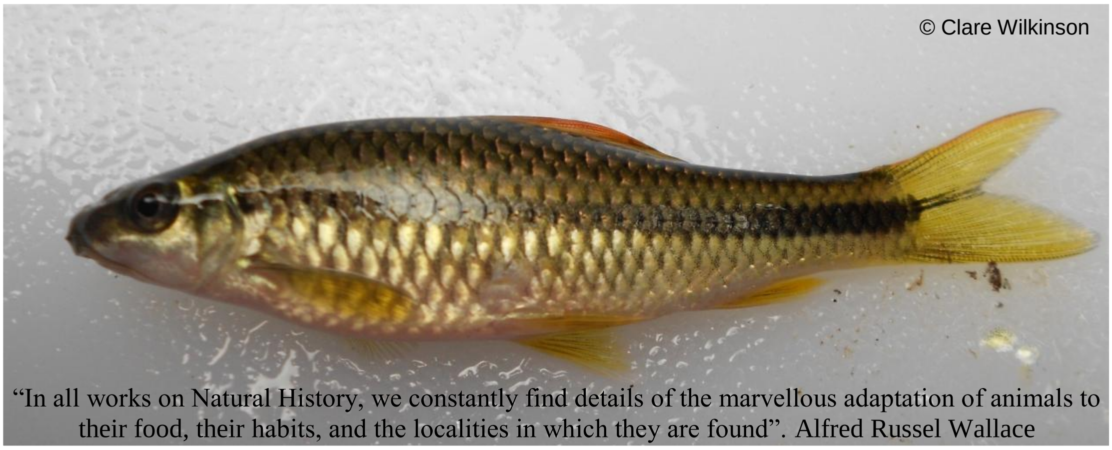
Composition and abundance of freshwater fish communities across a land use gradient in Sabah, Borneo



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A thesis submitted in partial fulfilment of the requirements for the degree of Master of Science and Diploma of Imperial College London

Declaration of own work

I declare that this thesis (insert full title)

Composition and abundance of freshwater fish communities across a land use gradient in Sabah, Borneo

is entirely my own work and that where material could be construed as the work of others, it is fully cited and referenced, and/or with appropriate acknowledgement given.

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LIST OF ACRONYMS

AIC	Akaike's Information Criterion
AICc	Akaike's Information Criterion corrected
ANOVA	Analysis Of Variance
CMR	Capture Mark Recapture
DBH	Diameter at Breast Height
DVCA	Danum Valley Conservation Area
GLM	Generalised Linear Model
LF	Logged Forest
OG	Old growth (forest)
OP	Oil Palm
PC1	Principle Component 1
PCA	Principle Component Analysis
PCoA	Principle Co-ordinates Analysis
PIT	Passive Integrated Transponder
SAFE	Stability of Altered Forest Ecosystems
VJR	Virgin Jungle Reserve

ABSTRACT

Malaysia has the highest levels of deforestation and production of palm oil around the world. Due to the paucity of data on the ichthyofauna of Sabah, understanding how this affects the diversity and abundance of freshwater fish, is of great interest as levels of deforestation and conversion to oil palm increase. This project used capture-mark-recapture to determine the abundance and dispersal of three focal taxa (*N. everetti, Tor dourensis, Rasbora*). Relative abundance was used to compare community similarity and absolute species turnover rates, across the land use gradient. 200m stream transects were established in old growth forest, logged forest and oil palm catchments as part of the Stability of Altered Forest Ecosystems project. Abundance of three focal taxa decreased as riparian vegetation and stream quality decreased; despite this the highest abundance estimate was in oil palm. Community analysis demonstrated a slight, albeit non-significant difference between sampling years ($p=0.054$) and land use ($p=0.098$) between sites, and relative abundance of species varied by year and land use. Results need to be treated cautiously due to low recaptures rates and a degree of over-dispersion in the data. The difference indicated over a land use gradient is corroborated, but alternate hypotheses are somewhat divided in the literature, as suggestions of differences at the mesohabitat scale and possible barriers to migration need to be further investigated. It is recommended that intensive research is conducted to obtain a full species list, for the area in order to fully assess how logging and conversion affects fish diversity and abundance in the short and long term. A critical assessment of trapping methodologies was undertaken and recommendations as to the use of trapping techniques are made accordingly.

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1. INTRODUCTION

Tropical rainforest ecosystems contain two-thirds of the world's terrestrial biodiversity (Gardener *et al.,* 2009) but are one of the most threatened ecosystems on the planet (FAO, 2006). Borneo lies within the Sunderland hotspot and regrettably has the highest deforestation rates around the world (Sodhi *et al.,* 2010), with annual rates of deforestation reaching 1.3% (FAO, 2010). This deforestation has resulted in a matrix of degraded forest and agricultural estates, with little primary forest existing outside forest reserves and protected areas (McMorrow and Talip, 2001). In addition a shift in agricultural practices for further economic gain now sees the highest levels of palm oil production in Malaysia and Indonesia, in the world. Oil palm plantations cover 1.2 million ha in the Malaysian state of Sabah in North Borneo, with more being created on degraded, selectively logged, secondary forest (McMorrow and Talip, 2001; Bradshaw *et al.,* 2009; Bruhl and Eltz, 2010). Forest degradation and the increasing demand for palm oil are accelerating forest loss. These factors, compounded with few studies addressing the effects on biodiversity, concerns conservationists and ecologists around the world (Laurance *et al.,* 2012).

1.1. Tropical freshwater fish diversity

The majority of the world's freshwater fish biodiversity is contained within tropical regions (Lowe-McConnell, 1987; Kottelat *et al.,* 1993; Kottelat and Whitten, 1996). Almost 10,000 species of freshwater fishes are currently recognised (Nelson, 1994) with many more species awaiting discovery and description (Kottelat and Whitten, 1996). The tropics of South-East Asia possess less fish species than that of South America, but have a greater diversity in the number of families (Kottelat *et al.,* 1993). Malaysia (including the states of Sabah and Sarawak in Borneo) is in the top 10 countries for highest freshwater fish diversity, with more than 600 described species (Kottelat and Whitten, 1996). Despite these figures, the ichthyofauna of Asia, and particularly Borneo, remains patchy because of concentrations of studies in easily accessible areas. The Kalabakan basin, where this project takes place, is not well documented due to a lack of appropriate species counts and descriptions (Dudgeon, 2000; Martin-Smith, 1998b).

1.2. Threats to freshwater fish in Borneo

As figures above state, logging and conversion to oil palm in Sabah is continuing and increasing at an unprecedented rate. A number of projects have studied forest fragmentation and deforestation (Lovejoy *et al.,* 1983; Casant *et al.,* 2002; Ritters *et al.,* 2000) but few have directly quantified the effects of fragmentation and land use change on tropical forest ecosystems (Ewers *et al,.* 2011), and fewer still have focused on fish. It is therefore essential to be able to draw comparisons between land uses, on the effects on biodiversity at the species and community level.

The removal of tropical forest cover during timber extraction represents an extreme form of disturbance, with potentially far-reaching effects on fish biodiversity. Positive and negative effects have been observed on freshwater fish abundance and community diversity across different land uses (Martin-Smith, 1998a; Iwato *et al.,* 2005; Nunakawa, 2005). A number of hypotheses are therefore presented: altered allochthonous inputs and solar regulation (Nunakawa, 2005), mesohabitats (Martin-Smith, 1998b) and significant barriers to migration (Martin-Smith and Laird, 1998). The lack of corroboration between studies strengthens the need to understand the impact land use change has on freshwater fish. Until now, no study has been conducted to quantify species abundance or community composition between streams covered by the SAFE project.

1.3 The Stability of Altered Forest Ecosystems (SAFE) Project

The SAFE project is one of the largest, established ecological experiments investigating responses of biodiversity to land use change and fragmentation (Ewers *et al*., 2011). Based in Sabah, Malaysia, the SAFE project's principal aim is to quantify the effects of logging, deforestation and fragmentation on the biodiversity and physical processes intrinsic to tropical forests. The project makes use of the planned expansion of oil palm activities over time. While the focus may be on the responses of forest fragmentation and conversion, the

SAFE project has been working on the biological impacts deforestation has on riparian strips. A group of six small, headstream catchments, currently in continuous logged forest have been instrumented and transects set up. The SAFE project has planned manipulations in the future riparian strip after conversion to oil palm, translating to differences in percentage forest cover in each catchment. The Brantian Tantulit Virgin Jungle Reserve, the nearby oil palm estates of Selangan Batu and Merbau, and continuous logged forest outside the experimental area will act as unlogged primary forest, established oil palm plantation and logged forest control sites, respectively.

1.4. Project Aims and objectives

This project aims to use capture-mark-recapture (CMR) to investigate the impact forest modification has on freshwater fish population abundance and composition. The main objectives are to:

1. Use CMR data to calculate the rate and extent of dispersal of fish to determine if such studies are justifiable.
2. Compare the abundance of three focal fish species across the land use gradient at the SAFE project.
3. Utilise previous data collected to compare the species turnover rates and community similarity of streams, across the land use gradient.
4. Critically assess the trapping methods used at the SAFE project (cast netting and bottle trapping), for catching fish.

2. BACKGROUND

2.1 Capture Mark Recapture of Freshwater Fish

2.1.1 Sampling and tagging methods

Scientists and managers of fish populations need to be able to collect reliable data, through all environmental conditions in order to understand populations and increase predictive capabilities for successful management (Barbour *et al.,* 2011). Sampling methods vary in skill and technical ability to catch fish, from electrofishing using two electrodes to pass a current through water, to traditional line and net methods. Electrofishing has short and long term problems, including immediate and delayed mortality (Neilson, 1998) which is not an option when sampling endangered species. Whereas, traditional methods may target specific species when attempting to monitor a whole stream community, through the chosen net size or the mesohabitat it is used in (Martin-Smith, 1998a; 1998b). Both of these methods enable CMR, one of the most widespread tools for monitoring to be used, and a variety of methods exist in order to do this (Barbour *et al.* 2011; Pine *et al.* 2003).

CMR methods are used for estimating the population size and survival parameters of fish populations (Pine *et al.,* 2003). CMR methods can take the form of chemical, internal (PIT, radio tags) or external marks (floy, streamer tags, fin clips and dyes) each with there own advantages and disadvantages. The methods vary over: the number of fish that can be tagged, the cost to implement such a study, the information that can be determined about the population and the fate of tagged fish (Barbour *et al.,* 2011; Pine *et al.,* 2003).

The majority of marking methods require individuals to be physically captured in order to be marked and recaptured to 'read' the tag. Traps can be set along stream or river transects, and need to be monitored over several trapping occasions. This may be carried out over successive days if determining abundance estimates or longer periods if further population parameters (for example survival and recruitment) are desired (Cooch and White, 2011). The simplest estimates of abundance can be calculated over two trapping occasions. On the first, individuals within a population are caught, marked, and released at the point of capture. After the determined time length (often the next day) the same method is used to re-sample the population. It is noted which animals already have a mark, and unmarked individuals are marked. Repeating this process over multiple trapping occasions allows capture histories (1's if encountered and 0's if not encountered for each capture occasion) to be created for each individual, forming the basis of CMR models.

As mentioned, many different methods can be used to tag fish. Passive integrated transponder (PIT) tags, introduced in the 1980s have substantially increased in popularity (Gibbons and Andrews, 2004). Tags are implanted internally, are available in a range of sizes (6-32mm length), allow for individual identification and are relatively low cost allowing for a high number of individuals to be tagged (Barbour *et al.,* 2011). PIT tags are ideal for long-term studies as they have no battery and very long life spans (Gibbons and Andrews, 2004). A benefit of PIT tags is that they give flexibility in recapture methods; through physical recapture or integration with telemetry or autonomous antenna detection systems and both have been successfully applied in freshwater environments (Jepsen *et al*., 2000; Muir *et al.,* 2001; Sandford and Smith, 2002). Disadvantages include low retention rates for some species (Barrowman and Myers, 1996) and sparse data outputs that are typical of traditional CMR studies if physical recapture is required (Adams *et al.,* 2006). The benefit of individual identification outweighs these negatives, making pit tagging a good option for conducting a mark recapture study in headstreams in Borneo.

2.1.2 Modelling fish abundance

Modelling the abundance of fish is dependent upon the efficiency of sampling methods and model assumptions being adhered to. Freshwater fish studies typically have low efficiency of sampling methods (Bayley and Austin, 2002) and very low recapture rates which affect the accuracy and precision of models (Pine *et al.,* 2003). It is because of this that studies need to clearly plan fieldwork, to maximise capture efficiencies, and ensure model assumptions are met.

Abundance of sampled populations can be explored and quantified using a variety of models that fall into two broad categories, open- or closed- capture models, each with different assumptions (Cooch and White, 2011). Closed-capture models assume the sampled population has no immigration, emigration, births or deaths during the study period (Otis *et al.,* 1978; White *et al*., 1982), whereas open-capture models allow the population to change in size and composition over the duration of the study (Lettink and Armstrong, 2003). Closed-capture models can be used over short time periods for monitoring freshwater fish, if the major assumption of population closure has to be met (Pine *et al.,* 2003; Lettink and Armstrong, 2003). Open-capture models can be used if closure cannot be met. The major assumptions of open models include: the equal catchability of marked and unmarked individuals at each sampling occasion, tags are not lost, tags are read properly, sampling is instantaneous, survival probabilities are the same for all individuals between each sampling occasion and the study area is constant (Cooch and White, 2012; Pine *et al*., 2003).

The parameters of simple open-capture models that are applied to sampled populations include: p – the probability of capture of both marked and unmarked animals, phi – the survival probability of both marked and unmarked animals between occasions, b – the probability of entrance into the population and t – time (Cooch and White, 2012). These parameters are used to estimate N, the initial population size of the sampled population. The parameters can be manipulated allowing the user to consider biologically relevant characteristics of animals. The most parsimonious model is sort, to find the optimal compromise between precision and bias, as this model will be the best fit to the data and only parameters that are useful in explaining the data will remain (Burnham and Anderson 2002). This is based on the model likelihood, Akaike Information Criterion (AIC) value and number of model parameters (Akaike, 1974; Burnham and Anderson, 2004). Multiple models may be plausible and the estimate of population abundance can be weighed in relation to these criteria.

2.2 Freshwater fish in Borneo

2.2.1 Overview of species

Despite the lack of information on freshwater fish in Borneo mentioned earlier, other catchments in Sabah adjacent to this project site have been sampled and species lists compiled (For example: Danum Valley by Martin-Smith and Tan, (1998); Kinabatangan by Lim and Wong, (1994); Lower Kinabatagan earlier this year near Danau Girang Field Centre). Martin-Smith and Tan (1998b) document finding 65 different species in the Danum valley, Lower Segama and Upper Kuamut Rivers with the Family Cyprinidae dominating the diversity. New taxa have been described from the Kalabakan catchment area (Tan, 2006) and a previous three month study at the SAFE project in 2011, caught over 2000 fish of 18 different species in 12 different genera (Bignet, Pers. Comms.). This indicates the high diversity in the region, as species accumulation curves have long tails, suggesting rare species can be found opportunistically over time (Martin-Smith, 1998b). However, significant differences are shown between regions and catchments separated by small geographical distances (Martin-Smith, 1998b).

The species diversity is dominated by the cyprinid family (Inger and Kong, 2002). Cyprinids vary in size and diet, from detritivores to active predators and are commonly caught in cast nets in smaller streams (Martin-Smith and Tan, 1998). The most interesting group of endemic fish are the ‘sucker fish’, found in rocky, fast-flowing streams. These fish feed on the algae on rocks and comprise the genera Gastromyzon, Glaniopsis, Protomyzon and Neogastromyzon (Inger and Kong, 2002).

Methods for capture of all species may be biased to certain species, for example cast netting for pool dwelling Cyprinids, but less so for ‘sucker fish’, due to the differences in vertical distribution within the water column of streams. Methods must therefore be diversified in order to obtain a picture of the whole stream community.

2.2.2 Role in tropical forest streams

The relationship between freshwater fish abundance and distributions, and the physical habitat, have been investigated in a range of temperate environments (Chipps *et al.,* 1994), but remain largely unexplored in tropical communities (Martin-Smith, 1998a). However, fish do play important roles in tropical streams (Chipps *et al.,* 1994), and it is accepted that detrital and algal-feeding fish are widespread in tropical streams which contrasts to the insectivorous fish that predominate in temperate streams (Flecker, 1992). These fish can affect the invertebrate assemblage by modifying the distribution and abundance of resources available to them (Flecker, 1992), thus impacting the whole stream biota. Martin-Smith (1998a) comments on the sharp discontinuities that can exist in small streams giving rise to distinct habitats, that fish species commonly show preferences towards, giving rise to differences in species assemblages.

Several additional roles have been suggested for freshwater fish. Firstly, that frugivorous fish have a possible role in seed dispersal in the neo-tropics (Horn *et al.,* 2011). Secondly, that fish could have a role as useful bio-indicators of water quality and pollution levels (Ahmad and Shuihami-Othman, 2010) as forests are logged and converted to oil palm.

These roles raise the profile of the potential use of freshwater fish in studies investigating stream quality, health and in studies comparing land use. This is coupled with a general lack of knowledge about tropical fish rarity, abundance and effects of land use change.

2.3 Effects of land use change on freshwater fish in Borneo

2.3.1 Logging

There are four main categories of human induced threat to fish: flow alteration or regulation, pollution, catchment alteration and overharvesting (Dudgeon, 2000). These cause streams to lose integrity, frequently resulting in less diversity and lower productivity of the ecological communities involved (Pringle *et al.,* 2000; Jackson *et al.,* 2001; Jungwirth *et al*., 2002). Logging and deforestation is a form of catchment alteration, causing changes in water flow and dramatic increases in sedimentation (Iwata *et al.,* 2003; Inoue *et al.,* 2003). These physical and hydrological modifications have been studied to some extent in the tropics, but the effects on freshwater fish remains poorly understood (Martin-Smith, 1998a).

Due to extensive deforestation in the tropics (Archard *et al.,* 2002), the last 25 years has seen a substantial research effort demonstrating that stream ecosystem maintenance is dependent upon land use regimes (Naiman *et al.,* 2000). Riparian forests have a strong influence on the abundance, distribution and assemblage of fish, particularly in headwater and low order streams (Inoue, 2005). Woody debris can alter the channel and provide cover and different habitats (Inoue and Nakano, 1998); whereas root systems reduce sediment inputs by providing bank stability (Tabacchi *et al.,* 1998), as high sediment loads have negative impacts on stream biota (Iwato *et al.,* 2005). The forest canopy intercepts solar radiation, reducing light intensity, lowering the autotrophic production providing a fixed food resource for fish (Inoue, 2005).

Focussing on deforestation in Borneo, Iwato *et al.* (2005) demonstrate that the habitat alteration caused by deforestation lowered the abundance of benthic fish and other taxa through sedimentation, but nektonic (free-swimming) fish did not suffer from these reductions. However, the practice of slash and burn agriculture provided long term degradation of streams which has a greater impact on vegetation and soil conditions than the reductions shown by logging regimes (Iwato *et al.,* 2005).

In contrast to this, Martin-Smith (1998a) found little or no evidence of biodiversity loss from streams in logged forest, compared to old growth forest, but shifts in community structure and dominance were recorded in some communities. Martin-Smith (1998b) illustrated that mesohabitats (pools and riffles) in streams were more important in determining community diversity than the logging history, which did not vary for pools or riffles when comparing logged and unlogged forest streams. However the time since logging activity did affect common cyprinids, but it was concluded the type of logging regimes studied had low impact on stream communities and species abundance through perceived methods of persistence and/or re-colonisation.

2.3.2. Conversion

Oil palm is one of the world's fasting growing crops and is significantly contributing to tropical deforestation (Fitzherbert *et al.,* 2008). Oil palm was first planted on the Malaysia peninsula in 1917 (Corley and Tinker, 2003). Since this period oil palm plantations moved to Sabah and Sarawak where more than 1 million hectares of forest were converted between 1990 and 2005 (Koh and Wilcove, 2008).

It is widely accepted that oil palm plantations support less biodiversity for terrestrial vertebrate and invertebrate taxa (Fitzherbert *et al.,* 2008; Chung *et al.,* 2000; Aratrakorn *et al*., 2006). Analyses by Downing *et al.* (1999) suggest primary productivity will increase by altering pristine tropical lands, leading to substantial alterations in freshwater communities. Despite this, less than 1% of publications in the literature on palm oil are related to biodiversity or conservation and call for further work to establish the threats for biodiversity (Turner *et al.,* 2008; Fitzherbert *et al.,* 2008;). No studies at present determine the effects of oil palm for freshwater fish, except to suggest that fish could be a useful bio-indicator of water quality and pollution levels (Ahmad and Shuihami-Othman, 2010). There are no studies on the effects of conversion of forest to palm oil. It would be interesting to determine to what extent palm oil plantations affect fish diversity in headstreams.

The conclusion of findings from logging and conversion to palm oil emphasise the importance and need of clear and implemented management practices (Iwato *et al.* 2005) of highly biodiverse forest habitats.

2.4 Study Site: A land use gradient in Eastern Sabah, Borneo

2.4.1. Yayasan Sabah Concession Area

Located in South-Eastern Sabah, the Yayasan Sabah Concession Area comprises one million hectares (ha) of logged forest. Extensive variation is shown in the quality and structure of the forest, representing the irregular logging intensities in the past 30 years (McMorrow and Talip, 2001). The forest has undergone a minimum of two rounds of selective logging with the majority undergoing additional exploitation. Extensive oil palm plantations at different stages of development surround the concession. Started in April 2013, part of the forest concession is being converted to oil palm plantation (The experimental area in Figure 2.1). The current conversion forms the basis of the SAFE project's fragmentation and conversion experiment. The project has established blocks and experimental stream catchments within an area of continuous forest (7,200ha). After the logging, clearance and conversion, the surrounding habitat will become oil palm plantation as it undergoes cultivation. For the extent of this project, the experimental stream catchments within the SAFE experimental area were comprised of continuous logged forest.

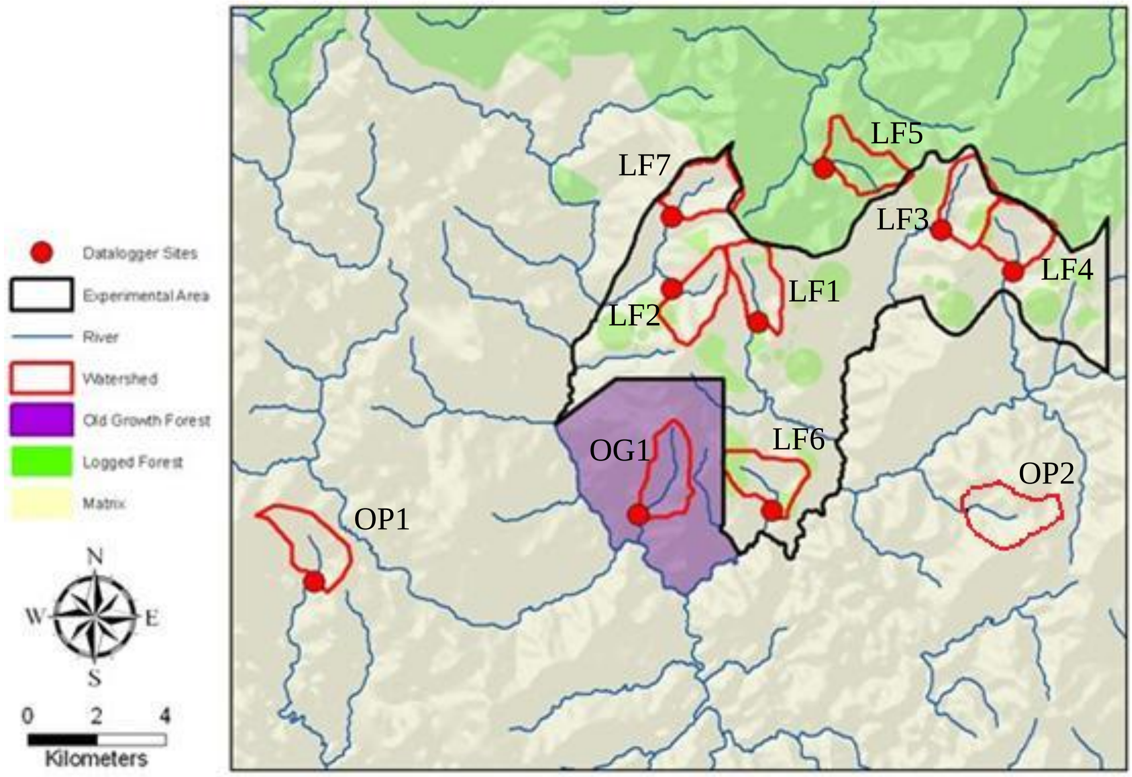

Figure 2.1. Map of study site. The SAFE project experimental area in Sabah, Borneo is delineated by the black line and experimental stream catchments by red lines. Catchments within the experimental area and to the North are logged forest, and the two outside are oil palm.

2.4.2 Virgin Jungle Reserve (VJR)

The Brantian Tantulit VJR is located adjacent to the South-East corner of the experimental area (Figure 2.1). VJRs are areas of old growth forest that have been protected and are intended for research and conservation by the Sabah Forest Department (http://www.parks.it/world/MY/Eindex.html). This reserve has been established as a control for old growth forest and has a stream catchment matched in area and slope, to those in the SAFE project experimental area (Ewers *et al.,* 2011).

2.4.3 Benta Wawasan Oil Palm Plantation

Similarly, adjacent to the Yayasan Sabah Concession Area, is the Benta Wawasan oil palm plantation that covers 45,601ha over 10 estates with oil palm varying in age (http://www.bentawawasan.com.my). The estates of Selangan Batu (South-West) and Merbau (South-East) have become established as control sites for oil palm catchments within the SAFE project framework.

3. METHODS

3.1 Framework

This project uses CMR to determine the abundance of freshwater fish species across the land use gradient provided by the SAFE project, ranging from old growth and logged forest to oil palm plantation. Dispersal rates of freshwater fish at the VJR were investigated to determine if CMR would provide sufficient recapture rates across a 200m transect. Capture histories for three focal species were subsequently created from trapping sessions at eight experimental streams. This data was compiled with measures of land use (stream characteristics and riparian vegetation quality), within linear models to evaluate how abundance varies across streams at the species level. CMR data was also utilised to critically assess the use of trapping methodologies for the three focal species across the land use gradient. At the community level, multivariate ordination and absolute species turnover rates were calculated to determine community similarity and how communities have changed over two years.

3.2 Data Collection

3.2.1 Project design and location

The SAFE project catchments have approximately equal area and slope. Catchments contain headstreams which are typically 2-3m wide, 2km long and enclosed by the forest canopy (Ewers *et al.,* 2011). Sampling was conducted at five SAFE project streams (logged forest: LF1-5) and three other streams outside the experimental area, chosen to match experimental catchments in size and slope, between April – June 2013. The other streams consisted of one in the VJR which is considered as old growth (OG1) forest, and two in palm oil plantations (OP1 and OP2). Sampling transects, 200m in length, were located within 100m of an in stream data logger for all streams except OP2 that does not have a data logger (data loggers are not directly adjacent to transects as they were installed after transects were established). The Function of the data logger is described below (Stream variables) and the location of all streams and data loggers can be seen in Figure 2.1.

3.2.2 Trapping

Fish were trapped using two methodologies, bottle trapping and cast netting, at all experimental streams. These methods were based on previous experimental trapping of fish at the SAFE project, September – November 2011, streams: LF1-7, OG1 and OP1. If possible, all streams were sampled for 6 consecutive days in order to obtain accurate recapture data to model population abundance. Due to time constraints on fieldwork and public holidays in Malaysia during the study period, LF5 and OP1 were sampled for 5 days and LF3 was sampled for 4 days, totalling 720 trap nights for bottle traps and 1009 cast nets were thrown. Sampling always started at the 0m point at the downstream end of the transect, moved upstream and cast netting was completed prior to the setting or checking of bottle traps (except on the first day of sampling, as the 200m transect had to be marked out), providing minimum disturbance to fish.

Bottle traps, made from recycled, plastic Blue Sky water bottles (Figure 3.1a), were set every 10m along the 200m transect, starting at the 10m point. Traps were placed facing upstream where possible and flush with the stream bed. Bread and fish were used to bait alternate traps, and traps were set at each of the experimental streams for five consecutive days. Traps were monitored every 24 hours; fish were removed and held in containers before release. The trap was re-baited and re-set in the same location. In the event of a flood in the previous 24 hours, causing traps to be lost, new traps were baited and set. In addition, fish were caught every morning of the six trapping days using a 9 foot cast net, with ¼ inch holes. Between 15-25 throws of the cast net were made along the 200m transect at each experimental stream, with the exact number determined by the number of pools appropriate for its use along the transect (Figure 3.1b). Fish were removed from the net by hand and held in containers prior to release. All fish captured were identified to genus level and to species if possible (using Inger and Kong, 2002 and a freshwater fish list by Hoek-Hui, 2013), body length measured, tagged if possible and released at the point of capture.

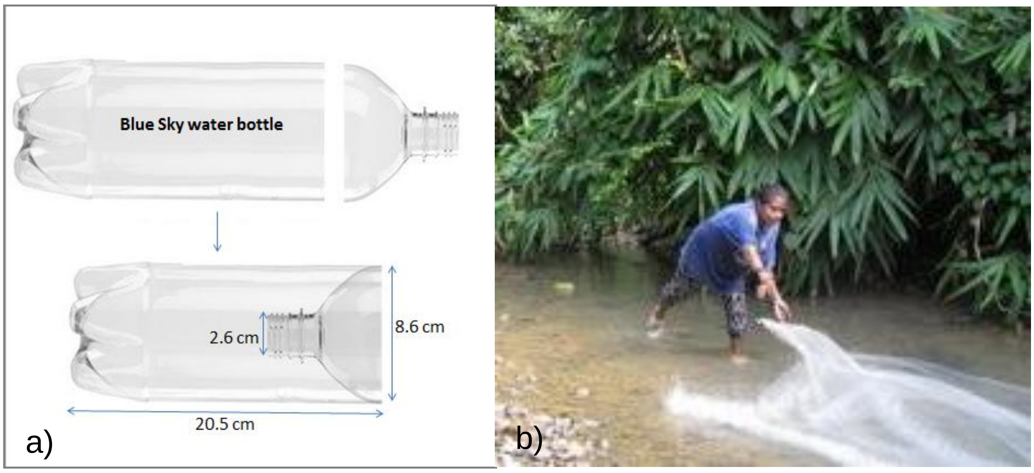

Figure 3.1. Fish trapping methods. a) Blue sky water bottles had the top removed and inverted to create a trap. Five holes were made in the bottom of the bottle to allow for some water flow and traps were tied to vegetation on the stream bank. b) Photo of Research Assistant, Maria, throwing a cast net at OP2 stream.

3.2.3 Tagging protocol

A pilot study was conducted (as recommended when conducting CMR studies in order to assess each species responses to anaesthesia, trapping, tagging, and the precision of estimated parameters (Pine *et al*., 2003)), with fish caught from a non-experimental stream. Several fish of each species were killed by an overdose of clove oil and dissected to determine an appropriate body cavity and method in which to insert PIT tags. It was decided to exclude all 'suckerfish', in the genera Protomyzon, Gastromyzon and Betta because their flattened morphology provided no obvious body cavity large enough in which to insert the tag. All other species with a body length (nose to tail tip) of 6cm or over were tagged, as a conservative measure of 5.5cm recommended in Baras *et al.* (2000).

The use of clove oil as an anaesthetic was tested on all species to be tagged to determine the appropriate concentration that provided sufficient anaesthesia coupled with rapid recovery as the effects of clove oil vary with water temperature, fish species, size of fish and actual eugenol concentration (Blackman, 2002; Coyle *et al.,* 2004; Dolezelova *et al.,* 2011). A concentration of 30mg/L of clove oil was used as time to reach anaesthesia was on average one minute for the different species and up to three minutes for recovery. This is consistent with recommended doses in other studies (40mg/L – Bayley and Austen (2002), and Blackman (2002); 30-120mg/L – Hajek *et al.* (2006)).

Not included in the pilot was the determination of survival rates because previous studies have shown a tag-related mortality level near zero (Ombredance *et al.,* 1998; Baras *et al.,* 1999 and 2000; Zydlewshi *et al.,* 2001). There was no need to investigate tag retention as the 'scar' from inserting the tag doubled as a secondary mark on the fish.

Fish to be tagged were held in containers after capture. The fish were transferred and placed in 5L of water containing 30mg/L of clove oil. Fish were held until total loss of equilibrium was reached, with slow but regular opercular rates indicating a level of surgical anaesthesia had been reached (Coyle *et al.,* 2004). A 3mm incision was made posterior and ventral to the pectoral fin, and an 11 x 2mm PIT tag (Biomark®) was inserted into the abdominal cavity and read with the aid of a portable reader. After tagging, fish were held in 5L of water until they had recovered, gained equilibrium and were released at the point of capture. Any fish that did not recover, or were unable to swim, were not released. There was no evidence of tag loss or tag failure during the project.

3.2.4 Forest quality variables in the riparian zone

Four forest quality variables were measured in the riparian zone in order to create an index of riparian vegetation quality that reflects levels of human disturbance and land use (Table 3.1). Riparian vegetation surrounding each stream was assessed every 50m, for 500m upstream from the 0m point on both banks. At each 50m point, measurements were taken approximately 10m up the bank, or the nearest area of level ground beyond that.

Table 3.1. Variables measured to indicate the riparian vegetation quality: How the variables were measured and the scales used to measure each.

Variable	Measure	Variable scale and description
Canopy openness	Densitometer	Percentage 0-100. Four measures taken: upstream, away from river, downstream and towards river
Percentage vine cover	Visually	Percentage: 0-100
Density of forest trees	2 Factor Relascope	The number of trees appearing larger than the viewing-window were counted, whilst standing at the measurement point and making a half turn movement starting upstream, turning away from the river.
Forest quality	SAFE project Forest quality scale	0= Oil palm 1= Very poor: No trees, open canopy with ginger/vines or low scrub. 2 = Poor: Open with occasional small trees over ginger/vine layer. 3 = OK: Small trees fairly abundant/canopy at least partially closed 4= Good: Lots of trees, some large, canopy closed 5 = Very good: Closed canopy with large trees, no evidence of logging

3.2.5 Stream variables

Data loggers and sensors are located in seven of eight of the experimental streams (excluding OP2), near the start of the 200m transect. Three sensors are attached to a metal gantry in the stream with cables leading to a Campbell Data Logger on the bank. The sensors record multiple variables, while analysis of monthly water samples provides levels of dissolved inorganic chemicals. Four stream characteristic variables (mean, minimum and maximum values) have been provided by A. Nainar (Pers. Comm.). Water level (stream discharge) and turbidity (suspended sediment concentrations) are measured every five minutes by the in stream sensors. Nitrate and phosphorus concentrations are obtained from analysed monthly water samples. These four measures are indicators of stream characteristics; and provide insight into the catchment allochthonous inputs to streams, allowing for comparisons across the land use gradient.

3.3 Data analyses

3.3.1 Dispersal

The distance travelled by fish indicates the rate and extent of dispersal, which could justify the use of mark-recapture to study fish over a 200m transect or prove that 200m is not an adequate distance. This data could also be essential information if attempting to protect endangered or protected fish or pescivorous species. OG1 was intensively re-sampled for three days, four weeks after the first 6 days of sampling. Bottle traps were set every 10m, and 55-60 cast nets thrown along a 600m transect (the experimental transect at 200-400m). Distance since last capture was calculated for each recaptured individual of the three focal species and plotted on histograms to compare dispersal rates. The null distribution was calculated from the probability of being able to be recaptured at each sampling point.

3.3.2 Abundance estimation

The two most common species and one genus (*Nematobramis everetti, Tor dourensis* and the genus *Rasbora*) were modelled using the programme MARK (White and Burnham, 1999) to obtain population size (N) estimates for each stream they were present in. These species and genus were chosen due to sufficient recapture data, and the genera grouped to prevent any mis-identification. Only streams providing recapture rates could be considered for modelling. Data input consisted of capture histories, compiled from both bottle trap and cast netting methods, detailing the unique ID of successive captures for each individual. Capture histories were input into the programme MARK and population estimates derived using the POPAN formulation of Jolly Seber models (Jolly, 1965; Seber, 1965) for live encounters and recaptures, using the variables: p, phi, b and t.

A fully time dependant model was first used to model the data (Table 3.2) as it consisted of the greatest number of parameters and was therefore tested for goodness of fit using RELEASE, a programme run within MARK. RELEASE uses data (in the form of live recaptures) to calculate a c-hat score, which measures the goodness of fit of the models and degree of dispersion of the data (Cooch and White, 2012). The results of tests 2 and 3 from RELEASE enabled the c-hat score to be calculated (c-hat = $\sum X^2 / \sum df$) and the models could then be adjusted to account for dispersion of the data (Cooch and White, 2012). Due to low recapture rates and low captures for some species and/or locations, several RELEASE goodness of fit tests were unable to be performed due to 'insufficient data'. Populations for each species were grouped and the test re-run to give a c-hat value applicable to each species. C-hat values were greater than one for *Rasbora* (1.524) and *N. everetti* (1.307) and were adjusted accordingly in Mark to account for over-dispersion in the data. A C-hat value of one was maintained for *T. dourensis* as the c-hat value was less than one (0.486), as there is no consensus within the literature to estimate under-dispersion of data (Cooch and White, 2012).

Additional models were run, varying time dependence of each or all parameters, by manipulation of the PIM charts (Table 3.2). The parm-specific link function was specified in the run menu, and in the design matrix: Sin for *p* and *phi* parameters, MLogit(1) for all *b* parameters as they need to sum to 1, and Log for *N*, the population parameter (Schwarz and Arnason, 2012). Models were compared using AIC values; the model with the lowest AIC is the most parsimonious and thus considered the closest to the 'true' scenario. Models were averaged to calculate a population estimate; accounting for all variation in the models, by taking a weighted average according to the models corrected AIC (AICc) weight and model likelihood. Models were averaged using *real* values, to calculate the *initial population size, N*, for each population that could be modelled for the three focal species. If no recapture data was available for the three focal species, the number of unique individuals was used as a measure of minimum population size.

Table 3.2. Models used to estimate abundance of *Rasbora*, *N. everetti* and *T. dourensis* in the programme mark. The variables in the models are: *p* – the probability of capture of marked and unmarked individuals, *phi* – the survival probability of marked and unmarked individuals, *b* – the probability of individuals entering the modelled area and *t* – time.

Model	**Description**
p(t), phi(t), b(t)	Time dependant catchability and survival
p(.), phi(t), b(t)	Constant catchability over time and time dependant survival.
p(.), phi(.), b(t)	Constant survival and catchability over time.
p(t), phi(.), b(t)	Constant survival over time and time dependant catchability.

3.3.3 Comparison of abundance across the land use gradient

Principle component analysis (PCA) was performed (as was all analysis in the statistical programme, R (R Development team, 2013)) on four variables of riparian vegetation quality and four variables of stream characteristics providing two indices across the land use gradient. To determine if the indices of land use effectively split up streams by land use, mixed effects models (with and without the fixed effect) were run on the Principle Component 1 (PC1) values (as a function of land use (fixed) effect and stream (random) effect). ANOVA was then conducted to determine if land use had a significant effect, indicating that the indices represent the land use gradient. Correlation was tested between riparian vegetation and stream characteristics to determine that they were different, even though they are not independent.

Poisson's Generalised Linear Models (GLMs) were performed for each focal species population abundance estimates to determine which variables (PC1 riparian vegetation, PC1 stream characteristics and/or Land use) predicted the data. Poisson's GLMs were used as the outcome variable (abundance) is count data, with a set of continuous predictor variables. GLMs were run, starting with the most parameterised model (*Abundance estimate ~ PC1value of riparian vegetation x Land use x PC1 value of streams*) and undergoing model simplification. The stream OP2, was removed from analysis as no data logger is present in the stream, resulting in no stream characteristics data.

The AIC value of the models was compared to that of the null model (~1) and ANOVA was conducted to determine if there was a significant difference to the null model.

3.3.4 Methodological comparison

Mixed effects models of standardised catch data (per trap, per day), with the abundance estimate from MARK as the fixed effect and stream as random the random effect were run. They were used to determine which trapping method, bottle traps or cast netting, was most appropriate in predicting the abundance of the three focal species. ANOVA was used to test the significance of models against the null model. The chi-squared value and slope of the model were used to compare the predictive power of models.

3.3.5 Community level comparisons across the land use gradient

Multivariate ordination (Principle Co-ordinates Analysis (PCoA) using the Bray distance function) in the vegan package of R (Oksanen *et al.,* 2013), was used to test the similarity between species and streams. A site-by-species matrix was created utilising catch data from this sampling season and from a previous sampling season, September - December 2011. Both indices of land use were fit to the analysis indicating the direction of greatest change.

MANOVA tests were run, to determine the effect of year, land use and the interaction on the Axis's co-ordinates of each stream to test for a temporal difference and/or land use effect. ANOVA were subsequently run to test the effects of year and land use on relative abundance of two genera, *Rasbora* and *Tor* (grouped to account for identification issues), and three species (*N. everetti*, *Protomyzon griswoldi* and *Systomus sealei*) all of which were distinct in the PCoA analysis.

Absolute species turnover rates (β, as a measure of beta diversity) could also be calculated across the land use gradient between the two sampling years indicating the change in species composition over time. $\beta = (s_1 - c) + (s_2 - c)$, where s_1 = total number of species in first community, s_2 = total number of species in second community and c = number of species in both communities.

GLMs were fitted of absolute turnover rates across the land use gradient indices of riparian vegetation quality and stream characteristics.

4. RESULTS

4.1 Fish Dispersal

Dispersal varied from 0 – 300m for all the fish that were recaptured from the three focal species at the OG1 stream (Figure 4.1). *T. dourensis* moved the least distance, but all fish tended to stay in the same locality over the study period, with 89% of fish recaptured less than 20m from the point of last capture. This is despite environmental conditions varying over the project duration, including regular flood events after periods of heavy rainfall. The lack of dispersal indicates that a mark-recapture study will be appropriate for fish in these headstreams and that a 200m transect is a sufficient size to capture a 'snapshot' of species abundance.

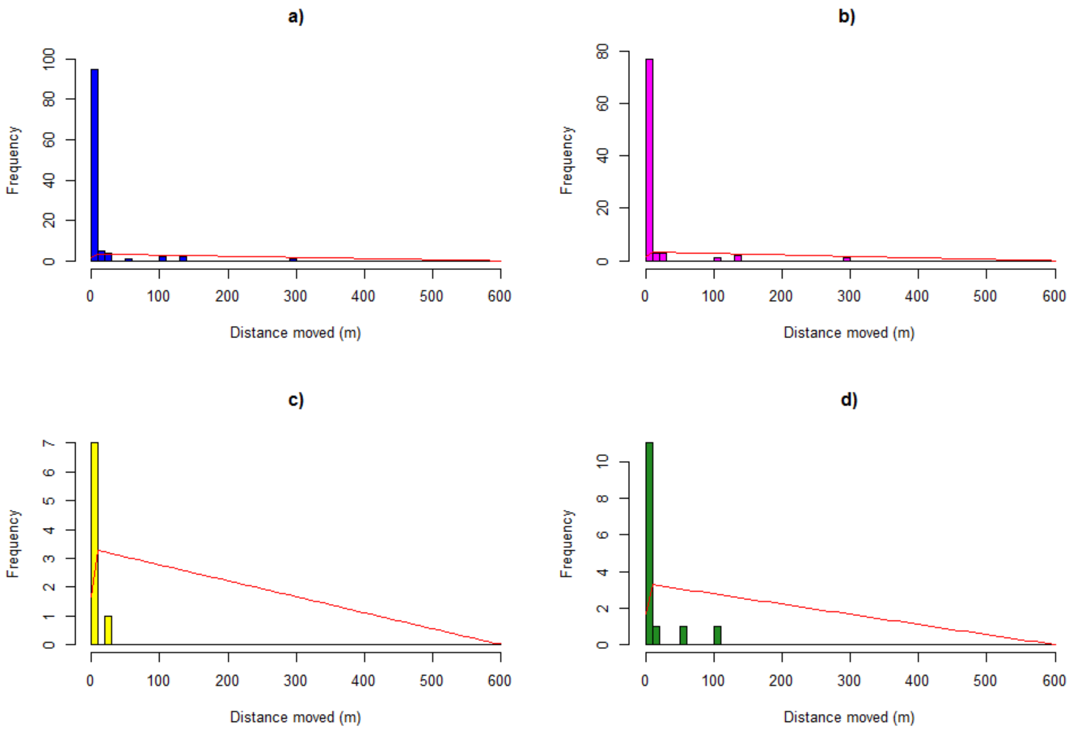

Figure 4.1. Dispersal of fish from all and the three focal species/genus. Dispersal is measured by the distance (metres) moved by the fish since the last capture, for a) all fish, b) *N. everetti* c) *T. dourensis* d) *Rasbora*. The red line illustrates the expected distribution of fish movement, calculated from the probability of moving each distance from known trapping points in the 600m transect.

4.2 Indices of land use gradient

For both indices, the PCA found a gradient across the land uses, from OG (negative numbers) to OP (positive numbers) with LF ranging between the two. However, the differences between the land uses was much smaller between OG and LF, than OP. ANOVA of mixed effects models of the riparian vegetation PC1 values, showed that the index of riparian vegetation effectively separated oil palm streams (Figure 4.2) from logged forest and old growth (p=0.027, chisq=7.188, df=2). The index of stream characteristics was also shown to effectively split up stream PC1 values in ANOVA tests of mixed effects models (p<0.001, chisq=21.559 df=2). However, the tests were inconclusive as there are only single values available for each stream and only one replicate of each OG and OP, providing little variation.

Correlation between the two indices was tested to determine the relationship between the land use variables. A weak positive correlation (r=0.283) was indicated as expected, but there was no significant correlation between the two variables (p=0.539, t=0.659, df=5), indicating responses to land use will be different for the two indices.

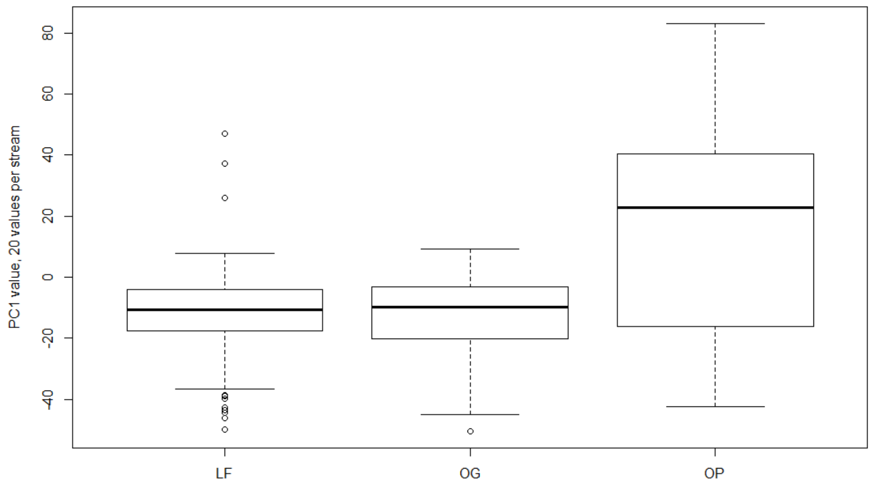

Figure 4.2. PCA of riparian vegetation variables effectively splits streams up into different land uses. Two models: PC1 values riparian vegetation index as a function of land use (fixed) effect and stream (random) and a null model excluding the fixed effect, were significantly different in an ANOVA test p = 0.02749 (chisq= 7.188, df= 2).

4.3 Population modelling and estimation

Only one stream had caught individuals of a focal species with insufficient data to calculate abundance. Three *T. dourensis* were caught at OP2 and this number was used as a minimum abundance estimate since abundance could not be modelled. Zeros indicate that individuals of the focal species/genus were not found.

Population abundance has large variation across streams for the three focal taxa (Table 4.1). The largest population modelled was *N. everetti* at OP2 and no fish of the three focal taxa were caught at two streams (OP1 and LF2). Despite this, the three focal taxa were present at all three land uses and subsequent models will explore the relationship of abundance across land use. The large standard errors for abundance estimations, is mainly due to low capture and recapture rates, but also the altered c-hat values for *Rasbora* and *N. everetti* indicating the level of over-dispersion in the data.

The model of best fit varied by species and stream, for 50% of estimates p(.), phi(.), b(t) was the best fit to the data. This model indicated a constant catchability and survival over time, suggesting that all assumptions of mark recapture modelling were satisfied. The best model for 25% of estimates was p(.),phi(t),b(t) indicating that survival was time dependant, but the assumption of equal survival of marked and unmarked animals at each sampling occasion is still met. The best model for the remaining 25% of estimates was p(t),phi(.),b(t) suggesting catchability varied between sampling occasions but the assumption of equal catchability of all animals at each sampling occasion is still met. The variation in catchability in sampling occasions may be related to water levels that subsequently reflect weather conditions and flood events.

Table 4.1. Abundance estimates (over 200m transect) for three focal species/genus at each site. Standard error (SE) and the abundance (Estimate) are calculated from model averaging in MARK. The output of each model, AIC value and model likelihood are detailed in Appendix I for each population.

Genus/Species	Stream	Estimate	SE	Best model
N. everetti	LF1	28.16	6.44	p(.),phi(.),b(t)
	LF4	108. 39	18.64	p(t),phi(.),b(t)
	LF3	0	0	-
	LF5	0	0	-
	OP2	794.16	420.13	p(t),phi(.),b(t)
	OG1	214.22	43.18	p(t),phi(.),b(t)
	OP1	0	0	-
	LF2	0	0	-
Rasbora	LF1	82.72	14.63	p(.),phi(t),b(t)
	LF4	128.89	65.91	p(.),phi(.),b(t)
	LF3	0	0	-
	LF5	357.91	129.72	p(.),phi(.),b(t)
	OP2	103.82	68.64	p(.),phi(.),b(t)
	OG1	137.66	45.06	p(.),phi(.),b(t)
	OP1	0	0	-
	LF2	0	0	-
T. dourensis	LF1	0	0	-
	LF4	347.02	165.5	p(.),phi(t),b(t)
	LF3	37.22	15.06	p(.),phi(.),b(t)
	LF5	0	0	-
	OP2	3	0	-
	OG1	677.67	249.21	p(.),phi(t),b(t)
	OP1	0	0	-
	LF2	0	0	-

4.4 Comparison of abundance across the land use gradient

As abundance of each species decreases, the quality of vegetation decreases (changes from old growth and logged forest to oil palm) and the quality of stream conditions decreases (changes from old growth and logged forest to oil palm), as shown in Figure 4.3. At a point of about zero, for each land use index, abundance estimates of the three taxa drop to zero. This suggests that once land use quality has deteriorated beyond that point, populations of this species cannot be sustained at that level of land use disturbance.

All poisons GLMs for the three focal taxa were significantly different to the null model ($p<0.001$). The model AIC value, deltaAIC and ANOVA test results are in Appendix II. The AIC values demonstrated that the model: *Abundance estimate ~ PC1value of riparian vegetation* ***x*** *Land use* **+** *PC1 value of stream characteristics* was most significant for *N. everetti* and *T. dourensis*. The model indicated that as the vegetation quality increases along the index gradient (OG to OP) the amount the abundance decreases is dependent upon the land use. However, the lack of data points for OG and OP prevent interpretation of land use interaction. There was no interaction with the PC1 of stream characteristics indicating that stream characteristics influences abundance in a linear relationship but is not influenced by the PC1 value of riparian vegetation or land-use.

In comparison, the most parameterised model (*Abundance estimate ~ PC1value of riparian vegetation* ***x*** *Land use* ***x*** *PC1 value of stream characteristics)* had the lowest AIC value for *Rasbora*. The model shows that as the vegetation quality increases (OG to OP) along the index gradient and stream quality increases (OG to OP) along the index gradient, the abundance decreases with respect to land use (as above, only the LF interaction can be explored). This interaction or inclusion of all response variables in the models was expected, as the variables are not independent and all influence the environment the fish inhabit.

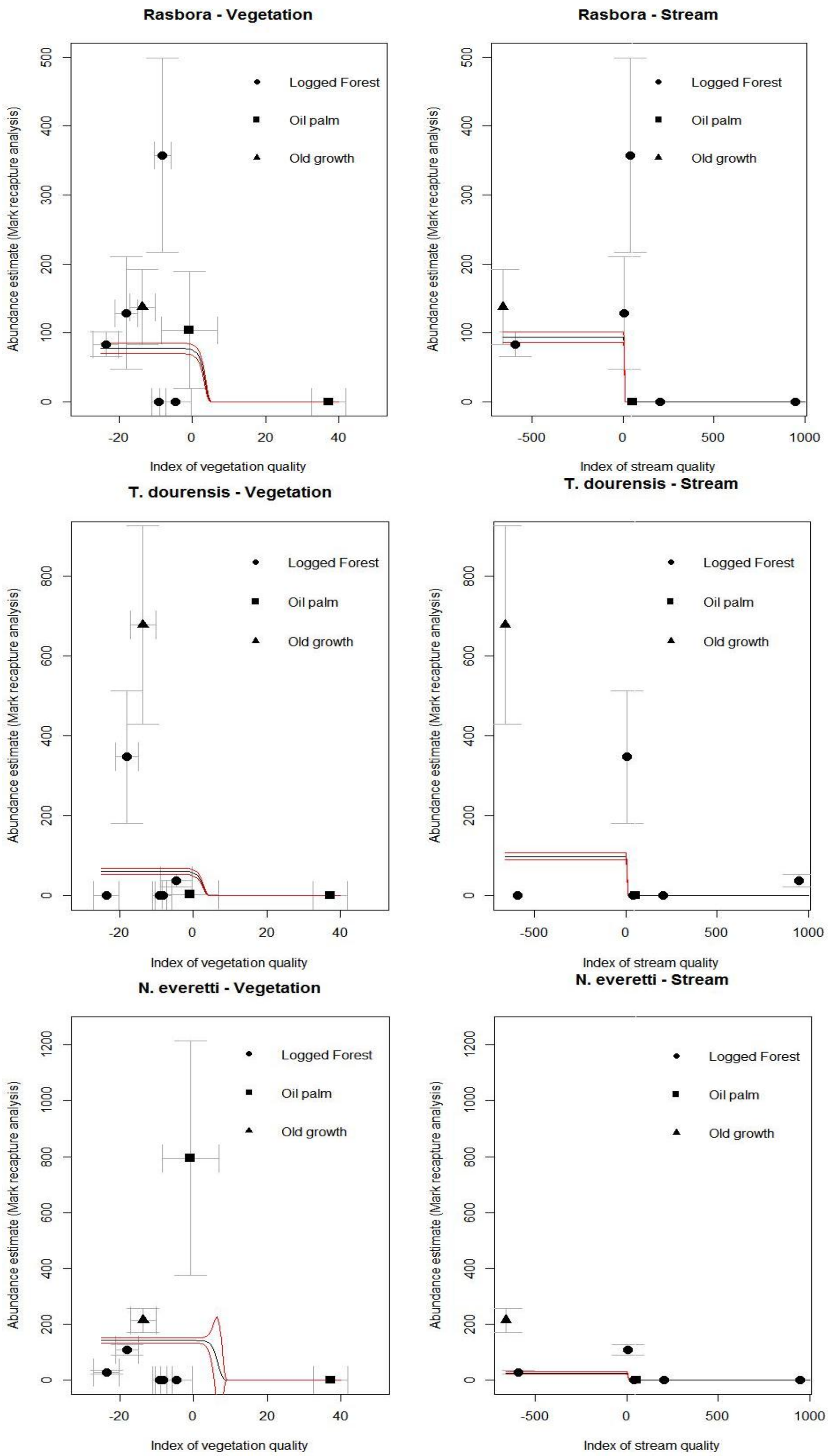

Figure 4.3. Modelled abundance against indices of riparian vegetation and stream quality, for three focal species. Points labelled by land use; black line indicates the glm *Abundance ~ PC1 riparian vegetation* and *Abundance ~ PC1 stream characteristics* and the red lines are the 95% confidence intervals to this model. Riparian vegetation index values are the axis one scores from a PCA of canopy openness, relascope count, percentage vine cover and the SAFE project forest quality scale, and reflect a gradient from high quality (negative values) to low quality (positive values) vegetation. Stream quality index values are the axis one scores from a PCA of water level, turbidity, nitrate concentration and phosphorous concentration, and reflect the same gradient for stream characteristics.

4.5 Methodological comparison

All linear models for bottle traps and cast netting of the three focal taxa were significantly different from the null model ($p<0.05$), indicating that the models sufficiently fit the data (Table 4.2). The only exception, was bottle traps for *N. everetti,* as no fish were caught ($p=1$).

The chi-squared value suggests the predictive power of the models, and the steepness of slope of GLMs fitted (Figure 4.4) allows the sensitivity of the different methods to be compared. The chi-squared for *Rasbora* and *T. dourensis* is lower for bottle traps than cast netting (Table 4.2) suggesting it has a better fit to the data for predicting population estimates. Despite this, the larger slope of mixed effect models for cast netting (Figure 4.4) suggest that it is easier to distinguish the effect of species abundance than when the slope is close to zero. For all three focal taxa the steepness of slope is greater for cast netting due to the higher capture rates indicating it is more sensitive to the response variable (abundance).

Table 4.2. Comparison of trapping methodologies, by ANOVA of mixed effects models (Standardised catch ~ Abundance (fixed) + Stream (random) with and without the fixed effect. Chi-squared of the model and slope of the GLMs (Figure 4.4), are included to compare methods.

Species	Method	Model AIC	Null AIC	Significant? (ANOVA)	Chi-squared	Slope
Rasbora	Cast	80.83	90.052	p<0.001	20.219	0.005
	Bottle	-145.16	-141.29	p=0.015	5.8719	<0.001
T. dourensis	Cast	46.984	54.989	p=0.001	17.034	0.001
	Bottle	-86.730	-71.696	p<0.001	10.005	<0.001
N. everetti	Cast	129.70	149.18	p<0.001	21.477	0.003
	Bottle	-145.16	-145.16	p=1	0	0

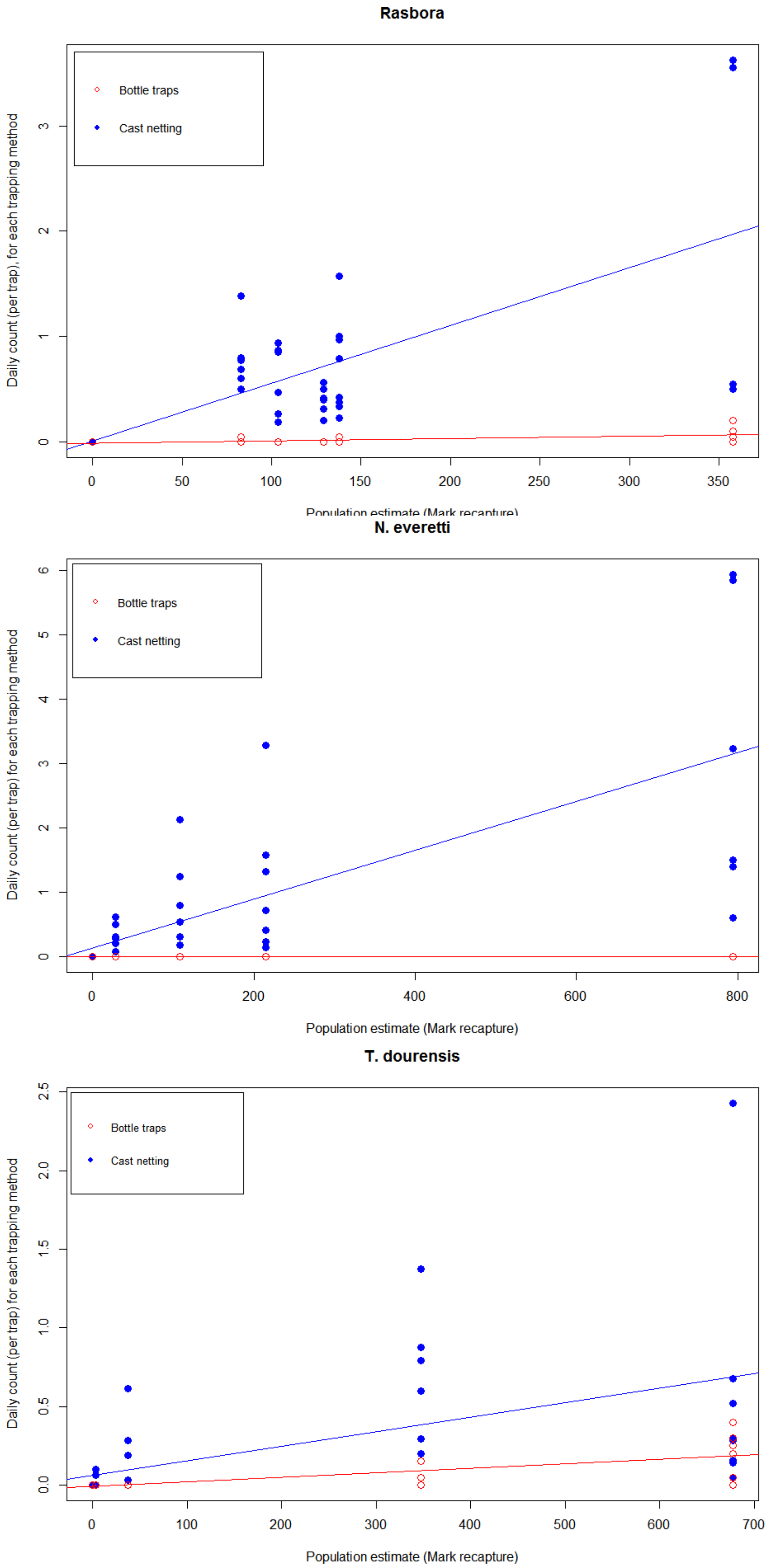

Figure 4.4. Comparison of trapping methods, bottle trapping and cast netting, for the three focal taxa. Methods are standardised to per trap per day (blue = cast netting, red = bottle traps). GLMs (catch ~ abundance) have been fit to the graph, as mixed effects models (used for analysis) cannot be predicted as the random effect value is unknown.

4.6 Community similarity and turnover rates

The most recent sampling in 2013 saw a total of 2779 fish captures from 11 genera, totalling 14 different species and the genus *Rasbora* which was not identified to species level. The 2011 sampling season saw a total of 2345 fish captures, of 19 different species in 12 genera. This disparity suggests some difference of community composition between sampling seasons.

Section 4.5 indicated that cast netting was more sensitive, and therefore a better model for relative species population abundance. The site by species matrix used standardised catch (per cast, per day) for all cast netting data from the two seasons. Multivariate ordination using PCoA, shows some degree of clustering of sites by land use (particularly OG) and between sampling seasons (Figure 4.5). The abundant species, which therefore could be suggested as being common species in the Kalabakan drainage basin are influenced by land use differences across the catchments indicated by the distinct separation in the PCoA (Figure 4.5). Unfortunately, nothing can be inferred about the clustered species, as all could be described as rare based on low catch numbers, however this could also be related to catching methodologies not targeting these species.

Environmental vectors, indices of riparian vegetation and stream characteristics were not significant (p=0.189 and p=0.819 respectively), when fit to the stream PCoA scores. The vectors correlation co-efficient indicate weak correlations for both indices, but is higher for riparian vegetation (r^2=0.170) than stream characteristics (r^2=0.024) suggesting a stronger correlation.

Ordination analysis is primarily described as a graphical tool (Oksanen, 2009). The PCoA displayed little similarity between sampling seasons at each site so two methods were used to compare between sampling years for streams and species. MANOVA conducted on the PCoA stream results (axis 1 and axis 2) showed no significant interaction between land use and year (p=0.468, approx. F number=0.916, df=4). Although no significant difference was shown between models with and without the interaction (p=0.469, approx. F number=0.91562, df=4), therefore the interaction was removed from the analysis. Subsequently, MANOVA conducted on the PCoA stream results suggested that there is a slight, albeit non-significant difference between sampling years (p=0.054, approx. F number=3.571, df=2) and land use (p=0.098, approx. F number=2.1441, df=4) for predicting the similarity distance between sites.

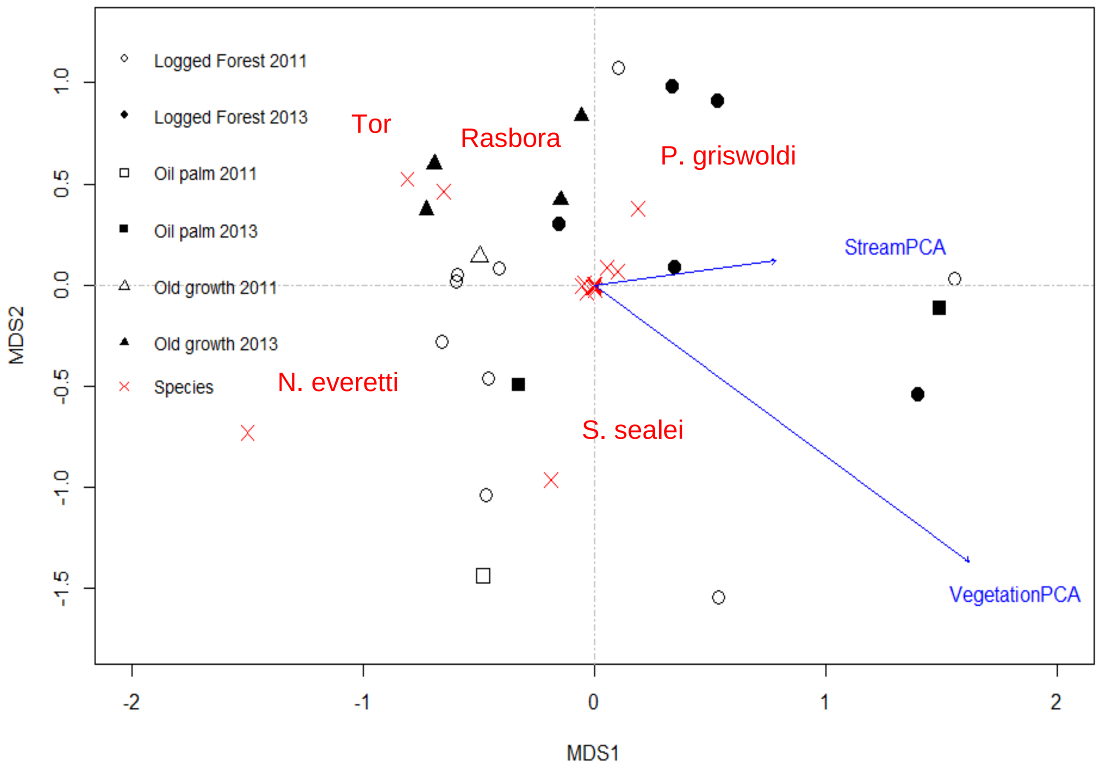

Figure 4.5. Multivariate ordination using PCoA, for two sampling seasons across all experimental streams. Stream and sampling date (black) and species or genus (red) are displayed along the first two axis's of the analysis. Blue arrows indicate direction of most rapid change for the two environmental indices: stream characteristics (StreamPCA) and riparian vegetation (vegetationPCA). All species names that can be read are abundant species (including the three focal species and others caught at most locations), all others are clustered around 0, 0 and are considered as less abundant.

Results of ANOVA of relative abundance by year and land use varied for each species or genus as all have different biological requirements (for example food and substrate). Relative abundance of S. sealei decreased from, 2011 to 2013 (p=0.005, F=10.631, df=1), increased from OG to OP (p=0.038, F=4.047, df=2), but proportionately to land use, with the greatest decrease in OP and smallest in OG (p=0.008, F=6.591, df=2). No significance for year or land use (p>0.05) was shown for the other four taxa, although this may be due to small sample sizes for OG and OP.

Relative abundance, indicated by standardised catch (per cast, per day), varied by year and land use (Appendix III). *Tor* and *Rasbora* had the largest populations in OG and lowest in OP, whereas the opposite was true for *P. griswoldi* and *N. everetti*. The effect of year was similar, with no clear difference for *Tor* and *Rasbora*, more caught in 2013 for *P. griswoldi* and more caught in 2011 for *N. everetti*. The difference between species response suggests that the effects of land use and sampling year cannot predict the community effect as different species can adapt to or may be resilient to different conditions.

Absolute species turnover rates indicating how stream communities have changed between seasons (Figure 4.6), were calculated for the seven streams sampled in both seasons (LF1-5, OG1 and OP1). Species turnover rates increase as riparian vegetation quality decreases, changes from OG/LF to OP, although it is not significant (p=0.338), and less than AIC two units separate experimental (39.1) and null (38.28) models. Species turnover rates decrease as stream characteristics change from OG/LF streams to that of OP although it is not significant (p=0.869), and less than two AIC units separate experimental (40.24) and null (38.28) models. The result concludes that species turnover rates vary between streams, but the reasoning cannot be confirmed.

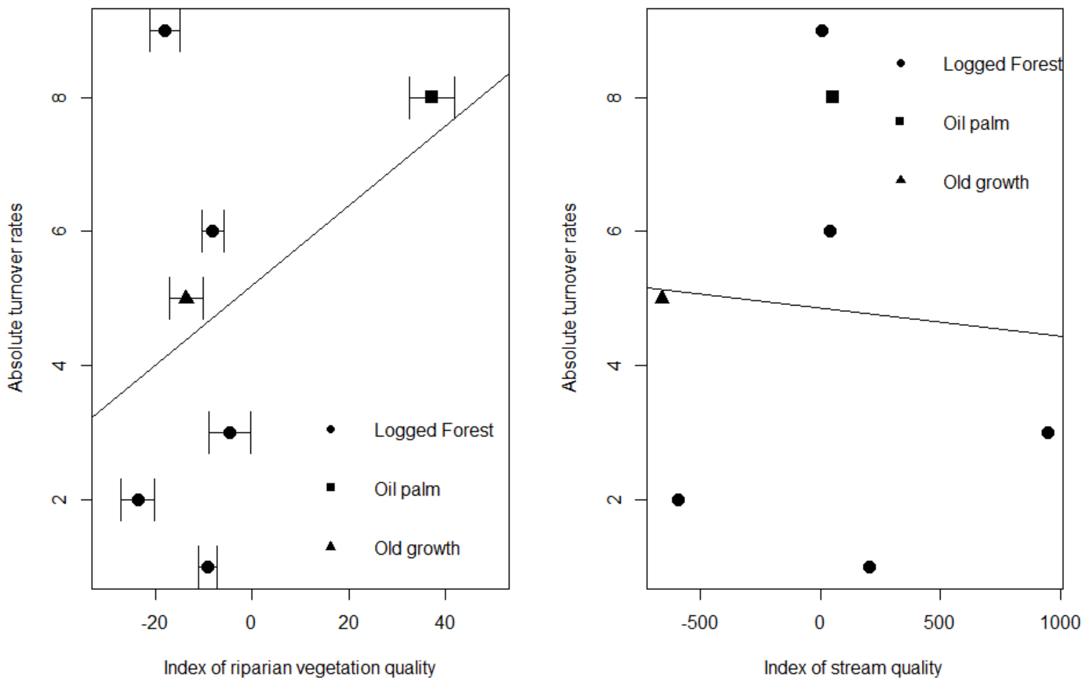

Figure 4.6. Absolute species turnover rates against indices of land use, riparian vegetation (left) and stream characteristics (right). Points are labelled by land use and where possible standard error bars fit. Riparian vegetation index values are the axis one scores from a PCA of canopy openness, relascope count, percentage vine cover and the SAFE project forest quality scale, and reflect a gradient from high quality (negative values) to low quality (positive values) vegetation. Stream quality index values are the axis one scores from a PCA of water level, turbidity, nitrate concentration and phosphorous concentration, and reflect a gradient from high quality (negative values) to low quality (positive values) stream conditions.

5. DISCUSSION

5.1 Response of fish to logging and conversion to oil palm

5.1.1 Species level

This project illustrates that fish species have varying responses to logging and conversion to oil palm. Abundance estimates of the three focal, common taxa decreased across the land use gradient (from old growth and logged forest to oil palm) for both riparian vegetation and stream characteristics. *Tor* and *Rasbora* had the largest populations in old growth forest whereas the largest populations for *P. griswoldi* and *N. everetti* were in oil palm. The value of logged forest and oil palm catchments is evident from the results in this project, as shown by the wide variety in abundance estimates for streams in the different land use areas.

The paucity of studies on tropical freshwater fish makes comparison to other areas difficult. Most studies focus on community assemblages and diversity rather than abundance, as was done in this work. Martin-Smith and Tan (1998b) recorded 65 different species of fish in DVCA, the Lower Segama and Upper Kuamut Rivers. However, Martin-Smith and Laird (1998) recorded that up to 30 species of fish were generally found in medium-sized rainforest streams across the land use gradient in Sabah. Currently, a total of only 20 species from 5 families have been recorded from the SAFE project area, although streams are narrow headstreams. This is in line with Martin-Smiths studies as more species can be expected, once more thorough surveys are conducted (Hoek-Hui, 2013).

The effects of logging vary widely, but depend upon the extent of logging in the region, the distribution of commercially valuable trees, and the local topography (Martin-Smith, 1998b). Typically, selective logging involves the removal of trees over 60 cm diameter at breast height (DBH). The act of logging and resultant skidding operations, to remove trees, can kill 30-50% of trees that are 1-60cm DBH (Pinard and Putz, 1996). In addition, the logging roads and skid marks created with logging activity have no designed drainage systems. Sidle *et al.* (2004) demonstrated how 78% of soil loss from the road system was delivered to a small headwater catchment, in Peninsular Malaysia. Once in the stream, much of the sediment was temporarily stored behind fallen woody debris, increasing the stream sediment load (Sidle *et al.,* 2004). This additional degradation exacerbates the problems of logging in the region, but it is the type and extent of logging practices that affect fish diversity and abundance.

Martin-Smith (1998a) demonstrated that there was little difference between abundance of species in old growth or logged forest, but there was a difference in abundance and biomass for three fish species in streams that had been recently logged (3-7 years) and old logged (17-18 years). The focal species of this project are different to those studied by Martin-Smith and the diversity of species at the SAFE project is considerably lower than that of DVCA. In this project logging occurred at all logged forest catchments in the 1970's and again between 2001-2008, whilst oil palm was planted in 2000 to 2006, dependent upon the estate (Ewers *et al.,* 2011), similar time periods to Martin-Smiths studies. A distinction across the logging gradient was indicated, but it is worth noting that the three focal species were not recorded at all streams, with differences across the LF streams. This indicates fish diversity and abundance is influenced by additional habitat level factors

Studies by Martin-Smith suggest logging does not affect species abundance or diversity over the long term, but this is confined to one region in Sabah. Further studies are needed to quantify the effects of logging on freshwater biota in a wider geographical context.

The effect of conversion to oil palm cannot be inferred from this project, only suggestions about post-conversion to oil palm can be discussed. The results of this project demonstrate that fish species can survive in streams within oil palm catchments, but abundance and species distribution is varied. One oil palm stream had the greatest abundance of a focal species and had the second highest level of diversity. The second oil palm stream had much lower species diversity and no recapture data was available due to very low catches or no catches of the focal species were caught. Unfortunately, the stream with higher catch was excluded from the majority of analysis as no in-stream data logger was present, resulting in little inference from the high level of diversity. The two oil palm streams differ as one has a narrow forest riparian strip along the majority of the experimental catchment compared to no riparian strip at the other (Pers. Obs).

Current environmental legislation in Malaysia states that a 30m buffer strip is left on either side of streams with permanent water flows (Ewers *et al.,* 2011). However, previous logging practices and conversion to oil palm often did not adhere to this, and there is no confirmation that 30m is sufficient to protect biodiversity and ecosystem services provided by headstreams. The future direction, to manipulate widths of experimental riparian strips (0–120m), at the SAFE project catchments will determine the short and long term effects of logging and conversion to oil palm (Ewers *et al*., 2011). The results provided by this large scale experiment at the SAFE project will demonstrate how fish species are affected by logging disturbance, forest clearance, and eventual conversion to oil palm plantation over time. As deforestation is going to continue into the future, results of this work will identify what width of riparian strip is optimal in maintaining freshwater diversity and abundance in Sabah.

5.1.2 Community level

Community composition varied across the land use gradient of the SAFE project. Results indicated decreased abundance and diversity as vegetation and stream quality decreased (old growth and logged forest to oil palm). There is a dearth of empirical evidence supporting the role of retained riparian forests in the maintenance of tropical fish assemblages (Dudgeon, 2000; Iwato *et al.,* 2005; Jackson and Sweeney, 1995). Despite this, literature is divided on the effects of logging and deforestation on tropical freshwater fish community assemblages. The few quantitative studies conducted on the communities and abundance of fish fauna in Sabah remains somewhat sporadic (Martin-Smith and Hoek-Hui, 1998), but several different hypotheses have arisen to explain the distribution and diversity of fish.

The removal of tropical forest cover during timber extraction represents an extreme form of disturbance, with potentially far-reaching effects on fish biodiversity. Martin-Smith *et al.* (1999) showed few long-term changes in abundance or species composition from streams around DVCA, with a diverse assemblage of more than 30 species of fish found in both sets of streams. As discussed, species can be affected by the time since logging which suggests logging has an effect on fish communities. Martin-Smith (1998b) suggested this effect of timber extraction is subtle, with changes being shown in abundance distributions rather than differences in community composition. Although, some communities did display short term (<18 months) absence or decrease in fish species after logging events.

In contrast to Martin-Smith but in agreement to this project, Inoue and Nunakawa (2005) demonstrated that riparian forests strongly affected the fish assemblage and abundance in low order streams. This can be explained by the obvious benefits provided by riparian vegetation to streams, in regulating allochthonous, sediment and light inputs to streams (Inoue and Nakano, 1998; Inoue and Nunakawa, 2005; Tabacchi *et al.,* 1998). Iwato *et al.* (2005) demonstrated that habitat alteration (through sedimentation, highly associated with logging and conversion of forest to palm oil) lowered the abundance and/or diversity of every benthic assemblage. In comparison, nektonic fishes were less affected, but the overall effects were detrimental, leading to a reduction in biodiversity of the stream communities.

A third explanation to the distribution and abundance of fish was demonstrated by Martin-Smith (1998a, 1998b). Martin-Smith (1998a) suggested that mesohabitats (riffles and pools) within streams and differences in stream size were more important in determining community structure of tropical freshwater fish communities in Sabah than logging history. Martin-Smith (1998a) concluded that the type of selective logging practices used locally had little impact on fish communities through mechanisms of persistance and rapid recolinisation. This project has not explicitly tested for possible differences in mesohabitats and therefore cannot be directly compared. The majority of sampling was conducted in pools, due to the methods involved in fish capture, so a difference in mesohabitats is not distinguishable. Explanations for the persistence of fish communities in the face of high disturbance pressure include high rates of re-colonisation, movement from undisturbed areas and avoidance or tolerance of disturbance effects (Martin-Smith,1998b; Martin-Smith *et al.,* 2010), proposing that communities are highly resilient.

In addition, the reported diversity of fish species at the SAFE project is currently considerably lower than that of DVCA. Martin-Smith and Laird (1998) studied depauperate communities (<10 species) in the Segama River region of Sabah. The streams either had substantial barriers to movement in the form of waterfalls and cascades, or unrestricted access but few species (Martin-Smith, 1998). Both of these environments compare to streams in this project, for example small waterfalls present in three logged forest streams (Pers. Obs). Martin-Smith and Laird (1998) showed abundance and biomass of fish above waterfalls were significantly lower than all other sites and fish were predominantly herbivores. Fishes in the other forest streams were from a variety of trophic groups. Fish caught in this project encompass all trophic levels but varied across the land use gradient. Food availability provides a possible explanation for the difference in abundance and community composition, both at the SAFE project and at the wider geographic scale. Further work should be conducted to confirm this assumption, to enable implications for management and future conservation efforts.

Shifting the focus to temperate freshwater fish, Inoue and Nunakawa (2005) demonstrated that substrate heterogeneity and competitor abundance were the most important variables in determining population abundance of two benthic species in Japanese streams. Riparian vegetation had no effect on food availability that limited fish populations, whereas this study showed that land use indices were a significant variable in estimating population abundance. In addition, a review of temperate stream-fish community studies concluded that stream and river fish communities were not resilient to logging in the absence of mitigation effects (Detenbeck *et al*.,1992). This is in contrast to what is seen in tropical freshwater fish communities (Martin-Smith, 1998b; Martin-Smith *et al.,* 1999).

The results also demonstrated diversity of species was influenced by sampling season. The confirmation of difference between sampling years and the lack of corroboration between species turnover rates across the land use indices could be explained by a number of factors, none of which can be confirmed within this study. This difference could relate to actual differences in community structure and/or diversity, the ability of research assistants employed to catch fish using a cast net, correct identification of species (although as far as possible this was controlled for by grouping species to genera for *Tor* and *Rasbora*), and the sampling duration at each stream (where possible this project sampled a stream for 6 consecutive days, in 2011 sampling was more random). This necessitates the cautious treatment of results and the inferences that can be drawn, but does not influence how the diversity varies across the land use gradient focused on in the project, as it is indicated in both years.

5.2 Dispersal

This project showed fish in headstreams are highly sedentary, which was an advantage to the project as it enabled fish to be re-caught across a 200m transect. This is expected as the streams sampled are steep and fast flowing, with many small waterfalls and drops. Streams are impassable in low water and a raging torrent in high water (Pers. Obs). As previously stated, Martin-Smith (1998a) suggested fish are specific to mesohabitats, confining them to these localities within streams. Martin-Smith (2010) suggested depauperate communities were present above waterfalls, a significant barrier to migration, preventing dispersal, although this project did not have the scope to confirm if this was the cause of limited dispersal.

5.3 Capture-Mark-Recapture of fish

This project was designed, where possible, to maximise fish catch and recapture rates, as fisheries recapture rates are typically low (Pine *et al.,* 2003). Recapture rates ranged from 0-35.6%, which potentially affected the accuracy and precision of population parameters, increased uncertainty of results at the lower end (Adams *et al.,* 2006). Recapture rates are high for studies that manually recapture fish, as typical recapture rates ranged from 2-14% (Cunjak *et al.,* 2004; Bayley and Austen, 2002), compared to studies with in-stream tag readers that have recapture rates of up to 84% (Barbour *et al.,* 2011; Stakenas *et al.,* 2008).

These results need to be treated cautiously due to the uncertainty in low recapture rates and as several issues presented themselves when validating the calculation of abundance. The project focused on a snapshot of fish abundance and diversity over a short time period (4-6 consecutive days sampling). Despite the short time period, open modelling was used to estimate abundance, in order to ensure all the assumptions of models were met (Cooch and White, 2011). For many species, recapture rates were zero at several streams or too low in order to calculate abundance, leading to a deficiency of data and uncertainty in the results (Pine *et al.,* 2003). The data may therefore not illustrate the true picture of abundance for the three focal species.

A second potential source of error was a trapping response; less fish were caught on subsequent days at the majority of streams. Although, nothing in this study proved what caused this effect, the field of fish psychology and the shy-bold continuum is being explored (Byrant, 2000; Ward *et al.,* 2004). Fish may become ‘trap shy’ due to disturbance in the stream prior to capture with a cast net. However, Stott (1970) suggested that fish did not become trap shy, so an alternative explanation may be valid for this study.

Thirdly, over- and under-dispersion was demonstrated, resulting in altered c-hat values. There is no consensus within the literature, to the significance of c-hat values of less than one (Cooch and White, 2011), resulting in the test lacking validation. In addition, The RELEASE programme, used to test the goodness of fit of models (through calculation of c-hat), is just one method of calculating goodness of fit of the data. Cooch and White (2011) suggest goodness of fit can be calculated using the median c-hat and bootstrapping goodness of fit tests as alternatives, but there is no agreement on the most appropriate method (Cooch and White, 2011), begging the question of the utility of these tests. Tests are also problematic when small populations and low recapture rates are modelled, such as in this project and other fish CMR studies. Test results indicated insufficient data to run simplified models compared to the full time-dependant model. Lettink and Armstrong (2003) proposed the use of mark recapture studies on small populations of 20-30 individuals. These results suggest that improvements need to be made to goodness of fit testing, in order to progress with population modelling.

5.4 Review of fish capture techniques at SAFE

Cast netting and bottle trapping was implemented in this project. There had been previous success with these methods, sampling in 2011 and this work provided the opportune time to look at a comparison between methods. Analysis of catch data suggest that cast netting is a better model of relative population abundance than bottle trapping, although cast netting is biased towards fish in pools, where it is most used (Pollock *et al*., 1990).

The most common technique to survey fish is electrofishing, using electricity to stun fish so they can be caught (Bohlin *et al.,* 1989). Electrofishing has many short and long term effects, including immediate and delayed mortality of fish (Neilson, 1998), and it tends to add bias in catch to fish of a certain size or species more susceptible to electric shock (Bayley and Austin, 2002).

The lack of knowledge on rare and endangered species at the study site (and most of SE Asia), necessitated the limitation of potential mortality to fish. It is therefore suggested that traditional capture methods such as cast netting should be used in future work. In addition to this, cast netting is inexpensive, requires no technical equipment or experience and can be easily implemented across many streams.

5.5 Project limitations and further research

5.5.1 Project design

Time constraints on this project led to unequal sampling at streams of different land uses. Three streams were unable to be sampled for the full six days and only one replicate was available for analysis in old growth and oil palm. Indices of riparian vegetation quality and stream characteristics were used to account for this unequal sampling. The indices allowed interpretation of the varying conditions at each of the logged forest streams, as there is irregular logging intensity across the SAFE project experimental area.

Many fish could only be identified to genus level due to similar external morphology, the presence of sub-species, and apparent differences between sampling seasons. For example, the genus *Rasbora* is comprised of eight different species, present in Sabah, which vary by the length of a lateral stripe, the extent of lateral line and distinctness of black ‘spots’ (Inger and Kong, 2002). The common species list created by Hoek-Hui (2013) was utilized to ID fish in the field, enabling the immediate release of more than 99% of fish caught, with few retained to be identified later, but additions to this list would only be of benefit to anyone entering the field of freshwater fish ecology in Borneo.

Future sampling in the Kalabakan basin should include all SAFE Project streams (two were not sampled this season), oil palm and old growth stream replicates. Further oil palm streams are available at the estates of Binuang, Gaharu, Keruing in the Benta Wawasan Estate, with different widths of riparian strip and time since conversion to Oil Palm plantation (S. Luke, Per. Comms., 2013). DVCA also provides an opportunity to sample additional old growth streams, as sampled by Martin-Smith (1998a; 1998b), Martin-Smith and Laird (1998), and Martin-Smith *et al.* (1999). As mentioned previously, it will be interesting to understand how the process of logging and conversion to oil palm affects all taxa in the experimental SAFE Project streams.

5.5.2 Land use variables

Four measures were compiled using PCA in order to explain the greatest variation and obtain an overall picture of riparian forest quality and stream characteristics for each catchment. Despite high levels of variance explained in the PC1 values (70.8% and 94.7% for riparian vegetation quality and land use characteristics, respectively) for each index, several problems arose with the data. Firstly, information was only available for 500m upstream of the 0m point, for riparian vegetation quality. It would have greater inference if the entire 2km catchment had been studied. This would enable a calculation of the entire catchments forest quality to give a complete view of the allochthonous inputs to the stream, not just the riparian strip. Secondly, the data for riparian forest quality was collected in September 2011 and August 2012, giving the potential for changes in quality to have occurred by the time this project was conducted.

The lack of correlation between the two land use quality variables suggest that the use of both indices is appropriate, for investigating fish diversity and abundance, because riparian vegetation is not the only factor influencing stream quality. However, this work was not able to account for one logged forest stream being severely affected by an extreme flood event in 2011, which was exacerbated by logging and altered the stream hydrology (A. Nainar, Pers. Comms., 2013) with unknown effects on stream biota. Additionally, the old growth stream, which is described as being unlogged dipterocarp forest, has previously suffered from illegal logging, evidenced by tree stumps (Per. Obs) and recorded logging around the area perimeter (Ewers *et al.,* 2011). This provides a possible explanation for the similar results depicted and lack of distinction between the logged forest and old growth streams in either land use index. However, these impacts were not accounted for in the riparian vegetation index.

Further sampling of upstream riparian vegetation quality and additional measures of stream quality would benefit the land use indices. This would allow a greater inference about the results of species abundance across the land use gradient. The results clearly indicate abundance is influenced by both land use indices and it would be interesting to determine which aspect of these variables affected fish population abundance and distribution.

5.5.3 Capture-Mark-Recapture

Possible extensions to this project include implementing the robust design model or using closed and open models to calculate all population parameters (Pine *et al.,* 2003; Cooch and White, 2011). The pilot study indicated that fish moved short distances, with 89% of fish moving less than 20m. This indication, in conjunction with the results of Pine *et al.* (2003) and Lettink and Armstrong (2003), suggest that closed models could be used over a short time period to calculate abundance of fish populations. However further research is necessary to confirm that the assumption of population closure could be met.

Another extension of this project, as has happened with other studies of fish mark-recapture (Adams *et al.,* 2006; Barbour *et al.,* 2011), would be the implementation of in-stream tag readers to get precise information on species movement and increase recapture data. However, low dispersal rates in the streams studied may be verified with this technique, leading to uninteresting results.

5.6 Conclusion

This project demonstrates the lack of data of data on freshwater fish in Sabah, Borneo, which extends across the majority of the tropics. This paucity is exacerbated by increasing pressures (economic and geographical) to convert forest to agricultural land. Many local extinctions of freshwater fish are not random with respect to ecological and life-history traits because entire populations are being removed through habitat degradation and loss (Giam *et al*., 2011). The presence of a large number of endemic and un-described species coupled with rapid habitat loss, and possible declines in species diversity and abundance across the land use gradient validates the need for the development of further field studies in this region (Claro-Garcia and Shibatta, 2013). This will enable hotspots of fish endemism to be identified and determine the species that are vulnerable to extinction (Giam *et al*., 2011). The incorporation of this data into management plans could aid freshwater conservation. However, if protection is not possible, partnerships could be established with management companies of logging concessions and plantations with the aim to lessen the impacts on fish.

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APPENDICES

Appendix I – Tables of Model selection and values from MARK for each species/genus estimate at each site. The model at the top of the list is the model that best fits the data, has the lowest AIC and highest model likelihood.

Table 1. LF1 – *N. everetti*

Model	AICc	Delta AICc	AICc Weight	Model Likelihood	No. Par.	Deviance
{p(.),phi(.),b(t)}	96.6716	0.0000	0.99992	1.0000	8	0.0000
{p(.),phi(t),b(t)}	115.9373	19.2657	0.00007	0.0001	12	0.0000
{p(.),phi(t),b(t)}	119.1142	22.4426	0.00001	0.0000	13	0.0000
{p(t),phi(t),b(t)}	157.0004	60.3288	0.00000	0.0000	17	0.0000

Table 2. LF1 – *Rasbora*

Model	AICc	Delta AICc	AICc Weight	Model Likelihood	No. Par.	Deviance
{p(.), phi(t), b(t)}	157.0833	0.0000	0.58905	1.0000	12	0.0000
{p(.), phi(.), b(t)}	157.9400	0.8567	0.38381	0.6516	8	0.0000
{p(t), phi(.), b(t)}	163.5126	6.4293	0.02366	0.0402	13	0.0000
{p(t), phi(t), b(t)}	167.3463	10.2630	0.00348	0.0059	17	0.0000

Table 3. LF2 – *T. dourensis*

Model	AICc	Delta AICc	AICc Weight	Model Likelihood	No. Par.	Deviance
{p(.), phi(.), b(t)}	55.3013	0.0000	0.97781	1.0000	6	0.0000
{p(.), phi(t), b(t)}	63.7912	8.4899	0.01402	0.0143	8	0.0000
{p(t), phi(.), b(t)}	64.8711	9.5698	0.00817	0.0084	9	0.0000
{1}	80.0311	24.7298	0.00000	0.0000	11	0.0000

Table 4. LF4 – *N. everetti*

Model	AICc	Delta AICc	AICc Weight	Model Likelihood	No. Par.	Deviance
{p(t), phi(.), b(t)}	170.1155	0.0000	0.52181	1.0000	13	0.0000
{p(.), phi(.), b(t)}	171.3056	1.1901	0.28779	0.5515	8	0.0000
{p(.), phi(t), b(t)}	172.1592	2.0437	0.18781	0.3599	12	0.0000
{p(t), phi(t), b(t)}	180.7282	10.6127	0.00259	0.0050	17	0.0000

Table 5. LF4 – *Rasbora*

Model	AICc	Delta AICc	AICc Weight	Model Likelihood	No. Par.	Deviance
{p(.), phi(.), b(t)}	76.7572	0.0000	0.99921	1.0000	8	0.0000
{p(.), phi(t), b(t)}	91.2008	14.4436	0.00073	0.0007	12	0.0000
{p(t), phi(.), b(t)}	96.1811	19.4239	0.00006	0.0001	13	0.0000
{p(t), phi(t), b(t)}	119.8773	43.1201	0.00000	0.0000	17	0.0000

Table 6. LF4 – *T. dourensis*

Model	AICc	Delta AICc	AICc Weight	Model Likelihood	No. Par.	Deviance
{p(.), phi(t), b(t)}	89.3862	0.0000	0.84341	1.0000	12	0.0000
{p(t), phi(.), b(t)}	92.7556	3.3694	0.15645	0.1855	13	0.0000
{p(t), phi(t), b(t)}	107.4585	18.0723	0.00010	0.0001	17	0.0000
{p(.), phi(.), b(t)}	109.6486	20.2624	0.00003	0.0000	8	0.0000

Table 7. LF5 – *Rasbora*

Model	AICc	Delta AICc	AICc Weight	Model Likelihood	No. Par.	Deviance
{p(.), phi(.), b(t)}	121.4755	0.0000	0.76756	1.0000	7	0.0000
{p(t), phi(.), b(t)}	123.9379	2.4624	0.22408	0.2919	11	0.0000
{p(t), phi(t), b(t)}	130.5144	9.0389	0.00836	0.0109	14	0.0000
{p(.), phi(t), b(t)}	79219.8192	79098.3437	0.00000	0.0000	10	78855.5250

Table 8. Merbau – *N. everetti*

Model	AICc	Delta AICc	AICc Weight	Model Likelihood	No. Par.	Deviance
{p(t). (phi(.). b(t))}	466.3645	0.0000	0.98767	1.0000	13	0.0000
{p(.). (phi(t). b(t))}	475.1319	8.7674	0.01233	0.0125	12	0.0000
{p(.). (phi(.). b(t))}	493.7207	27.3562	0.00000	0.0000	7	0.0000
{p(t). (phi(t). b(t))}	576.3812	110.0167	0.00000	0.0000	17	0.0000

Table 9. Merbau – *Rasbora*

Model	AICc	Delta AICc	AICc Weight	Model Likelihood	No. Par.	Deviance
{p(.). phi(.). b(t)}	67.1978	0.0000	0.99996	1.0000	8	0.0000
{p(.). phi(t). b(t)}	87.4410	20.2432	0.00004	0.0000	12	0.0000
{p(t). phi(.). b(t)}	92.4609	25.2631	0.00000	0.0000	13	0.0000
{p(t). phi(t). b(t)}	139.1951	71.9973	0.00000	0.0000	17	0.0000

Table 10. VJR – *N.everetti*

Model	QAICc	Delta QAICc	QAICc Weight	Model Likelihood	No. Par.	QDeviance
{p(t). phi(.) pent(t) N}	575.6362	0.0000	0.98777	1.0000	13	0.0000
{p(t). phi(t) pent(t) N}	584.4192	8.7830	0.01223	0.0124	17	0.0000
{p(.). phi(.) pent(t) N}	644.7363	69.1001	0.00000	0.0000	8	0.0000
{p. phi(t) pent(t) N}	648.3881	72.7519	0.00000	0.0000	12	0.0000

Table 11. VJR – *Rasbora*

Model	AICc	Delta AICc	AICc Weight	Model Likelihood	No. Par.	Deviance
{p(.).phi(.).b(t)}	109.3325	0.0000	0.98566	1.0000	8	0.0000
{p(t).phi(.).b(t)}	118.5320	9.1995	0.00991	0.0101	13	0.0000
{p(.).phi(t).b(t)}	120.1444	10.8119	0.00443	0.0045	12	0.0000
{p(t).phi(t).b(t)}	135.0620	25.7295	0.00000	0.0000	17	0.0000

Table 12. VJR – *T. dourensis*

Model	AICc	Delta AICc	AICc Weight	Model Likelihood	No. Par.	Deviance
{p(.). phi(t). b(t)}	127.6879	0.0000	1.00000	1.0000	12	0.0000
{p(t). phi(t). b(t)}	186.9473	59.2594	0.00000	0.0000	17	0.0000
{p(t). phi(.). b(t)}	196.1479	68.4600	0.00000	0.0000	12	0.0000
{p(.). phi(.). b(t)}	210.6485	82.9606	0.00000	0.0000	8	0.0000

Appendix II – Tables for each species of GLM results in modelling the abundance estimate.

Table 1. Results of GLM models for abundance of *T.dourensis.* Model in bold is the best fit of the data based on the lowest AIC value.

Model	AIC	Delta AIC (to null)	Anova test against null model (~1)	Notes (Merbau has been removed from all models as no stream data and cannot compare models using different subsets)
Estimate~PC1veg	2135	-388	P=2.2e-16 P<0.000	Significant
Estimate ~PC1Stream	1772	-751	P=2.2e-16 P<0.000	Significant
Estimate ~ Landuse	1020	-1503	P=2.2e-16 P<0.000	Significant
Estimate~PC1medianve g	2094	-429	P=2.2e-16 P<0.000	Significant
Estimate~PC1veg * Landuse	903.8	-1619.2	P=2.2e-16 P<0.000	Significant
Estimate~PC1veg + PC1 stream	1727	-796	P=2.2e-16 P<0.000	Significant
Estimate ~ Landuse + PC1stream	1021	-1502	P=2.2e-16 P<0.000	Significant
Estimate ~ PC1veg * Landuse + PC1stream	**31.5**	**-2491.5**	**P=2.2e-16 P<0.000**	**Significant. Has the lowest AIC value, that is at least 2 units different from all other models so is the most appropriate/best fit of the data.**
Estimate ~ PC1veg * Landuse * PC1stream	33.5	-2489.5	P=2.2e-16 P<0.000	Significant

Table 2. Results of GLM models for abundance of *N. evertti.* Model in bold is the best fit of the data based on the lowest AIC value.

Model	AIC	Delta AIC (to null)	Significance? (test against null model)	Notes (Merbau has been removed from all models as no stream data and cannot compare models using different subsets)
Estimate~PC1veg	604.3	-172.8	P= 2.2e-16 P<0.000	Significant
Estimate ~PC1Stream	429.4	-347.7	P= 2.2e-16 P<0.000	Significant
Estimate ~ Landuse	324.4	-452.7	P=2.2e-162.2e-16 P<0.000	Significant
Estimate~PC1medianve g	611.4	-164.7	P=2.2e-16 P<0.000	Significant
Estimate~PC1veg * Landuse	214.7	-562.4	P=2.2e-16 P<0.000	Significant
Estimate~PC1veg + PC1 stream	417	-360.1	P=2.2e-16 P<0.000	Significant
Estimate ~ Landuse + PC1stream	292.4	484.7	P=2.2e-16 P<0.000	Significant
Estimate ~ PC1veg * Landuse + PC1stream	**28.9**	**-748.2**	**P=2.2e-16 P<0.000**	**Significant. Has the lowest AIC value, that is 2 units different from all other models so is best fit of the data.**
Estimate ~ PC1veg * Landuse * PC1stream	30.9	-746.2	P=2.2e-16 P<0.000	Significant

Table 3. Results of GLM models for abundance of *Rasbora.* Model in bold is the best fit of the data based on the lowest AIC value.

Model	AIC	Delta AIC (to null)	Anova test against null model (~1)	Notes (Merbau has been removed from all models as no stream data and cannot compare models using different subsets)
Estimate~PC1veg	910.5	-138.5	P=2.2e-16 P<0.000	Significant
Estimate ~PC1Stream	957.8	-91.2	P=2.2e-16 P<0.000	Significant
Estimate ~ Landuse	831.4	-217.6	P=2.2e-16 P<0.000	Significant
Estimate~PC1medianveg	966.4	-82.6	P=2.2e-16 P<0.000	Significant
Estimate~PC1veg * Landuse	833.3	-215.7	P=2.2e-16 P<0.000	Significant
Estimate~PC1veg + PC1 stream	886.7	-162.3	P=2.2e-16 P<0.000	Significant
Estimate ~ Landuse + PC1stream	752.8	-296.2	P=2.2e-16 P<0.000	Significant
Estimate ~ PC1veg * Landuse + PC1stream	464.6	-584.4	P=2.2e-16 P<0.000	Significant
Estimate ~ PC1veg * Landuse * PC1stream	39.45	-871.05	P=2.2e-16 P<0.000	Significant. Has the lowest AIC value, that is at least 2 units different from all other models so is the most appropriate/best fit of the data.

Appendix III. How do Year and Land use variables affect relative abundance? Box plots indicate the effects by species: a) *Tor* b) *Rasbora* c) *P. griswoldi* d) *N. everetti* (for each variable, 1= year, 2= land use).