratios were similar in CON, averaging between 32% and 55%. During the dry season, TSP contributions to the TP pool averaged between 30% and 53% while the contribution averaged between 58% and 76% during the rainy season.

Ratios of TN:TP at upstream and downstream sites within each watershed are shown in Table 5-6. Dry season TN:TP values are generally higher than the wet season value at the same site although only the ratios at the downstream site in CON (CON04) and the downstream site in SWD (SWD04) were significantly different ($t=2.946$, $P=0.022$ and $t=2.484$, $P=0.048$, respectively). Overall, TN:TP ratios suggest waters are P-limited across all sites, though large standard deviations during the rainy season in YXL, SWD, and CON reflect reduced N concentrations during periods of high flow (see also Table 5-4).

Nutrient Fluxes and Flow Weighted Mean Nutrient Concentrations

Nutrient fluxes (kg/day) varied over time and across catchments within the Temash River (Figures 5-5 and 5-6). In all catchments, daily loads were low at the end

of the 2007 dry season. An increase, or pulse in daily N flux was observed at the onset of the 2007 rainy season, although its timing and duration varied across catchments. In CRS, the first increase in N flux, from <1 kg/day to almost 10 kg/day, was observed between the June and July sampling period (Figure 5-5). A second, much larger pulse of N occurred in CRS later in the rainy season, at the end of August. Subsequent N fluxes remained low throughout the rainy season in CRS (Figure 5-5). In YXL, a pulse of N was observed during June and persisted through the July sampling period. In SWD, N fluxes were low throughout the 2007 sampling period (Figure 5-5). In Conejo creek, the initial pulse of N was observed in June 2007, declined in July 2007, and then remained low throughout the year. The final sampling event (June 2008) occurred within 3 days of the onset of the 2008 rainy season and all study catchments showed a large change in nutrient flux at that time.

Daily phosphorus loads were low across all study watersheds, rarely exceeding 0.5 kg/day (Figure 5-6). A small increase was observed following the onset of the rainy season (late June 2007) in CRS, YXL, and CON. A large pulse of TP, TSP, and SRP was observed in CRS in late August, with a TP flux of ~2.0 kg/day. A series of sampling events in CRS after the onset of the dry season (January 2008) captured additional fluxes of P. P fluxes remained low through the dry season with the onset of the rainy season in June 2008 showing a small increase in P flux occurring at the end of the sampling (Figure 5-6) Flow weighted mean (FWM) concentrations for dissolved N (inorganic and total) varied little seasonally within catchments or between catchments (Figure 5-7). In CRS, no statistically significant difference between wet and dry FWM concentrations of DIN, TSN, or TN were detected. In YXL, dry season FWM TSN

concentrations were higher than wet season FWM concentrations (t = -2.154, P=0.049). Dry season FWM TN was also higher than wet season FWM TN in YXL (t = -3.889, P=0.012). In SWD, FWM TN was also higher in the dry season than the wet season (t = -5.516, P=0.005). There were no seasonal differences in FWM concentrations of N-species in CON (Figure 5-7). Flow weighted mean concentrations of P across all sampling sites were low and there were no significant differences between season P concentrations (Figure 5-8). Concentrations were generally below 0.001 mg L^{-1} in all catchments and samples included high variability, evident in the large standard deviations for each estimated FWMC (Figure 5-8).

Between-catchment differences in the annual FWMC were observed for some nutrients. The FWM DIN in CON was greater than the FWM DIN in YXL (t = -2.070, P=0.046) and SWD (t = -2.690, P=0.011). The FWM TSN in CON was also greater than the FWM TSN in SWD (t = -3.034, P=0.005). In YXL, the FWM TSN was greater than the FWM TSN in SWD as well (t=2.298, P=0.028). No other differences in total annual FWM nutrient concentrations were observed between catchments (Figures 5-7 and 5-8).

Relationships between annual flow weighted mean concentrations and land cover and soil classes were determined by linear regression (Table 5-7, 5-8). Flow weighted means from each of the 19 sample sites were regressed against percent land cover and percent soil class cover. Outliers were determined using the studentized residual and Cook's D statistic to determine degree of influence of a particular variable. FWM DIN concentrations were positively related to % silty clay alluvium (soil class 28) although the coefficient of determination was weak (r^2 = 0.28). FWM TSN was also positively

related to soil class 28, although again the relation was weak (r^2 = 0.28). When outliers were removed, TSN was negatively related to soil class 49 (sandy alluvium). Flow weighted mean concentrations of TN were positively related to % forest cover and negatively related to % agriculture. The strongest predictors of TN were % coastal loamy sand (soil class 24; r^2 = 0.56) and % sandy loam (soil class 49; r^2 = 0.47).

Land cover and catchment soils were generally weak predictors of FWM phosphorus concentrations (Table 5-8). SRP, TSP, and TP were positively related to % forest cover and negatively related to % agriculture in catchments, although the coefficients of determination (r^2) were weak. TSP was also positively related to the % clay soils in the study watersheds. (Table 5-8).

Discussion

After forests are cleared for milpa, decaying organic matter decomposes and remineralized nitrogen is able to enter the soil (Williams et al. 1997). The capacity of the soil to absorb this nitrogen is reduced because of the felling of trees and the dying of root systems. Consequently, leached nitrate accumulates and moves downslope. At the onset of the rainy season, this soil nitrate can be released as a pulse into adjacent waterbodies (Downing et al. 1999). The capacity of the riparian zone to minimize the amount of nitrate entering adjacent waterbodies via denitrification depends on the extent of land clearing and the presence of organic-rich soils (McClain et al. 1994). The burning of slash also leads to a pulse of dissolved N, able to reach adjacent waterbodies via subsurface flow. Although much of the available N in slash is released into the atmosphere during the burning, a portion of it will settle on the landscape. As the cycle of slash and burn continues, areas repeatedly used for slash-and-burn will

have reduced export of dissolved nitrogen because of a gradual decrease in N stocks (Downing et al. 1999).

In the four study watersheds, this initial pulse of N was observed at the onset of the rainy season in three of the four study watersheds and was dominated by dissolved N. Following this initial pulse, daily loads of N remained low throughout the rainy season (Figure 5-5). Concentrations also remain low well into the dry season. The dry season is a period of peak leaf litter fall in much of the tropics and leaf litter decomposition can contribute to N immobilization in both terrestrial and aquatic ecosystems (McDowell and Asbury 1994). These low N concentrations during the dry season differ from watersheds dominated by large-scale agriculture in that base flow conditions can often be associated with increases in dissolved N because of the increased contribution of groundwater to stream hydrology (Kemp and Dodds 2002).

The nitrogen/phosphorus ratios indicate the study watersheds are all P-limited. Phosphorus concentrations at all sites (except YXL01) were measured at or near the detection limit of the laboratory instrumentation. There is a weak correlation between TSP and clay soils in the study area. It is possible that any available P is derived from clay soils where P is adsorbed to metal oxides (Pacini et al. 2008). In YXL01 and CON01 higher SRP concentrations were observed during the dry season (Table 5-5). It is possible that P is released from the mineralization of accumulated organic matter during the dry season (Saunders et al. 2006).

The four study watersheds exhibited different patterns of N and P export (Table 5-9). CRS and CON exported the largest volume of N, relative to the other study watersheds. In CRS, DIN comprised approximately 46% of the TSN exported while in

CON, DIN accounted for more than 80% of the dissolved N exported from the watershed (Table 5-9). YXL and SWD both had low TSN export coefficients (0.53 and 0.18, respectively). Dissolved inorganic N comprised approximately 50% of the TSN exported from YXL while DIN accounted for ~65% of the TSN exported from SWD (Table 5-9). Phosphorus export was low across all study watersheds (Table 5-10). Total soluble P export was lowest from SWD and highest from CRS. Soluble reactive P contributed between 33% and 75% of the TSP exported.

When compared to other tropical watersheds, the export coefficients presented here are lower. This is most notable in YXL and SWD, where export coefficients for DIN and TSN are at least one order of magnitude less than other tropical systems (Table 5-9, 5-10). Overall, nutrient export from the study watersheds is likely underestimated because of the method used (Table 5-9). Daily and monthly loads of nutrients were derived from only one or two samples and assumed to be representative of the sampling interval. Discharge in these watersheds is correlated with rainfall (Table 5-3) and sampling was not conducted to develop a stronger estimate of export during storm events. It is likely that a large percentage of the annual export occurs during a few single events at the onset of the rainy season and finer sampling intervals would capture some of this export (McDowell and Asbury 1994)

Table 5-1. Catchment characteristics of the four study watersheds and the Temash River watershed (all data are for the catchment area of the farthest downstream sampling site).

	Crique Sarco (CRS 05)	Yax Cal (YXL 05)	Sunday Wood (SWD 04)	Conejo (CON 04)
area (ha)	1084	469	3204	1384
mean elevation (m)	53	26	41	28
mean slope (%)	6.6	2.9	5.4	2.9
Geology (%)				
Alluvial	0	0	60	27
Limestone	23	0	15	0
Clastic Sedimentary	77	100	25	73
Soils (%)				
Cambisol	79	95	30	52
Fluvisol	0	5	61	48
Leptosol	11	0	9	0
Leptosol-vertisol	10	0	0	0
Land Cover (%)				
Semi-decid./evergreen Forest	23	23	40	36
Lowland Swamp Forest	0	0	0	5
Agricultural Lands	75	77	60	59
Waterbody	2	0	0	0

Table 5-2. Comparison of dry and wet season discharge rates ($m^3 s^{-1}$) across all sampling sites (Mann Whitney U statistic, P<0.05)

Sampling Site	Dry Season Discharge ($m^3 sec^{-1}$)			Wet Season Discharge ($m^3 sec^{-1}$)				
	median	25%	75%	median	25%	75%	U-statistic	P-value
CRS01	0.047	0.017	0.248	0.353	0.088	0.642	15.0	0.037
CRS02	0.060	0.000	0.263	0.446	0.167	0.970	10.0	0.017
CRS03	0.123	0.000	0.234	0.680	0.257	1.541	10.0	0.017
CRS04	0.181	0.047	0.257	0.383	0.171	0.859	14.0	0.045
CRS05	0.288	0.116	0.539	0.365	0.142	0.926	23.5	0.613
YXL01	0.095	0.036	0.199	0.210	0.155	0.291	15.0	0.079
YXL02	0.037	0.019	0.088	0.148	0.056	0.246	16.0	0.046
YXL03	0.053	0.033	0.113	0.236	0.089	0.403	8.0	0.007
YXL04	0.088	0.035	0.133	0.382	0.137	0.716	8.0	0.007
YXL05	0.000	0.000	0.000	4.2×10^5	0.000	0.222	17.0	0.232
SWD01	0.000	0.000	0.000	0.031	0.000	0.080	13.5	0.029
SWD02	0.000	0.000	0.000	0.135	0.010	0.264	6.0	0.013
SWD03	0.019	0.011	0.083	0.250	0.100	0.299	7.0	0.006
SWD04	0.031	0.006	0.043	0.293	0.169	0.316	2.0	0.002
CON01	0.000	0.000	0.006	0.059	0.017	0.155	9.5	0.005
CON02	0.009	0.000	0.025	0.124	0.057	0.280	7.0	0.006
CON03	0.000	0.000	0.000	0.260	0.129	0.406	5.0	0.002
CON04	0.000	0.000	0.000	0.380	0.076	0.844	4.0	0.001

Table 5.3. Pearson's product moment correlation coefficients for instantaneous discharge and antecedent rainfall†

	Antecedent Rainfall			
Catchment	1 day	7 day	14 day	28 day
CRS*	0.73	0.53	0.41	0.39
YXL*	0.48	0.63	0.68	0.33
SWD^	0.52	0.34	0.30	0.26
CON^	0.47	0.61	0.59	0.62

† all correlations are significant at $P < 0.01$

*CRS and YXL correlated with the CRS rain gauge station

^SWD and CON correlated with the CON rain gauge station

Table 5-4. Seasonal mean nutrient concentrations for DIN, TSN, and TN for all sampling sites within each study watershed. Values are mean ± 1 standard deviation (sample size in parenthesis).

	DIN (mg/L)		TSN (mg/L)		TN (mg/L)	
	dry	wet	dry	wet	dry	wet
Crique Sarco						
CRS01	0.276 ± 0.442 (10)	0.142 ± 0.269 (13)	0.391 ± 0.401 (10)	0.207 ± 0.267 (13)	0.586 ± 0.298 (6)	0.097 ± 0.063 (3)
CRS02	0.291 ± 0.443 (11)	0.159 ± 0.342 (11)	0.351 ± 0.416 (11)	0.219 ± 0.330 (11)	0.496 ± 0.377 (6)	0.106 ± 0.065 (2)
CRS03	0.292 ± 0.452 (11)	0.178 ± 0.360 (11)	0.355 ± 0.421 (11)	0.218 ± 0.350 (11)	0.513 ± 0.364 (6)	0.108 ± 0.079 (3)
CRS04	0.270 ± 00.407 (12)	0.169 ± 0.355 (11)	0.406 ± 0.386 (11)	0.204 ± 0.346 (11)	0.517 ± 0.323 (7)	0.093 ± 0.051 (3)
CRS05	0.329 ± 0.434 (11)	0.218 ± 0.457 (9)	0.507 ± 0.374 (11)	0.275 ± 0.437 (9)	0.602 ± 0.275 (6)	0.124 ± 0.038 (2)
Yax Cal						
YXL01	0.154 ± 0.259 (12)	0.214 ± 0.511 (12)	0.179 ± 0.252 (12)	0.225 ± 0.508 (12)	0.378 ± 0.401 (7)	0.058 ± 0.026 (3)
YXL02	0.227 ± 0.374 (12)	0.204 ± 0.309 (12)	0.255 ± 0.360 (12)	0.227 ± 0.305 (12)	0.405 ± 0.408 (7)	0.040 ± 0.027 (3)
YXL03	0.216 ± 0.428 (12)	0.205 ± 0.293 (12)	0.314 ± 0.408 (12)	0.220 ± 0.296 (12)	0.446 ± 0.371 (7)	0.047 ± 0.040 (3)
YXL04	0.278 ± 0.483 (12)	0.191 ± 0.291 (12)	0.334 ± 0.459 (12)	0.222 ± 0.288 (12)	0.432 ± 0.401 (7)	0.047 ± 0.023 (3)
YXL05	0.272 ± 0.399 (12)	0.170 ± 0.315 (10)	0.357 ± 0.366 (12)	0.201 ± 0.306 (10)	0.491 ± 0.342 (7)	0.082 ± 0.055 (3)
Sunday Wood						
SWD01	0.120 ± 0.204 (11)	0.161 ± 0.401 (10)	0.143 ± 0.187 (11)	0.183 ± 0.394 (10)	0.424 ± 0.405 (6)	0.027 ± 0.009 (3)
SWD02	0.118 ± 0.190 (10)	0.163 ± 0.369 (11)	0.240 ± .0230 (10)	0.177 ± 0.364 (11)	0.431 ± 0.395 (6)	0.064 ± 0.052 (3)
SWD03	0.121 ± 0.198 (11)	0.219 ± 0.480 (11)	0.308 ± 0.163 (10)	0.246 ± 0.472 (11)	0.522 ± 0.300 (7)	0.057 ± 0.037 (3)
SWD04	0.143 ± 0.219 (11)	0.286 ± 0.630 (9)	0.213 ± 0.192 (11)	0.306 ± 0.623 (9)	0.494 ± 0.341 (6)	0.063 ± 0.039 (2)
Conejo						
CON01	0.178 ± 0.302 (11)	0.381 ± 0.723 (11)	0.353 ± .0477 (11)	0.402 ± 0.714 (11)	0.556 ± 0.369 (6)	0.071 ± 0.033 (3)
CON02	0.123 ± 0.191 (11)	0.932 ± 0.764 (10)	0.216 ± 0.207 (10)	0.432 ± 0.760 (10)	0.435 ± 0.348 (5)	0.087 ± 0.050 (3)
CON03	0.0117 ± 0.0171 (11)	0.313 ± 0.465 (10)	0.226 ± 0.181 (11)	0.348 ± 0.482 (10)	0.556 ± 0.328 (6)	0.080 ± 0.020 (3)
CON04	0.128 ± 0.176 (11)	0.303 ± 0.450 (10)	0.280 ± 0.167 (11)	0.338 ± 0453 (10)	0.636 ± 0.258 (6)	0.086 ±0.028 (3)

Table 5-5. Seasonal mean nutrient concentrations for SRP, TSP, and TP for all sampling sites within each study watershed. Values are mean ± 1 standard deviation (sample size in parenthesis).

	SRP (mg/L)		TSP (mg/L)		TP (mg/L)	
	dry	wet	dry	wet	dry	wet
Crique Sarco						
CRS01	0.001 ± 0.002 (9)	0.001 ± 0.001 (13)	0.004 ± 0.005 (10)	0.003 ± 0.003 (13)	0.009 ± 0.004 (6)	0.006 ± 0.003 (3)
CRS02	0.002 ± 0.002 (10)	0.002 ± 0.003 (11)	0.004 ± 0.003 (11)	0.005 ± 0.003 (11)	0.009 ± 0.003 (6)	0.010 ± 0.003 (2)
CRS03	0.001 ± 0.001 (10)	0.001 ± 0.001 (11)	0.003 ± 0.002 (11)	0.005 ± 0.005 (11)	0.007 ± 0.003 (3)	0.007 ± 0.003 (3)
CRS04	0.004 ± 0.004 (10)	0.005 ± 0.008 (11)	0.006 ± 0.004 (11)	0.009 ± 0.007 (11)	0.009 ± 0.004 (7)	0.015 ± 0.012 (3)
CRS05	0.005 ± 0.006 (10)	0.002 ± 0.001 (9)	0.010 ± 0.010 (11)	0.006 ± 0.003 (9)	0.023 ± 0.017 (6)	0.008 ± 0.001 (2)
Yax Cal						
YXL01	0.019 ± 0.008 (12)	0.006 ± 0.006 (12)	0.022 ± 0.008 (12)	0.010 ± 0.007 (12)	0.029 ± 0.010 (7)	0.011 ± 0.010 (3)
YXL02	0.004 ± 0.002 (11)	0.004 ± 0.005 (12)	0.006 ± 0.003 (12)	0.006 ± 0.004 (12)	0.011 ± 0.004 (7)	0.010 0.008 (3)
YXL03	0.002 ± 0.002 (11)	0.001 ± 0.001 (12)	0.006 ± 0.005 (12)	0.004 ± 0.002 (12)	0.011 ± 0.005 (7)	0.004 ± 0.002 (3)
YXL04	0.001 ± 0.001 (11)	0.001 ± 0.001 (12)	0.004 ± 0.003 (12)	0.004 ± 0.001 (12)	0.008 ± 0.003 (7)	0.005 ± 0.002 (3)
YXL05	0.005 ± 0.014 (11)	0.005 ± 0.014 (10)	0.007 ± 0.013 (12)	0.011 ± 0.015 (10)	0.011 ± 0.017 (7)	0.018 ± 0.023 (3)
Sunday Wood						
SWD01	0.002 ± 0.003 (10)	0.002 ± 0.002 (10)	0.004 ± 0.005 (10)	0.005 ± 0.003 (10)	0.009 ± 0.006 (6)	0.007 ± 0.003 (3)
SWD02	0.011 ± 0.022 (9)	0.002 ± 0.001 (10)	0.012 ± 0.025 (10)	0.003 ± 0.002 (11)	0.009 ± 0.012 (6)	0.006 ± 0.002 (3)
SWD03	0.007 ± 0.008 (9)	0.002 ± 0.002 (11)	0.014 ± 0.018 (10)	0.007 ± 0.004 (11)	0.020 ± 0.020 (7)	0.008 ± 0.003 (3)
SWD04	0.001 ± 0.001 (10)	0.002 ± 0.002 (9)	0.005 ± 0.002 (11)	0.005 ± 0.003 (9)	0.010 ± 0.005 (6)	0.009 ± 0.005 (2)
Conejo						
CON01	0.010 ± 0.019 (10)	0.004 ± 0.009 (11)	0.012 ± 0.020 (11)	0.005 ± 0.008 (11)	0.017 ± 0.021 (6)	0.005 ± 0.001 (3)
CON02	0.001 ± 0.002 (9)	0.002 ± 0.001 (10)	0.004 ± 0.003 (10)	0.014 ± 0.033 (10)	0.006 ± 0.001 (5)	0.039 ± 0.059 (3)
CON03	0.010 ± 0.020 (10)	0.003 ± 0.002 (10)	0.013 ± 0.018 (11)	0.006 (0.003) (10)	0.028 ± 0.010 (6)	0.007 ± 0.002 (3)
CON04	0.002 ± 0.003 (10)	0.002 ± 0.002 (10)	0.005 ± 0.004 (11)	0.004 ± 0.001 (10)	0.017 ± 0.009 (6)	0.007 ± 0.001 (3)

Table 5-6. Seasonal mean (± stdev) TN:TP ratios for upstream and downstream sampling sites within each study watershed.

		TN:TP	
		Dry Season	Rainy Season
Crique Sarco			
	upstream	167 ± 115	31 ± 8
	downstream	92 ± 85	33 ± 4
Yax Cal			
	upstream	33 ± 26	18 ± 17
	downstream	248 ± 174	19 ± 14
Sunday Wood			
	upstream	106 ± 99	9 ± 1
	downstream	109 ± 45	23 ± 23
Conejo			
	upstream	105 ± 61	32 ± 12
	downstream	90 ± 36	26 ± 6

Table 5-7. Linear regression parameters for annual flow weighted DIN, TSN, and TN concentrations with watershed characteristics of land cover and soil classification as independent variables.

Regression #	Dependent variable mg L^{-1}	r^2	*p* value	Intercept	Coefficient of Independent Variable: Land Cover For.	Land Cover Agr.	Soil Class 15	Soil Class 24	Soil Class 28	Soil Class 49	Outliers
1	DIN	0.0068	0.75	0.1246	-0.0004	-	-	-	-	-	CON03; CON04
2	DIN	0.051	0.39	0.1534	-	-0.0007	-	-	-	-	CON03; CON04
3	DIN	0.0008	0.91	0.1094	-	-	-0.0001	-	-	-	CON03
4	DIN	0.0043	0.8	0.1	-	-	-	0.0001	-	-	CON03
5*	DIN	0.28	0.03	0.0948	-	-	-	-	0.0018	-	CON03
6	DIN	0.21	0.075	0.115	-	-	-	-	-	-0.0009	CON03; CON04
7	TSN	0.0073	0.74	0.174	-0.0004	-	-	-	-	-	CON03; CON04
8	TSN	0.039	0.47	0.1979	-	-0.0007	-	-	-	-	CON03; CON04
9	TSN	0	-	-	-	-	0	-	-	-	YXL01
10	TSN	0.12	0.19	0.1142	-	-	-	0.0006	-	-	CON03; CON04
11*	TSN	0.26	0.03	0.1481	-	-	-	-	0.002	-	-
12*	TSN	0.39	0.01	0.1705	-	-	-	-	-	-0.0012	CON03; CON04
13	TN	0.37	0.01	0.0628	0.0036	-	-	-	-	-	YXL05
14	TN	0.35	0.016	0.3645	-	-0.003	-	-	-	-	CRS05; YXL05
15	TN	0.14	0.156	0.123	-	-	0.0016	-	-	-	CRS05; YXL05
16*	TN	0.56	<0.001	0.0184	-	-	-	0.0021	-	-	YXL01; YXL05
17	TN	0.092	0.24	0.1675	-	-	-	-	-0.0016	-	YXL05
18*	TN	0.47	0.003	0.1958	-	-	-	-	-	-0.0023	YXL05

soil class 15 = limestone/dolomite clay; class 24 = coastal loamy sand; class 28 = silty clay alluvium; class 49 = sandy clay alluvium

(*) indicates significant linear relationship between nutrient and cland cover or soil class

Table 5-8. Linear regression parameters for annual flow weighted SRP, TSP, and TP concentrations with watershed characteristics of land cover and soil classification as independent variables.

Regression #	Dependent variable mg L^{-1}	r^2	p value	Intercept	Coefficient of Independent Variable: Land Cover For.	Land Cover Agr.	Soil Class 15	Soil Class 24	Soil Class 28	Soil Class 49	Outliers
1*	SRP	0.32	0.017	0.0008	0.0001	-	-	-	-	-	YXL05
2*	SRP	0.29	0.025	0.0069	-	-0.0001	-	-	-	-	YXL05
3	SRP	0.19	0.08	0.002	-	-	0	-	-	-	YXL05
4	SRP	0.005	0.78	0.0022	-	-	-	0	-	-	YXL05
5	SRP	0.003	0.85	0.0025	-	-	-	-	0	-	YXL05
6	SRP	0.12	0.18	0.0029	-	-	-	-	-	0	YXL05
7*	TSP	0.3	0.027	0.0026	0.0001	-	-	-	-	-	YXL05; CON02
8	TSP	0.24	0.054	0.0115	-	-0.001	-	-	-	-	YXL05; CON02
9*	TSP	0.26	0.043	0.0042	-	-	0.0001	-	-	-	YXL05; CON02
10	TSP	0.005	0.79	0.0047	-	-	-	0	-	-	YXL05; CON02
11	TSP	0.06	0.36	0.0054	-	-	-	-	0	-	YXL05; CON02
12	TSP	0.1	0.23	0.0057	-	-	-	-	-	0	YXL05; CON02
13*	TP	0.28	0.028	0.0043	0.0002	-	-	-	-	-	CON02
14*	TP	0.238	0.047	0.0226	-	-0.0002	-	-	-	-	CON02
15	TP	0.003	0.83	0.0086	-	-	0	-	-	-	YXL01; CON02
16	TP	0.038	0.46	0.0071	-	-	-	0	-	-	CON02
17	TP	0			-	-	-	-	0	-	CON01; CON02
18	TP	0.21	0.067	0.011	-	-	-	-	-	-0.0001	CON02

soil class 15 = limestone/dolomite clay; class 24 = coastal loamy sand; class 28 = silty clay alluvium; class 49 = sandy clay alluvium

(*) indicates significant linear relationship between nutrient and cland cover or soil class

Table 5-9. A comparison of nutrient export coefficients (kg ha^{-1} yr^{-1}) from tropical watersheds

Stream/River	size (ha)	land cover	DIN	TSN	TN	SRP	TSP	TP	Location/Reference
			Yield kg ha^{-1} yr^{-1}						
Crique Sarco creek	1084	75% agr	1.04	2.25		0.03	0.09		southern Belize / this study
Yax Cal creek	469	77% agr	0.28	0.53		0.03	0.04		southern Belize / this study
Sundaywood creek	3204	60% agr	0.12	0.18		0.00	0.01		southern Belize / this study
Conejo creek	1384	59% agr	2.04	2.48		0.02	0.03		southern Belize / this study
Braco do Mota	23.4	80% agr	3.64	6.44	9.14	0.08	0.33	0.48	central Amazon (A)
Igarape de Mota	18	>95% forest	2.67	3.61	4.31	0.02	0.05	0.08	central Amazon (A)
Tempisquito	319	>95% forest	6.10				0.57		Costa Rica (B)
Tempisquito Sur	311	>95% forest	4.90				0.33		Costa Rica (B)
Kathia	264	>95% forest	5.60				0.34		Costa Rica (B)
Marilin	36	>95% forest	4				0.46		Costa Rica (B)
El Jobo	55	>95% forest	4.3				0.34		Costa Rica (B)
Zompopa	37	>95% forest	6				0.43		Costa Rica (B)
Icacos	326	>95% forest	3.2	8.01	9.8		0.07		Puerto Rico (C)
Sonadora	262	>95% forest	1.69	5.43	5.9		0.05		Puerto Rico (C)
Toronja	16.2	>95% forest	1.16	3.96	4.4		0.03		Puerto Rico (C)

(A) Williams and Melack 1997; (B) Newbold et al. 1995; (C) McDowell and Asbury 1994

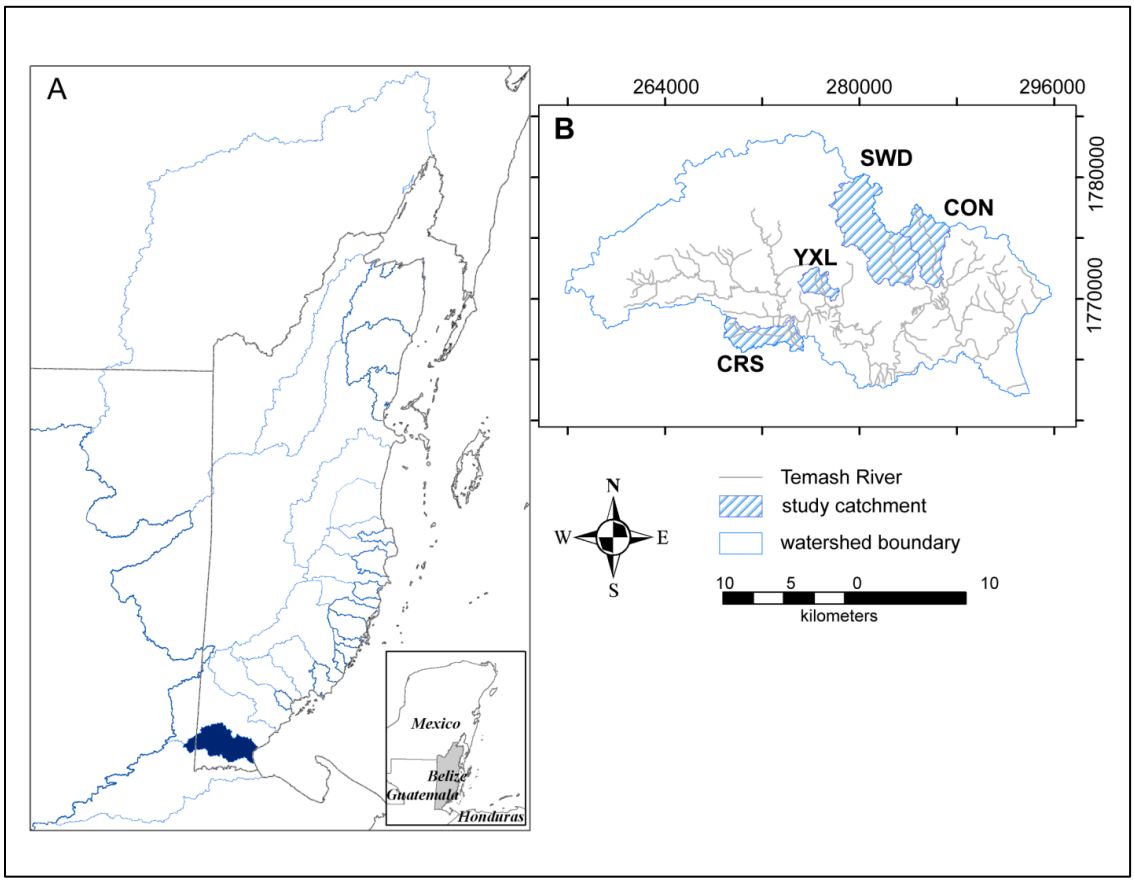

Figure 5-1. Map of study area, including (A) Belize and the location of the Temash River watershed in southern Belize. (B) Four catchments were selected for in-stream nutrient monitoring - Crique Sarco (CRS), Yax Cal (YXL), Sunday Wood (SWD), and Conejo (CON).

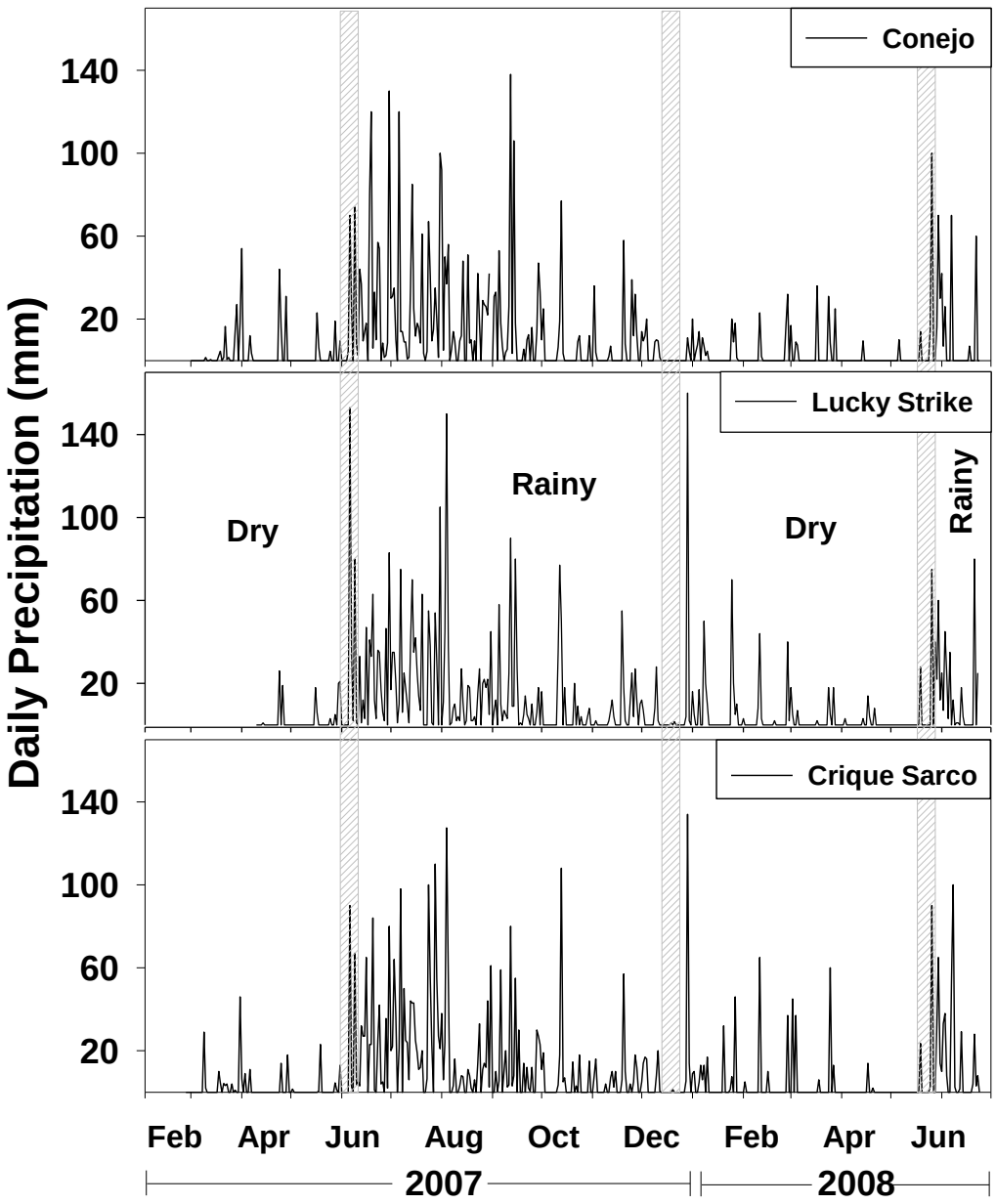

Figure 5-2. Precipitation record for three stations in the Temash River watershed from February 2, 2007 through June 24, 2008. Shaded vertical bars mark the seasonal boundaries between dry and rainy seasons.

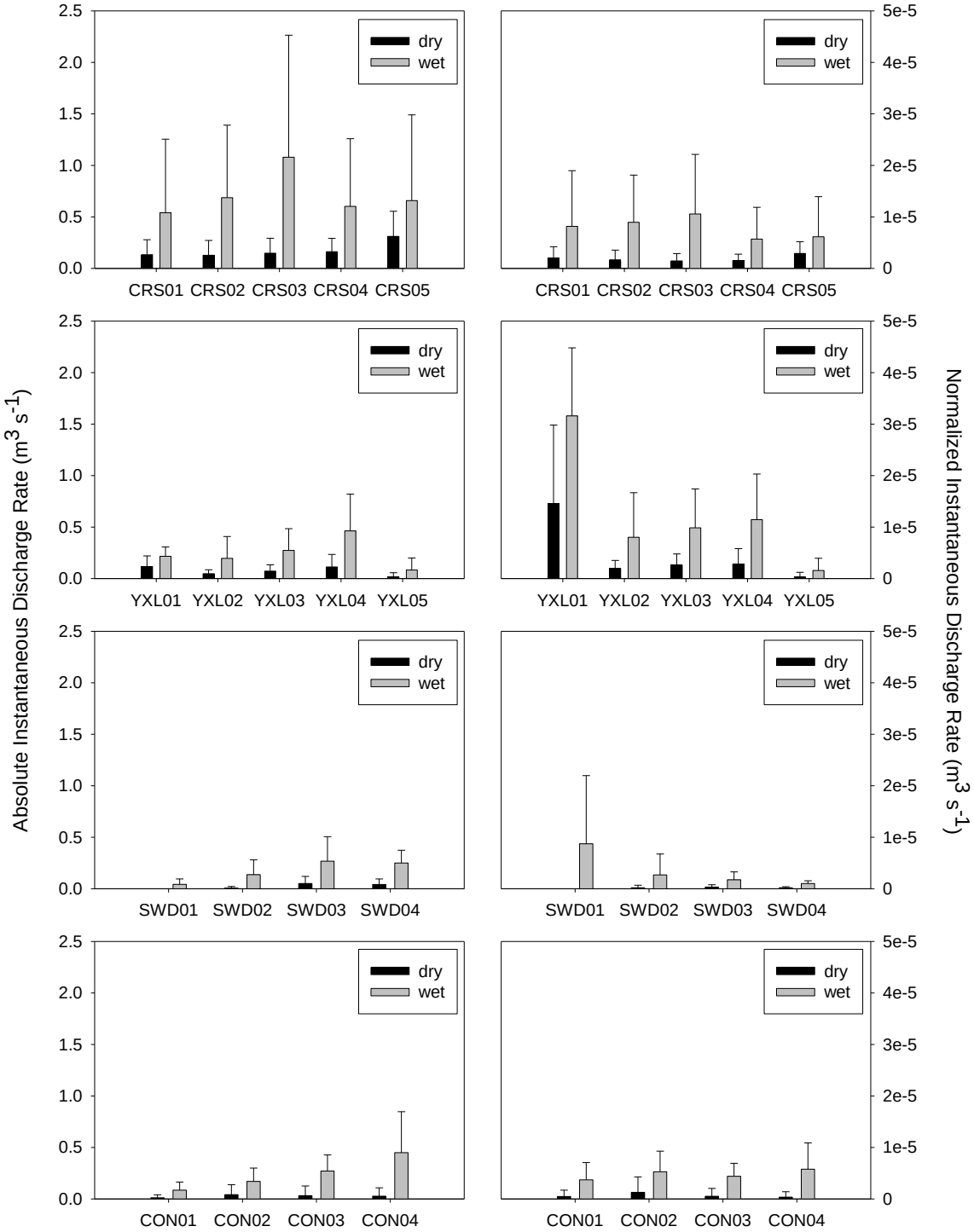

Figure 5-3. Absolute and normalized (by watershed area) discharge rates for the study watersheds.

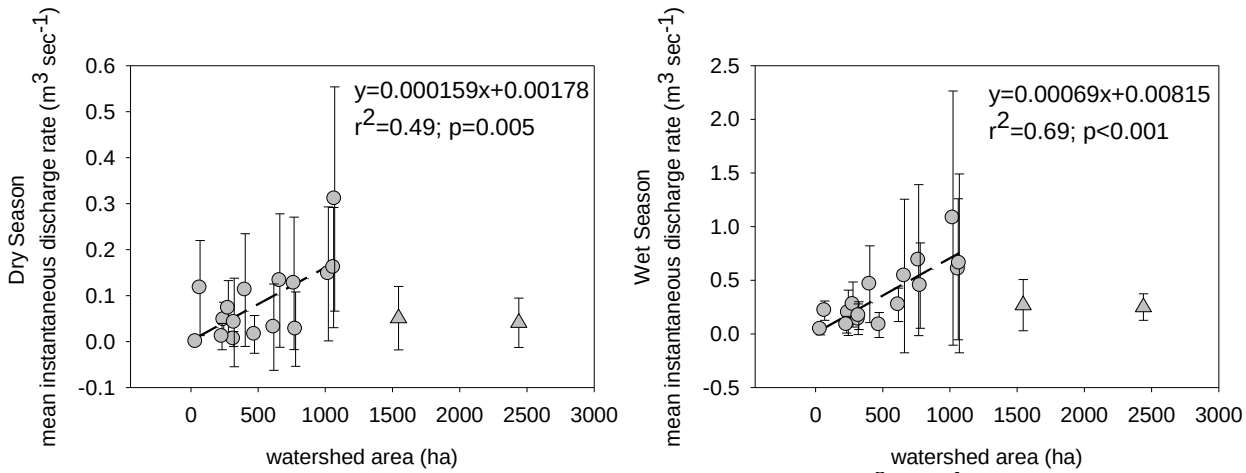

Figure 5-4. Seasonal mean instantaneous discharge rates (m^3 sec^{-1}) versus watershed area (ha). The two outliers (gray triangles) are YXL01 and SWD01.

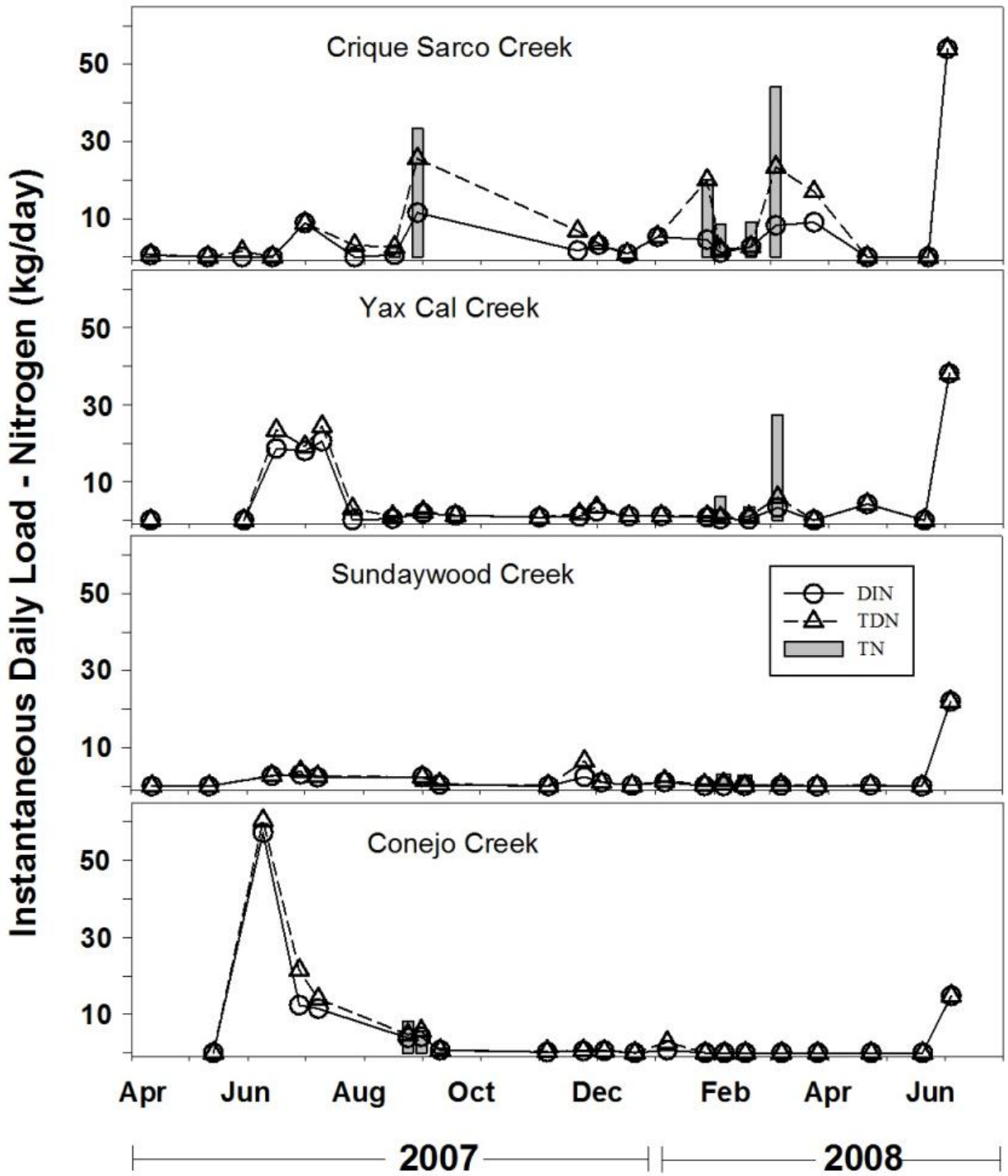

Figure 5-5. Daily nitrogen fluxes from the study watersheds

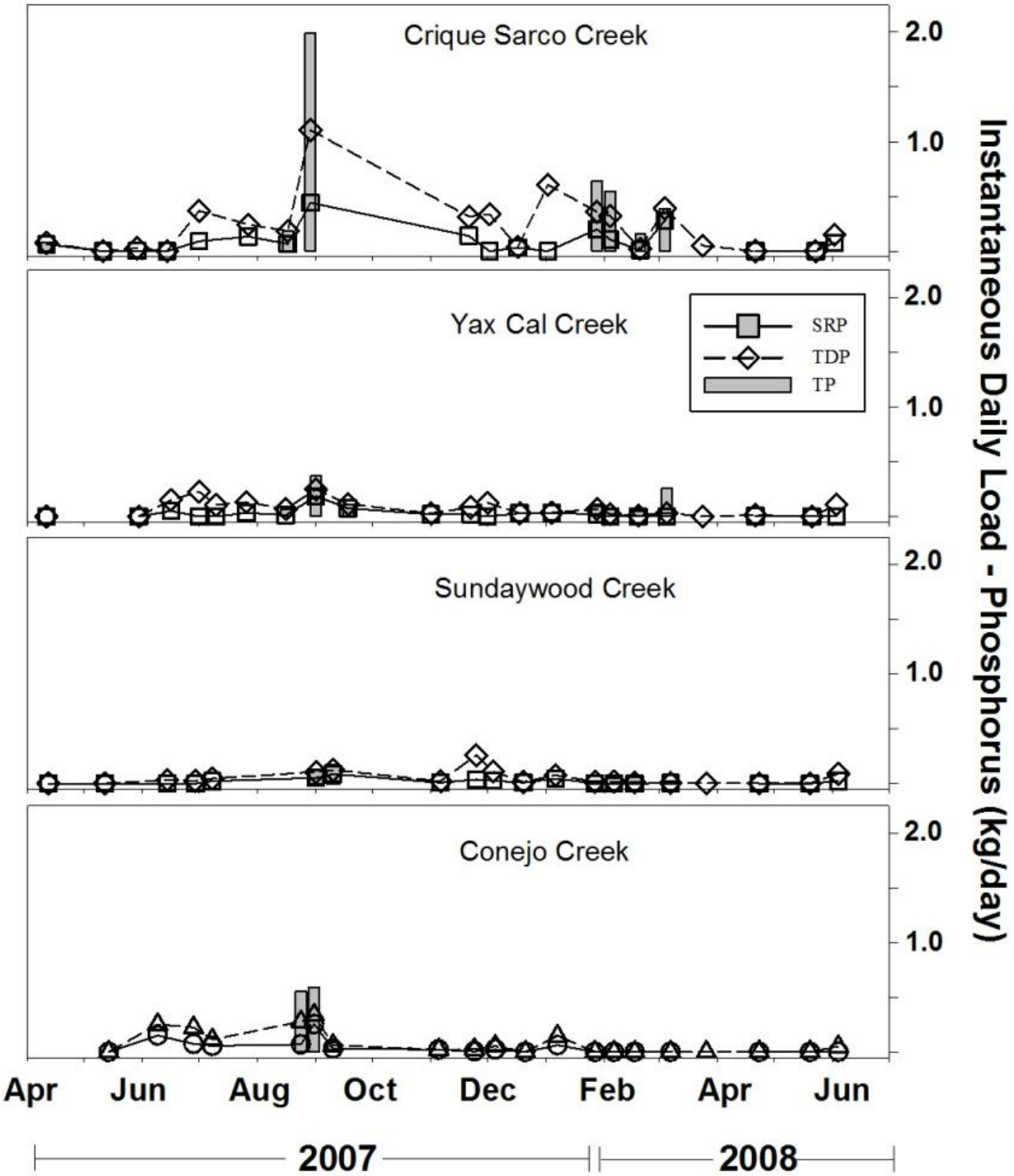

Figure 5-6. Daily phosphorus fluxes from the study watersheds

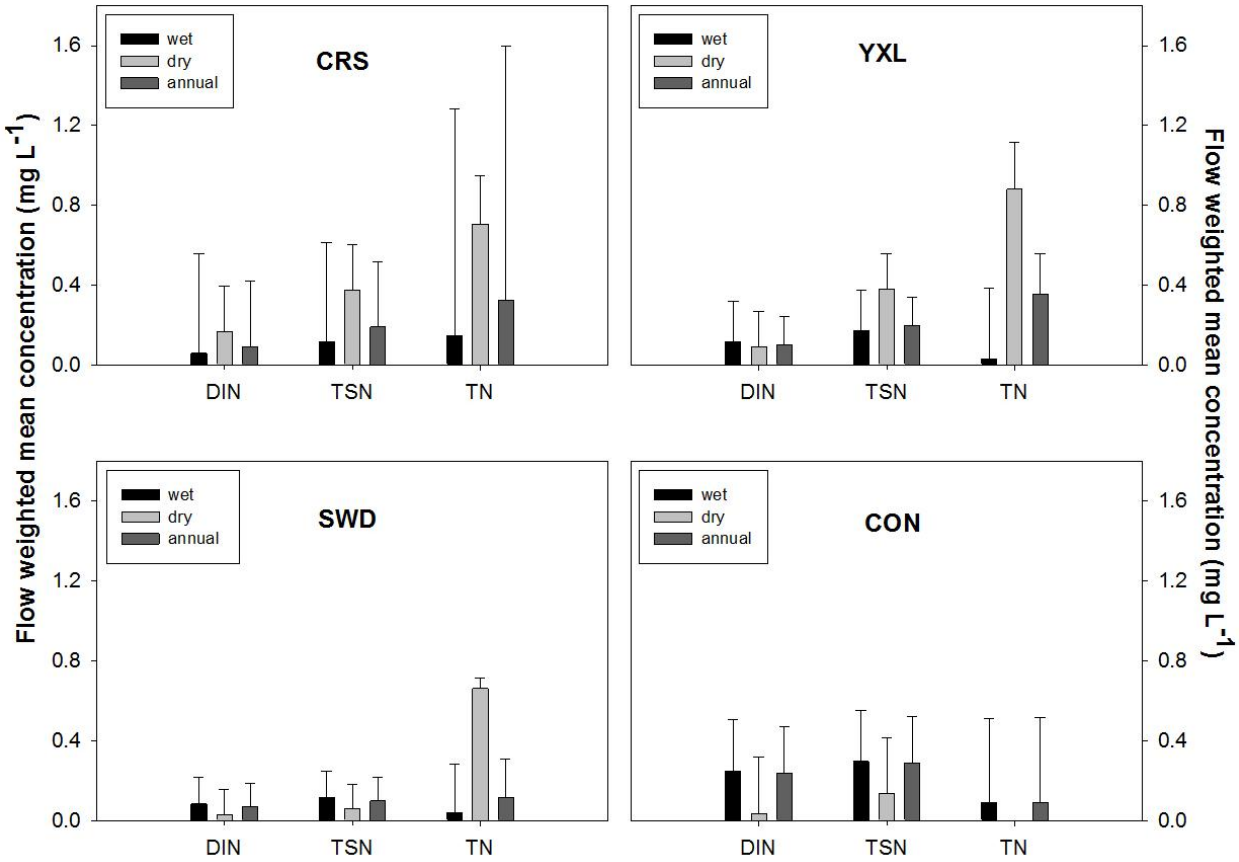

Figure 5-7. Flow weighted mean concentrations for DIN, TSN, and TN. Concentrations are averaged across seasonal (wet vs. dry) and annual (wet and dry combined) time periods.

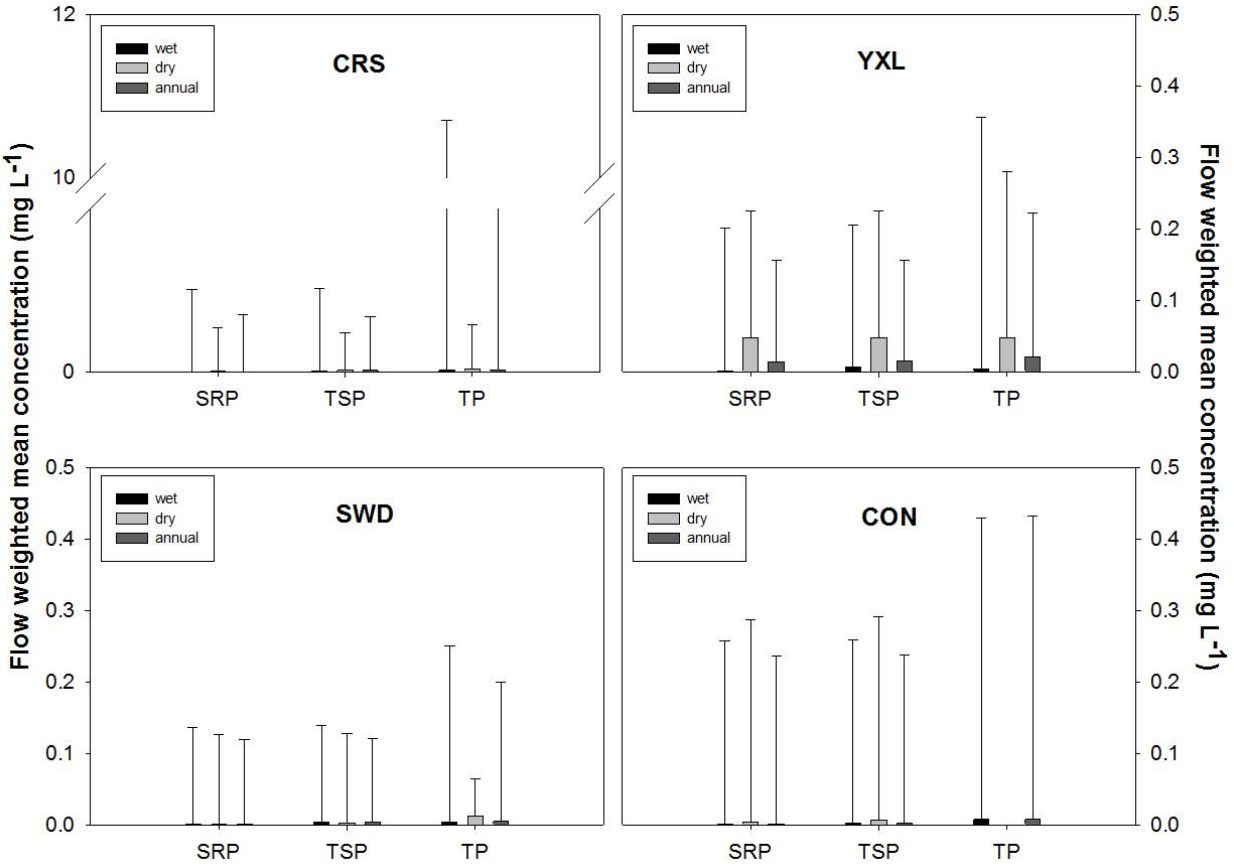

Figure 5-8. Flow weighted mean concentrations for SRP, TSP, and TP. Concentrations are averaged across seasonal (wet vs. dry) and annual (wet and dry combined) time periods.

CHAPTER 6
CONCLUDING REMARKS

Land use and land cover change occurs across varying spatial and temporal scales and can alter ecosystem structure and function in the lowland tropical watersheds of Belize. Local conditions within the riparian zone can alter thermal conditions and nutrient biogeochemistry in tropical rivers. Aquatic ecosystems are impacted most severely when land use change within riparian zones occurs rapidly across multiple river reaches. Human impact mapping within riparian zones is an important conservation tool that can detect trends in land use change and estimate the severity of stress stemming from these changes. The resulting maps are easily interpreted and provide spatially explicit data on aquatic ecosystem stresses useful for natural resource managers, policy decision makers, and the general public.

Human impact mapping represents a rapidly deployable and low-cost method for collecting spatially-explicit information about potential threats to aquatic ecosystems. However, one challenge with such rapid assessment techniques is that they do not accurately quantify threats not immediately observed during the mapping effort. In a more detailed assessment of riparian zone land use, interviews with small scale farmers revealed that agrochemicals represent an emerging threat to aquatic ecosystems. Whereas agrochemical use within plantation-scale farming systems (i.e., banana and citrus plantations) is well documented, my research in the Temash River watershed documented the common practice of agrochemical use within riparian zone farming plots. This research points to the need for more detailed studies of the impact of agrochemical use on aquatic ecosystems in Belize.

LULCC occurring at larger spatial scales can also influence aquatic ecosystem function. The dominant land use within the Temash River watershed is slash-and-burn agriculture and the survey of soils along the chronosequence revealed that, when practiced with a 15-year fallow period, this land use practice does not significantly impact soil nutrients. The slash-and-burn agriculture practiced by the Q'eqchí Maya of the Temash River watershed is not detrimental to the conservation of tropical forests. However, pasture expansion does threaten to alter land use practices within the watershed. Pasture requires large tracts of land and it permanently removes land from the traditional fallow rotation used for slash-and-burn. Carbon isotopic analysis of soil organic matter revealed that the introduction of C4 pasture grasses has altered the carbon isotopic signature of pasture soils relative to forest soils and the organic matter turnover rate in pastures varies with regard to the age of the pasture. Although C4 corn is the dominant crop that is cultivated within the Temash, soils under shifting cultivation still reflect the isotopic signature of C3-forest-derived organic matter.

Results from the in-stream nutrient analysis suggest that nutrient concentrations within the tributaries and main channel of the Temash River watershed are largely controlled by the seasonal fluctuations in hydrology. During periods of low flow (i.e., dry season), nutrient concentrations remain low and vary little across time and space. At the onset of the rainy season, nutrient concentrations peak, suggesting a pulse of nutrients being transported from terrestrial areas into the aquatic ecosystem. Following this initial peak, nutrient concentrations vary little throughout the remainder of the rainy season.

APPENDIX A
ELSI RANK SCORES FOR OBSERVED STRESSES

Stress	Sources	Source Code	Contrib.	Irrevers.	Rank	Rank Score
Sedimentation	No riparian buffer-commercial agriculture	NB (BAN)	H	H	H	10
	No riparian buffer-milpa	NB (MLP)	M	H	M	5
	No riparian buffer-old milpa	NB (OLDMLP)	L	H	M	5
	No riparian buffer-cattle grazing	NB (GRZ)	H	H	H	7.5
	No riparian buffer-buildings/residential	NB (house)	L	H	M	5
	Thin Buffer-commercial agriculture	TB (BAN)	M	H	M	5
	Thin Buffer-milpa	TB (MLP)	L	H	M	5
	Thin Buffer-old milpa	TB (OLDMLP)	L	H	M	5
	Thin Buffer-cattle grazing	TB (GRZ)	H	H	H	7.5
	Thin Buffer-buildings/residential	TB (house)	L	H	M	5
	In-stream gravel mining	GRV	V	H	V	10
	Channelization	CHN	H	H	H	7.5
	In-stream Grazing	GRZ	V	H	V	10
	Road access	RD	H	H	H	7.5
	Drainage ditch	DD	M	H	M	5
Nutrient loading	No riparian buffer-commercial agriculture	NB (BAN)	H	M	M	5
	No riparian buffer-milpa	NB (MLP)	L	M	L	2.5
	No riparian buffer-old milpa	NB (OLDMLP)	L	M	L	2.5
	No riparian buffer-cattle grazing	NB (GRZ)	H	M	M	5
	No riparian buffer-buildings/residential	NB (house)	L	M	L	2.5
	Thin Buffer-commercial agriculture	TB (BAN)	M	M	M	5
	Thin Buffer-milpa	TB (MLP)	L	M	L	2.5
	Thin Buffer-old milpa	TB (OLDMLP)	L	M	L	2.5
	Thin Buffer-cattle grazing	TB (GRZ)	M	M	M	5
	Thin Buffer-buildings/residential	TB (house)	L	M	L	2.5
	Laundry	CU	L	M	L	2.5

	In-stream Grazing	GRZ	H	M	M	5
	Drainage ditches	DD	V	M	H	7.5
Toxins/Contaminants	No riparian buffer-commercial agriculture	NB (BAN)	H	H	H	7.5
	No riparian buffer-milpa	NB (MLP)	M	H	M	5
	In-stream gravel mining	GR	L	H	M	5
	Thin Buffer-commercial agriculture	TB (BAN)	M	H	M	5
	Thin Buffer-milpa	TB (MLP)	L	H	M	5
	Pump house	PH	L	H	M	5
	Road access	RD	L	H	M	5
	Drainage ditches	DD	V	H	V	10
Altered flow regime	Water pumping	PH	M	M	M	5
	In stream gravel mining	GRA	L	M	L	2.5
	Drainage ditches	DD	L	M	L	2.5
Habitat alteration/ fragmentation	No riparian buffer-commercial agriculture	NB (BAN)	M	M	M	5
	No riparian buffer-milpa	NB (MLP)	M	M	M	5
	No riparian buffer-old milpa	NB (OLDMLP)	M	M	M	5
	No riparian buffer-cattle grazing	NB (GRZ)	M	M	M	5
	No riparian buffer-buildings/residential	NB (house)	M	M	M	5
	Thin Buffer-commercial agriculture	TB (BAN)	L	M	L	2.5
	Thin Buffer-milpa	TB (MLP)	L	M	L	2.5
	Thin Buffer-old milpa	TB (OLDMLP)	L	M	L	2.5
	Thin Buffer-cattle grazing	TB (GRZ)	L	M	L	2.5
	Thin Buffer-buildings/residential	TB (house)	L	M	L	2.5
	In-stream gravel mining	GR	H	M	M	5
	In-stream Grazing	GRZ	M	M	M	5
	Channelization	CHN	L	M	L	2.5
	Pump house	PH	L	M	L	2.5
	Sandbag dam	DAM	L	L	L	2.5
	Gravel mining	GRV	L	L	L	2.5

Thermal alteration	No riparian buffer-commercial agriculture	NB (BAN)	H	M	M	5
	No riparian buffer-milpa	NB (MLP)	H	M	M	5
	No riparian buffer-old milpa	NB (OLDMLP)	H	M	M	5
	No riparian buffer-cattle grazing	NB (GRZ)	H	M	M	5
	No riparian buffer-buildings/residential	NB (house)	H	M	M	5
	Drainage ditches	DD	L	M	M	5
	Thin Buffer-commercial agriculture	TB (BAN)	L	M	M	5
	Thin Buffer-milpa	TB (MLP)	L	M	M	5
	Thin Buffer-old milpa	TB (OLDMLP)	L	M	M	5
	Thin Buffer-cattle grazing	TB (GRZ)	L	M	M	5
	Thin Buffer-buildings/residential	TB (house)	L	M	M	5

APPENDIX B
MATAMBRE INTERVIEW QUESTIONAIRE

Matambre Practices in the Temash River Watershed, Belize

David G. Buck
School of Natural Resources & Environment, University of Florida

Informed Consent

Please read this consent document carefully before you decide to participate in this study.

My name is David Buck. I am a graduate student supervised by Dr. Mark Brenner, a professor at the University of Florida, in the USA. His address is P.O. Box 112120, University of Florida, Gainesville, FL 32611, USA. I would like to ask a few questions and talk with you about land use history and agricultural practices on your land and the overall landscape in this region.

It will take about one hour to ask these questions. Answering these questions will not affect you for better or worse. You do not have to answer any question you do not wish to answer. Your participation is completely voluntary and I can offer you no compensation for your participation in this survey. I have had other participants refuse before. You do not have to stop working to speak with us. If you would prefer, I can come back at another time. I will not write down your name and all answers or information you share with us will be private.

If you have any questions or concerns about your rights, they can be directed to the UFIRB office, P.O. Box 112250, University of Florida Gainesville, FL 32611, USA. If you have additional questions regarding the research, data analysis and/or use of the results I can be contacted locally by phone (669-4336). Do you have any questions? May I begin asking my questions? Remember, you can stop us at any time or we can schedule to meet some other time.

I have read the procedure described above. I voluntarily agree to participate in the procedure and I have received a copy of this description.

A. Interview Info

A1. Participant		A2. Village	
A3. Interviewer		A4. P.I.	

Notes: _____
_____

B. Household Info – data provided elsewhere

C. Matambre Questions

C1. Did you plant matambre this year? [Yes] [No]

C2. Where did you plant?
- a) along Temash River
- b) along creek [name:_____]
- c) on a hillside
- d) other (where: _____)

C3. How do you travel to your matambre field and how long does it take?
- a) Walk _____(time)
- b) Bike_____(time)
- c) Dory_____(time)
- d) Horse_____(time)
- e) Other _____ ; _____(time)

C4. If you were to walk to your field, how long would it take you?
_____ (time)

C5. What is the size of your field? _____ (circle one:acres / manzanas / hectares)

C6. How long did it take to clear? _____ (days)

C7. Did anyone help you clear the field? [Yes] [No]
If NO, proceed to C9

C8. If yes to C5, how many men helped you clear? _____ (# of men)

C9. Did anyone help you plant your field? [Yes] [No]
If NO, proceed to C11

C10. How many people helped you plant your field? _____ (# of men)

C11. Did you plant anything else other than corn in your field?
[Yes] _____, _____, _____, _____, _____ (specify)
[No]

C12. On average, do you reap more corn from your matambre or milpa? (circle one)
Average number of bags of unshelled corn/acre for matambre _____
Average number of bags of unshelled corn/acre for milpa _____

D. Field Maintenance/Use

D1. Will you weed your matambre after planting? [Yes] [No]
If NO, proceed to D3

D2. How frequently will you weed your field?
a) once c) three times
b) twice d) other _____ (# of times)

D3. Do you use herbicides on your field? [Yes] [No]
If NO, proceed to D6

D4. What type of herbicide do you use?
a) 2, 4-D c) Round-Up
b) Gramoxone d) other _____

D5. How frequently do you apply herbicides to your field?
a) once c) three times
b) twice d) other _____ (# of times)

D6. Have you received training in how to use agrochemicals?
[Yes] [No]
If NO, proceed to D8

D7. Did you receive this training from the Pesticide Control Board? If so, where?
[Yes] _____ [No]

D8. If you have excess corn from your matambre, will you sell it? [Yes] [No]
If NO, proceed to Section E. Riparian Zone

D9. How much will you sell it for? _____ (amount $BZ/pound)

D10. Will you sell it in the village or in town? [Punta Gorda] [village _____]

E. Riparian Zone/ Water Quality / Conservation

E1. Please tell me all of the animals that live in the Temash River and its creeks (Q'eqchi' or English)

a) _____
b) _____
c) _____
d) _____
e) _____
f) _____
g) _____
h) _____
i) _____
j) _____
k) _____
l) _____

E2. Is water in the Temash River clean? [Yes] [No]

E3. Are the creeks near your village clean? [Yes] [No]

E4. What makes the creeks and river dirty and not safe to drink?

a) cattle
b) clearing land
c) matambre
d) chemicals
e) other _____; _____; _____; _____

E5. Do other communities in the watershed effect the water near your village?

[Yes] How? _____
_____

[No]

E6. Have you ever gotten sick from drinking or swimming in the river/creek?

[Yes] _____ (what kind of sickness) [No]

E7. What caused your sickness?

_____

E8. Do you collect drinking water from the river, creek or spring?

[Yes]_____(specify which river/creek/spring) [No]

E9. Do you wash clothes/dishes in the river or creek?

[Yes]_____(specify river/creek) [No]

E10. Does anyone control access to the river or creeks? Who?

[Yes] _____ [No]

E11. Has there ever been any conflict in the village about water or water resources?

[Yes] [No]

briefly describe: _____
_____

E12. Who resolves those conflicts?

a) Alcalde
b) village council
c) government
d) other: _____

E13. When you farm, harvest timber, or clear for pasture, do you leave a buffer of trees along the edge of the stream or river?

[Yes] _____ (how wide – meters) [No]

E14. Would you participate in a conservation program that encouraged maintaining a buffer of forest along the edges of all creeks and the river in the Temash Watershed?

[Yes] [No]

E15 . Would you participate in a tree planting program that planted trees along the edges of the creeks and river?

[Yes] [No]

LIST OF REFERENCES

Achard, F., H. D. Eva, H.-J. Stibig, P. Mayaux, J. Gallego, T. Richards, and J.-P. Malingreau. 2002. Determination of deforestation rates of the World's humid tropical forests. Science **297**:999-1002.

Alcorn, J., and V. M. Toledo. 1998. Resilient resource management in Mexico's forest ecosystems: the contribution of property rights. Pages 216-249 *in* F. Berkes and C. Folke, editors. Linking social and ecological systems. Cambridge University Press, Cambridge, UK.

Allan, J. D. 2004. Landscapes and Riverscapes: The Influence of Land Use on Stream Ecosystems. Annual Review of Ecology, Evolution and Systematics **35**:257-284.

Allan, J. D., D. L. Erickson, and J. Fay. 1997. The influence of catchment land use on stream integrity across multiple spatial scales. Freshwater Biology **37**:149-161.

Ankers, C., D. E. Walling, and R. P. Smith. 2003. The influence of catchment characteristics on suspended sediment properties. Hydrobiologia **494**:159-167.

APHA. 1989. Standard methods for the examination of water and wasterwater, 17 edition. American Public Heath Association.

Arbuckle, K., and J. A. Downing. 2001. The influence of watershed land use on lake N:P in a predominantly agricultural landscape. Limnology and Oceanography **46**:970-975.

Arunachalam, A. 2002. Dynamics of soil nutrients and microbial biomass during first year cropping in an 8-year jhum cycle. Nutrient cycling in agroecosystems **64**:283-291.

Asmussen, L. E., A. W. White, E. W. Hauser, and J. M. Sheridan. 1977. Reduction of 2,4-D load in surface runoff down a grassed waterway. Journal of Environmental Quality **6**:159-162.

Assessment, M. E. 2005. Ecosystems and Human Well-being: Synthesis. Island Press, Washington, D.C.

Atran, S. 1993. Itza Maya tropical agro-forestry. Current Anthropology **34**:633-700.

Atran, S., D. Medin, N. Ross, E. Lynch, J. Coley, E. U. Ek', and V. Varnarsky. 1999. Folkecology and commons management in the Maya Lowlands. Proceedings of the National Academy of Sciences **96**:7598-7603.

Attiwill, P. M. 1994. The disturbance of forest ecosystems: the ecological basis for conservative management. Forest Ecology and Management **63**:247-300.

Awiti, A. O., M. G. Walsh, and J. Kinyamario. 2008. Dynamics of topsoil carbon and nitrogen along a tropical forest-cropland chronosequence: Evidence from stable isotope analysis and spectroscopy. Agriculture, Ecosystems and Environment **127**:265-272.

Azaroff, L. S., and L. M. Neas. 1999. Acute health effects associated with nonoccupational pesticide exposure in rural El Salvador. Environmental Research Section A **80**:158-164.

Bailey, M., P. H. Kirshen, E. Spang, S. Morrison, C. Pizarro, G. Kayser, G. Cid, and A. Parish. 2007. Rapid assessment of anthropogenic impacts on select transboundary watersheds of the Mesoamerican Barrier Reef System (MBRS) region. MBRS Project, National Oceanic and Atmospheric Administration and Tufts University.

Baillie, I. C., A. C. S. Wright, M. A. Holder, and E. A. Fitzpatrick. 1993. Revised Classification of the Soils of Belize. NRI Bulletin 59, Natural Resources Institute, Chatham, UK.

Balesdent, J., and A. Mariotti. 1996. Measurement of soil organic matter turnover using 13 C natural abundance. Mass Spectrometry of Soils:83–111.

Balesdent, J., A. Mariotti, and B. Guillet. 1987. Natural 13C abundance as a tracer for studies of soil organic matter dynamics. Soil Biology & Biochemistry **19**:25-30.

Berkes, F., and C. Folke. 2002. Back to the future: ecosystem dynamics and local knowledge. Pages 121-146 *in* L. H. Gunderson and C. S. Holling, editors. Panarchy: understanding transformations in human and natural systems. Island Press, Washington.

Bernard, H. R. 2006. Research methods in anthropology: qualitative and quantitative approaches, 4th edition. Altamira Press, New York.

Biggs, T. W., T. Dunne, T. F. Domingues, and L. A. Martinelli. 2002. Relative influence of natural watershed properties and human disturbance on stream solute concentrations in the southwestern Brazilian Amazon basin. Water Resources Research **38**:(10.1029/2001WR000271).

Biggs, T. W., T. Dunne, and L. A. Martinelli. 2004. Natural controls and human impacts on stream nutrient concentrations in a deforested region of the Brazilian Amazon basin. Biogeochemistry **68**:227-257.

Blanco, L. E., A. Aragón, I. Lundberg, C. Lidén, C. Wesseling, and G. Nise. 2005. Determinants of dermal exposure among Nicaraguan subsistence farmers during pesticide applications with backpack sprayers. The Annals of Occupational Hygiene **49**:17-24.

BMAF. 2006. Belize Ministry of Agriculture and Fisheries - Annual Report 2006.

Boles, E. 1998. Developing a Community-Based Rapid Watershed Assessment Protocol Focusing on the Sibun River Watershed in Belize as a Case Study. Jackson State University, Jackson, MS.

Boles, R., D. G. Buck, and P. C. Esselman. 2008. Synthesis of Water Resource Conservation, Management and Research Activities in the Watersheds of Belize. The Nature Conservancy - Belize program.

BPCB. 2008. Belize - Restricted Use Pesticides. *in*. http://www.pcbbelize.com/.

BPCB. unpublished data. Belize Pesticide Imports from 2002 to 2007. *in*. Belize Pesticide Control Board.

Bray, D. B., E. A. Ellis, N. Armijo-Canto, and C. T. Beck. 2004. The institutional drivers of sustainable landscapes: a case study of the 'Mayan Zone' in Quintana Roo, Mexico. Land Use Policy **21**:333-346.

Brenner, M., M. F. Rosenmeier, D. A. Hodell, and J. H. Curtis. 2002. Paleolimnology of the Maya Lowlands: long-term perspectives on interactions among climate, environment, and humans. Ancient Mesoamerica **13**:141-157.

Brubacher, D., J. T. Arnason, and J. Lambert. 1989. Woody species and nutrient accumulation during the fallow period of milpa farming in Belize, CA. Plant and Soil **114**:165-172.

Bruijnzeel, L. A. 2004. Hydrological functions of tropical forests: not seeing the soil for the trees? Agriculture, Ecosystems and Environment **104**:185-228.

Buck, D. G., P. C. Esselman, and R. Boles. 2008. Status, Needs and Future Directions for Effective Freshwater Conservation in Belize, Central America. The Nature Conservancy - Belize program.

Burnet, A. M. R. 1972. Effects of paraquat on invertebrates in a Canterbury stream, New Zealand. New Zealand Journal of Marine and Freshwater Research **6**:448-455.

Burns, C. J., W. M. Mahlburg, and J. P. Dutra. 2007. Pesticide exposure among farm workers. Environmental Research **105**:285-286.

Buschbacher, R., C. Uhl, and E. A. S. Serrão. 1988. Abandoned pastures in Eastern Amazonia. II. Nutrient stocks in the soil and vegetation. Journal of Ecology **76**:682-699.

Cantor, K. P., A. Blair, G. Everett, R. Gibson, L. F. Burmeister, L. M. Brown, L. Schoman, and F. R. Dick. 1992. Pesticides and other agricultural risk factors for non-Hodgkin's lymphoma among men in Iowa and Minnesota. Cancer Research **52**:2447-2455.

Caraco, N. F., and J. J. Cole. 1999. Human impact on nitrate export: an analysis using major world rivers. Ambio **28**:167-170.

Carpenter, S. R., N. F. Caraco, D. L. Correll, R. W. Howarth, A. N. Sharpley, and V. H. Smith. 1998. Nonpoint pollution of surface waters with phosphorus and nitrogen. Ecological Applications **8**.

Carr, D. L. 2002. Rural-Frontier Migration and Deforestation in the Sierra de Lacandon National Park, Guatemala. University of North Carolina.

Carr, D. L. 2004. Ladino and Q'eqchí Maya land use and land clearing in the Sierra de Lacandón National Park, Petén, Guatemala. Agriculture and Human Values **21**:171-179.

Carr, D. L. 2008. Farm households and land use in a core conservation zone of the Maya Biosphere Reserve, Guatemala. Human Ecology **36**:231-248.

Castillo, L. E., E. d. l. Cruz, and C. Ruepert. 1997. Ecotoxicology and pesticides in tropical aquatic ecosystems of Central America. Environmental Toxicology and Chemistry **16**:41-51.

Cayet, C., and E. Lightfouse. 2001. δ13C of plant-derived n-alkanes in soil particle-size fractions. Organic Geochemistry **32**:253-258.

Chomitz, K. M., and D. A. Gray. 1996. Roads, Land Use, and Deforestation: A Spatial Model Applied to Belize. The World Bank Economic Review **10**:487-512.

Chowdhury, R. R. 2006. Driving forces of tropical deforestation: the role of remote sensing and spatial models. Singapore Journal of Tropical Geography **27**:82-101.

Cowgill, U. M. 1962. An agricultural study of the Southern Maya Lowlands. American Anthropologist **64**:273-286.

Davis, C. O., and M. S. Simmons. 1979. Water chemistry and phytoplankton field and laboratory procedures.

Derrien, D., and W. Amelung. 2011. Computing the mean residence time of soil carbon fractions using stable isotopes: impacts of the model framework. European Journal of Soil Science **62**:237-252.

Desjardins, T., F. Andreux, B. Volkoff, and C. C. Cerri. 1994. Organic carbon and 13C contents in soils and soil size-fractions, and their changes due to deforestation and pasture installation in Eastern Amazonia. Geoderma **61**:103-118.

Desjardins, T., E. Barros, M. Sarrazin, C. Giradin, and A. Mariotti. 2004. Effects of forest conversion to pasture on soil carbon content and dynamics in Brazilian Amazon. Agriculture Ecosystems & Environment **103**:365-373.

Diekmann, L. O., D. Lawrence, and G. S. Okin. 2007. Changes in the spatial variation of soil properties following shifting cultivation in a Mexican tropical dry forest. Biogeochemistry **84**:99-113.

Diemont, S. A. W., and J. F. Martin. 2009. Lacandon Maya ecosystem management: sustainable design for subsistence and environmental restoration. Ecological Applications **19**:254-266.

Diemont, S. A. W., J. F. Martin, S. I. Levy-Tacher, R. B. Nigh, P. R. Lopez, and J. D. Golicher. 2006. Lacandon Maya forest management: Restoration of soil fertility using native tree species. Ecological Engineering **28**:205-212.

Døckersmith, I. C., C. P. Giardina, and R. L. Sanford. 1999. Persistence of tree related patterns in soil nutrients following slash-and-burn distrubance in the tropics. Plant and Soil **209**:137-156.

Downing, J. A., M. McClain, R. Twilley, J. M. Melack, J. Elser, N. N. Rabalais, W. M. Lewis, R. E. Turner, J. Corredor, and D. Soto. 1999. The impact of accelerating land-use change on the N-Cycle of tropical aquatic ecosystems: Current conditions and projected changes. Biogeochemistry **46**:109-148.

Dreiher, J., and E. Kordysh. 2006. Non-Hodgkin Lymphoma and pesticide exposure: 25 years of research. Acta Haematologica **116**:153-164.

Dudgeon, D., A. H. Arthington, M. O. Gessner, Z.-I. Kawabata, D. J. Knowler, E. Lévêque, R. J. Naiman, A.-H. Prieur-Richard, D. Soto, M. L. J. Staissny, and C. A. Sullivan. 2006. Freshwater biodiversity: importance, threats, status, and conservation challenges. Biological Reviews **81**:163-182.

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

Eisler, R. 1990. Paraquat Hazards to Fish, Wildlife, and Invertebrates: A Synoptic Review. U.S. Fish and Wildlife Service, Laurel, Maryland.

Ekström, A. M., M. Eriksson, L.-E. Hansson, A. Lindgren, L. B. Signorello, O. Nyrén, and L. Hardell. 1999. Occupational exposures and risk of gastric cancer in a population-based case-control study. Cancer Research **59**:5932-5937.

Emch, M. 2003. The Human Ecology of Mayan Cacao Farming in Belize. Human Ecology **31**:111-131.

ERMA. 2007. Draft National Integrated Water Resources Management Policy (Including Climate Change) for Belize. Consultants report to the Ministry of Natural Resources and the Environment, Belmopan, Belize.

Escobichon, D. J. 2001. Pesticide use in developing countries. Toxicology **160**:27-33.

Esselman, P. C. 2001. The Monkey River Baseline Study: Basic and Applied Research for Monitoring and Assessment in Southern Belize. University of Georgia, Athens, GA.

Esselman, P. C. 2009. Fish Communities and Conservation of Aquatic Landscapes in Northeastern Mesoamerica. University of Michigan, Ann Arbor, MI.

Esselman, P. C., and E. Boles. 2001. Status and future needs of limnological research in Belize. Pages 35-68 *in* R. G. Wetzel and B. Gopal, editors. Limnology in Developing Countries.

Esselman, P. C., and D. G. Buck. 2007. Hydrologic Assessment of the Monkey River Watershed, Belize. Phase 1: Human Impact Mapping Along the Monkey River and its Main Tributaries. Toledo Institute for Development and the Environment.

Esselman, P. C., M. C. Freeman, and C. M. Pringle. 2006. Fish-assemblage variation between geologically defined regions and across a longitudinal gradient in the Monkey River Basin, Belize. Journal of the North American Benthological Society **25**:142-156.

Evers, D. C. 2008. Assessing the global threat of mercury: A case study of the Mesoamerican Reef - with preliminary findings for sharks and groupers. Report BRI 2007. BioDiversity Research Institute, Gorham, ME.

Ewel, J., C. Berish, B. Brown, N. Price, and J. Raich. 1981. Slash and burn impacts on a Costa Rican wet forest site. Ecology **62**:816-829.

Ewel, J. J. 1986. Designing agricultural ecosystems for the humid tropics. Annual review of ecology and systematics **17**:245-271.

Farella, N., R. Davidson, M. Lucotte, and S. Daigle. 2007. Nutrient and mercury variations in soils from family frams of the Tapajós region (Brazilian Amazon): Recoomendations for better farming. Agriculture, Ecosystems and Environment **120**:449-462.

Fearnside, P. M., and R. I. Barbosa. 1998. Soil carbon changes from conversion of forest to pasture in Brazilian Amazonia. Forest Ecology and Management **108**:147-166.

Ferguson, B. G., J. Vandermeer, H. Morales, and D. M. Griffith. 2003. Post-agricultural succession in El Petén, Guatemala. Conservation Biology **17**:818-828.

Filoso, S., L. A. Martinelli, R. W. Howarth, E. W. Boyer, and F. Dentener. 2006. Human activities changing the nitrogen cycle in Brazil. Biogeochemistry **79**:61-89.

Filoso, S., L. A. Martinelli, M. R. Williams, L. B. Lara, A. Krusche, M. V. Ballester, R. Victoria, and P. B. deCamargo. 2003. Land use and nitrogen export in the Piracicaba River basin, Southeast Brazil. Biogeochemistry **65**:275-294.

Fitzgerald, D., D. S. Chanasyk, D. R. Neilson, D. Kiely, and R. Audette. 2001. Farm well water quality in Alberta. Water Quality Research Journal of Canada **36**:565-588.

Gagneten, A. M., and M. Marchese. 2003. Effect of paraquat on freshwater zooplankton and zoobenthos assemblages in enclosure experiments. International Journal of Ecohydrology and Hydrobiology **3**:389-398.

Galvao, L. A., J. A. Escamilla, S. Henao, E. Loyola, C. S. Castillo, and P. Arbelaez. 2004. Pesticides and Health in the Central American Isthmus. PAHO, Washington, D.C.

Garabrant, D. H., and M. A. Philbert. 2002. Review of 2,4-Dichlorophenoxyacetic acid (2,4-D) epidemiology and toxicology. Critical Reviews in Toxicology **32**:233-257.

Gavlak, R. G., D. A. Horneck, and R. O. Miller. 1994. Plant, Soil and Water Reference Methods for the Western Region. WREP 125.

Geist, H. J., and E. F. Lambin. 2002. Proximate Causes and Underlying Driving Forces of Tropical Deforestation. BioScience **52**:143-150.

Giardina, C. P., J. Robert L. Sandford, I. C. Døckersmith, and V. L. Jaramillo. 2000. The effects of slash burning on ecosystem nutrients during the land preparation phase of shifting cultivation. Plant and Soil **220**:247-260.

Giovannini, G., S. Lucchesi, and M. Giachetti. 1990. Effects of heating on some chemical parameters related to soil fertility and plant growth. Soil Science **149**:344-350.

Glynn, P. W., L. S. Howard, E. Corcoran, and A. D. Freay. 1984. The occurrence and toxicity of herbicides in reef building corals. Marine Pollution Bulletin **15**:370-374.

GOB. 2002. The Europoean Union Grants 6.00 Million Belize Dollars to the Banana Industry, Government of Belize, accessed 11/27/2008 (http://www.governmentofbelize.gov.bz/press_release_details.php?pr_id=2016). *in* Press Office, Government of Belize.

Gomez-Montes, C., and N. J. Bayly. 2010. Habitat use, abundance, and persistence of Neotropical migrant birds in a habitat matrix in northeast Belize. Journal of Field Ornithology **81**:237-251.

Gonzalez, V. J. B. 1980. A Limnological Investigation of a Tropical Fresh-Water Ecosystem: The Belize River, Belize, Central America. Western Michigan University, Kalamazoo, MI.

Gore, J. A. 2006. Discharge measurements and streamflow analysis. Pages 51-78 *in* F. R. Hauer and G. A. Lamberti, editors. Methods in Stream Ecology. Elsevier Inc., London.

Grandia, L. 2004. The Wealth Report: Q'eqchi' traditional knowledge and natural resource management in the Sarstoon-Temash National Park. Krishna Copy Center, Berkeley, California.

Grayson, R. B., C. J. Gippel, B. L. Finlayson, and B. T. Hart. 1997. Catchment-wide impacts on water quality: the use of 'snapshot' sampling during stable flow. Journal of Hydrology **199**:121-134.

Greenfield, D. W., and J. E. Thomerson. 1997. Fishes of the Continental Waters of Belize. University of Florida Press, Gainesville, FL.

GUARD. 2007. Monkey River Village Erosion Study: An Assessment and Proposed Mitigation. Monkey River Village Council and the Protected Areas Conservation Trust.

Hamilton, S., and E. F. Fischer. 2005. Maya farmers and export agriculture in highland Guatemala: implications for development and labor relations. Latin American Perspectives **32**:33-58.

Hardell, L. L. 1979. Malignant lymphoma of histiocytic type and exposure to phenoxyacetic acids or chlorophenols. Lancet **1**:55-56.

Harding, J. S., E. F. Benfield, P. V. Bolstad, G. S. Helfman, and E. B. D. Jones. 1998. Stream biodiversity: the ghost of land use past. Proceedings of the National Academy of Sciences **95**:14843-14847.

Herrera, A. 2004. Sartoon-Temash National Park Management Plan. *in*, Punta Gorda, Belize.

Hintze, J. 2001. NCSS and PASS. Number Cruncher Statistical Systems. *in*, Kaysville, Utah.

Hobbie, S. E., and P. M. Vitousek. 2000. Nutrient Limitation of Decomposition in Hawaiian Forests. Ecology **81**:1867-1877.

Holdridge, L. R. 1971. Forest Environments in Tropical Life Zones: A Pilot Study. Pergamon Press, New York.

Holland, B., N. Brorsen, and B. Johnson. 2003. Assessment of Geology and Soils in the Sarstoon Temash Region, Toledo District, Belize, Central America. Sarstoon-Temash Institute for Indigenous Management, Punta Gorda.

Hölscher, D., B. Ludwig, R. F. Möller, and H. Föster. 1997. Dynamic of soil chemical parameters in shifting cultivation in the Eastern Amazon. Agriculture, Ecosystems and Environment **66**:153-163.

Houghton, R. A. 1994. The Worldwide Extent of Land-Use Change. BioScience **44**:305-313.

Howard, P. H. 1989. Handbook of Environmental Fate and Exposure Data for Organic Chemicals. Lewis Publishers, Chelsea, MI.

Hughes, R. F., J. B. Kauffman, and V. J. Jaramillo. 1999. Biomass, carbon and nutrient dynamics of secondary forests in a humid tropical region of Mexico. Ecology **80**:1892-1907.

Humphries, S. 1993. The intensification of traditional agriculture among Yucatec Maya farmers: facing up to the dilemma of livelihood sustainability. Human Ecology **21**:87-102.

Hynes, H. B. N. 1975. The stream and its valley. Verh. Internat. Verein. Limnol **19**:1–15.

Johnson, L. B., and S. H. Gage. 1997. Landscape approaches to the analysis of aquatic ecosystems. Freshwater Biology **37**:113-132.

Johnson, L. B., C. Richards, G. E. Host, and J. W. Arthur. 1997. Landscape influences on water chemistry in Midwestern stream ecosystems. Freshwater Biology **37**:193-208.

Kaimowitz, D. 1995. The end of the hamburger connection? Livestock and deforestation in Central America in the 1980s and 1990s. *in* H. Cortes-Salas, R. d. Camino, and A. Conteras, editors. Readings of the workshop on government policy reform for forestry conservation and development in Latin America. Inter-American Institute for Cooperation on Agriculture, San Jose, CR.

Kammerbauer, J., and J. Moncada. 1998. Pesticide residue assessment in three selected agricultural production systems in the Choluteca River basin of Honduras. Environmental Pollution **103**:171-181.

Karper, J., and E. Boles. 2004. Human Impact Mapping of the Temash River, Belize. Sarstoon-Temash Institute for Indigenous Management.

Kauffman, J. B., D. L. Cummings, D. E. Ward, and R. Babbitt. 1995. Fire in the Brazilian Amazon: 1. Biomass, nutrient pools, and losses in slashed primary forests. Oecologia **104**:397-408.

Kegley, S. E., B. R. Hill, S. Orme, and A. H. Choi. 2008. PAN Pesticide Database. *in*. Pesticide Action Network, North America (San Francisco, CA, 2008), http://www.pesticideinfo.org.

Kemp, M. J., and W. K. Dodds. 2002. Spatial and temporal patterns of nitrogen concentrations in pristine and agriculturally-influenced prairie streams. Biogeochemistry **53**:125-141.

Keys, E. 2004. Commercial agriculture as creative destruction or destructive creation: a case study of chili cultivation and plant-pest disease in the southern Yucatan region. Land Degradation and Development **15**:397-409.

Keys, E., and R. R. Chowdhury. 2006. Cash crops, smallholder decision-making and institutional interactions in a closing frontier: Calakmul, Campeche, Mexico. American Geographer **5**:75-90.

Kleinman, P. J. A., R. B. Bryant, and D. Pimentel. 1996. Assessing ecological systainability of slash-and-burn agriculture through soil fertility indicators. Agronomy Journal **88**:122-127.

Knicker, H. 2007. How does fire affect the nature and stability of soil organic nitrogen and carbon? A review. Biogeochemistry **85**:91-118.

Kotto-Same, J., P. L. Woomer, M. Appolinaire, and Z. Louis. 1997. Carbon dynamics in slash-and-burn agriculture and land use alternatives of the humid forest zone in Cameroon. Agriculture, Ecosystems and Environment **65**:245-256.

Lambert, J. D. H., and J. T. Arnason. 1986. Nutrient dynamics in milpa agriculture and the role of weeds in initial stages of secondary succession in Belize, CA. Plant and Soil **93**:303-322.

Lambin, E. F., B. L. Turner, H. J. Geist, S. B. Agbola, A. Angelsen, J. W. Bruce, O. T. Coomes, R. Dirzo, G. Fischer, and C. Folke. 2001. The causes of land-use and land-cover change: moving beyond the myths. Global Environmental Change, Part A: Human and Policy Dimensions **11**:261-269.

Lawrence, D., and D. Foster. 2002. Changes in forest biomass, litter dynamics and soils following shifting cultivation in Southern Mexico: an overview. Interciencia **27**:400-408.

Lawrence, D., and W. H. Schlesinger. 2001. Changes in soil phosphorus during 200 years of shifting cultivation in Indonesia. Ecology **82**:2769-2780.

Lee, M. 2002. Human Impact Mapping in Crooked Tree Lagoon. School for International Training.

Lee, M. D., J. D. Stednick, and D. M. Gilbert. 1995. Belize Environmental Water Quality Monitoring - Final Report. Submitted as part of USAID - NARMAP project to the Government of Belize.

Levasseur, V., and A. Olivier. 2000. The farming system and traditional agroforestry systems in the Maya community of San Jose, Belize. Agroforestry Systems **49**:275-288.

Levy-Tacher, S. I., and J. D. Golicher. 2004. How predictive is traditional ecological knowledge? The case of the Lacandon Maya fallow enrichment system. Interciencia **29**:496-503.

Lewis, W. M., S. K. Hamilton, and J. F. Saunders. 1995. Rivers of northern South America. *in* C. E. Cushing, K. W. Cummins, and G. W. Minshall, editors. Rivers and stream ecosystems. Elsevier, Amsterdam.

Lewis, W. M., J. M. Melack, W. H. McDowell, M. McClain, and J. E. Richey. 1999. Nitrogen yields from undisturbed watersheds in the Americas. Biogeochemistry **46**:149-162.

Lide, D. R. 2004. CRC handbook of chemistry and physics. CRC Press, Boca Raton, FL.

Likens, G. E., and F. H. Bormann. 1974. Linkages between Terrestrial and Aquatic Ecosystems. BioScience **24**:447-456.

Lowrance, R., L. S. Altier, J. D. Newbold, R. R. Schnabel, P. M. Groffman, J. M. Denver, D. L. Correll, J. W. Gilliam, and J. L. Robinson. 1997a. Water quality functions of riparian forest buffers in Chesapeake Bay watersheds. Environmental Management **21**:687-712.

Lowrance, R., G. Vellidis, R. D. Wauchope, P. Gay, and D. D. Bosch. 1997b. Herbicide transport in a managed riparian forest buffer system. Transactions of the Ameican Society of Agricultural Engineers **40**:1047-1057.

Manguin, S., D. R. Roberts, R. G. Andre, E. Rejmánková, and S. Hakre. 1996. Characterization of *Anopheles darlingi* (Diptera: Culicidae) larval habitats in Belize, Central America. Journal of Medical Entomology **33**:205-211.

Marengo, J. A., D. Liebmann, V. E. Kousky, N. P. Filizola, and I. C. Wainer. 2001. Onset and end of the rainy season in the Brazilian Amazon Basin. Journal of Climate **14**:833-852.

Marques, M. N., A. I. Badiru, O. Beltrame, and M. A. F. Pires. 2007. Pesticide leaching and run-off hazard in the Ribeira de Iguape river basin in São Paulo state, Brazil. Journal of the Brazilian Society of Ecotoxicology **2**:179-185.

McClain, M. E. 2002. The application of ecohydrological principles for better water resources management in South America. Pages 193-209 *in* M. E. McClain, editor. The Ecohydrology of South American Rivers and Wetlands. IAHS Press, Wallingford.

McClain, M. E. 2008. Ecohydrology as a tool in the sustainable development of large tropical rivers. *in* D. Harper, M. Zalewski, and N. Pacini, editors. Ecohydrology: Processes, Models and Case Studies. CABI, Oxfordshire.

McClain, M. E., and R. E. Cossio. 2003. The use of riparian environments in the rural Peruvian Amazon. Environmental Conservation **3**:242-248.

McClain, M. E., J. E. Richey, and T. P. Pimentel. 1994. Groundwater nitrogen dynamics at the terrestrial-lotic interface of a small catchment in the Central Amazon Basin. Biogeochemistry **27**:113-127.

McConnell, R., and A. Hruska. 1993. An epidemic of pesticide poisoning in Nicaragua: implications for prevention in developing countries. American Journal of Public Health **83**:1559-1562.

McDowell, W. H., and C. E. Asbury. 1994. Export of carbon, nitrogen, and major ions from three tropical montane watersheds. Limnology and Oceanography **39**:111-125.

McGrath, D. A., M. L. Duryea, N. B. Comerford, and W. P. Cropper. 2000. Nitrogen and phosphorus cycling in an Amazonian agroforest eight years following forest conversion. Ecological Applications **10**:1633-1647.

McGrath, D. A., C. K. Smith, H. L. Gholz, and F. d. A. Oliveira. 2001. Effects of land-use change on soil nutrient dynamics in Amazônia. Ecosystems **4**:625-645.

Meerman, J., P. Herrera, and A. Howe. 2003a. Rapid Ecological Assessment: Sarstoon Temash National Park, Toledo District, Belize. Sarstoon-Temash Institute for Indigenous Management.

Meerman, J., and J. R. Wilson. 2005. The Belize National Protected Areas System Plan. Taskforce on Belize's Protected Areas Policy and Systems Plan, Belmopan, Belize.

Meerman, J. C., P. Herrera, and A. Howe. 2003b. Rapid Ecologial Assessment: Sarstoon Temash National Park, Toledo District, Belize. Sarstoon-Temash Institute for Indigenous Management.

Meybeck, M. 1982. Carbon, nitrogen, and phosphorus transport by world rivers. American Journal of Science **287**:401-450.

Mills, P. K., and R. C. Yang. 2007. Agricultural exposures and gastric cancer risk in Hispanic farm workers in California. Environmental Research **104**:282-289.

Moraes, J. M. d., A. E. Schuler, T. Dunne, R. d. O. Figueiredo, and R. L. Victoria. 2006. Water storage and runoff processes in plinthic soils under forest and pasture in eastern Amazonia. Hydrologic Processes **20**:2509-2526.

Moran, E. F., E. S. Brondizio, J. M. Tucker, M. C. d. Silva-Forsberg, S. McCracken, and I. Falesi. 2000. Effects of soil fertility and land-use on forest succession in Amazônia. Forest Ecology and Management **139**:93-108.

Murray, D. L. 1994. Cultivating crisis: the human cost of pesticides in Latin America. University of Texas Press, Austin, TX.

Murray, D. L., C. Wesseling, M. Keifer, M. Corriols, and S. Henao. 2002. Surveillance of pesticide-related illness in the developing world: putting the data to work. International Journal of Occupational and Environmental Health **8**:243-248.

Myers, N. 1991. Tropical forests: Present status and future outlook. Climatic Change **19**:3-32.

Naiman, R. J., and H. Decamps. 1997. The ecology of interfaces: riparian zones. Annual Review of Ecology and Systematics **28**:621-658.

Naiman, R. J., H. Decamps, and M. E. McClain. 2005. Riparia: Ecology, Conservation, and Management of Streamside Communities. Elsevier Academic Press, Burlington, MA.

Nations, J. D., and R. B. Nigh. 1980. The evolutionary potential of Lacandon Maya sustained-yield tropical forest agriculture. Journal of Anthropological Research **36**:1-30.

Neill, C., L. A. Deegan, S. M. Thomas, and C. C. Cerri. 2001. Deforestation for pasture alters nitrogen and phosphorus in soil solution and stream water of small Amazonian watersheds. Ecological Applications **11**:1817–1828.

Neill, C., J. M. Melillo, P. A. Steudler, C. C. Cerri, J. F. L. d. Moraes, M. C. Piccolo, and M. Brito. 1997. Soil carbon and nitrogen stocks following forest clearing and pasture in the southwestern Brazilian Amazon. Ecological Applications **7**:1216-1225.

Nelson, D. W., and L. E. Sommers. 1996. Total carbon, organic carbon and organic matter. Pages 961-1010 *in* J. M. Bigham, editor. Methods of soil analysis. Part 3: Chemical and microbiological properties. ASA, CSSA, SAAJ, Madison, WI.

Nepstad, D. C., C. R. d. Carvalho, E. A. Davidson, P. H. Jipp, P. A. Lefebvre, G. H. Negreiros, E. D. d. Silva, T. Stone, S. Trumbore, and S. Viera. 1994. The role of deep roots in the hydrological and carbon cycles of Amazonian forests and pastures. Nature **372**:666-669.

NRCC. 1978. Phenoxy Herbicides - Their Effects on Environmental Quality with Accompanying Scientific Criteria for 2,3,7,8 Tetrachlorodibenzo-p-Dioxin (TCDD). Ottawa, Canada.

Nunny, R., M. Santana, P. Stone, D. Tillet, and D. Walling. 2001. An investigation of the impact on reef environments of changing land-use in the Stann Creek district of Belize. The Watershed Reef Interconnectivity Scientific Study (WRIScS) Module 3. EU Project No. B7-6200/97-02/VII/ENV.

Nye, P. H., and D. J. Greenland. 1960. Soil Under Shifting Cultivation. Commonwealth Agricultural Bureaux.

Nye, P. H., and D. J. Greenland. 1964. Changes in the soil after clearing tropical forest. Plant and Soil **21**:101-112.

Pacini, N., D. M. Harper, V. Ittekkot, C. Humborg, and L. Rahm. 2008. Nutrient Processes and Consequences. Pages 30-45 *in* D. M. Harper, M. Zalewski, and N. Pacini, editors. Ecohydrology: processes, models and case studies. CAB International, Cambridge.

Parma de Croux, M. J., P. Arquiel, H. Ortego, and J. A. Lorente. 1999. Acute toxicity of paraquat to a common neotropical fish species (Apareiodon affinis) (Pisces, Hemiodidae). Polish Archive of Hydrobiology **46**:57-62.

Penagos, H. 2002. Contact dermatitis caused by pesticides among banana plantation workers in Panama. International Journal of Occupational and Environmental Health **8**:14-18.

Peterson, B. J., W. M. Wollheim, P. J. Mulholland, J. R. Webster, J. L. Meyer, J. L. Tank, E. Marti, W. B. Bowden, H. M. Valett, A. E. Hershey, W. H. McDowell, W. K. Dodds, S. K. Hamilton, S. Gregory, and D. D. Morrall. 2001. Control of nitrogen export from watersheds by headwater streams. Science **292**:86-90.

Popper, R., K. Andino, M. Bustamante, B. Hernandez, and L. Rodas. 1996. Knowledge and beliefs regarding agricultural pesticides in rural Guatemala. Environmental Management **20**:241-248.

Pringle, C. M., C. F. Rabeni, A. C. Benke, and N. G. Aumen. 1993. The role of aquatic science in freshwater conservation: cooperation between the North American Benthological Society and organizations for conservation and resource management. Journal of the North American Benthological Society **12**:177-184.

Pringle, C. M., and F. J. Triska. 1991. Effects of geothermal groundwater on nutrient dynamics of a lowland Costa Rican stream. Ecology **72**:951-965.

Puckett, L. J., and W. B. Hughes. 2005. Transport and fate of nitrate and pesticides: hydrogeology and riparian zone processes. Journal of Environmental Quality **34**:2278-2292.

Rainwater, T. R., B. M. Adair, S. G. Platt, T. A. Anderson, G. P. Cobb, and S. T. McMurry. 2002. Mercury in Morelet's Crocodile eggs in northern Belize. Archive of Environmental Contamination and Toxicology **42**:319-324.

Rainwater, T. R., T. H. Wu, A. G. Finger, J. E. Cañas, L. Yu, K. D. Reynolds, G. Coimbatore, B. Barr, S. G. Platt, G. P. Cobb, T. A. Anderson, and S. T. McMurry. 2007. Metals and organochlorine pesticides in caudal scutes of crocodiles from Belize and Costa Rica. Science of the Total Environment **373**:146-156.

Rao, P. S. C., and J. M. Davidson. 1980. Estimation of pesticide retention and transformation parameters required in nonpoint source pollution models. *in* M. R. Overcash and J. M. Davidson, editors. Environmental Impact of Nonpoint Source Pollution. Ann Arbor Science, Ann Arbor, MI.

Read, L., and D. Lawrence. 2003. Recovery of biomass following shifting cultivation in dry tropical forests of the Yucatán. Ecological Applications **13**:85-97.

Reddy, K. S., and R. P. Gambrell. 1987. Factors affecting the adsoprtion of 2,4-D and methyl parathion in soils and sediments. Agriculture, Ecosystems and Environment **18**:231-241.

Requena, E. L. 2008. Human Impact Mapping of the Belize River Watershed: Branch Mouth to the Coast Line. University of Belize, Belmopan.

Resh, V. H. 2007. Multinational, freshwater biomonitoring programs in the developing world: lessons learned from African and southeast Asian river surveys. Environmental Management **39**:737-748.

Roberts, D. R., S. Manguin, E. Rejmánková, R. Andre, R. E. Harbach, E. Vanzie, S. Hakre, and J. Polanco. 2002. Spatial distribution of adult *Anopheles darlingi* and *A. albimanus* in relation to riparian habitats in Belize, Central America. Journal of Vector Ecology **27**:21-30.

Roberts, D. R., J. Paris, S. Manguin, R. Harbach, R. Woddruff, E. Rejmánková, J. Polanco, B. Wullschleger, and L. Legters. 1996. Predictions of malaria vector distributions in Belize based on multispectral satellite data. American Journal of Tropical Medicine and Hygeine **54**:302-308.

Rosenberg, D. E., and T. P. Marcotte. 2005. Land-use system modeling and analysis of shaded cacao production in Belize. Agroforestry Systems **64**:117-129.

Sabo, J. L., R. Sponseller, M. Dixon, K. Gade, T. Harms, J. Heffernan, A. Jani, G. Katz, C. Soykan, J. Watts, and J. Welter. 2005. Riparian zones increase regional species richness by harboring different, not more, species. Ecology **86**:56-62.

Sader, S. A., C. Reining, T. Sever, and C. Soza. 1997. Human Migration and Agricultural Expansion: A Threat to the Maya Tropical Forests. Journal of Forestry **95**:27-32.

Sala, O. E., F. S. Chapin, J. J. Armesto, E. Berlow, J. Bloomfield, R. Dirzo, E. Huber-Sanwald, L. F. Heuenneke, R. B. Jackson, A. Kinzig, R. Leemans, D. M. Lodge, H. A. Mooney, M. Oesterheld, N. L. Poff, M. T. Sykes, B. W. Walker, M. Walker, and D. H. Wall. 2000. Global biodiversity scenarios for the year 2010. Science **287**:1770-1774.

Salas, O., and J. Meerman. 2007. Maya Mountain Marine Corridor Conservation Action Strategy. The Nature Conservancy - Belize program.

Salcedo, I. H., H. Tiessen, and E. V. S. B. Sampaio. 1997. Nutrient availability in soil samples from shifting cultivation sites in the semi-arid Caatinga of NE Brazil. Agriculture, Ecosystems and Environment **65**:177-186.

Sanchez, P. A. 1982. Nitrogen in shifting cultivation systems of Latin America. Plant and Soil **67**:91-103.

Saunders, D. L., J. J. Meeuwig, and C. J. Vincent. 2002. Freshwater protected areas: strategies for conservation. Conservation Biology **16**:30-41.

Saunders, T. J., M. E. McClain, and C. A. Llerena. 2006. The biogeochemistry of dissolved nitrogen, phosphorus, and organic carbon along terrestrial-aquatic flowpaths of a montane headwater catchment in the Peruvian Amazon. Hydrological Processes **20**:2549-2562.

Scheuerell, M. D., and D. E. Schindler. 2004. Changes in the spatial distribution of fishes in lakes along a residential development gradient. Ecosystems **7**:98-106.

Schwartz, N. 1990. Forest society: A social histroy of Peten, Guatemala. University of Pennsylvania Press, Philadelphia.

Seubert, C. E., P. A. Sanchez, and C. Valverde. 1977. Effects of land clearing methods on soil properties of an ultisol and crop performance in the Amazon jungle of Peru. Tropical Agriculture (Trinidad) **54**:307-321.

Shriar, A. J. 2001. The dynamics of agricultural intensification and resource conservation in the buffer zone of the Maya Biosphere Reserve, Peten, Guatemala. Human Ecology **29**:27-48.

Six, J., and J. D. Jastrow. 2002. Organic matter turnover. Pages 936-942 *in* R. Lal, editor. Encyclopedia of Soil Science. Taylor and Francis, London.

Sommer, R., P. L. G. Vlek, T. D. d. A. Sá, K. Vielhauer, R. d. F. R. Coelho, and H. Fölster. 2004. Nutrient balance of shifting cultivation by burning or mulching in the Eastern Amazon - evidence of subsoil nutrient accumulation. Nutrient cycling in agroecosystems **68**:257-271.

Steffan-Dewenter, I., M. Kessler, J. Barkmann, M. M. Bos, D. Buchori, S. Erasmi, H. Faust, G. Gerold, K. Glenk, S. R. Gradstein, E. Guhardja, M. Harteveld, D. Hertel, P. Hohn, M. Kappas, S. Kohler, C. Leuschner, M. Maertens, R. Marggraf, S. Mige-Kleian, J. Mogea, R. Pitopang, M. Schaefer, S. Schwarze, S. G. Sporn, A. Steingrebe, S. S. Tjitrosoedirdjo, S. Tjitrosoemito, A. Twele, R. Weber, L. Woltmann, M. Zeller, and T. Tscharntke. 2007. Tradeoffs between income, biodiversity, and ecosystem functioning during tropical rainforest conversion and agroforestry intensification. Proceedings of the National Academy of Sciences **104**:4973-4978.

Thomas, S. M., C. Neill, L. A. Deegan, A. V. Krusche, V. M. Ballester, and R. L. Victoria. 2004. Influences of land use and stream size on particulate and dissolved materials in a small Amazonian stream network. Biogeochemistry **68**:135-151.

TIDE. 2003. The Freshwater Initiative: River Assessment Project. Toledo Institute for Development and Environment.

Tiessen, H., I. H. Salcedo, and E. V. S. B. Sampaio. 1992. Nutrient and soil organic matter dynamics under shifting cultivation in semi-arid northeastern Brazil. Agriculture, Ecosystems and Environment **39**:139-151.

TNC, editor. 2000. The Five-S Framework for Site Conservation, 2nd edition. The Nature Conservancy, Arlington.

Toledo, V. M., B. Ortiz-Espejel, L. Cortés, P. Moguel, and M. d. J. Ordoñez. 2003. The multiple use of tropical forests by indigenous peoples in Mexico: a case of adaptive management. Conservation Ecology **7**:9.

Trumbore, S. E., E. A. Davidson, P. B. d. Camargo, D. C. Nepstad, and L. A. Martinelli. 1995. Belowground cycling of carbon in forests and pastures of Eastern Amazonia. Global Biogeochemical Cycles **9**:515-528.

Turner, B. L., S. C. Villar, D. Foster, J. Geoghegan, E. Keys, P. Klepeis, D. Lawrence, P. M. Mendoza, S. Manson, Y. Ogneva-Himmelberger, A. B. Plotkin, D. P. Salicrup, R. R. Chowdhury, B. Savitsky, L. Schneider, B. Schmook, and C. Vance. 2001. Deforestation in the southern Yucatán peninsular region: an integrative approach. Forest Ecology and Management **154**:353-370.

Uhl, C. 1987. Factors controlling succession following slash and burn agriculture in Amazonia. Journal of Ecology **75**:377-407.

Uhl, C., and C. F. Jordan. 1984. Succession and nutrient dynamics following forest cutting and burning in Amazonia. Ecology **65**:1476-1490.

USDA. 2004. Soil Survey Laboratory Methods Manual. USDA-Natural Resources Conservation Service, Lincoln, NE.

USEPA. 2008. List of Drinking Water Contaminants and Their MCLs (http://www.epa.gov/ogwdw/contaminants/index.html). *in*. United States Environmental Protection Agency.

van Wendel de Joode, B., I. A. M. de Graaf, C. Wesseling, and H. Kromhout. 1996. Paraquat exposure of knapsack spray operators on banana plantations in Costa Rica. International Journal of Occupational and Environmental Health **2**:294-304.

Vance, C., and J. Geoghegan. 2002. Temporal and spatial modeling of tropical deforestation: a survival analysis linking satellite and household survey data. Agricultural Economics **27**:317-332.

Veldkamp, E. 1994. Organic carbon turnover in three tropical soils under pasture after deforestation. Soil Science Society of America Journal **58**:175-180.

Vitousek, P. M., and R. L. Sanford. 1986. Nutrient cycling in moist tropical forest. Annual review of ecology and systematics **17**:137-167.

Weins, J. A. 2002. Riverine landscapes: taking landscape ecology into the water. Freshwater Biology **47**:501-515.

Wesseling, C., A. Aragón, L. Castillo, M. Corriols, F. Chaverri, E. d. l. Cruz, M. Keifer, P. Monge, T. J. Partanen, C. Ruepert, and B. v. W. d. Joode. 2001a. Hazardous Pesticides in Central America. International Journal of Occupational and Environmental Health **7**:287-294.

Wesseling, C., M. Corriols, and V. Bravo. 2005. Acute pesticide poisoning and pesticide registration in Central America. Toxicology and Applied Pharmacology **207**:S697-S705.

Wesseling, C., B. v. W. d. Joode, and P. Monge. 2001b. Pesticide-related illness and injuries among banana workers in Costa Rica: a comparison between 1993 and 1996. International Journal of Occupational and Environmental Health **7**:90-97.

Wesseling, C., B. van Wendel de Joode, C. Ruepert, C. León, P. Monge, H. Hermosillo, and T. J. Partanen. 2001c. Paraquat in developing countries. International Journal of Occupational and Environmental Health **7**:275-286.

Whitaker, M. J. 1993. The challenge of pesticide education and training for tropical smallholders. International Journal of Pest Management **39**:117-125.

Wiesenberg, G. L. B., J. Schwarzbauer, M. W. I. Schmidt, and L. Schwark. 2004. Source and turnover or organic matter in agricultural soils derived from n-alkane/n-carboxylic acid compositions and C-isotope signatures. Organic Geochemistry **35**:1371-1393.

Wilk, R. R. 1981. Agriculture, Ecology and Domestic Organization Among the Kekchi Maya. The University of Arizona.

Wilk, R. R. 1985. Dry season agriculture among the Kekchi Maya and its implications for Prehistory. Pages 47-57 *in* M. Pohl, editor. Prehistoric Lowland Maya Environment and Subsistence Ecology. Papers of the Peabody Museum, Cambridge.

Wilk, R. R. 1997. Household ecology: economic change and domestic life among the Kekchi Maya in Belize. Northern Illinois University Press, Dekalb, IL.

Williams, M. R., T. R. Fisher, and J. M. Melack. 1997. Solute dynamics insoil water and groundwater in a central Amazon catchment undergoing deforestation. Biogeochemistry **38**:303-335.

Williams, M. R., and J. M. Melack. 1997. Solute export from forested and partially deforested chatchments in the central Amazon. Biogeochemistry **38**:67-102.

Wright, A. C. S., D. H. Romney, and R. H. Arbuckle. 1959. Land in British Honduras, Report of the Land Use Survey Team. Colonial Research Publication (24) HMSO. 327pp & maps. , London.

Wu, T. H., T. R. Rainwater, S. G. Platt, S. T. McMurry, and T. A. Anderson. 2000. Organochlorine contaminants in Morelet's crocodile (*Crocodylus moreletii*) eggs from Belize. Chemosphere **40**:671-678.

Zahm, S. H., D. D. Weisenburger, P. A. Babbit, R. C. Sall, J. B. Vaught, K. P. Cantor, and A. Blair. 1990. A case-control study of non-Hodgkin's lymphoma and the herbicide 2,4-dichlorophenoxyacetic acid (2,4-D) in eastern Nebrask. Epidemiology **1**:349-356.

Zarin, D. J., E. A. Davidson, E. Brondizio, I. C. G. Viera, T. Sá, T. Fedpausch, E. A. G. Schuur, R. Mesquita, E. Moran, P. Delamonica, M. J. Ducey, G. C. Hurtt, C. Salimon, and M. Denich. 2005. Legacy of fire slows carbon accumulation in Amazonian forest regrowth. Frontiers in Ecology and the Environment **3**:365-369.

BIOGRAPHICAL SKETCH

David Buck is from Charlotte, North Carolina and attended the University of North Carolina at Chapel Hill from 1992 to 1996, graduating with a bachelor's degree in Latin American Studies and a minor in geology. Between undergraduate and graduate school, David worked as the field station manager for Monkey Bay Wildlife Sanctuary. At Monkey Bay, he was responsible for facilitating international study abroad programs that focused on the natural and cultural history of the Maya Forest region.

David came to the University of Florida in 2001 as a master's student in the School of Natural Resources and Environment (SNRE) and graduated in 2004 with a degree in Interdisciplinary Ecology. His master's research focused on the limnology and paleolimnology of hypersaline Lago Enriquillo, Dominican Republic. In 2004 David started in the PhD program in SNRE as an NSF-IGERT fellow in the Working Forests in the Tropics program. He conducted his PhD research in the Temash River watershed of southern Belize, Central America. He received his PhD from the University of Florida in the summer of 2012.

David and his wife Ellie Harrison-Buck have two children and currently live in Durham, NH, where his wife is an assistant professor of archaeology at the University of New Hampshire. David works for the Biodiversity Research Institute (BRI) in Gorham, Maine, as an aquatic ecologist and directs BRI's tropical research program.