EFFECTS OF AGRICULTURAL LAND USE ON STREAM HABITATS, RIPARIAN ZONES, WATER QUALITY, AND FRESHWATER BIODIVERSITY IN MOOREA, FRENCH POLYNESIA

MICHELLE CHANG

Integrative Biology, University of California, Berkeley, California 94720 USA

Abstract. Agricultural land use has been shown to negatively impact stream ecosystems. These human-induced disturbances on the stream ecology can be reduced with proper management of riparian zones. This study explored how agricultural land use affects riparian zones, water quality, and stream biodiversity in Moorea, French Polynesia. Furthermore, this study assessed whether riparian management is necessary on a tropical island that is continuously developing in agricultural production. Riparian zones' widths, canopy openness, and predominant species compositions were recorded for ten study sites. Water quality measurements were taken multiple times at each site. Stream biodiversity was examined through sampling benthic macroinvertebrates with the D-net and visually counting fishes because they both serve as good indicators of water quality. This study showed that there were significant differences between land use types and riparian zone width, canopy openness, temperature, conductivity, total dissolve solids, and salinity. There were also significant positive correlations between canopy and total species richness and abundance. Results were suggestive of differences between land use and total richness and abundance. This study provides implications for riparian management as a preventative measure to conserve the existing freshwater biodiversity.

Key words: agriculture; land use; community structure; Moorea, French Polynesia; riparian zone; water quality; freshwater biodiversity; stream habitat; canopy

INTRODUCTION

Ecological consequences of development have been examined worldwide (Meyer and Turner 1994, Paul and Meyer 2001, Allan 2012). With an increased demand for production, more land is being cleared for agriculture. Some issues caused by agriculture are deforestation, pollution, and soil degradation.

Agricultural land use practices have also been shown to affect streams and rivers by causing soil erosion and runoff of sediments, nutrients, and pesticides (Cuffney et al. 2000). Runoff increases pollutant loadings into streams and results in a decline in richness of algal, invertebrates, and fish communities (Paul and Meyer 2001). These anthropogenic impacts on the environment can be reduced with proper management of riparian zones (Moore and Palmer 2005). Studies have shown that riparian zones serve as buffers between land use and adjacent streams by filtering sediments, nutrients, pathogens, and metals from agricultural runoff, while controlling stream temperatures and productivity (Smith 1989, Sweeney 1993, Rutherford et al. 1999). The effectiveness of riparian buffers is affected by the type of vegetation and the corridor's width (Doyle et al. 1977, Lowrance et al. 1983, 1984, Magette et al. 1987, Dillaha et al. 1989). Thus, riparian management is important to stream health and freshwater biodiversity by alleviating ecological disturbances.

The effects of agriculture on stream ecosystems have been examined by assessing benthic macroinvertebrates and fishes because they are indicators of stream health (Barbour et al. 1999, Weijters et al. 2009, Johnson and Ringler 2014). By comparing forest and pasture reaches, it was suggested that deforestation, even at a very local scale, can change the taxonomic composition of benthic macroinvertebrate assemblages, reduce macroinvertebrate diversity, and eliminate the most sensitive taxa (Lorion and Kennedy 2009). Although there are advantages of using benthic macroinvertebrates for biomonitoring, other factors besides water quality can affect the distribution and abundance of these organisms (Resh 1995). Therefore, it is essential to also look at fish communities to evaluate water quality because fishes' longevity allows them to be good long term indicators of streams (Karr 1981).

Although there have been studies on the effects of agricultural land use, there are limited studies conducted on tropical islands. Islands serve as good model systems because they are isolated and biodiversity is relatively low compared to mainlands (MacArthur and Wilson 1967). For example, on Moorea, French Polynesia, species richness in streams is lower than continental streams of comparable size and elevation (Resh et al. 1990). In addition, Moorea's abundance of plantations and lack of riparian management make it an important place to examine the effects of agriculture on existing freshwater biodiversity (H. Murphy, pers. comm.).

The goal of this study was to assess the stream habitats with respect to land use types and explore how agricultural land use affects riparian zones, water quality, and freshwater biodiversity. This study tested the following hypotheses: (1) agricultural sites would have poorer habitats due to human disturbances, (2) agricultural land use would have shorter riparian zone widths and less canopy shading due to deforestation which is associated with agriculture, (3) agricultural land use would lead to high nutrient inputs into streams due to runoff of pollutants (Bu et al. 2014), and (4) agricultural land use would decrease biodiversity by eliminating the most sensitive taxa based on Lorion and Kennedy's study (2009). This study also explored an alternative hypothesis that increase canopy openness, as a result of deforestation, and nutrient loadings into streams would increase productivity and lead to an increase in richness based on previous studies (Rosenzweig and Abramsky 1993, Pearson and Connolly 2000, Vandermeulen et al. 2001). Therefore, this study also focused on correlating richness and abundance of freshwater fauna to canopy openness in order to assess whether riparian management is necessary in a developing tropical island.

METHODS

Study site

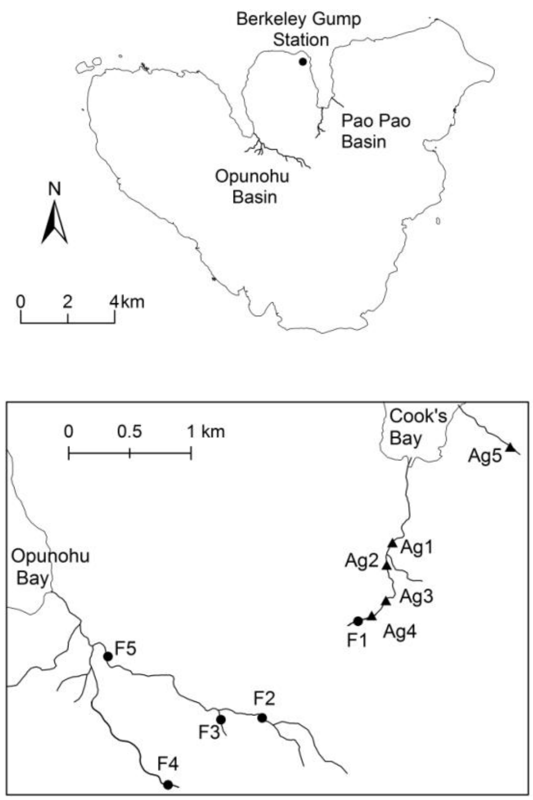

FIG. 1. Site locations on Moorea, French Polynesia. (Circles = forested (F) sites, triangles = agricultural (Ag) sites).

This study was conducted in Moorea, French Polynesia, on the northern watersheds that ultimately drain into Cook's Bay and Opunohu Bay (Fig. 1). Ten sites were sampled during October 30 to November 17, 2014 of the wet season. Sites were chosen based on land use type: agriculture or forest, which were identified using aerial photography, satellite imagery, or personal contact.

All land use sites were adjacent to a stream and 50 meters in length. A total of five agriculture sites and five forest sites were sampled for this study. Agriculture sites were defined as land use that included any cultivation of plants on at least one of the two banks. All five agriculture sites were located in Pao Pao Valley and drained into Cook's Bay. Forest sites were located in both Pao Pao Valley and Opunohu Valley.

Details on the locations, elevations, surface areas, and distances from the mouth of the two bays of the stream reaches are provided in Table 1.

Agriculture site 1 (Ag1) was a mixed field of bananas (on the right bank) and taro (on the left bank), with a shack on the left bank, a house on the right bank, and a bridge downstream. Agriculture site 2 (Ag2) was a mixed field of bananas and taro with a culvert immediately downstream and a house on the left bank upstream. Agriculture site 3 (Ag3) was a pineapple plantation with a row of three culverts immediately downstream of this site. Agriculture site 4 (Ag4) was a horticulture field with a bridge immediately upstream of this site and houses on both banks. Agriculture site 5 (Ag5) was another pineapple plantation with a channelized rock wall immediately upstream on the right bank of this site.

Forest sites were identified as areas with no agricultural land use and riparian zones dominated by trees. Forest site 1 (F1) was channelized with a rock wall immediately upstream on the left bank and a pile of garbage and litter on the left bank at the middle of the reach. Forest site 2 (F2) was channelized with a rock wall on the left bank and two culverts immediately upstream. Forest site 3 (F3) had a bridge downstream and a pasture beyond the left bank riparian corridor. Forest site 4 (F4) displayed no signs of human disturbances. Forest site 5 (F5) was surrounded by a barbed wire fence.

Stream habitat assessment

All ten sites were assessed within two days in order to minimize any inconsistency in habitat evaluation between sites. Each stream was assessed qualitatively with the Rapid Bioassessment Protocols (RBPs) Habitat Assessment Field Data Sheet for low gradient streams (Barbour and Stribling 1991, 1994). Nine descriptive parameters such as epifaunal substrate/available cover, pool substrate characterization, pool variability, sediment deposition, channel flow status, channel alteration, channel sinuosity, bank stability, and vegetative protection were rated on a numerical scale of 0 to 20 for each sampling reach based on visual observations, with 0 being the worst and 20 being the best habitat condition.

In addition, quantitative stream habitat measurements were recorded for the distance between agricultural land use and riparian

TABLE 1. Details of stream reaches

Site	Location	Elevation (m)	Surface Area[1] (m^2)	Distance from mouth (km)
Ag1	S 17°30.777', W 149°49.389'	15.0-18.0	267.4	0.78
Ag2	S 17°30.876', W 149°49.417'	18.0-19.8	172.8	0.99
Ag3	S 17°31.033', W 149°49.423'	27.4-28.0	82.6	1.37
Ag4	S 17°31.099', W 149°49.490'	26.5-28.3	122.7	1.54
Ag5	S17°30.361', W 149°48.839'	37.5-42.4	102.9	0.54
F1	S 17°31.123', W 149°49.554'	27.1-29.6	177.6	1.67
F2	S 17°31.531', W 149°50.003'	40.5-43.3	128.8	2.31
F3	S 17°31.551', W 149°50.194'	30.5-32.9	317.5	1.96
F4	S 17°31.836', W 149°50.443'	42.1-43.6	392.0	2.30
F5	S 17°31.263', W 149°50.711'	10.7-11.0	405.6	0.78

Note: [1.] Surface area is an averaged value.

zone, stream widths, and riparian zone widths. For agriculture sites, distances from site of land use to riparian zones were measured using a transect tape. Stream widths also were measured at every 10 meters for the whole sample reach and averaged. The endpoints of the riparian zones were marked by the physical change in species composition with increasing distance from the stream banks. Widths for both banks were measured with a transect tape at upstream or downstream points of sample reach. Actual widths were then converted to a value from 0 to 20 that matched the Habitat Assessment Field Data Sheet's description of riparian vegetative zone width.

Riparian zone assessment

Riparian vegetation found on each bank was used to guide the measurement of the widths of riparian zones. Riparian characteristics, such as predominant vegetation, species composition, and canopy openness, were recorded for each sample site. Canopy openness was measured using Strickler's 17-point modified convex spherical densiometer to prevent the overestimation of shading that occurs with the unmodified readings (Strickler 1959). A minor modification to Strickler's method was performed to incorporate five replicates within each reach (at every 10 meters) instead of one measurement at the center of the reach. Four measurements (upstream, downstream, left bank, and right bank) were recorded for each replicate and converted to a percentage by summing the values and multiplying with 1.47. After percentages of canopy openness for each of the five replicates were calculated, the average percent canopy openness of each site was computed to obtain a more representative measurement of the sample reach.

Water quality assessment

Water quality measurements were taken on different days between study sites from October 30 to November 17. Physical–chemical parameters, such as pH, temperature (°C), conductivity (µS/cm), total dissolved solids (TDS, ppm), salinity (ppm), and dissolved oxygen (DO, ppm), were assessed in–stream with a portable PCSTestr 35 Multi Probe System and Extech DO Stik II. Grab samples were taken for testing the concentration of nitrate (ppm) at the UC Berkeley Gump Station laboratory using Aquarium Pharmaceuticals Saltwater Master Test Kit. All physical-chemical parameters were measured in the morning from 7 A.M. to noon and water samples were tested for nitrate in the laboratory within nine hours of collection. Water quality measurements for each site were averaged.

Sampling stream biodiversity

There were two sampling phases for stream biodiversity during this study. A 3-day pilot study, also known as sample phase 1, was conducted prior to the actual sampling phase, also known as sample phase 2. Sample phase 2 for stream biodiversity occurred from November 7 to November 17 between 7 A.M. and noon. Biodiversity in the streams adjacent to agriculture and forest sites were examined by quantifying the diversity of fishes and macroinvertebrates. A visual count for fishes, prawns, and water striders was performed for ten minutes due to the difficulty in catching them.

For sampling macroinvertebrate communities, several methods were used in combination to avoid differences in fauna due to habitat preferences. Macroinvertebrates samples were taken along the stream reach applying two sampling methods for 10 minutes each: 1) collecting macroinvertebrates directly in their stream habitats from stones and exposed roots, and 2) using a D–frame net (500 µm mesh) for jab sampling at major habitats in proportional representation to the reach. For collection method 1, stones with approximate diameter of at least 8 cm were overturned. For collection method 2, macroinvertebrates were sampled by jabbing the substrate with the D-net and sweeping downstream at each point of disturbance. This technique was used for its effectiveness in sampling multi-habitat streams that varied in substrates and depth (MA DEP 1995, Mid-Atlantic Coastal Streams Workgroup 1996, Barbour et al. 1999, Hauer and Resh 2007)

Macroinvertebrates that were identifiable in the field were released after collection. Smaller macroinvertebrates were stored in freshwater and brought to the laboratory to be identified to the lowest possible taxonomical level using identification keys, Moorea Biocode database, and expert opinion.

Data analyses

All statistical analyses were performed using R (Oksanen et al. 2013, R Development Core Team 2013). Sampling phases were tested for any significant differences in sampling using the t-test.

Shapiro Wilk test for normality was performed for all habitat, riparian zone, and water quality parameter. For parameters that were normally distributed, Bartlett tests for equal variance were performed and followed by t-tests to determine whether there was a significant difference between each parameter and land use type. For parameters that were not normally distributed, the Kruskal-Wallis test was used to test for differences since they did not meet the requirements for the t-test.

For water quality measurements, averages were calculated for description.

Shannon-Wiener's diversity index along with evenness and richness were calculated for each of the 10 sample sites since diversity accounts for evenness, which compares similarity in population size of each species, and richness, which is the number of different species. Evenness was calculated as the logarithm of Shannon's diversity divided by the richness (Oksanen et al. 2013).

Total species abundance, which is defined as the total number of individuals of each species, was also examined for each site and for the two land use groups.

Differences in total community richness and abundance between land use groups were analyzed using t-test. Beta diversity between sites was visualized using Bray-Curtis dissimilarity cluster dendrogram (Oksanen et al. 2013).

Linear regressions were also performed to investigate the relationship between riparian canopy openness and stream biota richness and abundance within the two land use groups.

RESULTS

Stream habitat assessment

There was no significant difference between land use types and the following 8 habitat parameters: epifaunal substrate/ available cover ($t = -2.0$, $df = 4.56$, p-value = 0.097), pool substrate characterization (Kruskal-Wallis chi-squared = 0.11, $df = 1$, p-value= 0.74), pool variability (Kruskal-Wallis chi-squared = 0.10, $df = 1$, p-value= 0.75), sediment deposition (Kruskal-Wallis chi-squared = 2.29, $df = 1$, p-value= 0.13), channel flow status (Kruskal-Wallis chi-squared = 3.67, $df = 1$, p-value= 0.055), channel sinuosity ($t = -0.77$, $df = 8$, p-value = 0.46), bank stability ($t = 1.21$, $df = 8$, p-value = 0.26), and vegetable protection (Kruskal-Wallis chi-squared = 1.16, $df = 1$, p-value= 0.28).

The habitat parameter that showed a significant difference between agricultural and forested land use was riparian zone width ($t = -5.13$, $df = 8$, p-value = 0.0009, Fig. 2). Agricultural land use scores ranged from 2 to 7, while forested land use scores ranged from 10 to 20. The mean riparian score for agriculture was 3.2 (± SD 2.17), while the mean score for forest was 15.2 (± SD 4.76). Forested sites had greater riparian zone widths on average than agricultural sites; however, there was a greater variation in widths between forested sites than between agricultural sites.

Riparian zone assessment

Dominant plant species

Agricultural sites had riparian zones that were predominately composed of weeds, grasses, and vines. For example, Ag1 was dominated by morning glory. Ag2 was dominated by morning glory and cattail. Ag3 was dominated by grass, vines, weeds, morning glory, and cattail. Ag4 was dominated by trees, which included *Hibiscus tiliaceus* and bananas. Ag5 was dominated by trees and ferns, which included *Inocarpus fagifer*, *Angiopteris evecta*, and minimum *H. tiliaceus*.

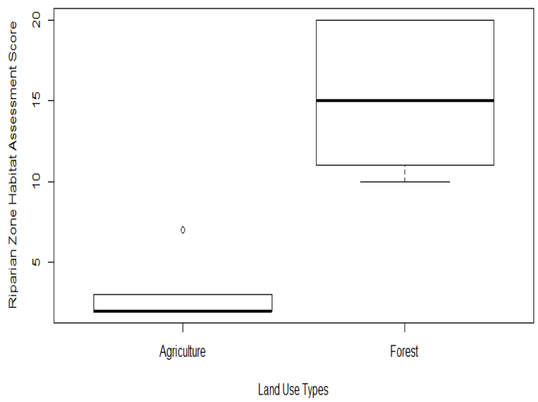

FIG. 2. Comparison of riparian zone width habitat assessment score between land use types. There was a significant difference between agricultural and forested land use in riparian zone width (t = -5.13, df = 8, p-value < 0.001). Outlier (circle) in agriculture category is Ag5 with a riparian zone habitat score of 7.

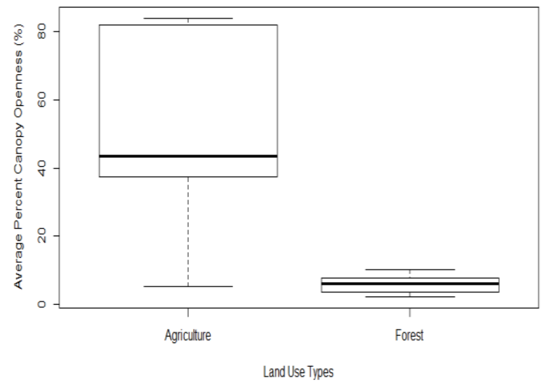

FIG. 3. Comparison of percent canopy openness between land use types. There was a significant difference between agricultural and forested land use in canopy openness (Kruskal-Wallis chi-squared = 3.94, df = 1, p-value < 0.05).

Forested sites riparian zones were predominately composed of native trees. For example, F1 was dominated by *Barringtonia asiatica*, *H. tiliaceus*, and *I. fagifer*. F2 and F3 were dominated by *I. fagifer* and *H. tiliaceus*. F4 was dominated by a mixture of trees and ferns, which included A. *evecta* and *I. fagifer* on both banks. F5 was dominated by *I. fagifer* and *H. tiliaceus* on the left bank and weeds on the right bank.

Canopy openness

There was a significant difference in canopy openness between land use types (Kruskal-Wallis chi-squared = 3.94, df = 1, p-value= 0.047, Fig. 3). Agricultural land use was more opened, but there was a great variation in canopy openness between the five agricultural sites. Forested sites were more covered and the differences between canopy openness between forested sites were very small. Average canopy openness for agricultural land use was 50.39% (± SD 33.04), while average canopy openness for forest was 6.00% (± SD 3.18).

Water quality assessment

Physical-chemical parameters of freshwater were averaged between study sites (Table 2).

On average, Ag5 had the highest pH value; Ag2 had the highest water temperature, conductivity, total dissolved solids, and salinity; Ag 1 had the highest dissolved oxygen concentration; and F2 had the highest concentration of nitrates.

There was no significant difference between land use and the following physical-chemical water parameters: pH (Kruskal-Wallis chi-squared = 0.098, df = 1, p-value = 0.75), dissolved oxygen (Kruskal-Wallis chi-squared = 0.53, df = 1, p-value= 0.46), and nitrate (Kruskal-Wallis chi-squared = 1.54, df = 1, p-value= 0.21). There was a significant difference between land use and temperature (t = 3.19, df = 8, p-value = 0.013, Fig. 4), conductivity (t = 3.86, df = 8, p-value = 0.0048, Fig. 5), total dissolved solids (t = 3.92, df = 8, p-value = 0.0044, Fig. 6), and salinity (t = 3.85, df = 8, p-value = 0.0049, Fig. 7).

Temperature was affected by land use type. Agriculture sites had higher water temperature, averaging around 24.94 °C (± SD 1.32), while forested sites averaged around

22.82 °C (± SD 0.68). Agricultural land use types had greater variability for temperature between sample sites than forested land use types.

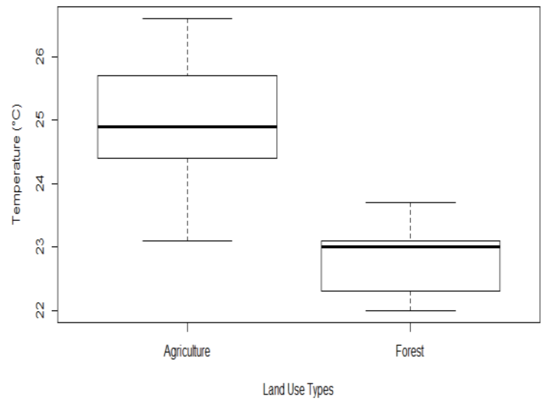

FIG. 4. Comparison of temperature between land use types. There was a significant difference between agricultural and forested land use in water temperature ($t = 3.19$, $df = 8$, $p\text{-value} < 0.02$).

Conductivity was affected by land use types. Agricultural land use had a mean conductivity of 222.8 µS/cm (± SD 15.90), while mean in forested land use was 157.9 uS/cm (± SD 34.09). There was higher variability between forested sites than between agricultural sites.

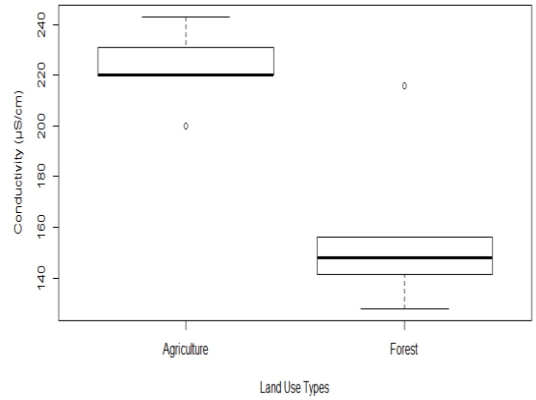

FIG. 5. Comparison of conductivity between land use types. There was a significant difference between agricultural and forested land use in conductivity concentration ($t = 3.86$, $df = 8$, $p\text{-value} < 0.005$). The outliers (circles) in agriculture and forest groups were Ag5 and F1 respectively.

Total dissolved solids were also affected by land use types. Agricultural land use had higher concentrations of TDS with a mean of 158.20 ppm (± SD 10.78) than forested land use with a mean of 111.96 ppm (± SD 24.11). There was a greater variability between forested sites than agricultural sites.

TABLE 2. Averaged for water quality measurements at each site

Site	pH	Temperature (°C)	Conductivity (µS/cm)	TDS (ppm)	Salinity (ppm)	DO (ppm)	Nitrate (ppm)
Ag1	7.26	24.60	206.83	146.00	100.40	10.16	10.00
Ag2	7.00	25.60	239.33	169.67	116.00	7.08	11.67
Ag3	7.71	24.67	217.33	153.67	105.50	6.35	5.00
Ag4	7.49	24.30	223.33	158.67	108.33	6.09	10.00
Ag5	7.91	22.95	204.50	145.50	96.55	6.67	20.00
F1	7.16	23.70	216.00	154.00	105.33	6.15	5.00
F2	7.58	23.06	138.45	98.23	68.60	6.72	47.50
F3	7.63	22.88	126.85	90.08	48.70	6.89	1.67
F4	7.88	22.63	143.87	101.67	70.73	7.10	0.00
F5	7.62	23.46	139.88	99.25	69.18	7.39	2.50

Note: Ag= agriculture, F= forest, number following Ag or F is the site number. TDS= total dissolved solids, DO= dissolved oxygen

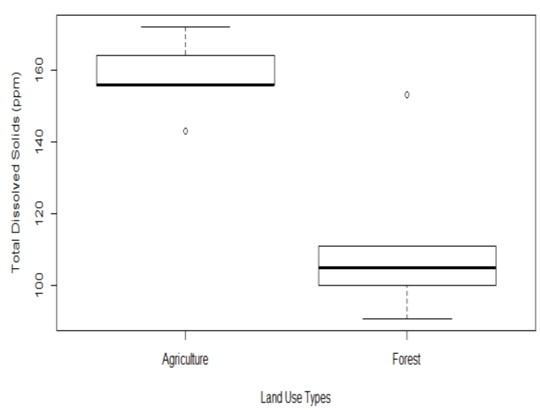

FIG. 6. Comparison of total dissolved solids between land use types. There was a significant difference between agricultural and forested land use in total dissolved solids concentration ($t = 3.92$, $df = 8$, p-value < 0.005). The outliers (circles) in the agriculture and forest groups were Ag5 and F1 respectively.

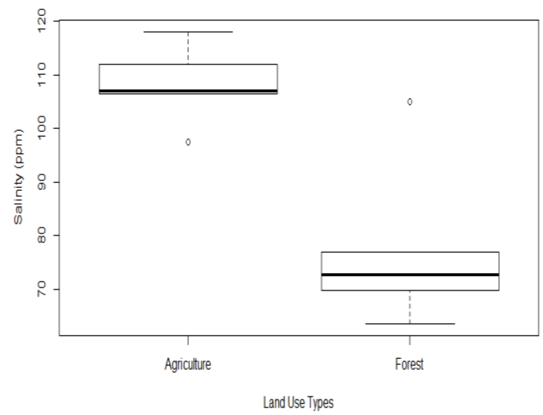

FIG. 7. Comparison of salinity between land use types. There was a significant difference between agricultural and forested land use in salinity concentration ($t = 3.85$, $df = 8$, p-value < 0.005). The outliers (circles) in the agriculture and forest groups were Ag5 and F1 respectively.

Salinity was also affected by land use types. Agricultural land use had a mean salinity concentration of 108.20 ppm (± SD 7.57), while forested land use had a mean concentration of 77.62 ppm (± SD 16.07). There was a higher variation between forested sites than between agricultural sites.

Sampling stream biodiversity

Sampling and weather

Richness was significantly different between sampling phases ($t = -2.43$, $df = 17.93$, p-value $= 0.026$). The mean richness of phase 1(the pilot study) and phase 2 were 8.2 (± SD 2.66) and 11.0 (± SD 2.49) respectively. Abundance was also significantly different between sampling phases (Kruskal-Wallis chi-squared $= 4.81$, $df = 1$, p-value $= 0.028$). The mean abundance of phase 1 and phase 2 were 74.6 (± SD 84.28) and 163.5 (± SD 122.44). Due to rainy weather and inconsistency in sampling techniques during phase 1, data was analyzed using phase 2 data.

Sampling freshwater fauna

A total of 37 species were identified (4 fishes and 33 macroinvertebrates). The total abundance, number of individuals of each species, from all sites was 1635.

Shannon's diversity index showed the variation in diversity between each site (Fig. 8). The average diversity index across all sites was 1.51 (± SD 0.42). Shannon's diversity index ranged from 0.75 (Ag3) to 2.17 (Ag4). The sites with the highest diversity index among land use types were Ag4 (2.17) and F3 (1.86). The sites with the lowest diversity index among land use types were Ag3 (0.75) and F4 (1.11). Ag 4 site had the highest diversity and Ag3 site had the lowest diversity despite both of them being adjacent to agricultural land use. Variance among agricultural sites was greater than variance among forested sites.

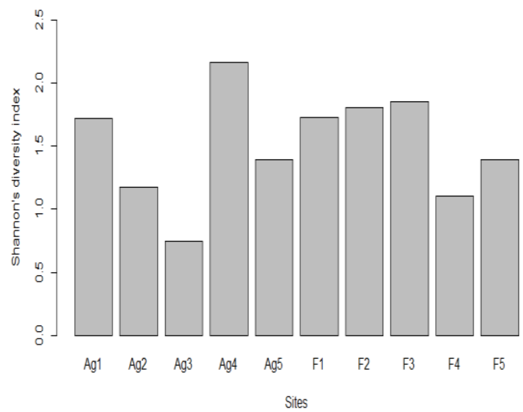

FIG. 8. Shannon-Wiener's diversity index among all ten sites (var = 0.18). Ag = agriculture and F = forest.

Evenness and richness

The average evenness across all sample sites regardless of land use was 0.64 (± SD 0.18). Evenness across all sites was shown to vary from 0.28 (Ag3) to 0.85 (F3) (Fig. 9). The sites with the highest evenness between land use types were Ag4 (0.82) and F3 (0.85). The sites with the lowest evenness between land use types were Ag3 (0.28) and F4 (0.50). Agricultural sites had higher variance (var = 0.049) than forested sites (var = 0.018).

The average richness across all sites regardless of land use was 11 (± SD 2.49). Richness from all sites varied from 8 (Ag5) to 15 (Ag2) species (var = 6.22, Fig. 10). The sites with the highest richness between land use types were Ag2 (15 species) and F1 (11 species). The sites with the lowest richness between land use types were Ag 5 (8 species) and F3, F4, and F5 (9 species at each site).

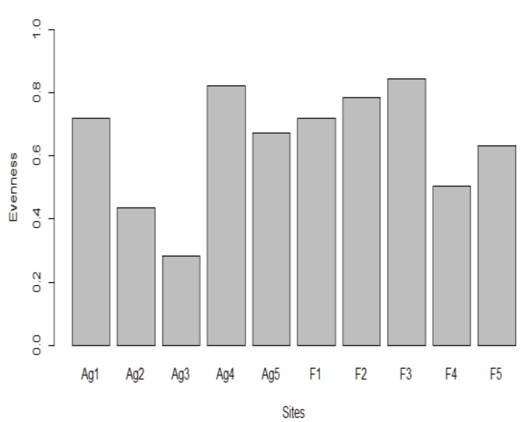

FIG. 9. Evenness across all sites (var = 0.033). Ag = agriculture and F = forest.

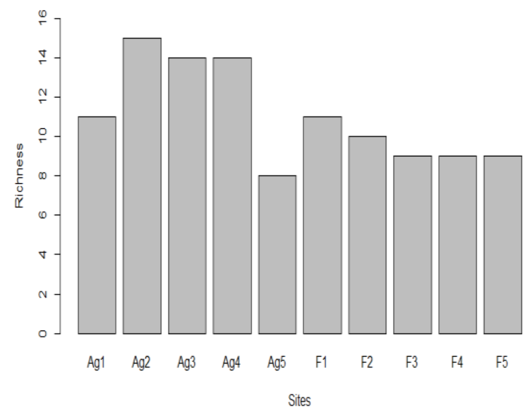

FIG. 10. Comparison of richness across all ten sites (var = 6.22). Ag = agriculture and F = forest.

Beta diversity

Bray-Curtis dissimilarity dendrogram showed clustering of similar community composition between all sites (Fig. 11). The sites that are most similar in community composition were Ag2 and Ag3, and then Ag4 and F1. Three agricultural sites (Ag 1, 2, and 3) were clustered together, while Ag4 and Ag 5 clustered with the forested sites.

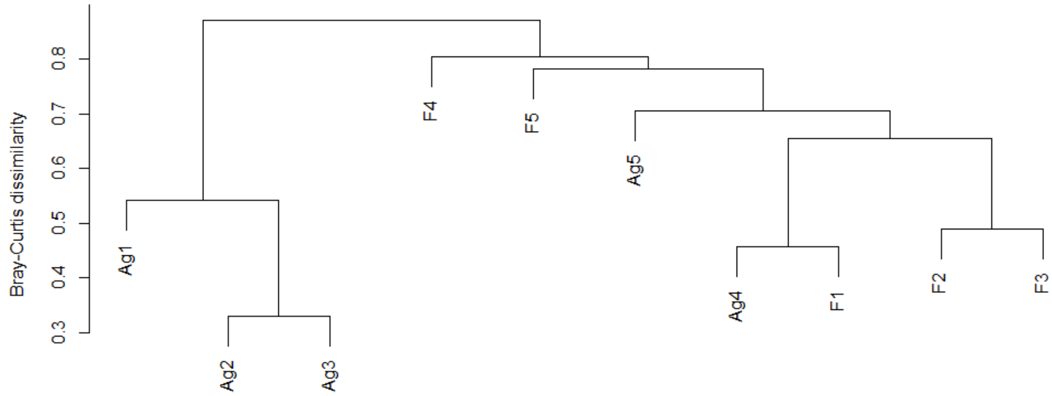

FIG. 11. Cluster dendrogram of agricultural and forested sites with similar communities. Ag = agriculture and F = forest.

The difference in richness between agricultural and forested land use was insignificant ($t = 2.08$, $df = 4.76$, p-value = 0.095, Fig. 12). The mean of richness for agricultural and forested land use were 12.4 (± SD 2.88) and 9.6 (± SD 0.89) respectively. Agricultural land use had a greater total richness but a larger variance between sites (var = 8.3) than forested land use (var = 0.8).

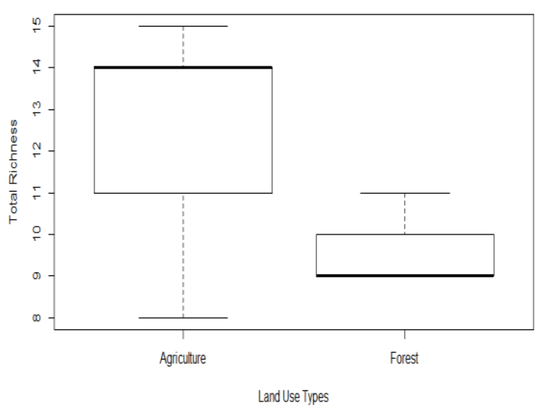

FIG. 12. Comparison of species richness between land use types. There was an insignificant difference between agricultural and forested land use in richness ($t = 2.08$, $df = 4.76$, p-value > 0.05)

The difference in abundance between agricultural and forested land use was insignificant ($t = 2.15$, $df = 8$, p-value = 0.063, Fig. 13). The mean of total abundance for agricultural and forested land use were 223.6 (± SD 132.81) and 93.0 (± SD 61.14) respectively. Agricultural land use had a greater total abundance but a larger variation between agriculture sites (var = 17638.2) when compare to the forested sites (var = 3738.2).

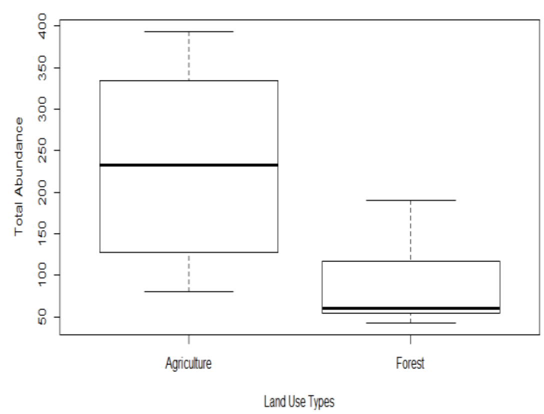

FIG. 13. Comparison of total abundance (number of individuals in each species) between land use types. There was an insignificant difference between agricultural and forested land use in total abundance ($t = 2.15$, $df = 8$, $0.06 < \text{p-value} < 0.07$).

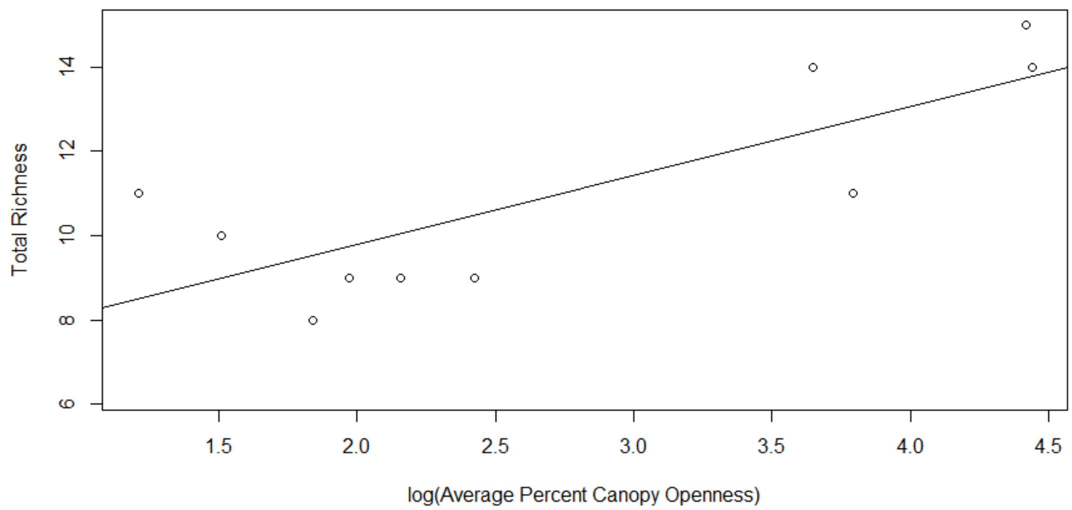

FIG. 14. Positive correlation between canopy openness and total richness amongst all sites (Multiple R-squared = 0.64, Adjusted R-squared = 0.59, F-statistic = 14 on 1, df = 8, t = 3.74, 0.005< p-value < 0.006).

Canopy openness

Richness

There was a significant positive correlation between canopy openness and richness (Multiple R-squared = 0.64, Adjusted R-squared = 0.59, F-statistic = 14 on 1, df = 8, t = 3.74, p-value = 0.0057, Fig. 14).

Abundance

There was also a significant correlation between canopy openness and total abundance (Multiple R-squared = 0.67, Adjusted R-squared = 0.63, F-statistic = 16 on 1, df = 8, t = 4.00, p-value = 0.0039, Fig. 15).

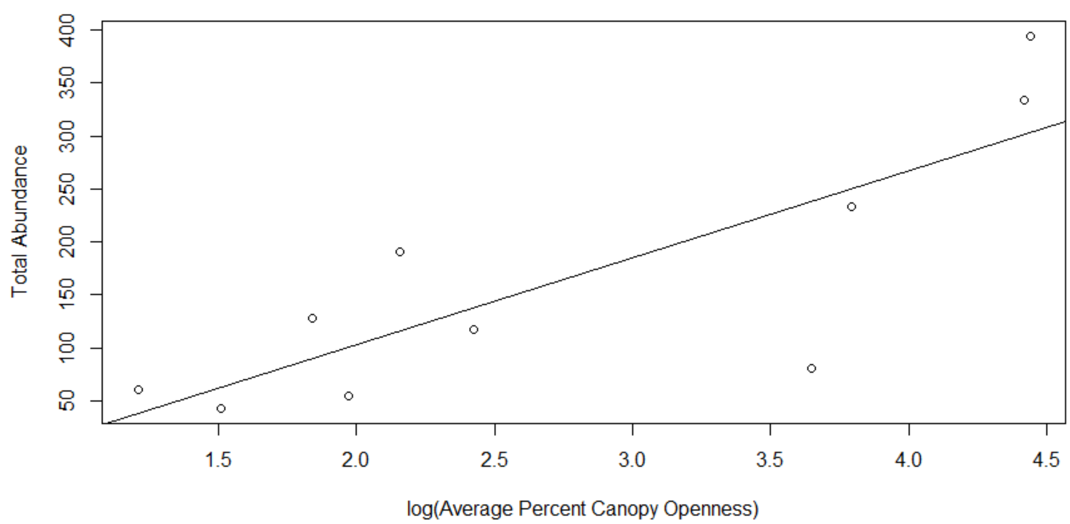

FIG. 15. Positive correlation between canopy openness and total abundance amongst all sites (Multiple R-squared = 0.67, Adjusted R-squared = 0.63, F-statistic = 16 on 1, df = 8, t = 4.00, 0.003 < p-value < 0.004)

Discussion

The purpose of this study was to compare stream habitats between two land use types (agriculture and forest) and examine the effects of agricultural land use on riparian zones, water quality, and freshwater biodiversity. This study also aimed to identify the correlation of canopy openness with freshwater biota's richness and abundance. This study showed that there was insignificant difference between land use types and all physical stream habitat parameters, except for riparian zone width. However, there were significant differences between land use types and canopy openness, temperature, conductivity, total dissolved solids, and salinity. In addition, this study suggested that land use types affect stream biodiversity, but it was not statistically significant. This study concluded that canopy openness is correlated to both species abundance and richness.

Stream habitat assessment

Riparian zone widths showed significant difference between land use types with agricultural sites scoring poor and forested sites scoring optimal. Thus, this result supports the causational relationship between the development of agricultural land with deforestation and physical disturbances on the environment (Dillaha et al. 1989). The negative anthropogenic impacts of agricultural land use were also supported by the results from the riparian zone assessment.

Riparian zone assessment

In agricultural sites, the dominant vegetations were weeds and grasses, while in forested sites the dominant vegetations were native trees and ferns. This physical change in plant species composition further supports the effects of agricultural land use on physical disruption of the natural stream habitat (Sweeney 1993, Meyer and Turner 1994).

In addition, agricultural sites had higher percentages of canopy openness, which indicates that they provide more energy to the stream ecosystem. The energy increase affects local stream conditions by directly increasing water temperature and indirectly increasing productivity (Quinn et al. 1997, Rutherford et al. 1999, Gücker et al. 2009).

The combination of differences in riparian width, predominant plant species, and canopy openness suggest that the microhabitats of streams are altered due to agricultural land use. These changes in riparian zones may be responsible for the differences found in water quality and freshwater biodiversity. It was noted that restoration of riparian zones by planting native vegetations would reestablish canopy shading in streams and alter the nutrient cycling, since light may lead to the growth of algae which would promote in-stream nutrient processing (Rutherford et al. 1999).

Water quality assessment

Land use types showed insignificant differences in pH, dissolved oxygen, and nitrates. Dissolved oxygen fluctuated greatly between and within sites, which may be due to differences in flow velocity, nutrient concentrations, riparian vegetation, temperature, and climate between different sampling days, times, and sites (Hem 1985, Mueller and Helsel 1999, Heartsill-Scalley and Aide 2003). Furthermore, nitrate levels at all sites, except F2, was consistent with the trend that agricultural sites had higher nitrate concentrations due to the usage of fertilizers and possibly even the alternations of the natural riparian zone habitat such as width, dominant vegetation, and canopy (Smith 1989, Rutherford et al. 1999). Nitrate levels should be evaluated more thoroughly because the result of this study was driven by one forested site (F2) that had the highest nitrate concentration, and there were no signs of nitrate input source upstream of this site.

Results also showed that agricultural sites were higher in temperature than forested sites. This result is consistent with the trend that increasing canopy openness and/or decreasing riparian widths yield(s) higher temperature readings (Smith 1989, Sweeney 1993, Rutherford et al. 1999). Since Ag5 had temperature readings consistent to forested sites, but a much smaller riparian width (15.5 meters), it is suggested that stream temperatures increase when widths are below 15 meters. This finding is supported by Davies and Nelson (1994)'s results that showed stream temperature increased only when riparian widths were below 10 meters in Australia.

There was a significance difference in conductivity, total dissolved solids, and salinity between agricultural and forested land use. High conductivity which is seen among agricultural sites means that there are high concentration of ions in the form of salts and inorganic compounds. Total dissolved solids account for these ions and dissolved organic matter, such as hydrocarbons and urea, while salinity accounts for the total concentration of all dissolved salts in water. Thus, this study showed that agricultural land use have higher concentrations of ions, dissolved organic matter, and dissolved salts than forested land use because the high nutrient soil characteristics associated with agriculture would increase conductivity, total dissolved solids, and salinity during runoff (Cooper et al. 1995).

Stream biodiversity

Shannon-Wiener diversity index, richness, and evenness within land use types

Shannon's diversity index showed that there was a great variation between agricultural sites. For example, Ag 4 site had the highest diversity, while Ag3 site had the lowest diversity despite both of them being adjacent to agricultural land use. Since the conditions for agriculture classification were not strict, this allowed for high variability between my sites within this land use category. This variability is seen even at a smaller scale when comparing richness and evenness, the two components of diversity (Figs. 8-9). Agricultural sites had greater variability than forested sites for both richness and evenness, which indicates that agricultural sites are different between each other. Moreover, through comparing both graphs, it was suggested that forested sites had greater relative evenness than relative richness when compared to all sites; however, there was an exception for Ag 4 and Ag5, which also followed this trend. Therefore, there are some indications for these two agriculture sites to be similar to forested land use types.

Moreover, both richness and evenness diagrams suggest that there is a higher number of species for agriculture sites, but there is a disproportionate abundance of a few species. A clear example is seen with Ag 3, in which there is high richness but low evenness. Graphically, it was suggested that the forested sites, as a land use type, had consistently higher evenness than agricultural sites, while the agricultural sites, as a land use type, had higher total richness.

Agricultural land use category was not well defined, which also resulted in variation of beta diversity between sites as seen with Bray-Curtis dissimilarity cluster (Fig. 10). Clustering of Ag4 and Ag5 with forested sites indicates that both agricultural sites look like forested sites in species composition. This may be due to their similarity in physical and chemical environment. Ag5 was the outlier within the agricultural land use category for riparian width, canopy openness, conductivity, total dissolved solids, and salinity and this site was comparable to forested sites in canopy openness, conductivity, total dissolved solids, salinity, and temperature. In addition, Ag 5 and Ag 4 were the only agricultural sites with native riparian vegetation, which increased canopy cover. The results from the Bray-Curtis dissimilarity cluster suggests that canopy cover and/or riparian species are influencing the beta diversity between each site.

This assumption is coherent with a study which found that buffer widths ranging from 8 to 27 meters supported similar stream invertebrate communities to those in native or mature plantation forest (Quinn et al. 2004).

Richness and abundance between land use types

There was insignificant difference in total richness of species and total abundance of all individuals between land use types, which may be due to the small sampling pool or the low biodiversity in tropical stream systems on islands when compared to mainlands

(MacArthur and Wilson 1967, Resh et al. 1990).

However, agricultural sites had greater species richness and abundance than forested sites, but lower evenness. This means that there were greater numbers of species and individuals in agricultural sites than forested sites, but agricultural sites were disproportionately dominated by a few species. This may be due to eutrophication of streams from agricultural runoff and increased canopy opening, which would increase the productivity of streams that only favors a few species (Rosenzweig and Abramsky 1993, Pearson and Connolly 2000, Vandermeulen et al. 2001, Weijters et al. 2009).

This logic is consistent with my observation and data. For example, the clustering of Ag 1, 2, and 3 together in the Bray-Curtis dendrogram shows that the communities within these sites were very similar. When integrating the physical and chemical environment into this analysis, it is indicative that Ag 1, 2, and 3 were not significantly different in habitat, riparian, and water quality parameters. These three sites had the highest percentages of canopy openness, shortest distances of riparian widths, and similar riparian vegetation. Due to a combination of these riparian zone characteristics, their water quality measurements were indifferent. The direct nutrient loadings into streams, due to short riparian widths, along with high canopy openness most likely resulted in the observed increase in algal growth that was not present at the other study sites. The increase in primary productivity of these three stream sites through algal growth created a new and optimal habitat for *Thiara granifera*, which is the species that was only found within all these three sites and in very high abundances, ranging from 110-330 individuals. *T. granifera* was able to become ecologically successful since they fed on algae, which was abundant at Ag1, 2, and 3 (Resh et al. 1990, Pointier et al. 1998)

Although there were statistically insignificant differences between land use types in total richness and abundance, graphically, it was suggestive that the two land use groups were different. Other studies have found that agricultural land use increases species richness and diversity (Moore and Palmer 2005, Johnson and Angeler 2014). This study is consistent with another research that showed a low intensity of agricultural practice and riparian thinning increases light, nutrients, and water temperatures, which in turn increase algal growth and macroinvertebrate abundance, without decreasing diversity (Quinn 2000). The latter study found that a decrease in diversity through the loss of the most sensitive taxa occurred only when intensified agricultural practices were performed.

Canopy openness

Finally, this study found that there was a significant positive correlation between canopy openness with total richness and abundance. Due to the high variability between my agricultural sites defined in this study, it is evident that canopy openness should be included in determining and identifying agricultural sites in the future in order to obtain more statistically significant results.

The implications of this study were that agricultural land use was affecting riparian zones, water quality, and steam biodiversity. The clear differences between agricultural and forested land use types suggest that there should be implementation of riparian management policies. Although there was no observed decrease in species richness as a result of agricultural practices, this study was limited by a small sample pool.

CONCLUSION

This study found significant differences between land use types and riparian zone width, canopy openness, temperature, conductivity, total dissolve solids, and salinity. There were suggestive differences between land use types in total species richness and abundance, but there was not enough statistical power to show a significant difference. The suggestive difference is supported by the significant positive relationship between canopy openness and richness and abundance, as canopy is a significantly different between land use types. However, more studies need to be conducted to show the direct causal relationships between land use and diversity. The effects of agriculture on tropical stream ecology should be evaluated in greater detail by well defining agricultural land use to include canopy openness and/or riparian vegetative widths. In addition, future studies should use a multi-habitat approach to examine the heterogeneity of streams adjacent to agricultural sites separately. For conservation management, future studies should also isolate riparian zones, water quality, and stream biota independently and conduct controlled experiments in order to examine the factors that affect biodiversity the most.

ACKNOWLEDGMENTS

I thank the Richard B. Gump Station and the University of California Berkeley for allowing me this opportunity to explore my passion for international field research in biology. I thank the professors: Vince Resh, Stephanie Carlson, Jonathon Stillman, and Brent Mishler and the graduate student instructors: Jason Hwan, Seth Kauppinen, and Jenna Judge for their support, advice, and guidance throughout the course. This project would not have been possible without the encouragement from my friends, family, and Hinano Murphy. Mauruuru roa!

LITERATURE CITED

Allan, J. D. 2012. Landscapes and riverscapes: the influence of land use on stream ecosystems. Annual Review of Ecology, Evolution, and Systematics **35**:257–284.

Barbour, M. T., J. Gerritsen, B. D. Snyder, and J. B. Stribling. 1999. Rapid bioassessment protocols for use in streams and wadeable rivers: periphyton, benthic macroinvertebrates and fish. Second edition. U.S. Environmental Protection Agency, Washington, D.C.

Barbour, M. T., and J. B. Stribling. 1991. Use of habitat assessment in evaluating the biological integrity of stream communities. in G. Gibson, editor. Biological criteria: research and regulation, proceedings of a symposium. U.S. Environmental Protection Agency, Washington, D.C.

Barbour, M. T., and J. B. Stribling. 1994. A technique for assessing stream habitat structure. Pages 156–178 Riparian ecosystems in the humid U.S.: functions, values and management. National Association of Conservation Districts, Washington, D.C.

Bu, H., W. Meng, Y. Zhang, and J. Wan. 2014. Relationships between land use patterns and water quality in the Taizi River basin, China. Ecological Indicators **41**:187–197.

Cooper, A. B., C. M. Smith, and M. J. Smith. 1995. Effects of riparian set-aside on soil characteristics in an agricultural landscape: Implications for nutrient transport and retention. Agriculture, Ecosystems and Environment **55**:61–67.

Cuffney, T. F., M. R. Meador, S. D. Porter, and M. E. Gurtz. 2000. Responses of physical, chemical, and biological indicators of water quality to a gradient of agricultural land use in the Yakima River Basin, Washington. Pages 259–270 in S. S. Sandhu, B. D. Melzian, E. R. Long, W. G. Whitford, and B. T. Walton, editors. Monitoring Ecological Condition in the Western United States. San Francisco.

Dillaha, T. A., R. B. Renear, S. Mostaghimi, and D. Lee. 1989. Vegetative filter strips for agricultural nonpoint source pollution control. Transactions of the American Society of Agricultural Engineers **32**:513–519.

Doyle, R. C., G. C. Stanton, and D. C. Wolf. 1977. Effectiveness of forest and grass buffer filters in improving the water quality of manure polluted runoff. American Society of Agricultural Engineers, St. Joseph.

Gücker, B., I. G. Boëchat, and A. Giani. 2009. Impacts of agricultural land use on ecosystem structure and whole-stream metabolism of tropical Cerrado streams. Freshwater Biology **54**:2069–2085.

Hauer, F. R., and V. H. Resh. 2007. Macroinvertebrates. Pages 435–454

Methods in stream ecology. Academic Press, San Diego CA.

Heartsill-Scalley, T., and T. M. Aide. 2003. Riparian vegetation and stream condition in a tropical agriculture – secondary forest mosaic. Ecological Applications **13**:225–234.

Hem, J. D. 1985. Study and interpretation of the chemical characteristics of natural water. U.S. Geological Survey Water-Supply Paper 2254. Third edition.

Johnson, R. K., and D. G. Angeler. 2014. Effects of agricultural land use on stream assemblages: Taxon-specific responses of alpha and beta diversity. Ecological Indicators **45**:386–393.

Johnson, S. L., and N. H. Ringler. 2014. The response of fish and macroinvertebrate assemblages to multiple stressors: A comparative analysis of aquatic communities in a perturbed watershed (Onondaga Lake, NY). Ecological Indicators **41**:198–208.

Karr, J. R. 1981. Assessment of biotic integrity using fish communities. Fisheries **6**:21–27.

Lorion, C. M., and B. P. Kennedy. 2009. Relationships between deforestation, riparian forest buffers and benthic macroinvertebrates in neotropical headwater streams. Freshwater Biology **54**:165–180.

Lowrance, R. R., R. L. Todd, and L. E. Asmussen. 1983. Waterborne nutrient budgets for the riparian zone of an agricultural watershed. Agriculture, Ecosystems, and Environment **10**:371–384.

Lowrance, R. R., R. L. Todd, J. Fail, O. Hendrickson, R. Leonard, and L. E. Asmussen. 1984. Riparian forests as nutrient filters in agricultural watersheds. Bioscience **34**:374–377.

MA DEP, M. D. of E. P. 1995. Massachusetts DEP preliminary biological monitoring and assessment protocols for wadable rivers and streams. North Grafton, Massachusetts.

MacArthur, R. H., and E. O. Wilson. 1967. The theory of island biogeography. Princeton University Press, Princeton.

Magette, W. L., R. B. Brinsfield, R. E. Palmer, J. D. Wood, T. A. Dillaha, and R. B. Reneau. 1987. Vegetated filter strips for agriculture runoff treatment. United States Environmental Protection Agency Region III, Philadelphia.

Meyer, W. B., and B. L. Turner. 1994. Changes in land use and land cover: a global perspective. Page 537. Second edition. Cambridge University Press, New York.

Mid-Atlantic Coastal Streams Workgroup, M. 1996. Standard operating procedures and technical basis: macroinvertebrate collection and habitat assessment for low-gradient nontidal streams. Dover, Delaware.

Moore, A. A., and M. A. Palmer. 2005. Invertebrate biodiversity in agricultural and urban headwater streams: implications for conservation and management. Ecological Applications **15**:1169–1177.

Mueller, D. K., and D. R. Helsel. 1999. Nutrients in the nation's waters-too much of a good thing? U.S. Geological Survey Circular 1136. National Water-Quality Assessment Program.

Oksanen, J., F. G. Blanchet, R. Kindt, P. Legendre, P. R. Minchin, R. B. O'Hara, G. L. Simpson, P. Solymos, M. Henry, H. Stevens, and H. Wagner. 2013. Vegan: community ecology package. R package.

Paul, M. J., and J. L. Meyer. 2001. Streams in the urban landscape. Annual Review of Ecology and Systematics **32**:333–65.

Pearson, R. G., and N. M. Connolly. 2000. Nutrient enhancement, food quality and community dynamics in a tropical rainforest stream. Freshwater Biology **43**:31–42.

Pointier, J. P., S. Samadi, P. Jarne, and B. Delay. 1998. Introduction and spread of Thiara granifera (Lamarck, 1822) in Martinique, French West Indies. Biodiversity and Conservation **7**:1277–1290.

Quinn, J. M. 2000. Effects of pastoral development. Pages 208–29 in K. J. Collier and M. J. Winterbourn, editors. In New Zealand Stream Invertebrates: Ecology and Implications for Management. Christchurch, NZ.

Quinn, J. M., I. K. G. Boothroyd, and B. J. Smith. 2004. Riparian buffers mitigate effects of pine plantation logging on New

Zealand streams: 2. Invertebrate communities. Forest Ecology and Management **191**:129–146.

Quinn, J. M., A. B. Cooper, R. J. Davies-Colley, J. C. Rutherford, and R. B. Williamson. 1997. Land use effects on habitat, water quality, periphyton, and benthic invertebrates in Waikato, New Zealand, hill-country streams. New Zealand Journal of Marine and Freshwater Research **31**:579–597.

Resh, V. H. 1995. Freshwater benthic macroinvertebrates and rapid assessment procedures for water quality monitoring in developing and newly industrialized countries. Pages 167–177 in W. S. Davis and T. P. Simon, editors. Biological Assessment and Criteria: Tools for Water Resource Planning and Decision Making. Lewis Publishers, Boca Raton.

Resh, V. H., J. R. Barnes, and D. A. Craig. 1990. Distribution and ecology of benthic macroinvertebrates in the Opunohu river catchment, Moorea, French Polynesia. Annales de Limnologie - International Journal of Limnology **26**:195–214.

Rosenzweig, M. L., and Z. Abramsky. 1993. How are diversity and productivity related? Pages 52–65 in R. E. Ricklefs and D. Schluter, editors. Species Diversity in Ecological Communities. University of Chicago Press, Chicago.

Rutherford, J. C., R. J. Davies-Colley, J. M. Quinn, M. J. Stroud, and A. B. Cooper. 1999. Stream shade: towards a restoration strategy. Department of Conservation, Wellington, New Zealand.

Smith, C. M. 1989. Riparian pasture retirement effects on sediment, phosphorus and nitrogen in channelised surface run-off from pastures. New Zealand Journal of Marine and Freshwater Research **23**:139–146.

Strickler, G. S. 1959. Use of the densiometer to estimate density of forest canopy on permanent sample plots. USDA Forest Service. USDA Forest Service, Pacific Northwest Forest and Range Exp. Sta. Research Note 180, Portland, Oregon.

Sweeney, B. W. 1993. Effects of streamside vegetation on macroinvertebrate communities of White Clay Creek in Eastern North America. Proceedings of The Academy of Natural Sciences of Philadelphia **144**:291–340.

Vandermeulen, M. A., A. J. Hudson, and S. M. Scheiner. 2001. Three evolutionary hypotheses for the hump-shaped productivity – diversity curve. Evolutionary Ecology Research **3**:379–392.

Weijters, M. J., J. H. Janse, R. Alkemade, and J. T. A. Verhoeven. 2009. Quantifying the effect of catchment land use and water nutrient concentrations on freshwater river and stream biodiversity. Aquatic Conservation: Marine and Freshwater Ecosystems **19**:104–112.