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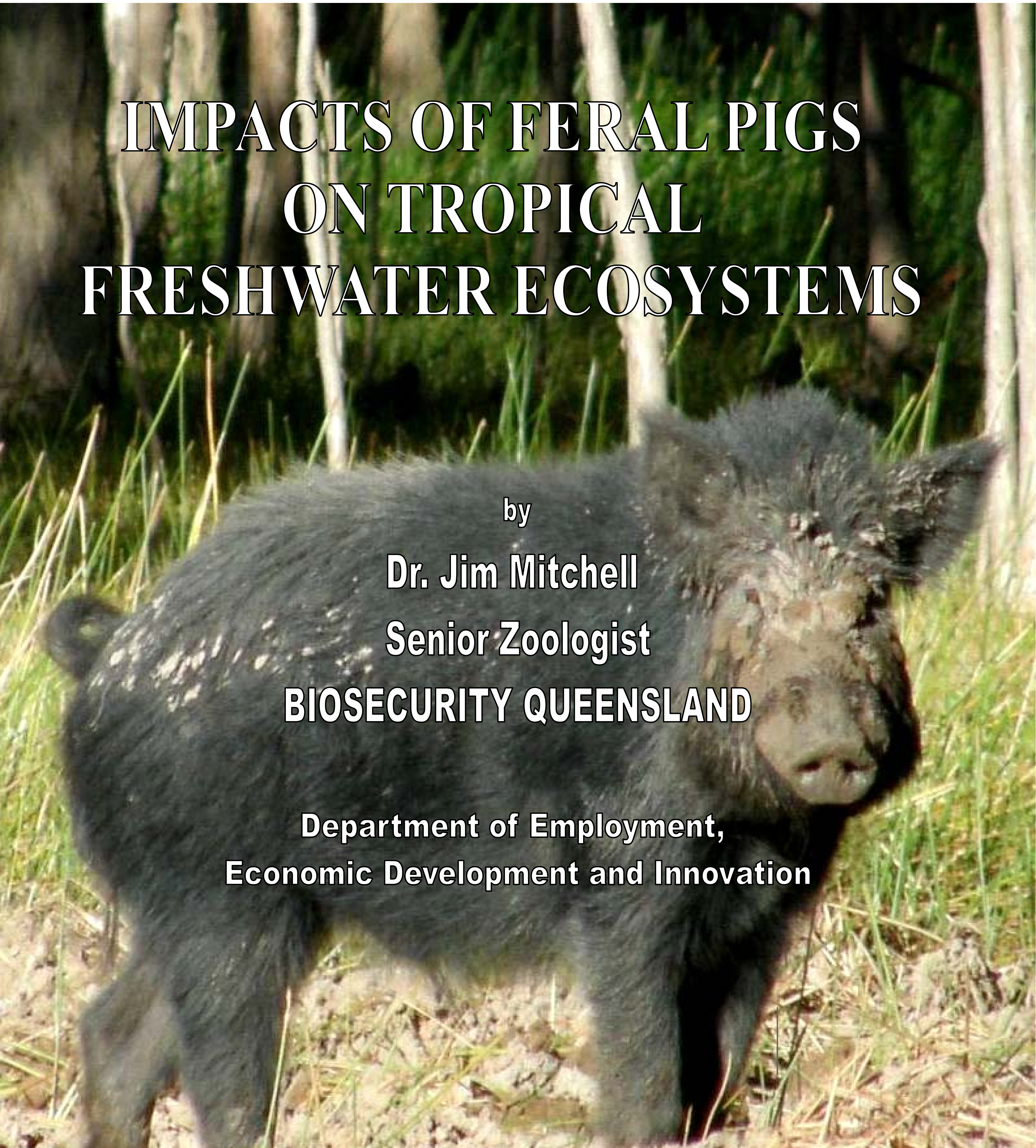

IMPACTS OF FERAL PIGS ON TROPICAL FRESHWATER ECOSYSTEMS

by

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Final Report – May 2010

Experimental Research to Quantify the Environmental Impact of Feral Pigs within Tropical Freshwater Ecosystems

To

Environmental Biosecurity Section, Department of the Environment, Water, Heritage and the Arts.
Australian Government.
Tender 083/0607 DEW

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Department of Employment, Economic Development and Innovation.

Foreword

In Australia, the general perception by the community is that feral pigs are doing substantial ecological damage and pose a threat to ecological values of many regional ecosystems. They have also been listed as a key threatening process under the EPBC Act (July 2001) and an associated threat abatement plan “Predation, Habitat Degradation, Competition and Disease Transmission by Feral Pigs” has been developed (Department of Environment and Heritage, 2005).

The plan developed a number of threat abatement objectives one of which (Objective 4) was to quantify the impacts feral pigs have on biodiversity (especially nationally listed threatened species and ecological communities) and to determine the relationship between feral pig density and the level of damage. It also listed a number of rare or endangered species and ecosystems that are regarded as threatened or suspected of being threatened by feral pig impacts, although for many of these there is a distinct lack of quantitative information.

To help address objective 4 of the threat abatement plan, the Department of Environment and Water Resources (now the Environmental Biosecurity Section of the Department of the Environment, Water, Heritage and the Arts) undertook a tender process. Research organisations were invited to submit proposals that would investigate and obtain scientific evidence to quantify:

- The impact that feral pigs have on elements of Australia’s native biodiversity.
- The relationship between feral pig density and the level of damage.

This report presents the findings of a successful tender that focussed on seasonal freshwater habitats of the dry tropics. Besides satisfying the scope of the tender, additional information was also collected to assess the influence landscape features have on feral pig impacts and the diet of feral pigs in this wetland habitat. These reports will be presented separately.

The agreement (083/0607DEW, Com ID 63969) was signed on the 8th June 2007. The study was conducted in north Queensland at Lakefield National Park, an area of high conservation significance and renowned for its vast river systems and spectacular wetlands. The initial setup of this study commenced in April 2007. Data collection continued throughout the wet and dry seasons of 2008 and 2009.

Prepared by
Dr. Jim Mitchell. June 2010.
Biosecurity Queensland: Department of Employment, Economic Development and Innovation. Queensland Government.

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- Shane Campbell and Barbara Madigan who helped with preparing the final report.

Executive Summary

- The Department of the Environment, Water, Heritage and the Arts (Australian Government) funded a study titled "Experimental Research to Quantify the "Environmental Impact of Feral Pigs within Tropical Freshwater Ecosystems" to investigate and obtain scientific evidence to quantify:

  (a) Feral pig impacts on the biodiversity of ephemeral lagoons

  (b) The relationship between pig ecological impacts and pig abundance.

- The study was conducted in Lakefield National Park in the tropical savannas area of eastern Cape York, an area of high conservation values. Lakefield N.P. is representative of freshwater systems throughout the Cape and Gulf regions of Queensland and the wetlands of the Northern Territory. Lakefield is renowned for its vast river systems and spectacular wetlands.

- A range of ecological indicators found in freshwater habitats were used to quantify feral pig impacts on elements of biodiversity. These indicators were measured for two years in both unprotected ephemeral freshwater lagoons and those protected from pig impact by fencing. The sequential measurements of these ecological indicators as the lagoons draw down gave a guide to the consequences of feral pig impacts on biodiversity.

  Overall, feral pig activity had a negative impact on the ecological condition of the ephemeral lagoons studied with the major impacts related to destruction of habitat and a reduction in water clarity. Pig foraging activities caused major destruction to aquatic macrophyte communities which were the preferred food resource. The visual differences in the proportion of bare ground and aquatic macrophytes between protected and unprotected lagoons were dramatic. Protected lagoons had significantly more macrophyte coverage. The destruction of macrophyte communities and upheaval of wetland sediments in unprotected wetlands significantly reduced the water clarity and had subsequent effects upon key water quality parameters such as dissolved oxygen availability. Other water quality parameters such as nutrients were also strongly affected by pig activity and although the effects were not able to be statistically realised, it appeared that pig activity contributed to an increase in nutrient levels in the unprotected lagoons.

- Despite these dramatic impacts, the composition and diversity of macrophyte, invertebrate and fish communities were not statistically affected by protection from pig foraging. Each lagoon, whether fenced or unfenced, harboured its own distinct community assemblage and the effect of the rapid drying of lagoons over the dry season showed stronger effects upon the diversity and composition of biotic communities than did the exclusion of pigs via fencing.

- The relationship between pig damage and pig density is poorly understood. This study quantified relationships between pig abundance and the extent of impacts they cause within tropical freshwater ecosystems. Feral pig population abundance around selected wetland systems were artificially manipulated by aerial shooting to enable the response of feral pig impact between sites that have varying pig abundance levels to be compared. There was a demonstrated positive association of pig abundance (frequency of occurrence of sign and frequency of occurrence of diggings) with the extent of impact (recent pig diggings that occurred). Thus the level of impacts caused by pig diggings was positively related to the level of pig abundance: more pigs, more diggings.

Our study identified an exponential “concave up” curve to the abundance / impact relationship. Impacts increased exponentially with increasing pig abundance. When pig abundance is high, a minimal level of population control will substantially reduce impacts. When pig abundance is low a substantial level of pig control is required for a minimal decrease in impact levels. The point on this relationship curve where the minimal level of population control required to maximise impact reduction appears to be approximately 50% visitation frequency on plots.

There was no clear overall statistical association of pig abundance with other ecological impact indicators although a number of sites within the lake systems had significant associations between pig diggings and a number of ecological indicators. Graphical examination of the data concluded that there is a demonstrated strong effect of pig impacts on water quality although the design did not lend itself to an ability to statistically detect the effects that actually occurred. The specific example of improved water quality after pigs were shot out of Caulders Lake is good evidence of a non-statistical water quality effect, the ecological variability between the sites tended to smother any overall effect being detected.

- There was a significant positive association of the Aquatic Vegetation Index with two pig abundance indices; pig abundance increased with increasing aquatic vegetation abundance. This illustrated that resource availability may be the dominant influence on the level of pig impacts. Pigs concentrate their impacts where adequate resources are present.

- The amount of new pig diggings on transects was 0.93% of the soil surface being disturbed on a daily basis. Thus hypothetically, in just over 100 days the total perimeter of the studied wetlands would be disturbed by feral pig diggings.

- A separate study conducted through a student honours project (James Cook University) was completed in 2009 at Caulders Lake, one of this projects study sites. It was titled *“Dietary impacts of feral pigs on ecological processes in Lakefield National Park”* and examined the diet of feral pigs during the wet (April /May) and dry seasons (August) of 2009. Stomach and colon contents of 95 feral pigs were collected from around Caulders Lake. Overall composition of the diet for all seasons was 0.47 % invertebrate matter, 0.56 % vertebrate matter, 8 % seed, 43 % leaf and stem, and 48% water lily bulb and nonda plum. Diet composition was related to seasonal conditions for all diet categories. Two of the diet categories were correlated with body weight. Plant matter occurred in 100 % of all diets, while animal matter occurred in 87 % of diets. Pigs were selective for some food items however opportunistic behaviours were evident. Body condition indices did not significantly differ between seasons, showing pigs are able to maintain body condition throughout the year. This was attributed to their ability to diet shift. The results suggest that feral pigs at LNP potentially have an ecological impact on water lilies, nonda plum trees and snails, and indirect effects on the comb-crested jacana.

- This study had no true control in the sense of an ecological reference point without feral pig disturbance since feral pigs are known to have been in the Lakefield region for 100 years or more. It follows that because the wetlands in this region have been disturbed by feral pigs for very many years, then all may be altered to some extent by this prolonged disturbance and any truly pig-sensitive species may have been eliminated prior to this study.

- We have demonstrated that feral pigs do have significant impacts upon wetlands in the tropical environments we studied. However, we have also demonstrated that there are significant natural disturbances also operating in these ecosystems that should be taken into account when assessing impacts to wetlands. Validation of these observations requires measuring how these effects are influenced by the seemingly greater annual disturbance regime of variable flooding and drying in this tropical climate. After the commencement of the wet season the wetland systems are "reset" back to their original condition.

- Further analysis of the data will quantify the influence that micro-landscape features have on the population dynamics of pig populations in this environment. A number of landscape features including soil type and depth of water have shown associations with pig impacts. This analysis will be present separately.

- A scientific manuscript reporting on the first year results of the pig impacts on biodiversity study has been published:
R. G. Doupe, J.L. Mitchell, M. J. Knott, A.M. Davis and A.J. Lymbery (2009). Efficacy of exclusion fencing to protect ephemeral floodplain lagoon habitats from feral pigs (Sus scrofa). Wetlands Ecology and Management. ***18*** *(1) 69-78.*

- A side project conducted by ACTFR on the migration of long necked turtles within the fenced exclosures has also been published:
R G. Doupe, J. Schaffer, M. J. Knott and P. W. Dicky. (2009). A Description of Freshwater Turtle Habitat Destruction by Feral Pigs in Tropical North-eastern Australia. Herpetological Conservation and Biology. ***4*** *331-339.*

CHAPTER 1.

Experimental Research to Quantify the Environmental Impact of Feral Pigs within Tropical Freshwater Ecosystems:
Study Summary

1.1 INTRODUCTION

Environmental impacts of feral pigs have not been studied intensively; very little quantitative information on the ecological impacts caused by the feral pig throughout Australia is available. The perception by the community is that feral pigs are doing substantial ecological damage and pose a threat to ecological values in the tropics. There is a distinct lack of information on a number of threatened ecosystems and in particular there is a scarcity of information in relation to the dry tropics seasonal freshwater habitats. The Threat Abatement Plan (Anon 2005) has proposed a number of rare or endangered species and ecosystems that are regarded as threatened or suspected of being threatened by feral pig impacts. A number of these species and ecosystems occur in the dry tropics and in particular the tropical freshwater ecosystems.

The primary aim of this study was to quantify the threat feral pigs pose to the health of tropical freshwater ecosystems. This study examined the two primary objectives outlined in the research proposal. A general outline of each study is presented below.

(A) Ecological impacts of feral pigs on biodiversity

Ecological indicators found in freshwater habitats were used to quantify feral pig impacts on elements of biodiversity. These indicators were measured for two years in both unprotected ephemeral freshwater lagoons and those from which pig impact was excluded. The sequential measurements of these ecological indicators as the lagoons draw down gave a guide to the consequences of feral pig impacts on biodiversity and a guide to the timing of recovery from these impacts if the level of impact is reduced. This study was conducted (under contract) with the Australian Centre for Tropical Freshwater Research, based at James Cook University, Townsville.

(B) Relationship of ecological impacts with feral pig density

The relationship between pig damage and pig density is poorly understood. This study attempts to quantify relationships between pig abundance and the extent of impacts they cause within tropical freshwater ecosystems. Feral pig population abundance around selected wetland systems in the study site was artificially manipulated by aerial shooting to enable the response of feral pig impact between sites that have varying pig abundance levels to be compared. This research design approach followed the adaptive approach to management outlined in the Feral Pig Threat Abatement Plan – Objective 4 (Anon 2005).

In addition, further research opportunities to complement this study became available. A study of the diet of feral pigs at Caulders Lake, one of the research lake systems was conducted during the wet (April /May) and dry seasons (August) of 2009. Data was also collected on the relationship of landscape features with impact levels; these two studies are reported separately.

1.2 FERAL PIG ENVIRONMENTAL IMPACTS

The negative effects of feral pigs in a range of natural habitats throughout the world have been well documented, both overseas and in Australia (Choquenot *et al.* 1996; Anon 2005). Many of these studies suggest feral pigs have a significant negative effect on ecological processes within most environments due to the soil disturbance caused by their digging activities. Feral pig impacts can be described as consumptive (e.g. predation of animals, soil invertebrates, insects and plants), or destructive (e.g. ground disturbance causing plant removal or uprooting of plant species). Most destructive impacts are an indirect consequence of foraging activity, so consumptive and destructive impacts may be closely related.

Degradation of habitats is probably the most obvious environmental impact caused by feral pigs with soil disturbance caused by feral pig diggings the most visual impact. This soil disturbance may also cause “hidden” ecological impacts, such as disrupting nutrient and water cycles, changing soil micro-organism and invertebrate populations, changing plant succession and species composition patterns and causing erosion. Diggings may also spread undesirable plant and animal species and plant diseases.

Cooray and Mueller-Dombois (1981), Stone and Taylor (1984), Aplet *et al*. (1991) and Diong (1982) found that weed expansion was clearly related to ground disturbance by pigs with weed seeds germinating rapidly in the disturbed soils of pig diggings. The distribution of soil arthropods and fungi was also caused by soil fragments adhering to the feet of introduced animals, including pigs, in Hawaii National Park (Spatz and Mueller-Dombios 1975).

The extensive montane bog systems in Hawaii have the highest number of threatened or endangered plant species for Hawaii. Feral pigs cause the only significant disturbance to these freshwater habitats and are a significant disruptive factor (Stone and Scott 1985). Loope and Scrowcroft (1985) found that pigs severely damage these fragile and limited montane bog communities on all Hawaiian Islands.

Alexiou (1983), in his study near Canberra, described significant changes in density and cover of a wide range of plant species following disturbance by pigs. Hone (1998) suggested that species richness is inversely related to the amount of pig digging disturbance. If diggings are less then 25% of the area, there will be a short-term effect, if pig diggings cover more then 25% of the area there will be a rapid reduction of species richness. Lacki and Lancia (1983) found soil chemical properties were influenced by pig diggings, and the longer the duration of digging the greater the effect. Kotanen (1994) found concentrations of mineral nitrogen tended to be higher in pig diggings then in undisturbed areas. He also identified that daytime soil surface temperatures averaged 10^0C warmer in pig diggings than in undisturbed sites. Bratton (1974) found pig diggings reduced the herbaceous understorey of American hardwood forests to less then 5% of the expected value. Singer *et al*. (1984) and Bratton *et al.* (1982) found a significant increase in plant biomass when pigs were excluded. Damage to forest seedlings and young trees has been reported in New Zealand (Challies 1975; Massei *et al*. 1997) and in the USA (Lucas 1977). Singer *et al*. (1984) found, in their USA study, that intensive diggings influence nutrient cycling by eliminating the litter layer, which greatly reduced the concentrations of nutrients in the soil and litter component. Lacki and Lancia (1983), also in USA, described how digging increased the cation exchange capacity and acidity by incorporating organic matter into the soil.

Diggings by pigs may also be an important source of erosion but the extent of this impact is unquantified. Diggings around creeks, rivers, billabongs and lakes in the tropics are subjected to significant rainfall and flooding effects during the tropical wet season. Loose soil in pig diggings may be transported downstream into the Great Barrier Reef lagoon leading to silting of the reef systems. Although this source of erosion may be minor compared to normal erosion processes during significant rainfall events from tropical cyclones in this area, the influence of pig diggings to encourage higher erosion rates may be significant and warrants investigation.

Feral pigs have been shown to have direct negative impacts on tropical freshwater ecosystems. There is some evidence to suggest a strong correlation between pig diggings and soil moisture, soil friability and the associated abundant soil invertebrate populations and plant bulbs (Mitchell and Mayer 1997; Hone 2002; Mitchell 2002). Feral pigs have been recorded predating on freshwater crayfish, frogs and northern snake necked turtles (*Chelodina rugosa*) which attain high densities in ephemeral swamps in the wet-dry tropics of northern Australia. Studies have shown that the survival of northern snake necked turtle of all sizes and stages of maturity was compromised by feral pig predation. Pigs are also responsible for severe damage to aquatic vegetation found in the freshwater ecosystems. Vegetative bulbs and rhizomes of water lilies formed the majority of feral pig diets in the Northern Territory and northern Queensland. Frogs may also be a common food item for pigs in tropical freshwater environments. Photographic evidence of over 150 frogs in a single pig stomach in the Cape York region suggest that pigs may be a significant predator of frogs in some areas. The role of feral pigs in competing for the available food resources with native animals has not been quantified. However, at the end of the tropical dry season when food resources are minimal this competition could be significant, especially within the declining freshwater sources.

A full literature review of feral pig impacts can be found in Anon (2005) and Choquenot *et al.* (1996). However, few of these studies provide quantitative data on direct environmental impacts. The perception in the general community is that feral pigs cause substantial ecological damage and pose a significant threat to the conservation value of many biogeographical regions of Australia. The studies in this report are an attempt to quantify the impact of feral pigs on tropical freshwater ecosystems and show whether this perception is correct. The results will also provide a basis for further research into feral pig impacts and also any recovery of ecological indicators in the presence of reducing pig impacts.

1.3 STUDY SITE DESCRIPTION

This study was conducted in Lakefield National Park in the tropical savannas area of eastern Cape York known as the Laura lowlands, an area of high conservation value (Slatter and Williams 1999). Abrahams *et al*. (1995) calls Lakefield National Park an area of significant wetlands with 743 spp. of native taxa found there (Qld EPA). The study site is characterised by gently undulating alluvial plains with many old stream channels, shallow lagoons and lakes. The mean maximum/minimum temperatures for November are 35.5° /20.8°C and for June, 29.6°/15.0°C. Mean rainfall for November is 57.7 mm and for June, 9.8 mm; annual rainfall range is 1000-1400 mm (Bureau of Meteorology 2009).

Lakefield National Park (LNP) is located northeast of the township of Laura in the Cape York Peninsula (Lakefield Ranger Station - 15^{o} 55.601 S, 144^{o} 12.104 E). Lakefield, Cape York's largest park (537,000 ha) and Queensland's second largest National Park, is representative of freshwater systems throughout the Cape and Gulf regions of Queensland and the wetlands of the Northern Territory. Lakefield is renowned for its vast river systems and spectacular wetlands. In the wet season the Normanby, Morehead and North Kennedy rivers and their tributaries join to flood vast areas, eventually draining north into Princess Charlotte Bay. During the dry season, rivers and creeks shrink, but leave large permanent waterholes, lakes and lagoons which attract a diversity of animals, particularly waterbirds.

Lakefield's flora is representative of the tropical savannas coastal lowlands of eastern Cape York. Low open woodland dominated by paperbark tea-tree (*Melaleuca viridiflora*) is widespread throughout Lakefield. Grey bloodwood (*Corymbia clarksoniana*) and/or Moreton Bay ash (*C. tessellaris*) woodland is common around the wetlands. Small areas of Molloy red box (*Eucalyptus leptophleba*) woodland grow on the floodplains of the Hann and Morehead rivers and *M. citrolens* grows along many drainage lines, particularly in the south and east. Small patches of *Neofabricia myrtifolia* heath grow on dunes. *Sorghum* spp. and/or *Themeda arguens* closed tussock grassland are widespread on the plains in the south. Small patches of *Panicum spp*. and *Fimbristylis spp*. tussock grassland and patches of *Eriachne spp*. closed tussock grassland grow in drainage depressions.

Wetland areas may be occupied by floating plants such as *Nymphoides* spp. and *Monochoria* spp. and bottom rooted species such as *Ludwigia perennis* and water lilies (*Nymphaea* spp.). Emergent grasses (e.g. *Oryza rufipogon*) and sedges (e.g. *Bulkurru, Eleocharis* spp.) are also common. Where water in perennial floodplain lakes is deeper than 1.7 m, there may be sparse populations of free floating plants such as ferny azolla (*Azolla pinnata*) and native water hyacynth (*Monochoria cyanea*) and fully submerged plants such as eelgrass (*Blyxa* sp.) and water nymph (*Najas tenuifolia*). Where the water is shallower, bottom rooted emergents such as *Nelumbo nucifera, Eleocharis* spp., water lilies (*Nymphaea* sp.), *Oryza rufipogon* and *O. australiensis* form a mid-dense fringe. On the lake edges there are narrow bands of northern tea-tree and/or swamp tea-tree (*Melaleuca dealbata*) woodland.

Due to the size of the Lakefield National Park (537 000 ha) and the presence of extensive riparian thickets along most of the waterways, this area is considered to have a high conservation value in terms of protection of the habitat and feeding

Plate 1.1 Examples of freshwater lagoons within Lakefield National Park.

grounds of the estuarine crocodile (*Crocodylus porosus*). The freshwater crocodile (*C. johnstoni*) also occurs in permanent water bodies of the park. The rare and endangered Lakeland Downs mouse (*Leggadina lakedownensis*) has been recorded near a lagoon on the edge of the saline flat between the North Kennedy and Bizant rivers in *Eucalyptus acroleuca* open forest. The following frogs have been recorded in the area: striped rocketfrog (*Litoria nasuta*), northern sedgefrog (*L. bicolor*), common green tree frog (*L. caerulea*), ruddy tree frog (*L. rubella*), bumpy rocketfrog (*L. inermis*), northern laughing treefrog (*L. rothii*), pallid rocketfrog (*L. pallida*), chirping froglet (*Crinia deserticola*), greenstripe frog (*Cyclorana alboguttata*), eastern snapping frog (*C. novaehollandiae*), marbled frog (*Limnodynastes convexiusculus*), ornate burrowing frog (*L. ornatus*) and mimicking gungan (*Uperoleia mimula*). The freshwater snake (*Tropidonophis mairii*) and northern snake-necked turtle (*Chelodina rugosa*) have been recorded here. Fish recorded at German Bar include oxeye herring (*Megalops cyprinoides*), catfish (*Arius sp.*), catfish (*Neosilurus sp*.), hardyhead (*Craterocephalus sp*.), rainbow fish (*Melanotaenia sp*.), barramundi (*Lates calcarifer*), spangled perch (*Leiopotherapon unicolor*), barred grunter (*Amniataba percoides*), sooty grunter (*Hephaestus fulignosus*), jungle perch (*Kuhlia sp*.), mouth almighty (*Glossamia aprion*), empire gudgeon (*Hypseleotris compressa*) and northern trout gudgeon (*Mogurnda mogurnda*) (Herbert, B. pers. comm.). Two bird species also found in this freshwater habitat are nationally listed as rare species: the Cotton pygmy goose (*Nettapus coromandelianu)s* and the Burdekin duck (*Tadorna radjah).*

Feral pigs, causing substantial habitat damage, also inhabit this high value freshwater ecosystem (Mitchell 1998, 1999). The magpie goose (*Anseranas semipalmate),* Australian river prawn (*Macrobrachium australiense*), red claw yabbies (*Cherax quadricarinatus)* and numerous insects, amphibians, snakes and lizards living in freshwater habitats are believed to be threatened by feral pig impacts. The survival of northern snake necked turtles, which attain high densities in ephemeral swamps in the wet-dry tropics of northern Australia is also compromised by feral pig predation (Fordam *et al.* 2008). However, the true ecological impact of the damage done by feral pigs has not yet been quantified.

The observed spatial and temporal variations in pig environmental impacts that occur in this site are determined primarily by rainfall. During the wet season, heavy rainfall can inundate large tracts of the site and impact levels are minimal. As the water recedes during the dry season (drawdown), the pigs follow this receding water and cause damage. This type of sequential impact subsequently affects the whole wetland area and the measured pig impacts are not restricted to edge effects.

A number of previous feral pig research projects have been completed either in this specific study site or in similar habitats nearby (Mitchell 1998, 1999). Aerial population densities have been calculated for this study site (approximately 4 .3 pigs / km^2) and aerial shooting operations have been conducted. Other organisations (Natural Resources and Water, Environmental Protection Authority, Cape York Peninsular land Use Study and James Cook University) have extensively studied this region, so a broad range of environmental data is available to complement this study.

A map of Lakefield National Park (Figure 1.1) shows the location of all research sites used in this study. The fenced exclosures were used in study A and the lake systems were used in study B. A permit for Scientific Purposes (WITK04788007) was obtained from the Environmental Protection Agency prior to the study's commencement.

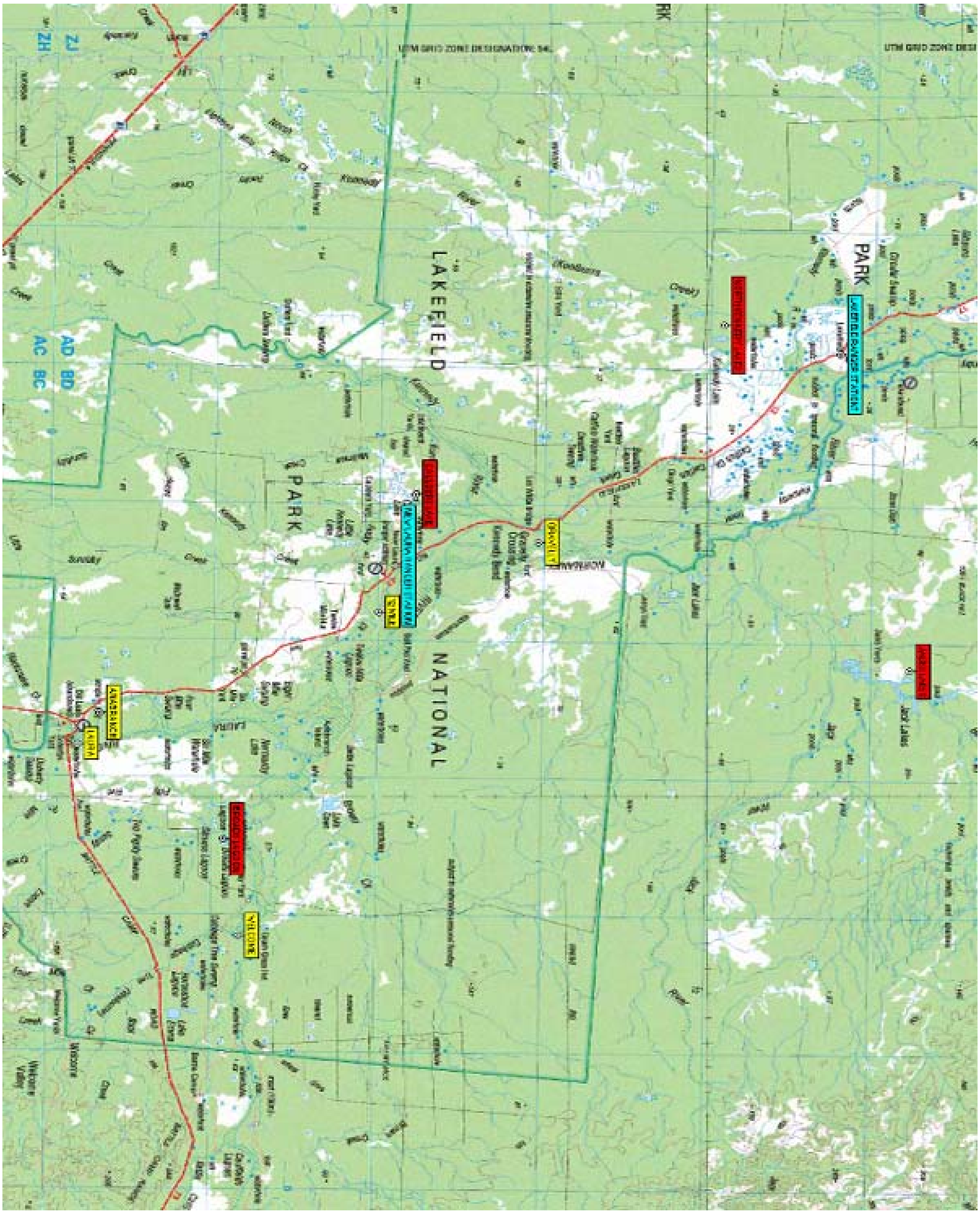

Figure 1.1 Map of study sites within Lakefield National Park. Yellow labels are the locations and names of the fenced and associated unfenced lagoons (study A); red labels are the locations and names of the lake systems (Study B).

CHAPTER 2.

Experimental Research to Quantify the Environmental Impact of Feral Pigs within Tropical Freshwater Ecosystems:

Study A: Ecological impacts of feral pigs on biodiversity

Prepared by D.W. Burrows, R. Doupe, J. Schaffer, M. Knott and A. Davis

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2.1 INTRODUCTION

Feral pigs (*Sus scrofa*) are among the most notorious introduced environmental pests in Australia (Choquenot *et al.* 1996). They have been recorded as predators of turtles, frogs and a variety of other animals. Through their significant digging and disturbance of soils, they have been widely implicated in significant alterations to the nature and character of soils, plant biomass and diversity, nutrient cycling and erosion. Their impacts upon freshwater ecosystems have not been experimentally investigated but are commonly perceived as a significant issue, especially in the dry tropics of northern Australia where they congregate around water sources in the dry season. Determining the extent and nature of pig impacts upon freshwater ecosystems requires separating the impacts of pigs from that of the substantial natural background variability and disturbance regimes that are often experienced in locations where pigs are prevalent.

The structure and function of wetland ecosystems in northern Australia are strongly influenced by seasonal changes in water level, and this drives natural disturbance regimes there (Junk *et al.* 1989; Mitsch and Gosselink 2000). Natural disturbance plays an important role in maintaining ecosystem biodiversity (Grime 1973), and alterations to the type and amount of disturbance can alter successional pathways (Sousa 1984) and thus ecological communities and processes (Hobbs and Huenneke 1992; Schmid *et al.* 2001). Throughout the world, ecological systems are becoming increasingly affected by non-native species whose activities can either introduce new forms of disturbance, or enhance or suppress existing disturbance regimes (Mack *et al.* 2000). This presents a considerable challenge for natural resource managers, who attempt to maintain native biodiversity while controlling the spread and impact of exotic species (Cushman *et al.* 2004).

Natural seasonal fluctuations in wetland water levels are sometimes accompanied by the encroachment of terrestrial organisms during the drying portion of the hydrologic cycle (Kirkman and Sharitz 1994) and this represents a potentially important secondary disturbance. For example, feral cattle may transform wetland plant communities through grazing and trampling of the receding littoral zone (Reimold *et al.* 1975; Walker 1995). The similar effects wrought by feral pigs are thought to be at least as detrimental (Arrington *et al.* 1999), because a substantial portion of their omnivorous diet is obtained by foraging or digging for roots, bulbs and other below-ground material (Howe *et al.* 1981; Baber and Coblenz 1987; Giménez-Anaya *et al.* 2008), both in and out of the water. This means that unlike cattle which graze at the wetland periphery, there is a concurrent disturbance of both the wetland substrate and water body, suggesting that pigs can strongly affect vegetation assemblages and so too wider ecological processes (Lavorel *et al.* 1997; Cushman *et al.* 2004; Tierney and Cushman 2006). There remain, however, very few examples of studies attempting to quantify the effects of feral pig foraging behaviours on freshwater wetland ecosystems (e.g. Dardaillon 1987; Kirkman and Sharitz 1994; Arrington *et al.* 1999), but these are clearly required for both effective wetland conservation and management of this global pest species (Lever 1985; Mayer and Brisbin 1991).

Aquatic macrophytes are a key component of habitat in wetlands, providing both an important food resource and structural complexity to the waterscape for associated biota (Thomaz *et al.* 2008). Since foraging feral pigs show a preference for macrophytes in moist habitats of northern Australia's wet-dry tropics (Hone 1990; Caley 1993, 1997), it implies that the ephemeral lagoons and their ecological communities in this region may be especially susceptible to significant disturbances by them (also see Bowman and Panton 1991; Mulrennan and Woodroffe 1998). Recently, Fordham *et al.* (2008) argued that fencing of ephemeral wetland lagoons in northern Australia was required to protect tropical freshwater lagoon habitats from "the destructive effects of feral pigs" (see also Bowman and McDonough 1991), but neither the impacts of pigs nor the effectiveness of fencing as a control method against them have been experimentally tested in these environments (Reddiex and Forsyth 2006).

The ecological effects of feral pigs have been listed as a key threatening process under the EPBC Act. The threat abatement plan developed as a result of that listing (Anon 2005) recommends studies on the impacts of feral pigs upon biodiversity. This study investigated the ecological effects of feral pig foraging in tropical floodplain lagoons and the efficacy of pig exclusion fencing as a means of limiting negative impacts of feral pigs.

2.2 MATERIALS and METHODS

2.2.1 Study Area and Experimental Design

Lakefield National Park is situated on Cape York Peninsula in north-eastern Australia and has been the site of previous pig research projects (Mitchell 1998). The region experiences a wet-dry monsoonal climate with a mean annual rainfall of about 1,200 mm falling predominantly during December to April. The floodplain landscape there is characterised by tropical savanna woodlands interspersed by numerous ephemeral, and some significant permanent, lagoons. As its name implies, Lakefield National Park contains a vast array of wetlands. In the wet season, several rivers draining through the park join to flood vast areas of the landscape. These floodwaters recede to form numerous wetlands, many of which dry out during the long hot dry season. This study is focused on ephemeral lagoons, which are abundant in Lakefield National Park and experience significant pig disturbance.

This study consisted of 'paired' lagoons at each of four locations (see Figure 1) - Twelve-mile (S 15°10′ E 144°21′), Anabranch (S 15°20′ E 144°26′), Laura (S 15°20′ E 144°27′) and Welcome (S 15°15′ E 144°35′). Like most wetlands in the area, these lagoons contain a wide array of submerged, emergent and floating aquatic plants. The abundance of aquatic plants and their generally shallow, accessible nature make them attractive to feral pigs, predominantly during the dry season. At each location one lagoon was enclosed by a pig-proof fence constructed at least 4 m above the demarcation between the wetland margin and the surrounding savannah woodland. The associated lagoon was fenced with a 4 plain wire fence to exclude feral cattle. Fences were erected in April 2007. Pig fencing was 1.1 m in height consisting of a plain top wire 200 mm above 900 mm of 150 mm × 150 mm netted wire mesh (200 mm^2; see also Hone and Atkinson 1983) and reinforced steel post corners. Bottom wires were barbed and secured into the earth to deter pig burrowing (Plate 2.14).

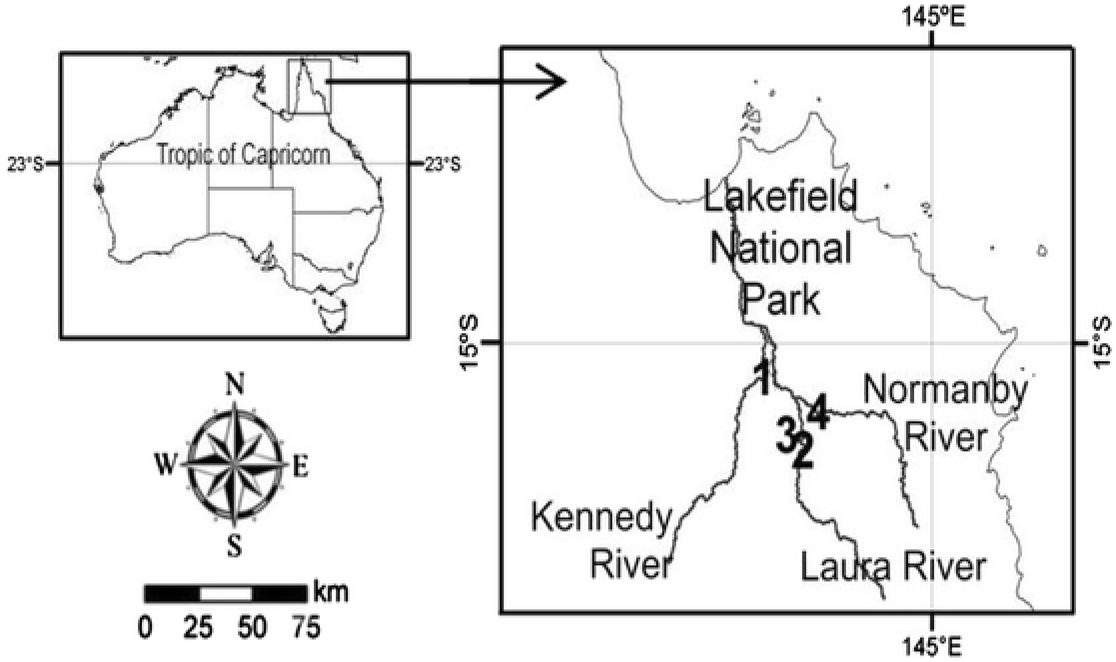

Figure 2.1 Map showing approximate positions of the Twelve-mile (1), Anabranch (2), Laura (3) and Welcome (4) waterholes at Lakefield National Park

Lagoon sizes ranged from approximately 0.15–0.5 hectares. Lagoon 'pairs' were chosen for their proximity to each other (about 200–350 m apart) and their dish or saucer-like shape which provided for a similar geomorphology and hydrology, rather than for similarities in size or floristic composition. This means they were not selected to be paired treatments in a statistical sense. As there is an underlying assumption that proximal 'paired' lagoons would be exposed to similar levels of pig foraging activity and natural levels of drawdown and disturbance, the experimental design simply focused on whether lagoons were fenced or not fenced.

2.2.2 Sampling and Data Collection

Data collection began in May 2008, about 12 months after fence construction. In both 2008 and 2009, we undertook three field trips, beginning in May of each year, when the lagoons were fully hydrated and accessible. Notionally, this was before any significant pig disturbance had occurred since the annual wet season flooding of the area. Lagoons were sampled twice more over the course of the dry season (when pig disturbance is greatest) but before they had dried out to such a point that their impending desiccation dominates their character. In 2008, these trips were conducted in July and September and in 2009, in July and October. By October 2009, most lagoons had dried out or nearly so, curtailing sampling of several aquatic parameters on that trip.

The deepest point of each lagoon was marked by fixing a permanent stake. On each sampling trip, a water quality multi probe (Hydrolab DataSonde) was attached 200 mm below the surface to concurrently record water pH, temperature (°C), dissolved oxygen (mg L^{-1} and % saturation) and electrical conductivity (µS cm^{-1}) at 30-min intervals for a 24-h period. Proximal lagoons were measured in tandem. Also at this deepest point we measured water depth and secchi depth (mm), and in the 2008 trips only, water was sampled for total and dissolved components of nitrogen and phosphorus, ammonia (all in µg L^{-1}) and turbidity (in Nephelometric Turbidity Units or NTU). Water samples were filtered on site where necessary, and then frozen and returned to the laboratory for assay using standard methods (APHA 2005). In 2008, in addition to the three scheduled trips for this project, we also were able to collect a fourth water quality sample in October 2008.

We also established between four and six permanent transects at 15–25 m intervals in each lagoon, beginning at the wetland margin and traversing the basin to finish at a similar point opposite. Quadrats of 4 m^2 were located at 10–20 m intervals along each transect, providing between 12 and 29 quadrats at each lagoon for a study total of 159. Although not individually marked, the approximate positions of all quadrats were easily located by measurement and so were more or less permanent. Within each quadrat, emergent (and submersed where possible) macrophytes were identified and percentage cover estimated by eye. Plant coverage as a percentage of lagoon surface area also allowed us to estimate the comparative extent of open water we expected to see increasing in the unfenced lagoons over time due to vegetation disturbance by pigs. Similarly, the extent of increasing bare ground was also estimated as an index of pig foraging activity.

At each lagoon a long-handled net with a triangular head covered with 250 µm mesh was used to sample for aquatic invertebrates at between three and six of the most dominant habitats. For example, in open water and submerged macrophyte habitats it was moved in a zigzag motion from the water surface to benthos, whereas in emergent macrophyte communities the net was vigorously forced from the bases of the plants to the water surface. An equivalent sampling effort was made over all lagoons. Samples were washed thoroughly in the net and then placed in 100% ethanol; in the laboratory they were stored in 70% ethanol until being identified to family, subfamily or class and counted.

To sample freshwater fishes on each trip, we set ten box-style baited fish traps (40x25x25cm with 1.5 mm mesh) in replicate habitats for a 24-h period at each site. Captured fish were identified, counted and returned to the water. We preferred this technique over either electro-fishing or seine netting since macrophyte coverage tended to hide stunned fish, and netting operations potentially disrupted and/or displaced aquatic plants and so may have confounded the effects of pigs. The added potential threat from estuarine crocodiles also prevented the use of several other netting techniques.

2.2.3 Data Analysis

Lagoon hydraulic residence time was estimated as the proportional loss of water between May and September [(end depth − start depth)/start depth]. Differences in arcsine transformed hydraulic residence time among fencing treatments were analysed using a one-way *t*-test.

The optical properties of lagoons were estimated using secchi depth and turbidity estimates. Secchi depths compared the visual clarity of different lagoons using the vertical contrast attenuation coefficient (K_c) from the relationship $K_c = 9/Z_{SD}$, where Z_{SD} is the depth at which the secchi disc disappears from view and gives a higher K_c value with decreasing secchi depth (see Kirk 1986). Turbidity, (NTU - nephelometric turbidity units), measures the distance that light is scattered within the water body, considered to be equivalent to scattering coefficient values in m^{-1} (Kirk 1986). We estimated the particulate fraction of nitrogen and phosphorus for each lagoon by subtracting the levels of dissolved nutrients from total nutrients. For lagoon water dissolved oxygen, temperature and pH, we divided their 24-h measurements obtained from the HYDROLAB Datasondes between day (0600–1800 hours) and night (1800–0600 hours) periods to estimate the respective biological effects of lagoon production and respiration. The amount of oxygen available for respiration was estimated by counting the number of hours at which percentage saturation values were below either chronic sub-lethal (75%) or acute toxic (30%) levels (see Sprague 1985; Butler and Burrows 2007) for each 24-h period. Since electrical conductivity was not expected to show any diurnal variation due to biological activity, we used the mean differences for each 24-h period. Because measures of water parameters for each lagoon at different sampling times were not independent, we tested for differences among fencing treatments and sampling times using a repeated measures analysis of variance, with treatment (fenced or unfenced) as the between subject (lagoon) effect and time (May, July or September) as the within-subject (lagoon) effect.

For each lagoon, macrophyte species abundance was estimated by total percentage plant cover, macrophyte species diversity was estimated by the number of species and macrophyte species diversity by the Shannon-Wiener diversity index (Shannon 1948). Aquatic invertebrate and fish relative abundances were estimated by absolute counts per unit sample. Aquatic invertebrate and fish species richness and diversity were estimated as for macrophytes. Macrophyte percentage cover was arcsine transformed and differences in all abundance, richness and diversity parameters among fencing treatments and sampling times were tested by repeated measures analyses of variance.

The similarities in macrophyte, invertebrate and fish species composition among lagoons were estimated using the Bray-Curtis similarity coefficient (Bray and Curtis 1957) following square-root transformation. The significance of differences in species composition among all lagoons, lagoons in different fencing treatments and lagoons at different sampling times was tested by a series of permutation procedures applied to each pair wise similarity matrix, using the ANOSIM procedure in the software package PRIMER 5.0 (Clarke and Gorley 2001).

2.3 RESULTS

2.3.1 The Physical Environment

Water depth in all lagoons decreased over time (Figure 2.2) and there was no significant difference in hydraulic residence times among the different fencing treatments ($t_6 = -0.007$, $P = 0.9$). These data confirm that the lagoon pairs have similar hydrologic regimes and geomorphologies, making them suitable as experimental study sites.

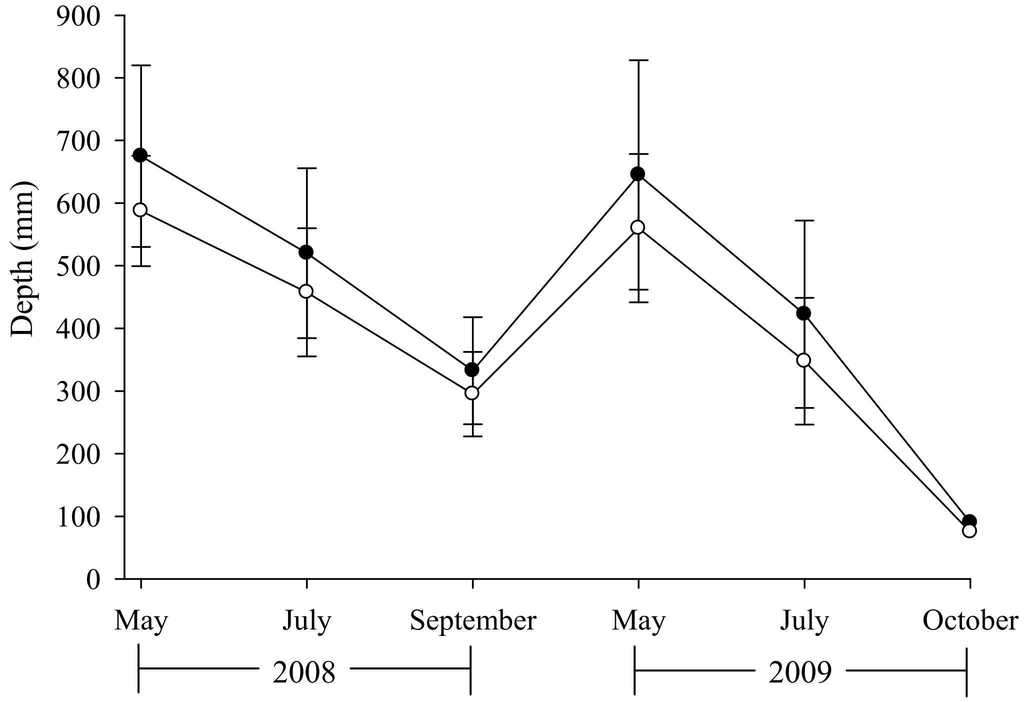

Figure 2.2 Variation in water depth at fenced (■) and unfenced (□) lagoons - 2008 and 2009.

The most obvious potential effect of pigs upon water quality and lagoon ecology is through reduction of water clarity, which we measured via secchi depth and turbidity. Secchi depth readings are summarised in Table 1

Table 2.1 Summary of mean secchi depth (cm) for each sampling trip.

	2008							
	Trip 1	Trip 1	Trip 2	Trip 2	Trip 3	Trip 3	2008 Overall	
	Fenced	Unfenced	Fenced	Unfenced	Fenced	Unfenced	Fenced	Unfenced
Secchi depth	612.5	472.5	467.5	323.75	332.5	151.7	470.83	330.91
	2009							
	Trip 1	Trip 1	Trip 2	Trip 2			2009 Overall	
	Fenced	Unfenced	Fenced	Unfenced			Fenced	Unfenced
Secchi depth	455.0	420.0	365.0	175.0			410.0	297.5

All lagoons developed significantly shallower secchi depths and therefore higher attenuation coefficients over time ($F_{2,12} = 6.90$, $P = 0.01$) regardless of fencing treatment ($F_{1,6} = 2.27$, $P = 0.18$). Water clarity became strongly affected by pig activity in the unfenced lagoons, which developed significantly higher turbidity than the fenced lagoons ($F_{1,6} = 3.92$, $P = 0.04$; Figure 2b) and increased over time ($F_{2,12} = 3.80$, $P = 0.05$). This effect was similarly apparent in both years and represents one of the strongest effects detected in this study.

On over half of the sampling occasions, (23 out of 40), the secchi depth was the same as the maximum depth. This occurs when the secchi disk can still be seen when on the bottom of the lagoon. When this occurs, the entire lagoon is considered to be within the euphotic zone of most aquatic macrophytes. However, statistically, it means that the true degree of clarity of that water is not apparent (i.e, how much greater would the secchi depth have been if the water depth was greater). Despite this limitation, significant effects were still apparent due to the magnitude of the effect on water clarity.

The scattering of light and heat reflection that occurs with more turbid water could be expected to alter water temperature. Despite this, we measured no significant differences in either day ($F_{1,6} = 0.33$, $P = 0.58$) or night ($F_{1,6} = 0.09$, $P = 0.77$) temperatures due to fencing, however there were significant seasonal effects for fenced and unfenced lagoons as the drying lagoons increased in temperature over the course of the year (for day $F_{2,12} = 17.92$, $P = 0.0002$; for night $F_{2,12} = 8.64$, $P = 0.004$).

2.3.2 Habitat Variables

Neither fencing treatment ($R = 0.08$, $P = 0.09$) nor sampling time ($R = 0.02$, $P = 0.31$) significantly affected macrophyte species composition, however there was a highly significant effect of lagoon ($R = 0.63$, $P = 0.001$) meaning that the wetlands themselves each differed in their macrophyte communities. We recorded 25 macrophyte species in the lagoons and while the number of species in each was not affected by fencing treatment ($F_{2,12} = 0.13$, $P = 0.72$), their numbers decreased significantly over time across all lagoons ($F_{2,12} = 11.47$, $P = 0.001$). Macrophyte species diversity was not significantly affected by fencing treatment ($F_{1,6} = 0.16$, $P = 0.85$), however there was a trend toward a temporal change ($F_{2,12} = 3.08$, $P = 0.08$). The percentage of macrophyte cover was not quite significantly affected by fencing treatment ($F_{1,6} = 5.23$, $P = 0.06$), but there was a significant effect of time on percentage cover ($F_{2,12} = 15.82$, $P = 0.0004$) and a significant interaction between fencing treatment and time ($F_{2,12} = 10.66$, $P = 0.002$). This was due to a decrease in macrophyte coverage over time in the unfenced, but not fenced treatments (Figure 2.3). This effect is better understood by the greater temporal increase in bare ground (Figure 2.4) and open water (Figure 2.5) in the unfenced lagoons than in the fenced lagoons, presumably due to pig foraging. Repeated measures analyses of variance showed that both proportion of bare ground and proportion of open water were significantly affected by fencing treatment (for bare ground, $F_{1,6} = 17.36$, $P = 0.005$; for open water, $F_{1,6} = 11.660$, $P = 0.01$) and sampling time (for bare ground, $F_{2,12} = 24.116$, $P < 0.0001$; for open water, $F_{2,12} = 5.18$, $P = 0.02$).

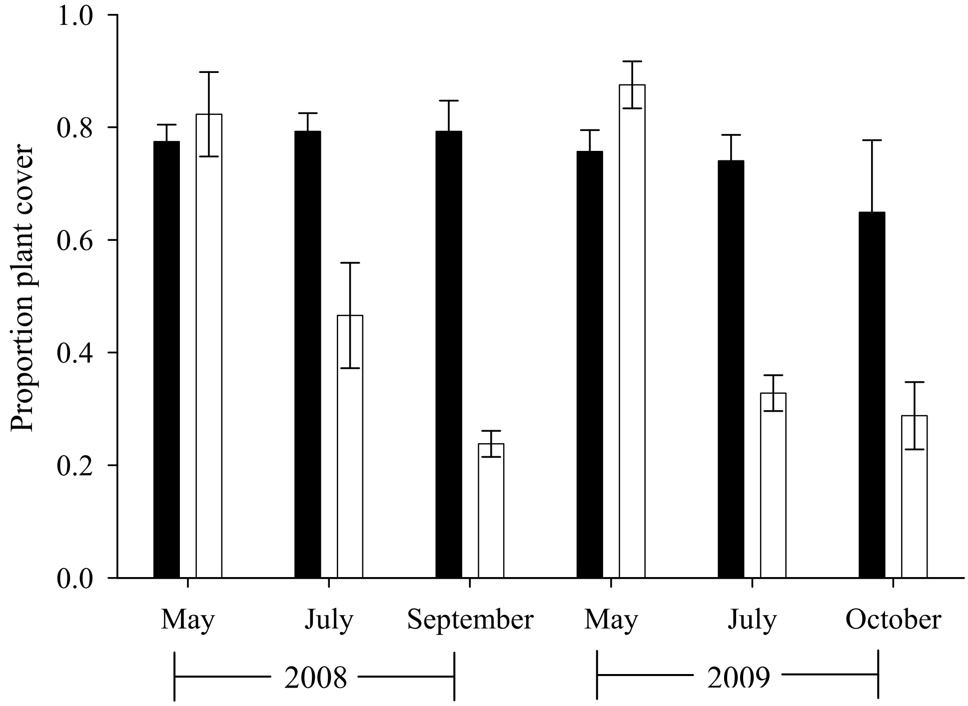

Figure 2.3. Changes in plant cover at fenced (■) and unfenced (□) lagoons - 2008 and 2009.

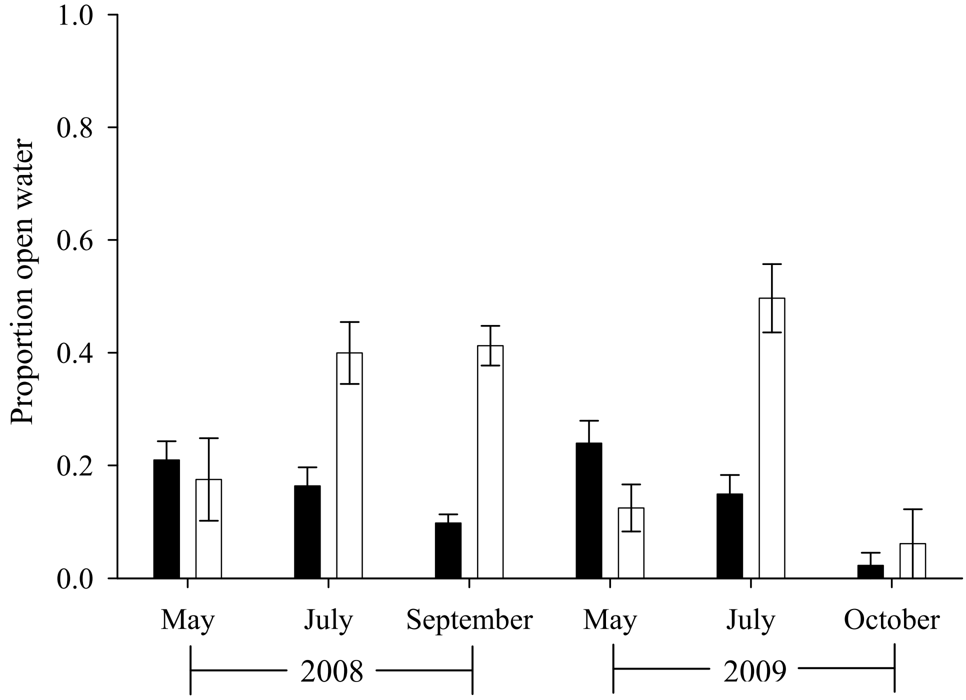

Figure 2.4. Changes in proportion of open water areas in fenced (■) and unfenced (□) lagoons - 2008 and 2009.

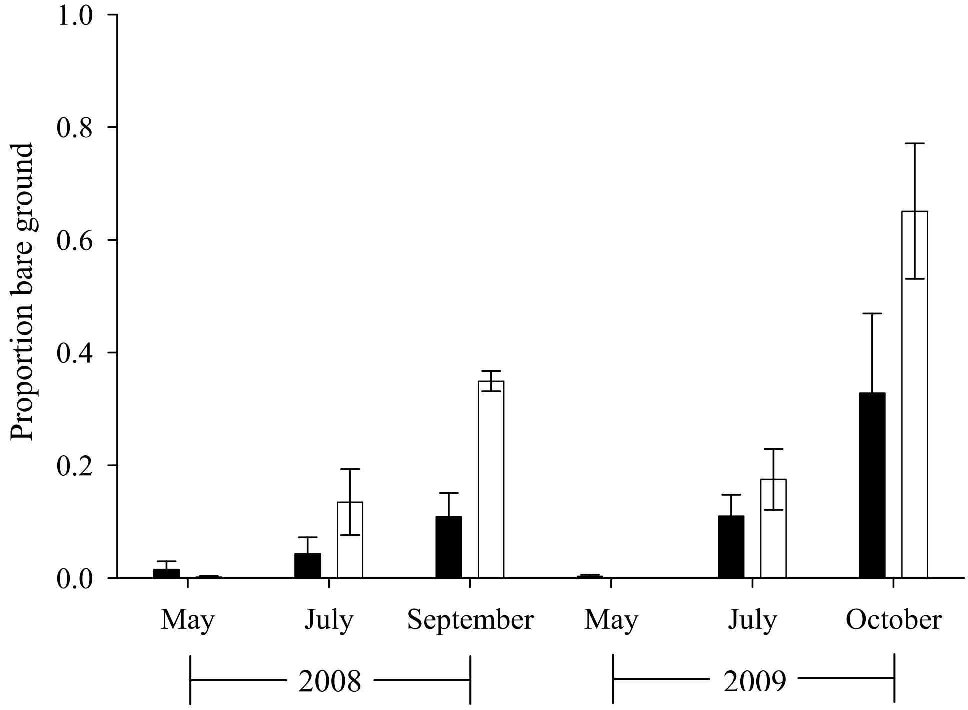

Figure 2.5. Changes in proportion of bare ground in fenced (■) and unfenced (□) lagoons - 2008 and 2009.

The visual changes in the four fenced, and four unfenced, lagoons over each sampling trip during 2008 are presented in plates 1-8. Similar changes were apparent in 2009.

(a)

(b)

(c)

Plate 2.1. Anabranch waterhole (fenced) in (a) May, (b) July and (c) September 2008.

(a)

(b)

(c)

Plate 2.2. Anabranch waterhole (unfenced) in (a) May, (b) July and (c) September 2008.

(a)

(b)

(c)

Plate 2.3. Laura waterhole (fenced) in (a) May, (b) July and (c) September 2008.

(a)

(b)

(c)

Plate 2.4. Laura waterhole (unfenced) in (a) May, (b) July and (c) September 2008.

(a)

(b)

(c)

Plate 2.5. Twelve-mile waterhole (fenced) in (a) May, (b) July and (c) September 2008.

(a)

(b)

(c)

Plate 2.6. Twelve-mile waterhole (unfenced) in (a) May, (b) July and (c) September 2008.

(a)

(b)

(c)

Plate 2.7. Welcome waterhole (fenced) in (a) May, (b) July and (c) September 2008.

(a)

(b)

(c)

Plate 2.8. Welcome waterhole (unfenced) in (a) May, (b) July and (c) September 2008.

Examples of pig disturbance and the physical environment of various water bodies within Lakefield National Park are presented in plates 2.9- 2.13.

Plate 2.9. Examples of complete turnover of lagoon substrate by feral pigs.

Plate 2.10. Examples of lagoon margins with extensive aquatic macrophyte beds.

Plate 2.11. Effect of pigs upon wetland margins and water clarity.

Plate 2.12. Pig diggings around margins of shallow lakes.

Plate 2.13. Examples of natural incipient dryness late in the dry season.

2.3.3 Aquatic Invertebrates

A total of 67 invertebrate taxa including 60 families were sampled during this study. The numbers of taxa sampled was almost the equal between the fenced and unfenced treatments and between the two years examined (Table 2). Raw data from invertebrate sampling is presented in Appendix B.1 and B.2.

Table 2.2. The number of aquatic invertebrates and invertebrate taxa in fenced and unfenced lagoons for 2008 and 2009.

	2008		2009		Combined Years	
	Fenced	Unfenced	Fenced	Unfenced	Fenced	Unfenced
Total No. Taxa	55	59	54	52		
Av. No. Taxa	29.7	28.2	30.0	28.2	29.8	28.2
No. Individuals	9039	8932	4400	5496	13439	14428

Neither treatment ($R = 0.01$, $P = 0.39$) nor sampling time ($R = 0.063$, $P = 0.12$) affected aquatic invertebrate species composition, however there was a highly significant effect of lagoon ($R = 0.59$, $P = 0.001$), indicating that each lagoon contained a distinct aquatic invertebrate community. The number of aquatic invertebrate species was not affected by fencing treatment ($F_{1,6} = 0.10$, $P = 0.76$) but did significantly decrease over time as mobile species dispersed ($F_{2,12} = 7.72$, $P = 0.007$). Lagoon fencing made no significant difference to either aquatic invertebrate diversity ($F_{1,6} = 0.35$, $P = 0.57$) or abundance ($F_{1,6} = 0.01$, $P = 0.95$) and neither was affected by time (for diversity $F_{2,12} = 1.79$, $P = 0.20$; for abundance ($F_{2,12} = 0.24$, $P = 0.78$).

2.3.4 Freshwater Fish

A total of 11 freshwater fish species were found in the lagoons; 10 of these were found in unfenced lagoons and 9 in fenced lagoons (Table 3). All raw data from the fish sampling at each site on each sampling trip are presented in Appendix A. Most species occurred in low abundances as the trapping method utilised is not the most effective for quantitative sampling of many species that are present in the lagoons. The two most abundant species, which are more suited to trap sampling methods, are the sailfin glassfish *Ambassis agrammus* and the hyrtl's tandan (an eel-tailed catfish) *Neosilurus hyrtlii*, both of which were much more abundant in fenced compared to unfenced lagoons.

Fish species composition was not affected by either fencing treatment ($R = 0.04$, $P = 0.23$) or sampling time ($R = -0.04$, $P = 0.74$), but there were significant differences in fish species composition among the lagoons ($R = 0.34$, $P = 0.001$). Lagoon fencing had no significant effect on either the number of fish species ($F_{1,6} = 0.62$, $P = 0.46$), fish species diversity ($F_{1,6} = 0.13$, $P = 0.72$) or fish abundance ($F_{1,6} = 3.14$, $P = 0.46$), and no parameter significantly changed over time (for species richness $F_{2,12} = 1.66$, $P = 0.23$; for diversity $F_{2,12} = 0.06$, $P = 0.12$; for abundance $F_{2,12} = 0.69$, $P = 0.51$). However, it should be noted that the fish catch data are variable, meaning that only very large effects could be detected. As an example, for sailfin glassfish individually, there was no significant difference in abundance between fenced and unfenced lagoons ($F_{1,22} = 2.66$, P=0.11), despite these fish being more than twice as abundant in fenced lagoons.

Table 2.3. Abundances of fish between fenced and unfenced lagoons – 2008 and 2009.

	Fenced	Unfenced	Total
Ambassis agrammus	1106	541	1663
Amniataba percoides	1	0	1
Glossamia aprion	16	10	26
Leiopotherapon unicolor	7	16	23
Melanotaenia splendida	59	21	80
Mogurnda adspersa	0	2	2
Nematalosa erebi	0	6	6
Neosilurus ater	5	3	8
Neosilurus hyrtlii	185	55	240
Oxyeleotris lineoloata	24	76	100
Toxotes chatareus	6	2	8
No. Species	**9**	**10**	**11**
No. Fish	**1418**	**729**	**2147**

There was no pattern of decline in fish abundance over the course of each dry season (Table 4). There was also no significant decline in sailfin glassfish ($F_{2,9} = 0.63$, P=3.01) or hyrtl's tandan ($F_{2,9} = 0.81$, P=3.01) individually. Hyrtl's tandan are particularly patchy in their distribution. For instance, on trip 1 in 2008, 56 of the 95 hyrtl's tandan's caught on that entire trip came from just one site and on trip 1 of 2009, 71 of the 72 caught came from just one site. In fact, these two samples made up 52% of the entire catch of hyrtl's tandan across the whole study.

Table 2.4. Number of fish caught on each trip in 2008 and 2009.

		2008				2009	
	Trip 1	Trip 2	Trip 3	2008 Total	Trip 1	Trip 2	2009 Total
Ambassis agrammus	477	201	503	1181	241	241	482
Amniataba percoides	0	0	0	0	1	0	1
Glossamia aprion	4	1	4	9	12	5	17
Leiopotherapon unicolor	5	3	5	13	7	3	10
Melanotaenia splendida	9	20	23	52	9	19	28
Mogurnda adspersa	0	0	1	1	1	0	1
Nematalosa erebi	3	0	0	3	3	0	3
Neosilurus ater	5	0	2	7	2	0	2
Neosilurus hyrtlii	95	49	13	157	72	1	73
Oxyeleotris lineoloata	12	21	36	69	8	23	31
Toxotes chatareus	0	0	0	0	8	0	8
No. Species	**8**	**6**	**8**	**9**	**11**	**6**	**11**
No. Fish	**610**	**295**	**587**	**1492**	**354**	**292**	**646**

2.3.5 Water Chemistry

The raw data from laboratory-analysed water samples collected during this study are presented in Table 2.5. Sampling for these parameters only occurred in 2008. Nutrient concentrations increased over the course of the season, especially in unfenced lagoons. The highest values were recorded in Welcome unfenced lagoon in September. Here, the ammonia level was very high and potentially at toxic levels (see discussion in section 2.4.2). As discussed earlier, exclusion of pigs had a significant beneficial effect on the secchi depth of fenced waterholes. Turbidity is a related parameter, measuring the scattering of light in the water. The turbidity of fenced and unfenced waterholes over 2008 is graphed in 2.6. As for secchi depth, all waterholes had similar turbidity at the beginning of the dry season (May) but unfenced waterholes clearly developed much higher turbidity over the course of the dry season.

Plate 2.14. Example of pig activity outside the exclusion fencing.

Table 2.5. Laboratory-analysed water quality parameters.

Site	Trip	Turbidity (NTU)	Total Nitrogen (µg N/L)	Total Filterable N (µg N/L)	Ammonia (µg N/L)	Total Phosphorus (µg P/L)	Total Filterable P (µg P/L)	Particulate N (µg N/L)	PN proportion of TN	Particulate P (µg P/L	TN:TP Ratio
12 Mile Fenced	May	11.3	1487	690	7	182	91	797	53.6%	91	18.1
12 Mile Unfenced	May	9.6	901	605	6	95	90	296	32.9%	5	21.0
Anabranch Fenced	May	5.6	752	350	13	49	11	402	53.5%	38	33.9
Anabranch Unfenced	May	14.6	671	378	8	40	7	293	43.7%	33	37.1
Laura Fenced	May	11.2	741	416	12	85	10	325	43.9%	75	19.3
Laura Unfenced	May	14.8	1048	318	5	48	9	730	69.7%	39	48.3
Welcome Fenced	May	7.3	896	634	9	42	13	262	29.2%	29	47.2
Welcome Unfenced	May	10.4	749	488	13	27	11	261	34.8%	16	61.3
12 Mile Fenced	July	4.7	1524	1239	3	215	92	285	18.7%	123	15.7
12 Mile Unfenced	July	160	1151	909	6	249	32	242	21.0%	217	10.2
Anabranch Fenced	July	7.6	622	525	9	58	22	97	15.6%	36	23.7
Anabranch Unfenced	July	31	1643	1224	19	534	309	419	25.5%	225	6.8
Laura Fenced	July	6.8	820	501	5	111	16	319	38.9%	95	16.3
Laura Unfenced	July	40	850	606	6	152	23	244	28.7%	129	12.4
Welcome Fenced	July	6.2	799	676	6	57	19	123	15.4%	38	31.0
Welcome Unfenced	July	34	1107	891	25	119	20	216	19.5%	99	20.6
Anabranch Fenced	Sep	23	992	954	5	64	14	38	3.8%	50	34.3
Anabranch Unfenced	Sep	33	2197	1970	12	250	50	227	10.3%	200	19.4
Laura Fenced	Sep	10	828	668	13	63	5	160	19.3%	58	29.1
Laura Unfenced	Sep	99	1782	1096	9	346	5	686	38.5%	341	11.4
Welcome Fenced	Sep	22	2085	1507	59	128	17	578	27.7%	111	36.0
Welcome Unfenced	Sep	4048	7419	3784	1830	910	33	3635	49.0%	877	18.0

Another way of viewing this data is to plot the ratio of the difference between each paired lagoon for each parameter on each sampling occasion. For turbidity, this is shown in Figure 2.7. A ratio of 1.0 shows no difference in turbidity between a paired (fenced and unfenced) site. Ratios greater than 1.0 indicate higher values for the unfenced lagoon in each pair and ratios below 1.0 indicate higher values for the fenced lagoon in each pair. Figure 2.7 clearly shows the increasing ratio (turbidity) of unfenced lagoons as the season progresses.

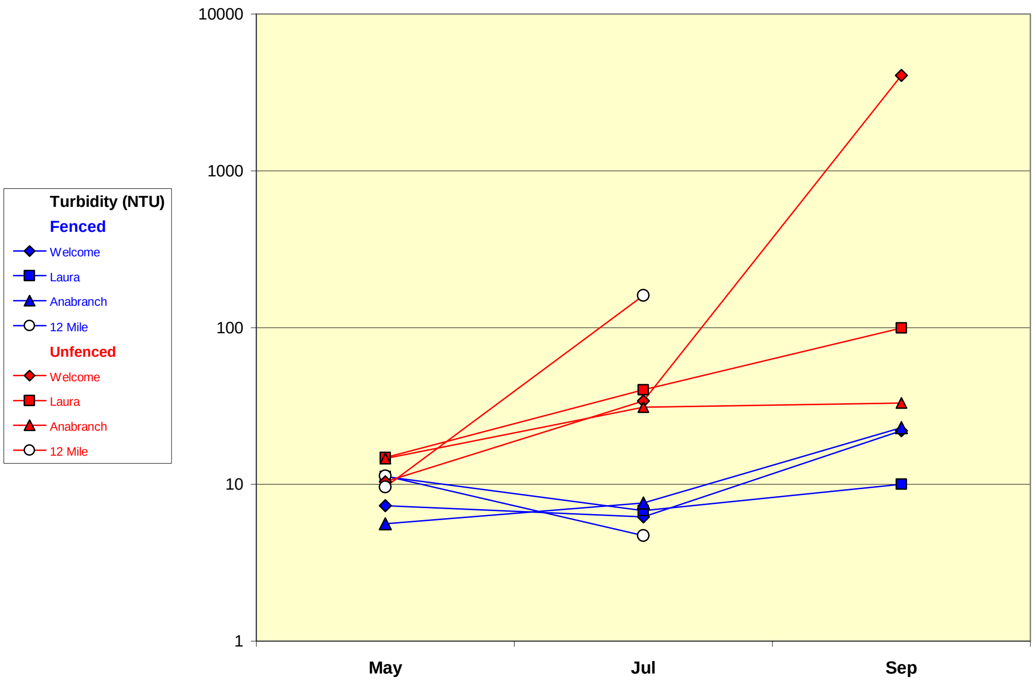

Figure 2.6. Effects of fencing on the turbidity of the experimental lagoons.

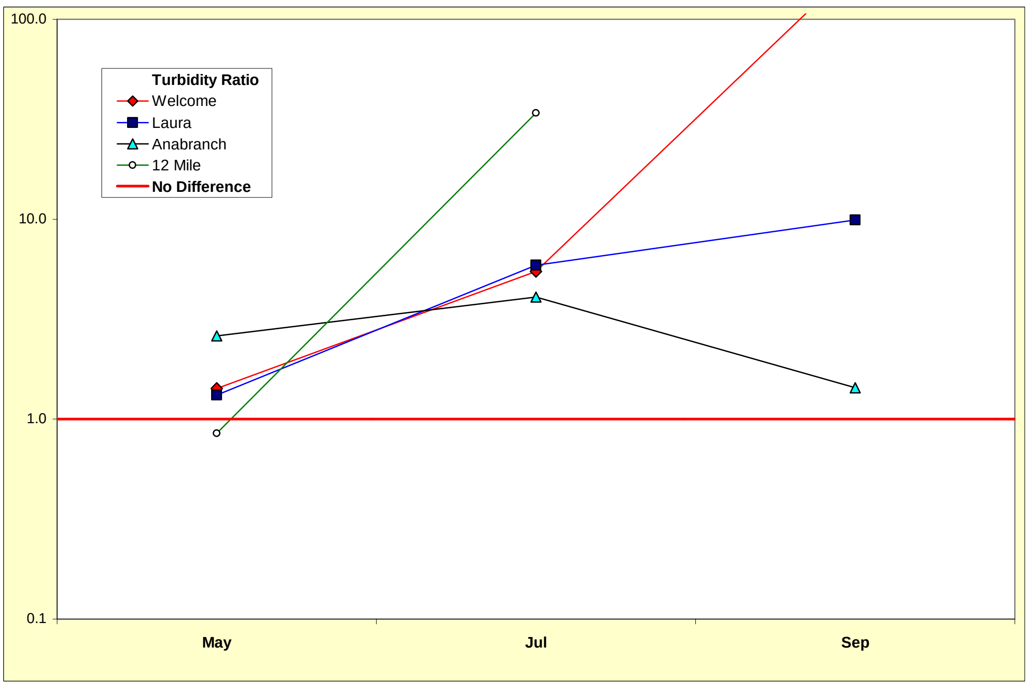

Figure 2.7. Effects of fencing on the between-site turbidity ratio of paired experimental lagoons.

Statistically, fencing treatment did not significantly affect either total nitrogen ($F_{1,6} = 0.26$, $P = 0.63$), total dissolved nitrogen ($F_{1,6} = 0.19$, $P = 0.68$), ammonia ($F_{1,6} = 0.01$, $P = 0.95$), total phosphorus ($F_{1,6} = 2.48$, $P = 0.16$) or total dissolved phosphorus ($F_{1,6} = 0.29$, $P = 0.61$) (Figure 2.8). However, time significantly affected the concentrations of total nitrogen ($F_{2,12} = 5.38$, $P = 0.02$), total dissolved nitrogen

($F_{2,12} = 13.46$, $P = 0.0009$) and total phosphorus ($F_{2,12} = 4.62$, $P = 0.03$), but not ammonia ($F_{2,12} = 1.04$, $P = 0.38$) or total dissolved phosphorus ($F_{2,12} = 0.10$, $P = 0.39$). Fencing made no significant difference to either particulate phosphorus ($F_{1,6} = 5.34$, $P = 0.06$) or particulate nitrogen levels ($F_{1,6} = 1.27$, $P = 0.30$), however particulate phosphorus levels did significantly increase with time ($F_{2,12} = 4.74$, $P = 0.03$) (Figures 8a-8g).

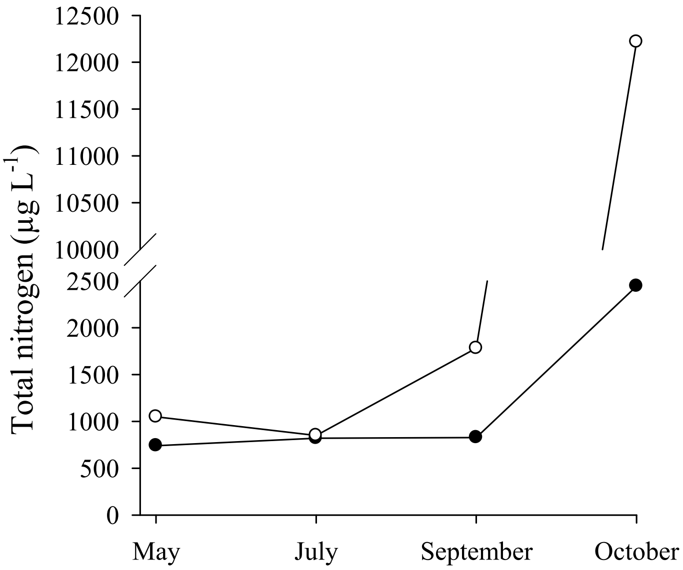

Figure 2.8a. Comparative effects of pig diggings on total nitrogen within exclusion fenced (●) and non-fenced (o) lagoons.

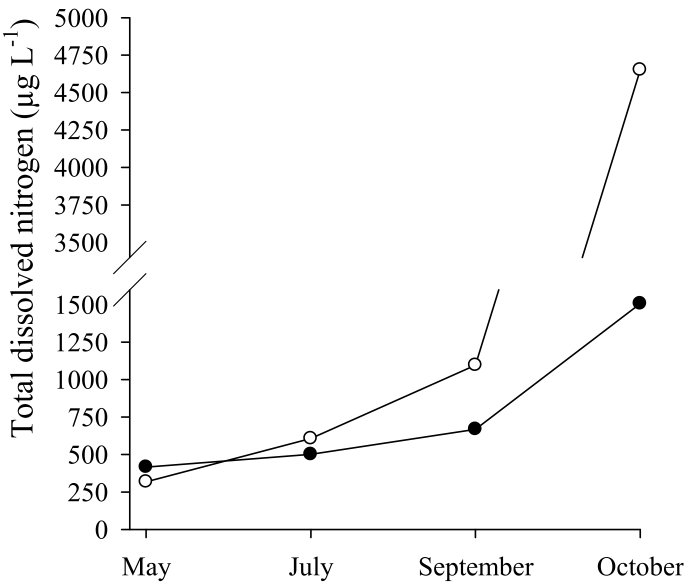

Figure 2.8b. Comparative effects of pig diggings on total dissolved nitrogen within exclusion fenced (●) and non-fenced (o) lagoons.

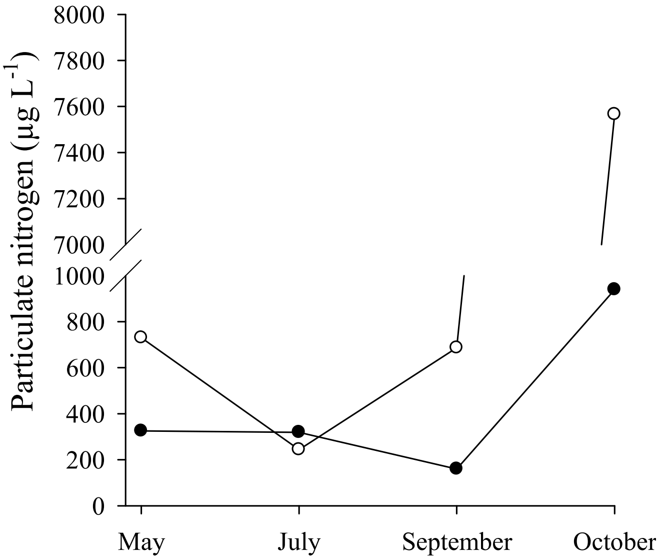

Figure 2.8c. Comparative effects of pig diggings on particulate nitrogen within exclusion fenced (●) and non-fenced (o) lagoons.

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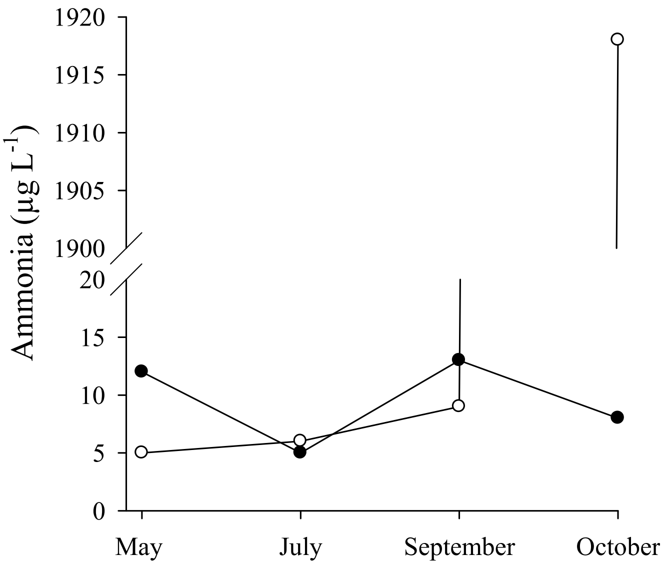

Figure 2.8d. Comparative effects of pig diggings on ammonia within exclusion fenced (●) and non-fenced (o) lagoons.

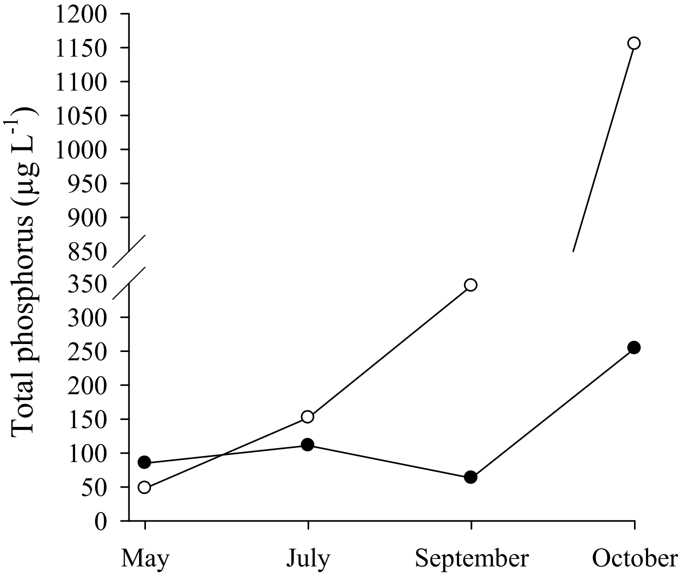

Figure 2.8e. Comparative effects of pig diggings on total phosphorus within exclusion fenced (●) and non-fenced (o) lagoons.

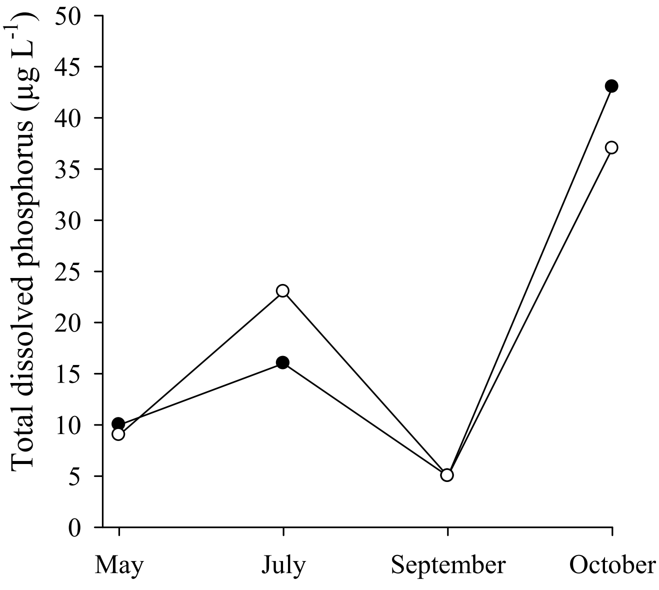

Figure 2.8f. Comparative effects of pig diggings on total dissolved phosphorus within exclusion fenced (●) and non-fenced (o) lagoons.

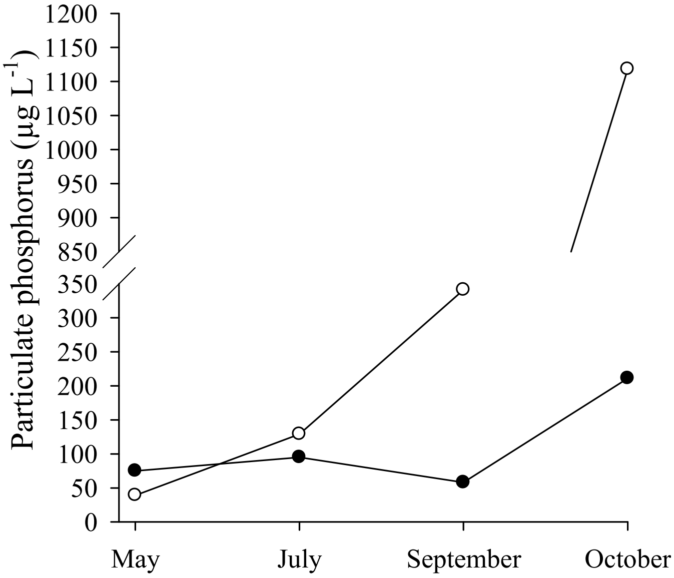

Figure 2.8g. Comparative effects of pig diggings on particulate phosphorus within exclusion fenced (●) and non-fenced (o) lagoons.

Although the nutrient parameters did not yield a statistically significant result for the effect of fencing, the level of replication (n=4) was low for such variable data. Visual plots of total nitrogen, total phosphorus and ammonia data, using the ratio of data from each unfenced and fenced pair of lagoons on each sampling occasion, are provided in Figure 2.9a-2.9c. These provide a compelling picture of the effect of fencing on nutrient concentrations in waters of these lagoons. In all cases (bar one for ammonia), the ratios are approximately 1.0 in the May samples (beginning of the dry season) but much greater than 1.0 by the September samples, indicating an ecologically (even if not statistically) significant effect of pigs upon these parameters.

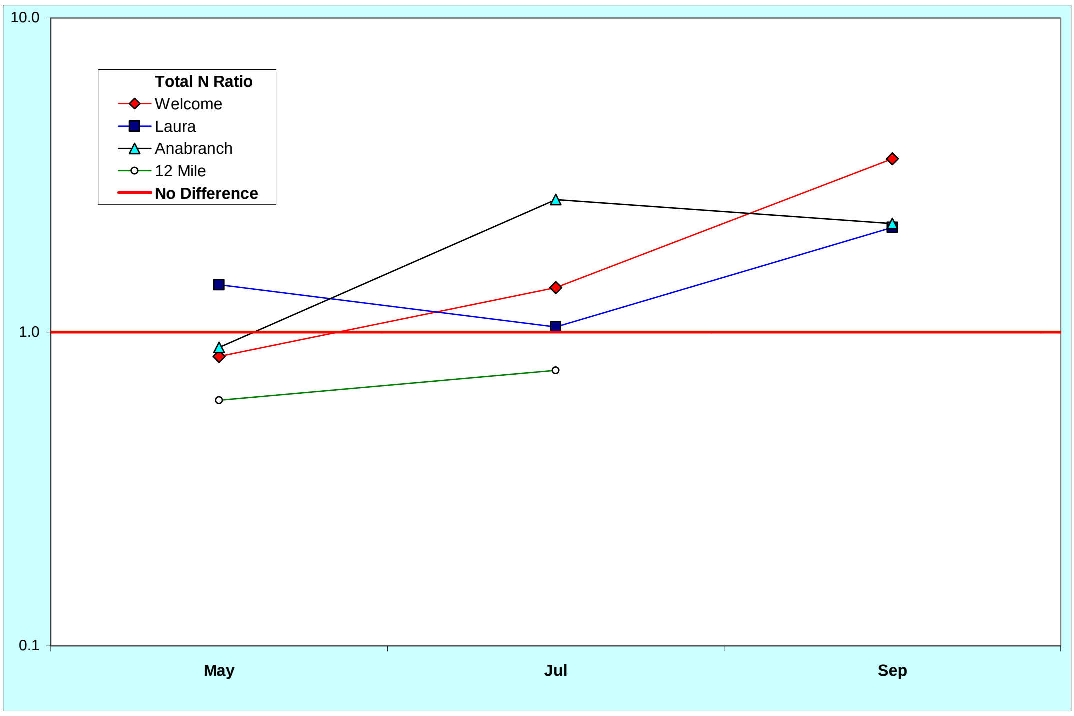

Figure 2.9a. Effects of fencing on the between-site total nitrogen ratios of paired lagoons.

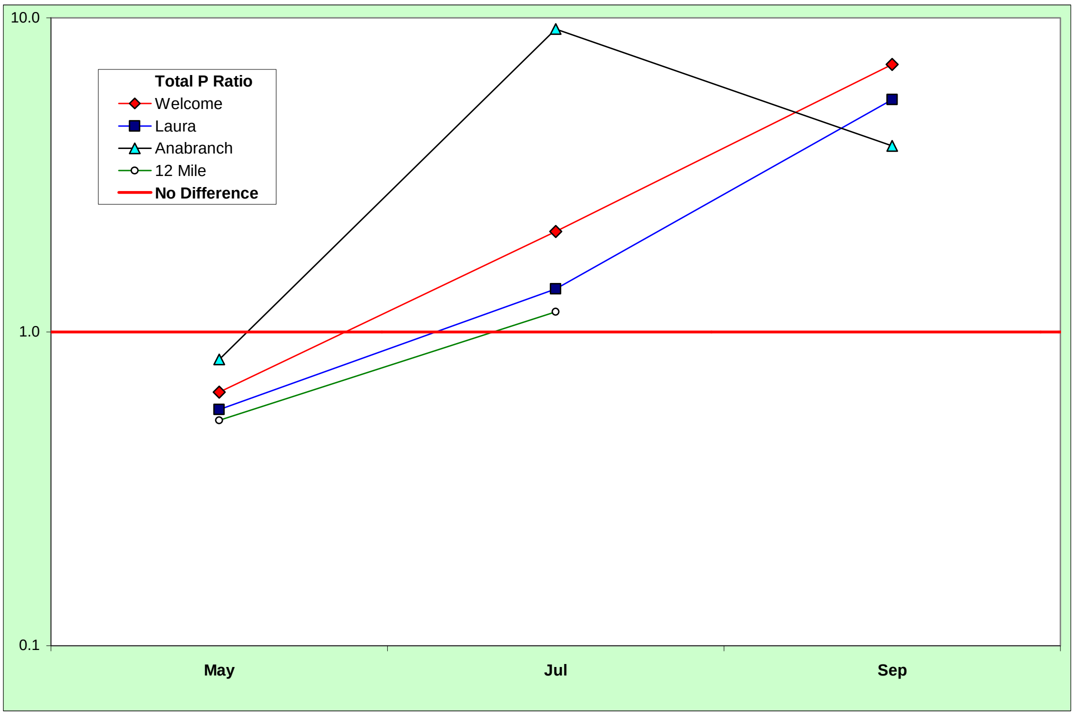

Figure 2.9b. Effects of fencing on the between-site total phosphorus ratios of paired lagoons.

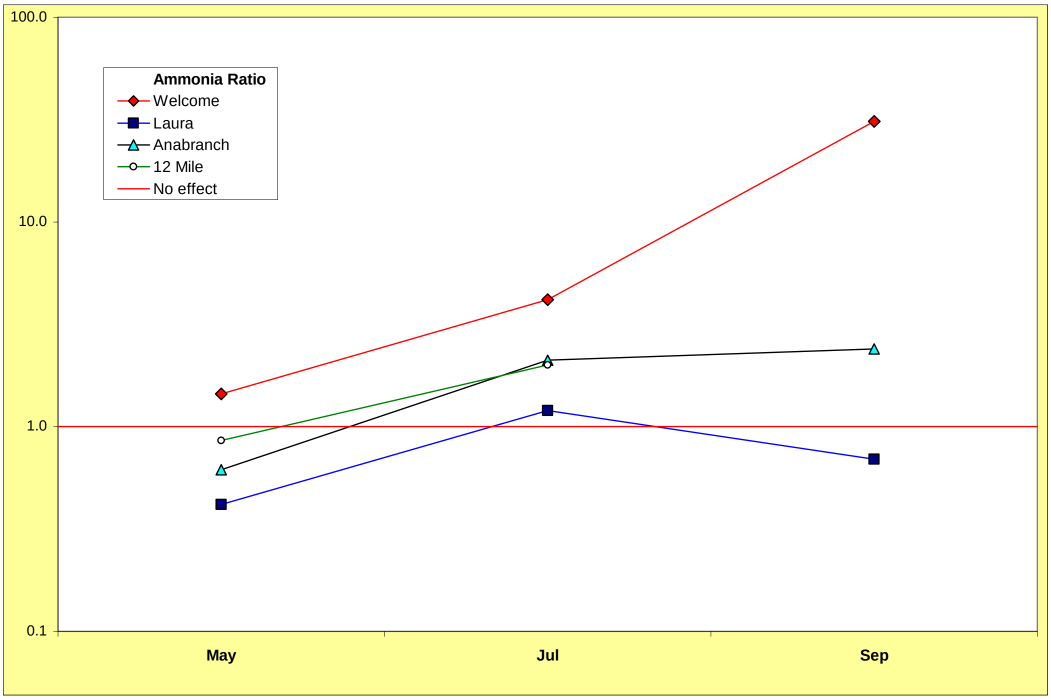

Figure 2.9c. Effects of fencing on the between-site ammonia ratios of paired lagoons.

Fenced lagoons had markedly higher dissolved oxygen levels during the day ($F_{1,6} = 34.39$, $P = 0.001$) and night ($F_{1,6} = 13.27$, $P = 0.0008$), and this increased over time for both day ($F_{2,12} = 4.94$, $P = 0.02$) and night ($F_{2,12} = 13.78$, $P = 0.0008$) readings (Figures 2.10a and 2.10b). Percentage saturation of dissolved oxygen followed this same trend, becoming significantly lower in the unfenced lagoons during the day ($F_{1,6} = 20.59$, $P = 0.003$) and night ($F_{1,6} = 7.72$, $P = 0.03$), and this also increased over time (for day $F_{2,12} = 5.03$, $P = 0.02$; for night $F_{2,12} = 7.86$, $P = 0.006$). Dissolved oxygen levels in the unfenced lagoons also recorded significantly more hours below both the 30% saturation ($F_{1,6} = 6.88$, $P = 0.03$) and 75% saturation ($F_{1,6} = 27.25$, $P = 0.002$) levels. There was a significant interaction between fencing treatment and time ($F_{2,12} = 7.06$, $P = 0.009$) for the 75% threshold, indicating that unfenced treatments recorded more time below this level throughout the dry season. The unfenced lagoons became more acidic, with a significantly lower pH during the day ($F_{1,6} = 12.46$, $P = 0.01$) and night ($F_{1,6} = 25.15$, $P = 0.002$), and this was maintained over time (for day $F_{2,12} = 17.65$, $P = 0.0003$; for night $F_{2,12} = 18.20$, $P = 0.0002$).

Fencing treatment had no significant effect on electrical conductivity ($F_{1,6} = 0.01$, $P = 0.93$), however all wetlands increased conductance over the dry season as water levels receded ($F_{2,12} = 9.00$, $P = 0.004$).

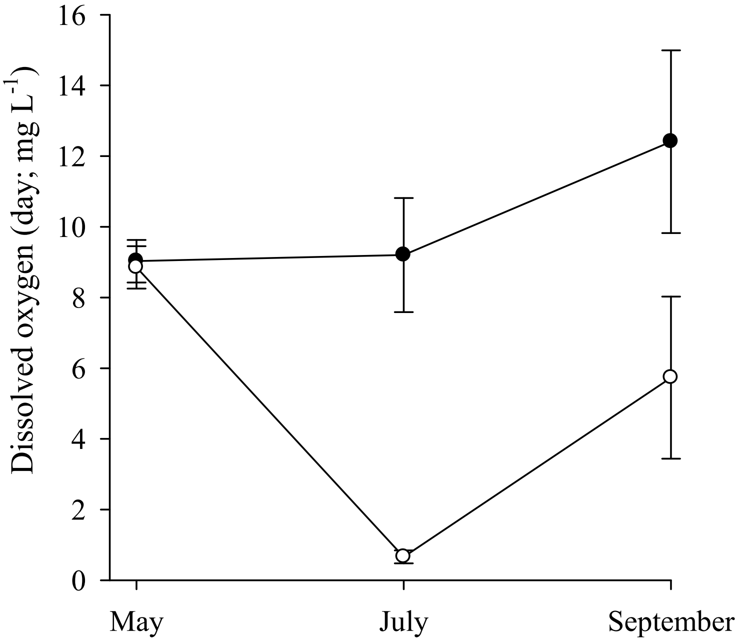

Figure 2.10a. Comparative effects of pig exclusion fencing (●) and non-fenced (o) lagoons on daytime dissolved oxygen. Values for dissolved oxygen levels in mg L^{-1} are 12-hour means ± S.E.

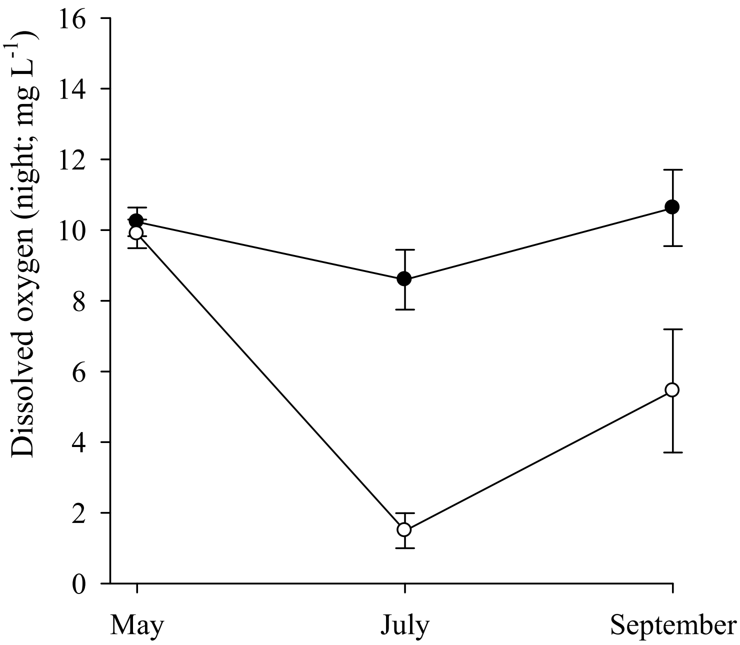

Figure 2.10b. Comparative effects of pig exclusion fencing (●) and non-fenced (o) lagoons on night time dissolved oxygen. Values for dissolved oxygen levels in mg L^{-1} are 12-hour means ± S.E.

2.4. DISCUSSION

Fencing is commonly accepted as an environmental management tool for large ungulates such as cattle and sheep and a number of studies have examined various aspects of the benefits of fencing in different environments both here and in Australia (e.g. Taylor 1996; Burrows 2000a, 2000b, 2003; Prior 2000). To our knowledge however, this is the first study examining the effects of fencing as a management tool for reducing the ecological effects of feral pigs in Australia. Although this study is meant to provide a more general test of the ecological benefits of fencing, by necessity, the benefits and management applications of fencing will differ in different wetland environments.

2.4.1 Habitat Variables

The lagoon 'pairs' had similar water depth and rates of water level decline, indicting their experimental suitability for this study. Although seasonal water level decline is the major driver of the ecology of these lagoons (i.e. all will go dry late in the dry season), these effects were equal across the lagoon pairs. The fencing treatments were effective as we did not observe pig disturbance within fenced lagoons, despite evidence of pigs circling the fenced boundaries of these lagoons.

Pig disturbance created an obvious disturbance of the lagoons. The entire substrate around the margin of all unfenced lagoons was turned over as a result of pig rooting activities. This effect became extreme later in the dry season. This pig rooting resulted in progressively decreasing aquatic plant cover and increasing amounts of open water and bare ground. This loss of plant cover also occurred to a significant degree in the fenced lagoons, demonstrating that it is a natural feature of these lagoons, but that pig disturbance heightened the effect. Although water clarity also naturally declines over the course of the dry season, this effect was further exacerbated as a result of pig disturbance. This loss of aquatic plant cover and decline in water clarity were the strongest effects detected in this study.

2.4.2 Water Quality

Fencing treatment did have a large affect upon water clarity as already discussed. Although statistically, no effect was noted on any of the nutrient parameters analysed, graphical examination of the data showed a compelling benefit of fencing on nutrient concentrations. Nutrient concentrations did increase over the course of the season as would be expected with decline in water level but the effects of pigs greatly increased the nutrient levels beyond their natural increase.

The elevated ammonia value at Welcome unfenced lagoon in Sep 2008 (1810 ug N/L) was not high enough to be toxic to aquatic fauna at the time of the survey mainly because the ambient pH levels were low (maximum value 6.9). The ANZECC guideline trigger value for protection against ammonia toxicity at a pH of 6.9 is 2360 ug N/L. However, if the pH values were to rise to levels above 7.3 (and that is undoubtedly possible) the ANZECC guideline would be <1800 ug N/L meaning that the existing ammonia concentrations would be in exceedance. Ammonia toxicity increases rapidly with increasing pH and at a pH of 9 the ANZECC guideline is only 180 ug N/L (i.e. 10 times lower than the concentration in question here). Based on local toxicity tests (Pearson *et al.* 2003) the ammonia concentration at Welcome unfenced lagoon could become toxic enough to mortally injure sensitive aquatic fauna within 6 to 24 hours if pH was to rise to values above 8.5. pH increases of this kind could occur if a cynaobacteria bloom was to develop (and the nutrient levels are certainly high enough to support this). They could also develop if the suspended solids in the water column were to settle out sufficiently to allow the submergent parts of plants and/or algae to photosynthesise. Note that in this region the normal daytime pH values of clear waters that support healthy assemblages of submerged plants and algae are typically alkaline, ranging from about 7.3 up to levels in excess of 9.4. The low pH levels observed at the unfenced Welcome site in September suggest that photosynthetic activity within the water column was inhibited (due to excessive water turbidity) and that there was significant decomposition of organic matter (resulting in the generation of carbonic acid which lowers pH).

Dissolved oxygen conditions progressively deteriorated over the course of the season but this affect was heightened in unfenced lagoons compared to fenced lagoons. Thus pig disturbance is implicated in negatively impacting dissolved oxygen availability. Along with water clarity, dissolved oxygen is probably the most ecologically important water quality parameter in tropical environments. Dissolved oxygen availability could have been affected by both increased consumption and decreased production. The decreased water clarity caused by pig rooting activities and reduced availability of aquatic plants (and presumably algal production) decrease photosynthetic production whilst increased soil disturbance and input of pig wastes increases oxygen consumption.

Unfenced lagoons frequently experienced dissolved oxygen levels between 30 and 70% saturation, which provide at least chronic sub-lethal effects for the associated biota (Sprague 1985; Butler and Burrows 2007). We didn't find evidence for toxic effects on either fish or invertebrates, although this may be because levels above 30% are not thought to be lethal to freshwater fishes (Dean and Richardson 1999) and aquatic surface respiration by fishes may compensate for periodically anaerobic conditions (McNeil and Closs 2007). Little is known of how low dissolved oxygen levels affect invertebrate faunas.

Electrical conductivity of the water was not affected by fencing treatment but did increase over the course of the season. However, the levels always remained within limits not known to cause harm to biota.

2.4.3 Aquatic Invertebrates and Macrophytes

A total of 25 aquatic plant species were recorded in the lagoons during this study. The species composition of the macrophyte communities was not affected by fencing but did differ significantly between the lagoons. Thus, the species composition of each lagoon was different, regardless of pig disturbance. This reflects the individual character of discrete lagoons in seasonal tropical environments. Fencing treatment (and thus pig disturbance levels) did not affect the number of plant species in each lagoon, but this diversity did decline over the course of both seasons. Again, decreasing species diversity over the course of the season is a natural pattern for these lagoons, but this factor was not affected by pig disturbance. Pig disturbance did reduce the amount of plant cover but not the number of species present or their species composition.

Like for aquatic macrophytes, for aquatic invertebrates, fencing did not affect species composition, but this did differ between lagoons, again indicating that the aquatic invertebrate composition of each lagoon was naturally different, regardless of the presence of pigs. Fencing treatment made no difference to aquatic invertebrate species diversity or abundance, although species diversity declined significantly over the course of each season as the dry season progressed. Invertebrate abundance did not decline over the season, presumably because some species actually become more abundant as habitat conditions deteriorate (e.g. due to loss of higher level predators as water quality declines). It should be noted here that abundance is relative, referring to the abundance of invertebrates in each sample. Given that pig rooting decreased the amount of plant cover in unfenced lagoons, this would obviously translate to a reduction of invertebrates within unfenced lagoons as a whole.

That water clarity could be so strongly affected by pig disturbance but aquatic macrophyte and invertebrate species diversity and community composition be not affected must reflect the adaptability of species living in these variable environments. The dry season in Lakefield imposes substantial stress upon biota, especially as the lagoon water levels decline very rapidly as does the ecological character of the lagoons. The species present are clearly used to rapid and substantial changes in their habitat.

2.4.4 Freshwater Fish

As occurred for both aquatic macrophytes and aquatic invertebrates, for freshwater fish, fencing did not affect fish community composition or the number of fish species, but these factors differed naturally between lagoons and both of these parameters were strongly affected by the natural seasonal water level decline. As for aquatic invertebrates, there was no apparent decline in fish abundance over the course of the dry season. It should be noted however, that fish may become easier to catch as water level declines. Actual fish abundance is not possible to accurately measure without substantial effort being deployed.

We sampled a total of 11 fish species in this study. The conditions for fish sampling were awkward. The wide, shallow, muddy margins made boat deployment and operation very difficult. This difficulty would have been heightened in the drier months. Manual netting was inoperable due to dense macrophyte stands and the danger from crocodiles. Quantitative fish sampling of the entire fish community in such freshwater environments is notoriously difficult. Each species have different susceptibility to different methods and the efficacy of each method alters greatly as the water level changes over the course of the season. Hence, most studies rely on a few key indicator species rather than abundance changes across the whole fish community. In this case, the trap method most suited sailfin glassfish and eel-tailed catfish, both of which were more abundant in fenced lagoons than unfenced lagoons, although these significant differences were not statistically significant. For hyrtl's tandan, the lack of significant difference can be attributed to 52% of specimens caught coming from just two samples (both in fenced lagoons), whereas 20 other samples had either none or just one individual present. Hyrtl's tandan prefer muddy to sandy substrates in still water lagoons and often shoal in groups (Pusey *et al.* 2004), reflecting the ‘many or none’ catch statistics in this study.

The data for sailfin glassfish more evenly balanced than that for hyrtl’s tandan, nonetheless, it was variable enough to mask statistical significance even though they were twice as abundant in fenced lagoons on average. Sailfin glassfish prefer microhabitats that provide plenty of cover such as aquatic macrophytes (Pusey *et al.* 2004) so any reductions in macrophyte cover would be expected to negatively impact upon their abundance.

More than 11 fish species can be expected to be present in freshwater lagoons in Lakefield National Park. Kennard (1995) sampled six lagoons on the Normanby floodplain within Lakefield National Park on two occasions each. He found a total of 21 fish species across the six lagoons, though no more than 17 species were found in any individual lagoon. A total of 26 fish species have been found across freshwater habitats in the vicinity of our study sites (Burrows and Perna 2006). However, our study lagoons were smaller than the lagoons sampled in previous fish studies of Lakefield National Park so would not be expected to house the same number of species as found in other studies. The use of additional sampling methods would also likely have found additional species, but the safe application of these was questionable.

2.4.5 Ecological and Management Implications

This study has both general and specific implications for fencing as a tool for managing the ecological impacts of feral pigs. Our ‘paired’ lagoons functioned in a hydrological similar way and this has allowed us to make inferences about the benefits of exclusion fencing to protect against foraging by feral pigs. However, our finding that each lagoon that we surveyed harbor distinct macrophyte, fish and invertebrate communities is also significant both in terms of unravelling the effects of pigs from natural variation in biological communities and also in the wider application of fencing across a landscape. In addition to this, whilst the study had a balanced replication of fenced and unfenced sites, we had no true control in the sense of an ecological reference point without feral pig disturbance since feral pigs are known to have been in the Lakefield region for 100 years or more. It follows that because the wetlands in this region have been disturbed by feral pigs for very many years, then all may be altered by, and even adapted to some extent, to this prolonged disturbance and any truly pig-sensitive species may have been eliminated from the test lagoons well before this study began (see also Kotanen 1995).

In the absence of pre-invasion data, this study nevertheless demonstrates that the foraging activities of feral pigs in these floodplain lagoons disrupt their physical, chemical and biological environments. Pig-mediated disturbance in the unfenced lagoons significantly affected their water clarity by dramatically increasing turbidity. The degree to which this may have altered primary productivity is unknown, however we have clearly linked pig foraging to the destruction of aquatic macrophytes, and the proliferation of bare ground and open (but turbid) water in these lagoons. In contrast to other studies (Kotanen 1995; Arrington *et al.* 1999), we have found no evidence that feral pig rooting has affected either the number or diversity of plant, invertebrate or fish species and nor have we found evidence for the invasion of exotic vegetation types as a consequence of pig disturbance. In north American grasslands and prairies, for example, foraging by pigs is believed to provide an important dispersal opportunity for exotic plants (Vitousek 1986; Cushman *et al.* 2004; Tierney and Cushman 2006).

Fencing against intrusions by exotic ungulates has been used in Hawaii for several decades (Loope and Scowcroft 1985; Stone *et al.* 1992; Katahira *et al.* 1993), but it is expensive and requires ongoing maintenance (Katahira *et al.* 1993; Reidy *et al.* 2008). We are unaware of pig exclusion fencing being used in a tropical floodplain ecosystem and the results presented here indicate that fencing clearly contributes to wetland habitat maintenance in these environments. Questions remain, however, given the finding that although the four 'pairs' of lagoons had similar water depth and seasonal rates of water level decline, they were still ecologically dissimilar to each other regardless of this and their close proximity to each other. This and the common result in our repeated measures analyses of most biological parameters being significantly affected by the temporal influence of season, rather than the fencing treatment, have implications for both the choice of lagoon for fencing and the practicality of this management tool. For example, how would we apply fencing on a wider scale in Lakefield given the ecological diversity of these eight lagoons if they are representative of the region? Moreover, it appears from our data that any effect of feral pigs on wetland biota may be dwarfed by seasonal climatic effects.

Feral pigs pose a serious ecological and economic threat in many parts of the world, including Australia (Tisdell 1982). We argue, however, that their true ecological effects might be best measured in a landscape-specific framework because their effects probably depend on the biology and disturbance history of the affected community (Denslow 1980; Hobbs and Huenneke 1992; Byers *et al.* 2002) and pigs are problematic in a very wide variety of wetlands across Australia.

At Lakefield and indeed across monsoonal northern Australia, there are, in most years, two predictable primary disturbances that annually affect the ecological communities of those floodplains. One is the complete inundation of the floodplain for several months followed by a desiccation of them. At Lakefield, juxtaposed on this natural and predictable seasonal disturbance regime is the secondary disturbance of feral pig intrusion which increases as the wetlands dry. Despite the statistical evidence for temporal effects strongly influencing many of the various (especially biological) parameters we have measured, there will be inter-annual differences in these climatic effects and they may concurrently increase or decrease the pig foraging disturbance (see also Euliss *et al.* 2004).

2.5 CONCLUSIONS

We have demonstrated that feral pigs do have significant impacts upon wetlands in the tropical environments we studied and that exclusion fencing can be successfully utilised to reduce these impacts. However, we have also demonstrated that there are significant natural disturbances also operating in these ecosystems that should be taken into account when assessing impacts to wetlands. The further application of fencing requires some planning as to where best it should be applied and ideally, this would be applied at a landscape scale, rather than just trying to protect individual wetlands. Applying fencing at this scale will require more detailed understanding of how feral pigs affect a wider variety of wetland types.

This study focused on ephemeral lagoons as these are an abundant water body type in northern Australia that is particularly impacted by feral pigs. It is recommended that future work include more permanent lagoons and wetlands. Permanent water bodies are arguably more ecologically important aquatic habitats and also have the advantage that the seasonal effects of lagoon drying are less extreme, thus making delineation of the effects of pigs more achievable. The impacts of pigs upon such habitats are likely to be quite different to that studied here as well. Given the effects on variability of seasonal drying and significant natural variation between lagoons, it may also be advisable to actually divide lagoons in half which an impermeable barrier, with one side being fenced from pigs and the other exposed to pig damage. This set-up has experimental benefits although from a management point-of-view, fencing entire lagoons may be more desirable.

Another aspect of pig disturbance that we have not seen considered is the effects of pig disturbance on water body permanence. The very shallow lagoons and wetlands with broad wetland margins that are common in Lakefield are subject to extreme rates of evaporation in the dry season. The pig diggings effectively increase the surface area of the water, and thus potentially increase the evaporation rate. Destruction by pigs of water lilies in these shallow water bodies may further enhance this evaporation effect. Drying of water bodies is clearly their greatest threat and any factor that increases the rate of water loss will have a significant ecological impact.

CHAPTER 3.

Experimental Research to Quantify the Environmental Impact of Feral Pigs within Tropical Freshwater Ecosystems:

Study B: The Relationship between Feral Pig Density and the Level of Damage

3.1. INTRODUCTION

Knowledge of the type and spatial and temporal scope of pig impacts and what factors determine the extent of these impacts needs to be further defined. Many studies have shown that pest abundance is one of the variables related to the extent of damage or impact by introduced pest animals. In theory, it is often assumed that there is a positive relationship between the number of pests and the damage they cause (Hone 2007), although the form of this relationship is generally unclear, particularly for feral pig studies. This study attempts to quantify relationships between pig abundance and the extent of impacts they cause within tropical freshwater ecosystems.

The extent of environmental impact by pigs is not just a simple function of factors such as pig abundance or food resource availability. A unit change in abundance will not necessarily give a unit change in impact. The assumption of a linear relationship between feral pig activity, density and damage is overly simplistic (Choquenot *et al.* 1996). To quantify the relationship between pest damage and pest abundance, and more specifically the response of pest damage to changes in pest abundance has been seen as a high priority (Hone 1994, Anon 2005).

The relationship between pig damage and pig density can be estimated from observational and experimental studies. Observational studies rely on data collected from different sites with varying levels of pig damage and pig densities. The disadvantage with this approach is that the level of pig damage and or density may vary even within limited range. Hone (1995) has found that pig damage generally consists of limited damage in many sites and large damage in few sites; this can make deriving a relationship difficult. Within this LNP study site, however, the level of pig damage and pig density is assumed to be uniform over the site.

Therefore an experimental approach was implemented in this study. Feral pig abundance surrounding large freshwater wetland systems was manipulated by aerial shooting. Changes in indices of impacts could then be related to the changes in indices of feral pig abundance. Methods of estimating pig abundance levels were first investigated. The impacts of these abundance levels on a range of impact indicators were then observed and a model of the association between these two factors was created.

The measure of pig population abundance was derived from three indices based on the presence of pig sign (or activity) on plots. The indices were the frequency of occurrence of pig sign, frequency of occurrence of pig diggings and the proportion of ground disturbed by pig diggings (see below). Aerial counts conducted during the aerial shooting process were also used as a measure of total pig abundance.

Feral pig impact levels were quantified by firstly measuring the amount of ground disturbance caused by pigs. Although animal signs do not necessarily correlate with population density or activity (Hone 1988b), rooted ground or the extent of pig diggings has been used as an index of the impact of feral pigs on various environments (Bratton 1975; Alexiou 1983; Ralph and Maxwell 1984; Hone 1988a). This soil disturbance caused by their digging or rooting when searching for food has been shown in many studies to have a range of ecological impacts (Lackie and Lancia 1983; Singer *et al.* 1984; Kotanen 1994; Spatz and Mueller-Dombois 1975; Bratton 1974, 1975; Hone 2002).

The ecological impact of pigs was also measured by monitoring a number of water quality and biodiversity indicators. The ecological impacts of feral pigs are often difficult to quantify and in some cases difficult to detect. Any changes in ecological processes may occur over significant time frames or may not be perceptible without detailed and complex monitoring. It is difficult and often even impossible to characterize the functioning of a complex ecological system by means of direct measurements. The size of the ecological system in this study and the complexity of the interactions involved necessitated the use of ecological indicators. Ecological indicators are easily measured surrogates for underlying properties or responses of a system that are difficult to measure accurately and reliably. This study used two types of ecological parameters to quantify the "health" of the lake systems to demonstrate the ecological impacts caused by feral pig activity.

3.1.1. Water quality parameters have been used extensively to measure the health of water sources. Feral pigs are thought to have a negative impact on water quality variables. The five water quality indicators selected in this study to describe feral pig impacts are described below.

Ammonia

Ammonium ions are a waste product of animal metabolism. Ammonia is toxic to fish and aquatic organisms, even in very low concentrations. Ammonia is reportedly the leading cause of fish stress, breaking down immunity system and leading to bacterial disease. The danger ammonia poses for fish depends on the water's temperature and pH, along with the dissolved oxygen and carbon dioxide levels, the higher the pH and the warmer the temperature, the more toxic the ammonia. Also, ammonia is much more toxic to fish and aquatic life when water contains very little dissolved oxygen and carbon dioxide. Feral pigs urinating or defecating near or in the water may be a significant source of ammonia. Decaying uneaten food, plant material and leaves also contribute to ammonia accumulations.

pH

The normal pH range for healthy fish populations is 5 to 9. Slight changes in pH in a 24-hour period are normal; however rapid changes in pH will result in stress to fish. Heavily vegetative systems will have more severe pH swings due to the plants giving off oxygen during the day and taking away oxygen at night and giving off carbon dioxide. pH also affects the toxicity of ammonia; the higher the pH, the more toxic any measurable amount of ammonia will be to fish. When waters with low pH values come into contact with certain elements, they often make them more toxic than normal. For example, fish that usually withstand pH values as low as 4.8 will die at pH 5.5 if the water contains 0.9 mg/L of iron. Feral pig foraging activity is thought to influence pH by the digging action which mixes soil elements with the water, for example digging in acid sulphate soils. Rotting vegetation due to pig feeding and associated destruction of aquatic vegetation is another source of changing water pH.

Algae

Algae - phytoplankton biomass is frequently used in monitoring of freshwater as they occupy a fundamental role in food chains and ecosystem functions. Algae biomass is difficult to measure in the field, however chlorophyll-a concentrations are an indicator of phytoplankton abundance and biomass. They can be an effective measure of trophic status, are potential indicators of maximum photosynthetic rate and are a commonly used measure of water quality. Elevated concentrations of chlorophyll-a can reflect an increase in nutrient loads and increasing trends can indicate eutrophication of aquatic ecosystems. High levels often indicate poor water quality and low levels often suggest good conditions. It is natural for chlorophyll-a levels to fluctuate over time. Chlorophyll-a concentrations are often higher after rainfall, particularly if the rain has flushed nutrients into the water. Higher chlorophyll levels are also common during the summer months when water temperatures and light levels are high because these conditions lead to greater phytoplankton numbers. Feral pig activity may influence algae abundance by helping to alter water quality parameters e.g. high nitrogen levels can cause algae blooms.

Total Dissolved Solids

Total dissolved solids cause toxicity through increases in salinity, changes in the ionic composition of the water and toxicity of individual ions. While an appropriate concentration of salts is vital for aquatic plants and animals, salinity that is beyond the normal range for any species of organism will cause stress or even death to that organism. Salinity also affects the availability of nutrients to plant roots. Increases in salinity have been shown to cause shifts in biotic communities, limit biodiversity, exclude less-tolerant species and cause acute or chronic effects at specific life stages. Changes in the ionic composition of water can exclude some species while promoting population growth of others. Feral pigs are thought to release salts during their foraging or digging activity within soils.

Turbidity

Turbidity is caused by the light-blocking properties of suspended particles in the water. High concentrations of particulate matter can modify light penetration and smother benthic habitats. As particles of silt, clay, and other organic materials settle to the bottom, they can suffocate larvae and fill in spaces between rocks which are some aquatic organisms' habitats. Fine particulate material also can clog or damage sensitive gill structures, decrease resistance to disease, prevent proper egg and larval development and potentially interfere with particle feeding activities. If light penetration is reduced significantly, macrophytes growth may be decreased which would in turn impact those organisms which depend on them for food and cover. Reduced photosynthesis can also result in a lower daytime release of oxygen into the water. Very high levels of turbidity for a short period of time may not be significant and may even be less of a problem than a lower level that persists longer. Feral pigs are thought to cause turbidity by their digging action on the banks or in the water when feeding on aquatic vegetation or searching for aquatic fauna.

3.1.2. Biodiversity Indicators are a good indicator of water health and have been used extensively to measure the health of water sources. Feral pigs are thought to have a negative impact on the biodiversity variables. The two biodiversity indicators used in this study to describe feral pig impacts were chosen for their ease of monitoring and are described below.

Macroinvertebrates
Population levels of macroinvertebrates provide an ideal measure of the response of the benthic community to environmental perturbations for many reasons. Macroinvertebrates are primarily sedentary and, thus, have limited escape mechanisms to avoid disturbances. Benthic invertebrates are relatively easy to monitor and tend to reflect the cumulative impacts of environmental perturbations, thereby providing good indications of the changes in an ecosystem over time. Benthic assemblages are often comprised of a variety of species (across multiple phyla) that represent a range of biotic responses to potential pollutant impacts.

Macrophytes
Macrophytes are plants which grow in or near water. They may be emergent, submergent, or floating. Macrophytes are beneficial to freshwater ecosystems as they provide cover for fish, substrate for aquatic invertebrates and food for some fish and other wildlife. They also produce oxygen, which assists with overall lake functioning. The absence of macrophytes may indicate water quality problems as a result of excessive turbidity, pollution, or salinity. However, an overabundance of macrophytes can result from high nutrient levels and may interfere with lake processing and detract from the aesthetic appeal of the system. Macrophytes are excellent indicators of watershed health because they respond to nutrients, light, toxic contaminants, metals, herbicides, turbidity, water level change and salt.

3. 2. METHODS

3.2.1. Study Sites within Lakefield National Park

Four large wetlands systems (termed lake systems) were selected within the study site (Figure 1.1 and Plate 3.1 to 3.4) and were a minimum of 30 km apart. Each lake system consisted of a series of small to medium freshwater lakes with associated lagoons and creek systems and was predominately backfilled from the flooding rivers during the wet season. The four lake systems selected had similar hydrology, floristic structure, topography and feral pig densities, but as they were located on different river systems, they were independent of each other. The pig populations within each lake system were also assumed to be independent due to the distance between them.

1. Caulders Lake / Little Kennedy - (15º 10.193’ S, 144 17.461 E)
Caulders Lake is situated approximately 7 km east of the New Laura Ranger Station. This shallow ephemeral lake, with a maximum depth of 3 m, is one of the larger lakes in LNP with an 8 km circumference. The lake was completely dry by October 2008 and almost dry in October 2009. Little Kennedy Lake is 5 km from Caulders Lake and connected to it by Stony Creek. Little Kennedy has a 3 km circumference and an average depth of 2 m. A small portion of the lake is a permanent lagoon. During the wet season these two lakes appear continuous.

Plate 3.1. Caulders Lake

Plate 3.2. North Kennedy Lake

Plate 3.3. Jacks Lakes

Plate 3.4. Broads / Strauss Lagoon

2. North Kennedy Lakes - (14º 59.614 S, 144º 11.854 E)
The North Kennedy Lakes are situated approximately 15 km southwest of the Lakefield Ranger Station. These lakes are a series of ephemeral lakes and swamps which are filled during the wet season from the North Kennedy River. The first lake selected is one of the larger lakes in the system and is approximately 4 km in circumference. This lake is very shallow being less then 2 m deep when full. The second lake is less then 1 km from the first (continuous with the first during the wet season), 3 km in circumference and up to 3 m deep in some places. The third selected lake is a permanent lagoon approximately 5 km from the second lake and is 2 km long and up to 500 m wide. This lake appears to be over 4 m deep in places when full. All of the selected lakes are linked during the wet season through the North Kennedy River.

3. Jacks Lakes - (14º 53.527 S, 144º 25.89 E)
Jacks Lakes comprise an extensive system of three open freshwater lakes (termed the lower, middle and top lakes) and paperbark swamps approximately 24 km east of Lakefield Ranger Station. The Jacks Lakes Aggregation is listed and described in the Directory of Important Wetlands in Australia (Environment Australia, 2001). The top lake is approximately 7 km in circumference and over 4 m deep in the middle when full. The middle lake is smaller with a circumference of 4 km and shallower. Both these lakes also support extensive *Melaleuca* swamp systems. The bottom lake was not utilised in this study.

4. Broads / Strauss lagoon system - (15º 15.972 S, 144º 31.765 E.).
This lake system is located in the southern end of LNP and is approximately 12 km northeast of the Old Laura Station site. The system consists of three lagoon systems. The largest, Broads Lagoon is a permanent lagoon with a 4 km circumference. The second lagoon, Strauss Lagoon, is also permanent and has a 2 km circumference. The third lagoon is the smallest at 1.5 km circumference and is ephemeral and was dry by October in 2008 and September 2009. All three lagoons systems are linked to the Normandy River system during the wet season.

These lake systems were monitored over a two year period, 2008 and 2009, by a series of nine surveys: May, June, July and September in 2008 and April, May, June, August and September in 2009. Surveys were commenced as soon after the wet season as access to the lake systems was possible (approx April/May) and concluded each year when the lake systems water levels were at very low levels or dry (September).

3.2.2. Pig population manipulation

Attempts were made to maintain the pig abundance at each lake system at a different level, ranging from a low population (for the site) to the normal pig abundance levels. Activity transects (see below) were used to derive the population abundance indices during each survey. This information was then used to determine what level of pig control was needed at respective lake systems to maintain populations at the desired level. Arial shooting was used to artificially manipulate pig abundance levels. The level of population control required at each lake system was adapted as required to maintain the differences in pig abundance levels between the treatments during the study.

The different levels of pig abundance and the aerial shooting regimes undertaken to attempt to maintain these different levels were as follows:

Caulders Lake -	0% to 25% of the normal population (shoot all pigs seen)
North Kennedy Lakes -	25% to 50% of the normal population (shoot 1 in 2 seen)
Jacks Lakes -	50% to 75% of the normal population (shoot 1 in 4 seen)
Broads Lagoon	75% to 100% of the normal population (public hunting)

Each lake system was extensively aerially searched. The number of pig sightings per hour of hunting time for each lake system was recorded and used as a pig abundance index. This level of aerial searching was assumed to be sufficient to estimate a total count of the total pig population inhabiting each lake system. Shooting was conducted up to 1 km surrounding the lakes and continued until no more pigs could be found. Generally, shooting was conducted within 3 hours of sunrise or sunset and in most cases each lake system was aerially searched on two consecutive days within each shooting program. A Pest Management Form (2007/2) was submitted and approved by the Environmental Protection Agency prior to the commencement of aerial shooting.

An additional control program (ground baiting) was implemented at Caulders Lake in September 08. This was in response to the abnormally high abundance of pigs during the July 08 survey as shown by the activity indices and by observational counts. Activity indices significantly decreased in the subsequent September 08 survey (see appendices for the full report).

3.2.3. Abundance Indices (Activity Transects)

The pig population abundance for each lake system for each of the nine survey periods (approximately at 2 month intervals) during the two year study period was quantified by using a series of abundance indices. Three population indices were derived from recording recent (up to three day old) pig activity or signs, such as footprints, diggings / wallowing, foraging activity and dung, on a series of 10 m long x 1 m wide plots established on the water's edge. A series of five consecutive plots spaced 20 m to 50 m apart were termed an individual activity transect. Individual transects were spaced approximately 100 to 200 meters apart. Three consecutive activity transects were grouped together as one of three sites; each site was established a minimum of 1 to 10 km from the others. Thus there were three sites, each with three transects, each with five plots (45 plots in total) for each lake system.

During flooding events all of the lake sites were continuous, but by the time access was possible for the initial survey each year, the sites were generally independent bodies of water. However, in the Caulders Lake system, sites 1 and 2 were not separate bodies of water but were on opposite sides of the large lake, 5 km apart. Similarly in Jacks lake system, sites 1 and 2 were also on opposite sides of the main lake, separated by 3 km. For the purpose of this study each lake site was termed an independent body of water with activity at one site having no influence on other sites.

During each survey, individual transects were located as close as possible to the waters edge at their original locations. The soft moist soil on the water's edge provided an ideal substrate to observe pig signs. Transects were initially cleared of pig sign by using the wheel tracks of all terrain vehicles (ATV) to mask old pig

Plate 3.5. Examples of activity plots (10m x 1m). Wheel tracks of two ATV's were used to mask old pig signs. Plots were inspected 3 days later for fresh pig activity signs. Bottom photograph shows pig activity (diggings wallowing and tracks) on a plot.

activity signs. Each transect was then inspected three days later and recordings were made of three abundance indices:

1. Visit Frequency (VF) - frequency of occurrence of sign.
The proportion of available plots (5) within each transect which recorded the presence of any pig signs - footprints, diggings or wallowing, tusking on trees, dung etc.

2. Digging Frequency (DF) - frequency of occurrence of diggings.
The proportion of available plots (5) within each transect that recorded the presence of ground disturbance or diggings caused by pigs foraging in the soil surface or by wallowing in the soil.

3. Digging Area (DA) - proportion of area disturbed by pig diggings.
A measuring tape was placed on the centre line of each plot (10 m in length). A line intercept method was used where the start and end of diggings were measured directly under the tape. The length of diggings was recorded and the proportion of the centre line of each plot that was disturbed by pig diggings was calculated and averaged for each transect. This index was also used as an indicator of impact as digging has often been associated with ecological impact

3.2.4. Ecological Impact Indicators

Two types of ecological indicators were sampled – a range of water quality parameters and aquatic plant and invertebrate biodiversity.

(a) Water parameters

Water sampling sites were located at the first and third activity transects within each of the three sites in each lake system. Two 50 ml water samples were collected at each of these six sites. Samples were collected 5 m from the water edge using an extension pole (plate 4.6). Samples were placed in eskies when collected to keep cool and remain excluded from light. They were then kept refrigerated in vehicles until they could be analysed each afternoon. Five indicators of water quality were measured at each lake system during every survey. Turbidity, ammonia and chlorophyll levels were measured using hand held fluorometers. Total dissolved solids and pH levels were measured using hand held instruments.

(1) Chlorophyll
We measured the *In vivo* fluorescence of the samples using a hand-held fluorometer (Aquafluor 8000, Turner Designs). The fluorometer measures the fluorescence of the light re-emitted from the chlorophyll within the algal cells at >665 nm of an excitation beam of 430 nm (Falkowski and Kiefer 1985). We standardised the fluorescence measurement prior to each survey using distilled water and a solid-state standard (part number 8000-952; Turner Designs) which we arbitrarily set to 10. We read each sample twice and averaged the readings for the two samples.

(2) Ammonia
We measured the *In vivo* fluorescence of the sample using a hand-held fluorometer (Aquafluor 8000, Turner Designs). The fluorometer measures the fluorescence of the ammonia at >420 nm of an excitation beam of 375 nm. We standardised the fluorescence measurement prior to each survey using a ammonia standard (ACR analytic reagent, ammonia at 1000 mg/L) and a solid-state standard (part number

8000-952; Turner Designs) which we arbitrarily set to 100. We read each sample twice and averaged the readings for the two samples.

(3) Turbidity
We measured the *In vivo* turbidity of the samples using a hand-held fluorometer (Aquafluor 8000, Turner Designs). It measures the intensity of a 515-nm beam at 90° of scatter as a measure of turbidity (i.e. by the nephelometric principle). We standardised the turbidity with distilled water and a 100-NTU (Nephelometric Turbidity Unit) solution. We arbitrary set the top value at 500 NTU as the values exceeding 500 could not be measured accurately on the turbidity instrument.

(4) Total Dissolved Solids (TDS)
TDS was measured by using a handheld instrument (Monarch Pool Systems TDS 9781). The instrument was calibrated prior to each survey using the calibration solution supplied.

(5) pH
The pH was measured by using a handheld instrument (Monarch Pool Systems pH 9780). The instrument was calibrated prior to each survey using the calibration solution supplied.

(b). Biodiversity Indicators

Two measures of biodiversity were recorded at each of the six sites used for the water sampling within each lake system.

(1) Aquatic Vegetation Index (AVI)
High quality photos of the macrophyte population present from the waters edge to 10 metres out at each site were taken at each survey period. These were used to identify the number and abundance of each macrophyte species present and the percentage cover at each survey period. This information was then incorporated into the AVI scoring system (Anon 2009).

(2) Stream Invertebrate Grade Number Average Level (SIGNAL).
A small bait trap was set at each of the water sampling sites at each lake system. Traps were left for three nights. All fish and insect species were identified and the abundance of each species present was recorded. Each taxon had been given a sensitivity score from 1 to 10 based on its modelled sensitivity to pollution. A SIGNAL score was derived for each sample site by averaging the pollution sensitivity scores of the macro-invertebrate groups at each sample (Signal 2 scoring method).

3.2.5. Analysis

These indices were used to monitor changes in the water health over time and in relation to the differing pig population abundance indices. Two-way ANOVA was used to indicate differences between the lake systems and between the survey periods for all ecological indicators and abundance indicators. Statistical analysis was primarily undertaken by fitting response lines and reporting these responses graphically for each measured ecological indictor. A repeated measure ANOVA best fit model was also fitted to the ecological response data to indicated trends and interactions over time. Correlation analysis was used to indicate the associations of abundance indicators with ecological indicators.

3.3. RESULTS

3.3.1. Aerial Shooting

During this study, seven aerial shooting programs were conducted. A total of 1051 pigs were observed over 25 hours of searching (42 pigs per hour) (Table 3.1). A total of 878 pigs were subsequently shot at an average shooting time of 35 pigs / hr of searching (not all pigs seen were shot).

Table 3.1. The number of pigs observed per hour of searching for each lake system during all of the aerial shooting programs.

	Caulders Lake	Jacks Lakes	North Kennedy Lakes
Jul-07	37.9	23.0	6.6
Dec-07	12.0	20.0	70.0
May-08	86.5	40.9	37.5
Aug-08	64.7	96.1	6.6
Apr-09	18.0	23.3	90.0
May-09	55.3	-	-
Aug-09	34.2	9.3	66.0
Average	**44.1**	**35.4**	**46.1**

The population abundance at each lake system was unable to be maintained at the theoretical levels established at the beginning of the study mainly due to excessive and unexpected pig abundance levels at Caulders Lake. Abundance estimation (index-manipulation-index method) was performed for the Caulders Lake system in June 08 by using the results from the activity transects and the counts from the aerial shooting program. A population of 2180 pigs was calculated to inhabit this lake system. A range of aerial count abundance levels throughout the survey periods was recorded to meet the aims of the study (Figure 3.2). No aerial counting was conducted at the Broads / Strauss lake system as no population manipulation (aerial shooting) was conducted at this site.

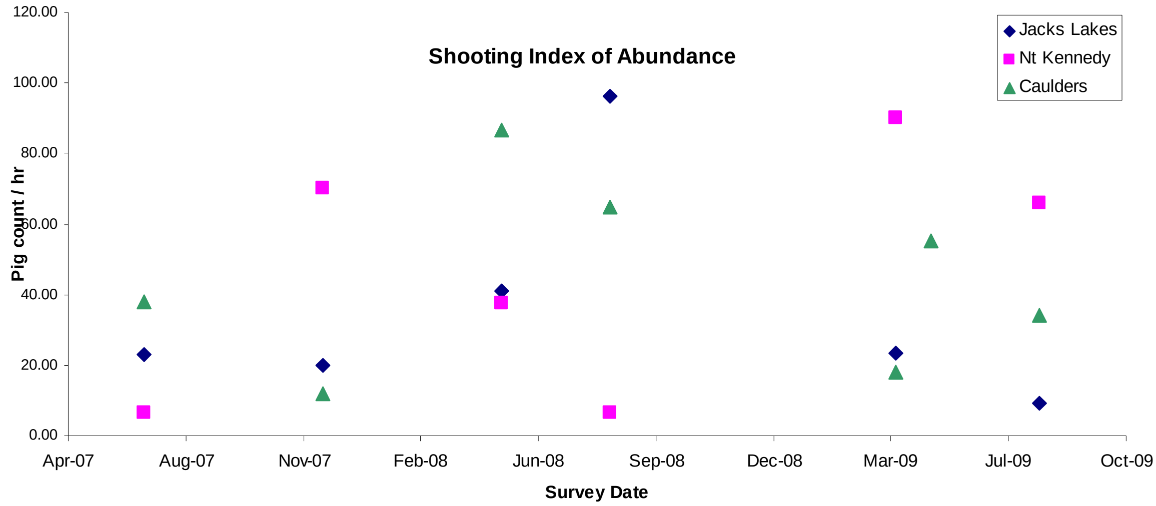

Figure 3.2. The range of pig abundance levels (count / hr) recorded from all aerial surveys, for all lake systems.

There was a significant positive curved relationship (Figure 3.3) between pig abundance (sightings / hr) and the visit frequency ($F_{1,5}$ = 12.29, $p < 0.05$; $R^2 = 0.91$). The visitation frequency index derived from the activity transects was therefore deemed to accurately represent the level of pig abundance as the aerial surveys were derived from a total pig population count of the lake system (albeit from only a one day count)

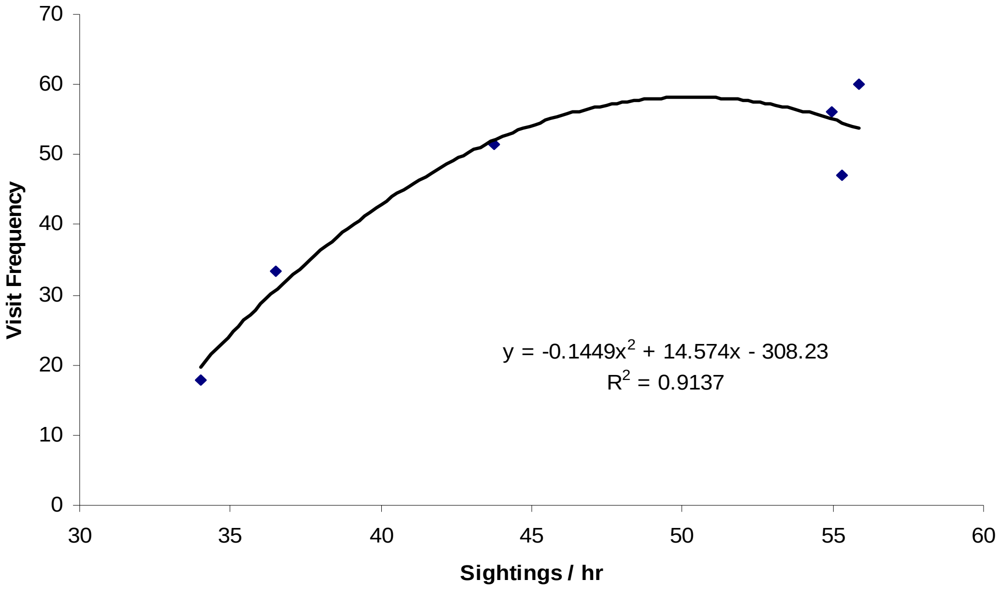

Figure 3.3. The best fit curve relationship between the visitation frequencies, recorded from the activity transects, with the abundance counts (sightings / hour) recorded during the aerial shooting component.

3.3.2. Abundance indices

The three activity indices for each survey period were calculated from the means of each transect within each site for each lake system. Overall, for all lake systems for all surveys, an average of 46.1% of plots were visited by pigs during each survey, 14.8% of plots had some diggings and 2.8% of the plot area was disturbed by pig diggings. The mean activity indices for all surveys are presented in Figure 3.4 and the mean of the three activity indices combined for all surveys for the four lake systems are presented in Figure 3.5. A significantly low level of pig activity is shown in September 08 for all three abundance indices. This corresponds with the end of the dry season.

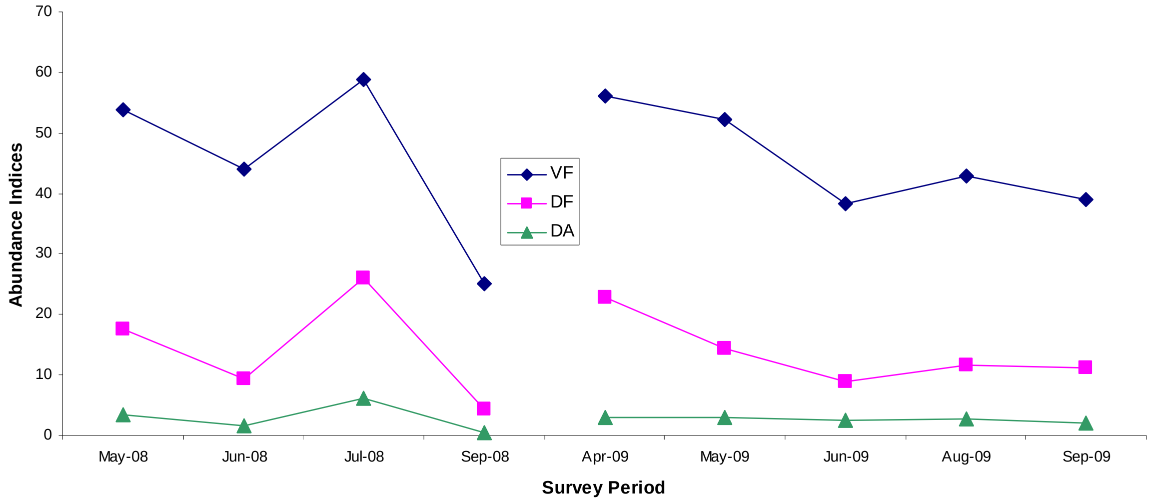

Figure 3.4. The mean of the three abundance indices for all lake systems combined, for each survey period.

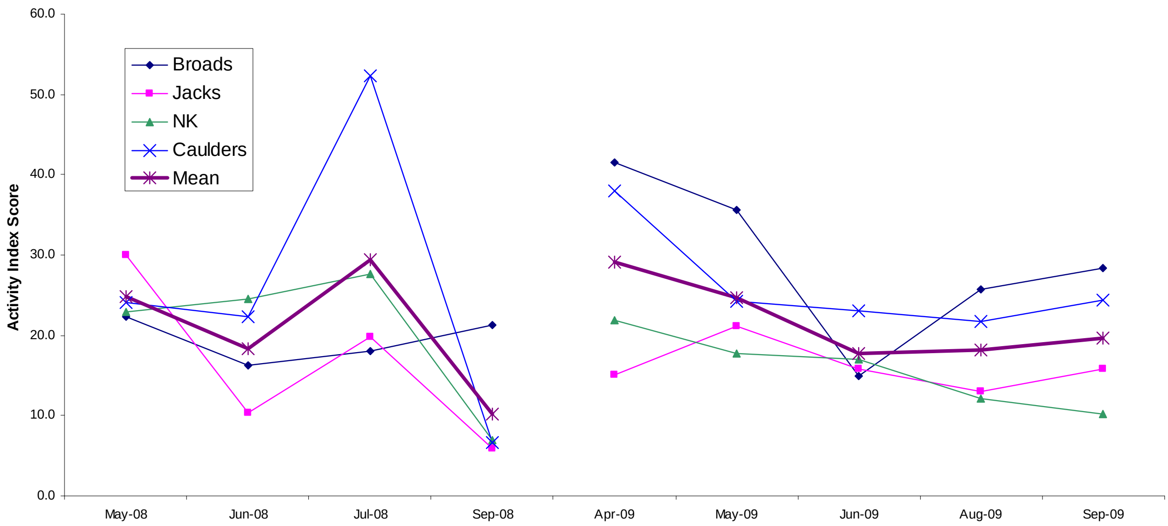

Figure 3.5. The abundance indices for each lake system for each survey period. The overall mean for all abundance indices for all lake systems is also presented.

(a) Visitation Frequency

The visitation frequency (VF) for all surveys was significantly different between the lake systems ($F_{3,24}$ = 5.28, $p < 0.05$); with Caulders Lake (53.2) and Broads lagoons (53.2) having significantly more visitations then Jacks Lakes (37.8) and North Kennedy Lakes (32.7). There was a significant difference in visitations for all lake systems over the survey periods ($F_{8,315}$ = 4.44, $p < 0.01$) and a significant lake by time interaction ($F_{24,315}$ = 2.09, $p < 0.01$).

The main difference in visitation frequency was between the lake systems in April 09 with Broads Lagoons (84.4) and Caulders Lake (77.8) having significantly ($F_{3,24}$ = 8.44, $p < 0.01$) higher visitation rates then Jacks Lakes (33.3) and North Kennedy Lakes (28.9). These same differences were almost significant (p = .067) in the next (May 09) survey period.

(a) Digging Frequency.

There were no significant differences between the lake systems for the digging frequency index ($p > 0.05$). However, there were significant differences for all lake systems over the survey periods ($F_{8,315} = 3.63$, $p < 0.01$) and a significant lake by survey interaction ($F_{24,315} = 1.78$, $p < 0.05$). There were no significant differences in the digging frequency between the lake systems within any survey period.

(b) Digging Area.

There were no significant differences between the lake systems for the digging area index ($p > 0.05$). However, there were significant differences for the lake systems over the survey periods ($F_{8,315} = 2.64$, $p < 0.05$) and a significant lake by survey interaction ($F_{24,315} = 2.27$, $P < 0.05$). There were no significant differences in the digging area between the lake systems within any survey period.

3.3.3. Ecological Impact indicators

(a) Diggings

The overall mean percentage of the area of all plots dug up by pigs was 2.8%. This equates to 50.4 m^2 of the 1800 m^2 surface area available for each survey that was disturbed over the three day monitoring period. The mean Digging Area index for all lake systems for each survey period is present in Figure 3.6. There was a significant ($F_{8,315} = 2.64$, $p < 0.05$) peak in digging activity in the June 08 survey at the Caulders Lake system (16.8%). Only Jacks Lakes in the June and September 08 surveys recorded no diggings were present.

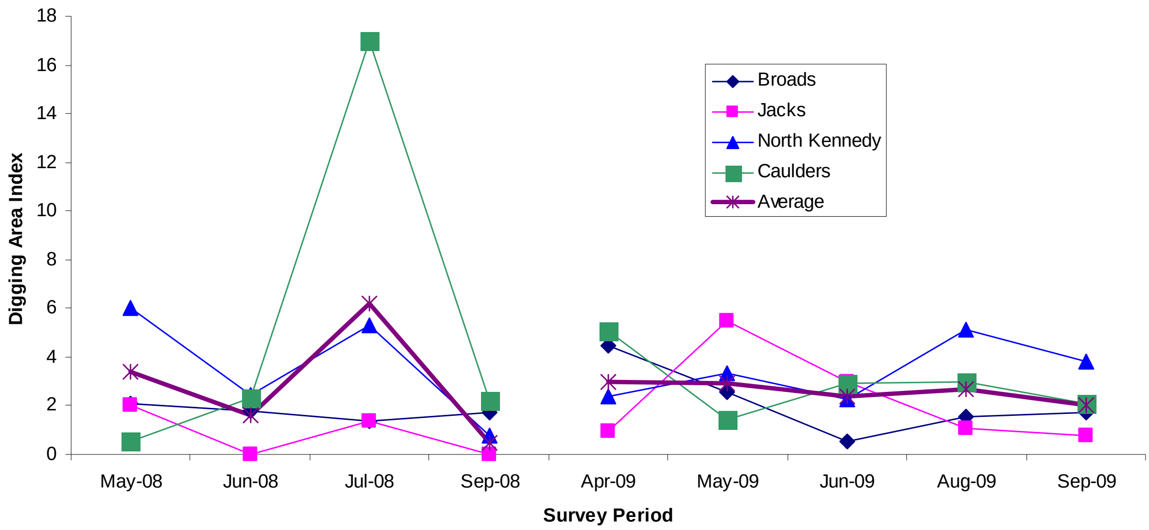

Figure 3.6. The mean percentage of the surface area of plots disturbed by pig diggings for all lake systems for each survey period.

There was no significant difference in digging area between the lake systems; Caulders Lake had a mean digging area of 3.9%, North Kennedy Lakes 3.2%, Jacks Lakes 2.0% and Broads Lagoon 1.9%. The digging area index between the lake systems varied during the survey periods from a high of 6.2 % to a low of 0.5 %. The hydrological cycle and associated ecological changes were the primary cause of this variation.

There was a significant ($F_{1,35}$ = 6.2, $p < 0.05$) association between the digging area and the frequency of visits for all lake systems over all survey periods. There was a significant curvature in this association, (Figure 3.7) so a significant exponential model was fitted ($DA = 0.24 + 0.14^{1.05\ VF}$, R^2 = 41.9%). There were also fitted significant (R^2 = 53.5) curved associations for individual lake systems except for Jacks lake where a straight line was fitted (Figure 3.8).

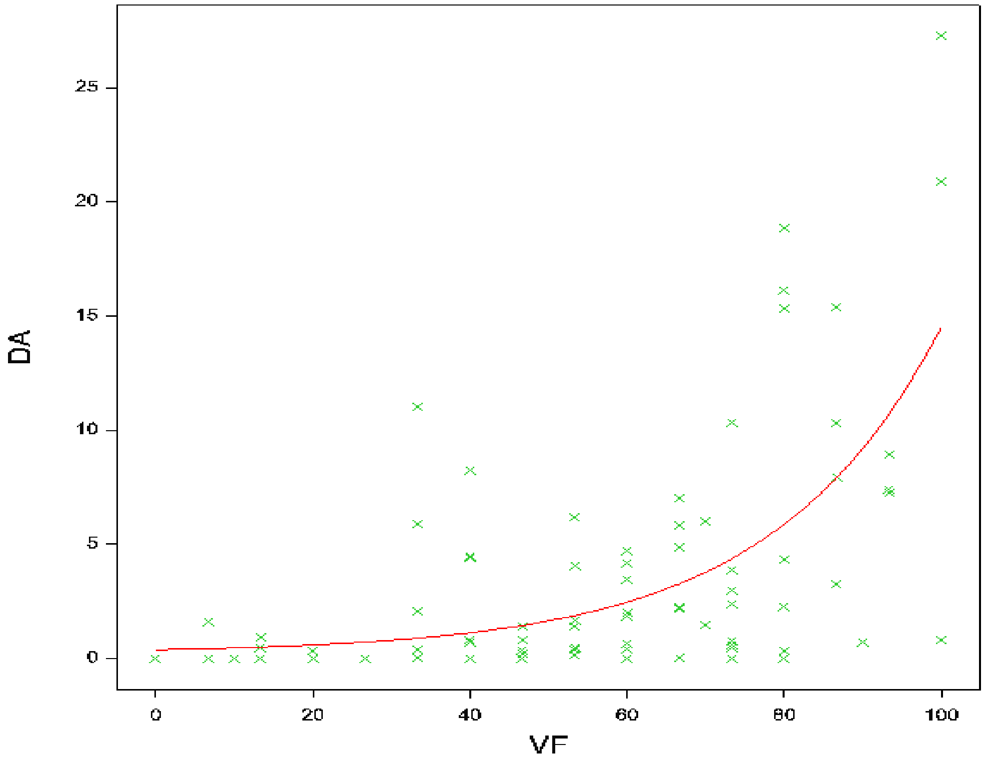

Figure 3.7. A fitted exponential model of the association of digging area with visit frequency for all lake systems over all survey periods.

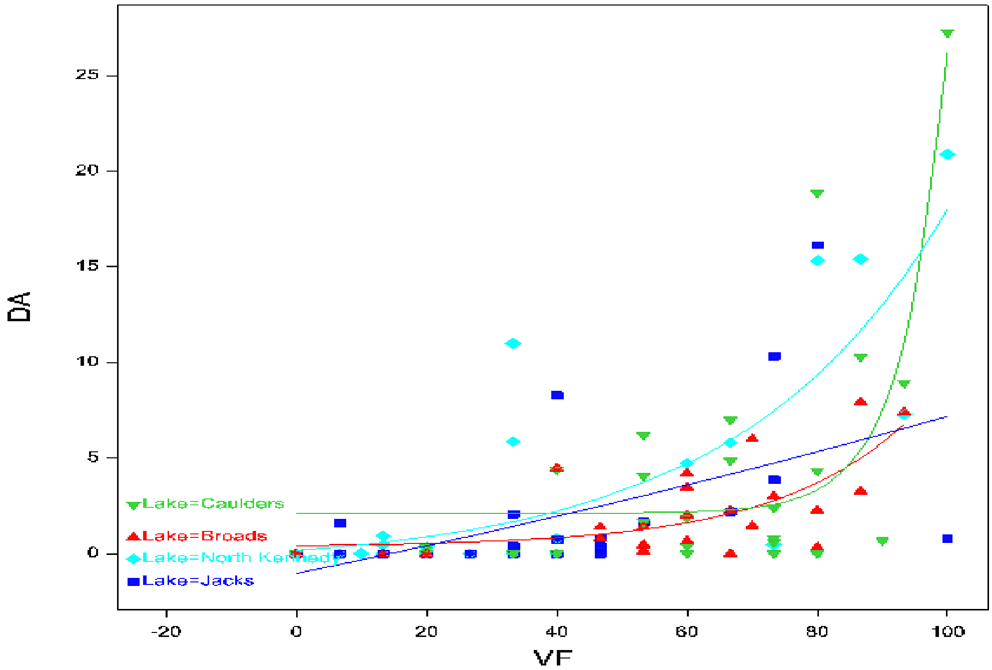

Figure 3.8. Fitted models of the association of digging area with visit frequency for the individual lake systems over all survey periods.

There were also significant ($F_{1,35} = 54.2$, $p < 0.01$, $R^2 = 0.37$)) associations between the digging area and the frequency of diggings and also a significant ($F_{1,35} = 24.7$, $p < 0.01$; $R^2 = 0.44$) association between the frequency of diggings and the frequency of visits (Figure 3.9 and Figure 3.10 respectively).

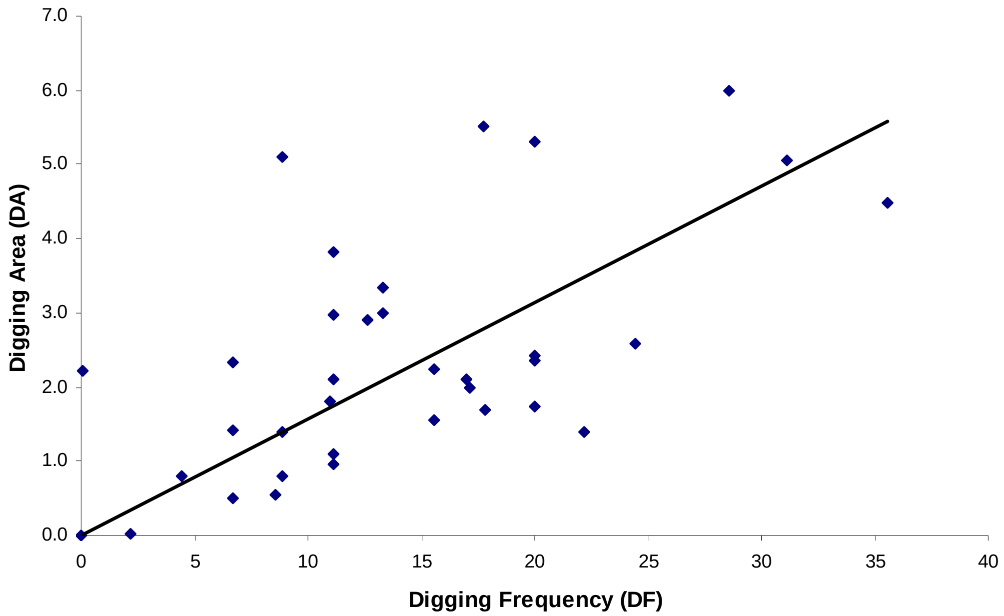

Figure 3.9. The association of digging area with digging frequency. A regression line is fitted to the data.

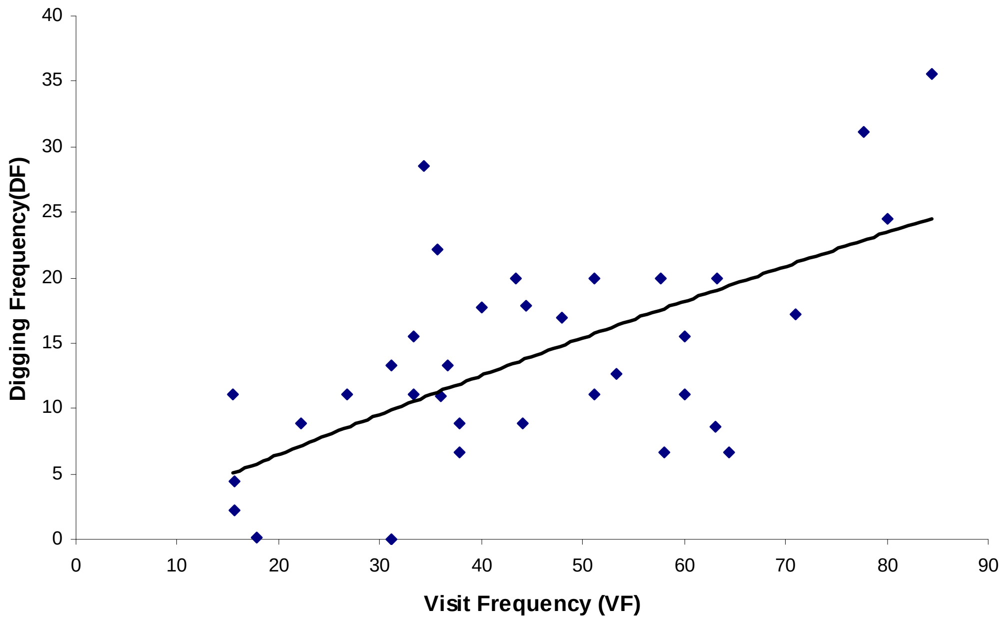

Figure 3.10. The association of digging frequency with visit frequency. A regression line is fitted to the data.

(b) Ecological indicators

For all lakes systems combined for all survey period's, the only significant association of ecological indicator with abundance indices was the significant ($p < 0.05$) aquatic vegetation index indicator association with the digging frequency. There were some significant associations with ecological indicators and abundance indices within some lake systems. For example a significant ($p < 0.05$) negative association between total dissolved solids and visitation frequency and a highly significant ($p < 0.01$) positive association between turbidity and digging area was found within the Caulders Lake system. Total dissolved solid levels were almost ($p < 0.1$) significantly associated with all abundance indices in the north Kennedy lakes system.

(1) Aquatic Vegetative Index (AVI)

The average AVI score for all lake systems over all survey periods was 41.2. There was no significant difference in AVI levels between the lake systems over all of the survey periods. There was a significant difference in the AVI between the survey periods ($F_{8,213} = 114.76$, $p < 0.01$) and a significant lake by survey interaction ($F_{24,213} = 7.55$, $p < 0.01$). There were significant differences in AVI levels between the lake systems within some of the survey periods. For May 09, Caulders Lake and Broads Lagoons had significantly higher AVI levels then Jacks Lakes and for June 09, Broads's lagoons, North Kennedy Lakes and Jacks Lakes had significantly higher AVI levels then Caulders Lake.

There was a significant positive correlation ($p < 0.05$) between the aquatic vegetation index and the digging frequency for all lake systems over all survey periods combined (Figure 3.11) and an almost significant correlation ($p = 0.08$) with the visitation frequency index. The mean AVI for all lakes showed a distinct decreasing trend over the dry season water draw down cycle. This trend was mirrored in the decreasing visitation frequency index (Figure 3.12). There is a positive correlation of decreasing pig abundance associated with decreasing aquatic vegetation availability.

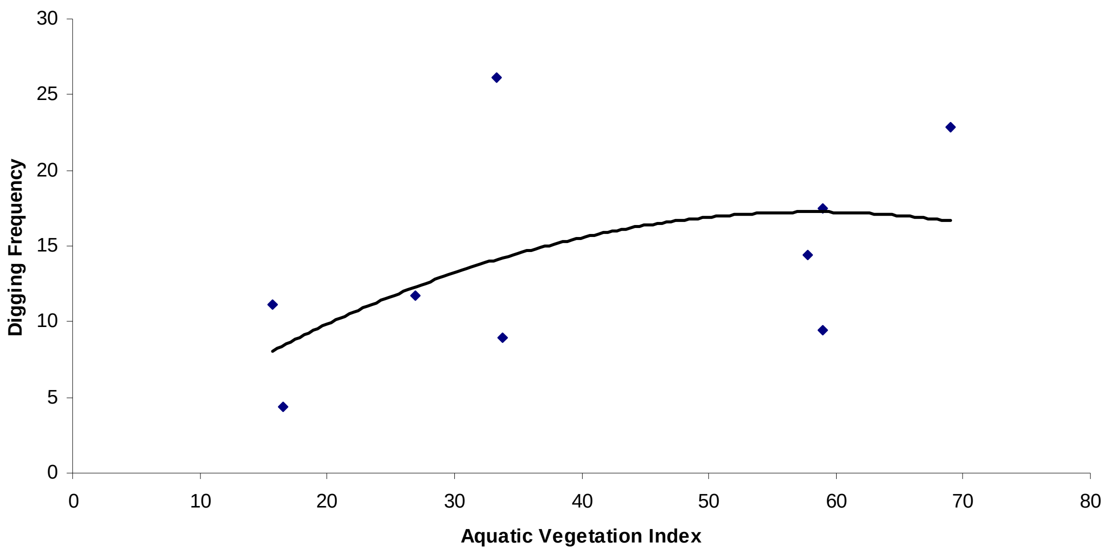

Figure 3.11. The fitted relationship between the digging frequency and the Aquatic Vegetation index.

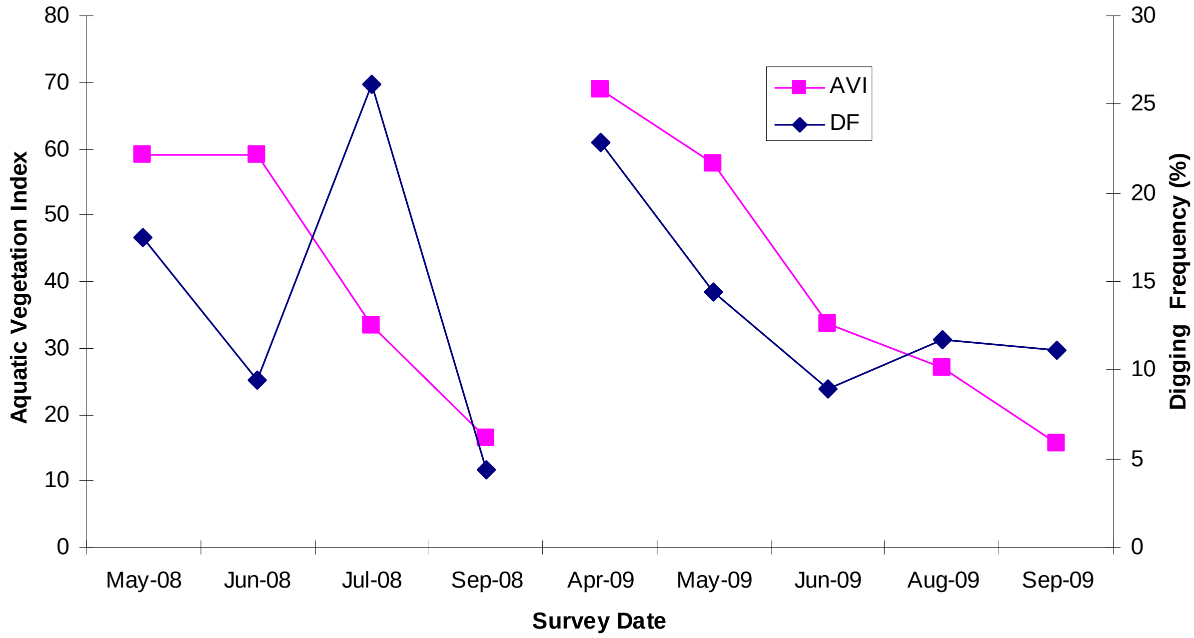

Figure 3.12. The mean Aquatic Vegetation Index and visitation frequency for all lake systems over all survey periods.

(2) Ammonia (AM)
The average AM fluorescence score for all lake systems over all survey periods was 275. There was a significant difference in the AM index between the lakes ($F_{3,12}$ = 5.75, $p < 0.05$), with Jacks lakes having significantly higher average AM levels (347.3) then the other lake systems. There was also a significant difference in AM levels between the survey periods ($F_{8,213}$ = 8.14, $p < 0.01$) and a significant lake / survey interaction ($F_{24, 213}$ = 2.44, $p < 0.05$).

There were significant differences in AM levels between the lake systems within some of the survey periods. In May 08, Caulders Lake and Jacks Lakes had significantly ($p < 0.05$) higher AM levels. For June 08, Caulders Lake, Jacks Lakes and Broads Lagoons had significantly higher AM levels then North Kennedy Lakes and in July 08, May 09 and June 09 Jacks Lakes had significantly higher AM levels then the other lake systems.

There was no overall significant association observed between ammonia levels and pig abundance. However, within the individual sites within the lake systems there were significant associations of ammonia with the abundance indices. For example North Kennedy site 3, Jacks Lakes site 3 and Broads Lagoon site 1 and 3 had significant associations of ammonia levels with visitation frequency. North Kennedy site 3 and Broads Lagoon site 1 also had significant associations with digging frequency and a significant association with digging area. For each year of the surveys, ammonia levels remained relatively constant during the draw down process, although ammonia levels in 2009 were generally at a higher level then in 2008. A graph of the association of AM levels with the main abundance index (visitation frequency) is presented to illustrate trends in the association (Figure 3. 13).

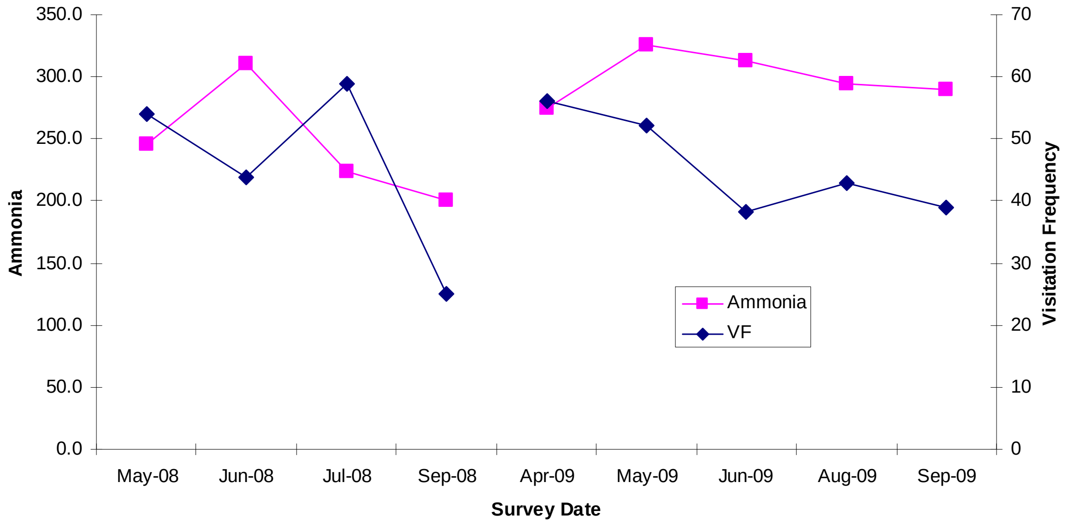

Figure 3.13. The mean ammonia index and visitation frequency for all lake systems over all survey periods.

(3) Chlorophyll (CL)

The average CL fluorescence score for all lake systems over all survey periods was 195. There was no significant difference in CH levels between the lake systems over all of the survey periods. There was a significant difference in the CH levels between the survey periods ($F_{24,213} = 9.65$, $p < 0.01$) For May 08, Jacks Lakes had significantly ($F_{8,12} = 20.17$, $p < 0.01$) higher CH levels then the other lake systems.

There was no overall significant association observed between chlorophyll levels and pig abundance. Only North Kennedy site 3 had a significant association with visitation frequency and digging frequency. For each year of the surveys, chlorophyll levels tended to increase throughout the draw down process and tended to be negatively associated with pig abundance. A graph of the association of CL levels with the main abundance index (visitation frequency) is presented to illustrate trends in the association (Figure 3.14).

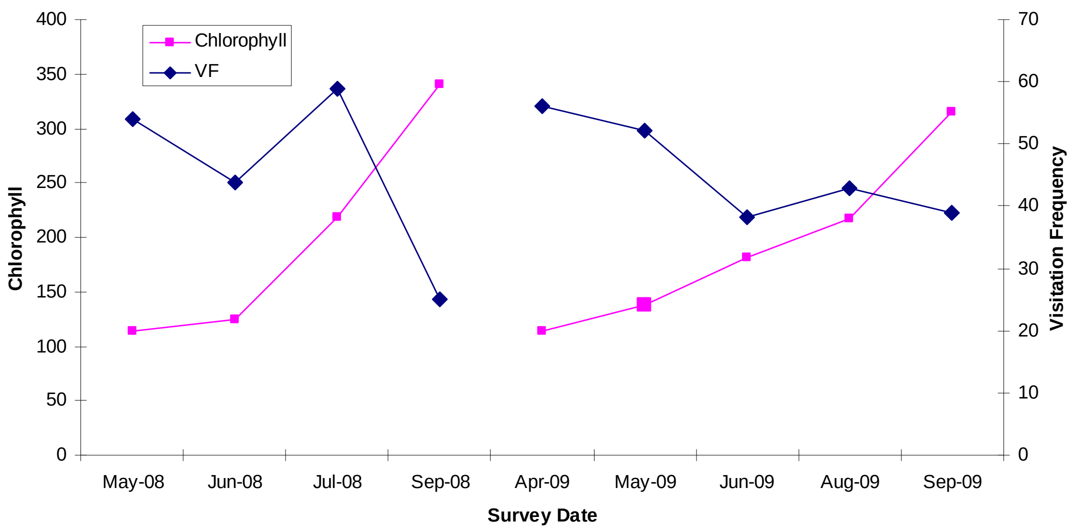

Figure 3.14. The mean chlorophyll index and visitation frequency for all lake systems over all survey periods.

(4) Turbidity (TU)
The average TU score for all lake systems over all survey periods was 90.8 NTU. There was no significant difference in TU levels between the lake systems over all of the survey periods. There was a significant difference in the TU levels between the survey periods ($F_{8,213}$ = 13.9, $p < 0.01$) and a significant lake by survey interaction ($F_{24,213}$ = 2.28, $p < 0.05$). There were no significant differences in TU levels between the lake systems within any of the survey periods.

There was no overall significant association observed between turbidity levels and pig abundance. However, within the individual sites, Broads Lagoon site 3 had a significant association with digging frequency and Caulders lake site 2 had a significant association with digging area. For each year of the surveys, turbidity levels tended to increase throughout the draw down process and were relatively negatively associated with pig abundance. A graph of the association of TU levels with the main abundance index (visitation frequency) is presented to illustrate trends in the association (Figure 3.15).

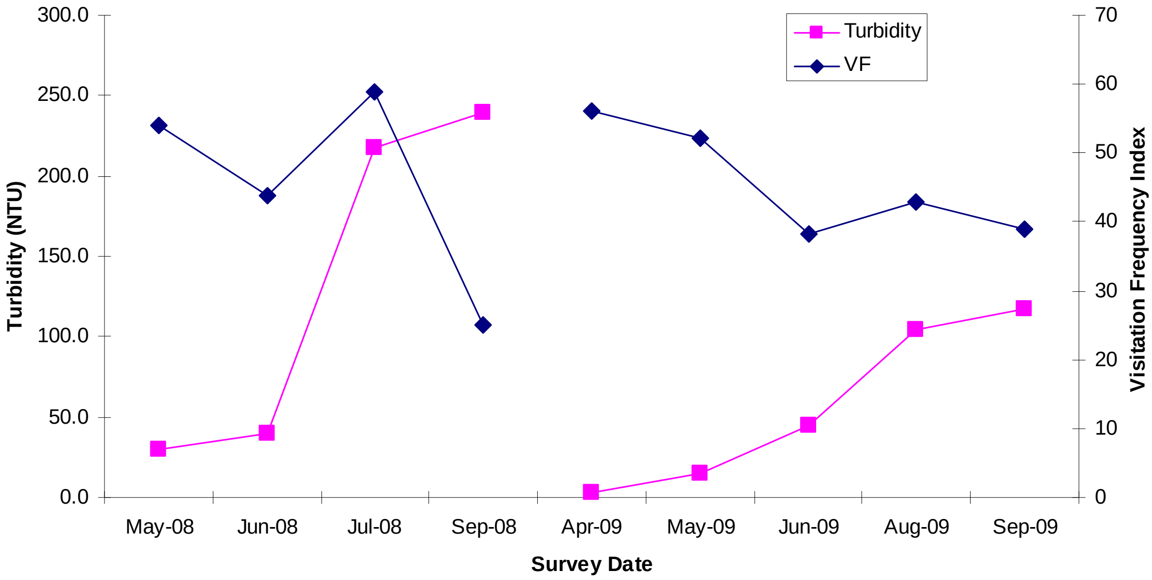

Figure 3.15. Mean turbidity and visitation frequency for all lakes for all survey periods.

(5) Total Dissolved Solids (TDS).
The average TDS score for all lake systems over all survey periods was 98.5. There was no significant difference in total dissolved solids levels between the lake systems over all of the survey periods. There was a significant difference in the TDS between the survey periods ($F_{8,213}$ = 15.15, $p < 0.01$) and a significant lake by survey interaction ($F_{24,213}$ = 5.43, $p < 0.01$). There were no significant differences in TDS levels between the lake systems within any of the survey periods. There was no significant association observed between TDS levels and pig abundance. However, within the individual sites, Caulders Lake site 1 had a significant association with visitation frequency. For each year of the surveys, TDS levels tended to increase throughout the draw down process and were relatively negatively associated with pig abundance. A graph of the association of TDS levels with the main abundance index (visitation frequency) is presented to illustrate trends in the association (Figure 3.16).

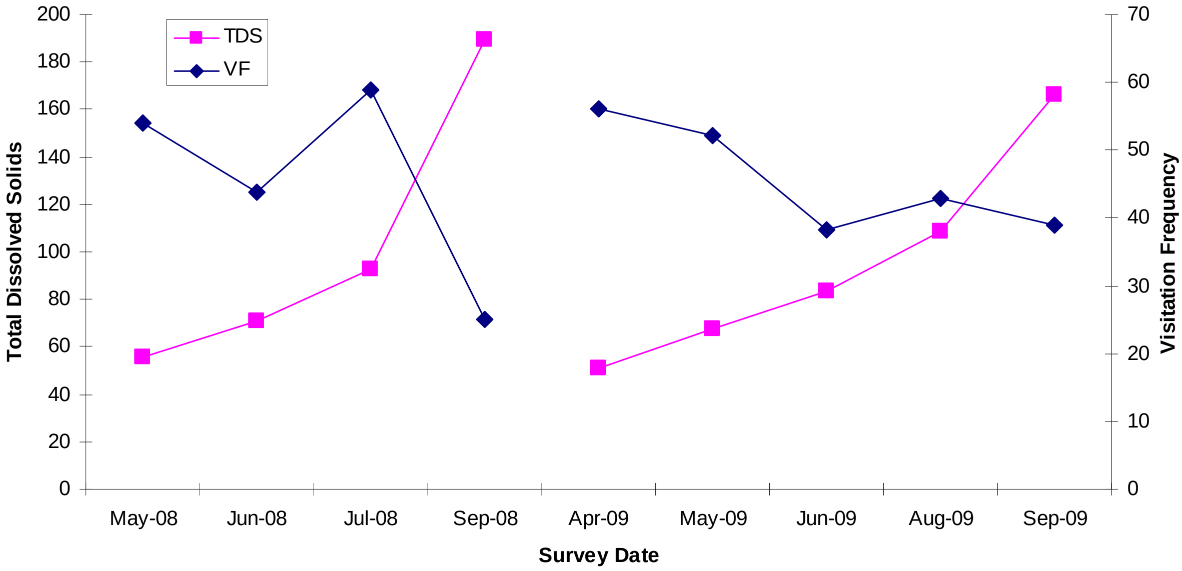

Figure 3.16. The mean total dissolved solids and visitation frequency for all lake systems over all survey periods.

(6) pH.
The average pH for all lake systems over all survey periods was 6.6. There was no significant difference in pH levels between the lake systems over all of the survey periods. There was a significant difference in the pH between the survey periods ($F_{8,213} = 11.75$, $p < 0.01$) and a significant lake by survey interaction ($F_{24,213} = 4.23$, $p < 0.01$). There were significant differences in pH levels between the lake systems within some of the survey periods. For April 09, Caulders Lake and May 09, North Kennedy Lakes had significantly higher pH levels then the other lake systems.

There was no overall significant association observed between pH levels and pig abundance. However, within the individual sites, Broads Lagoon site 1, North Kennedy site 2 and 3 had significant associations with visitation frequency and North Kennedy site 3 with digging frequency. For each year of the surveys, pH levels decreased throughout the draw down process and were positively associated with pig abundance. A graph of the association of pH levels with the main abundance index (visitation frequency) is presented to illustrate trends in the association (Figure 3.17).

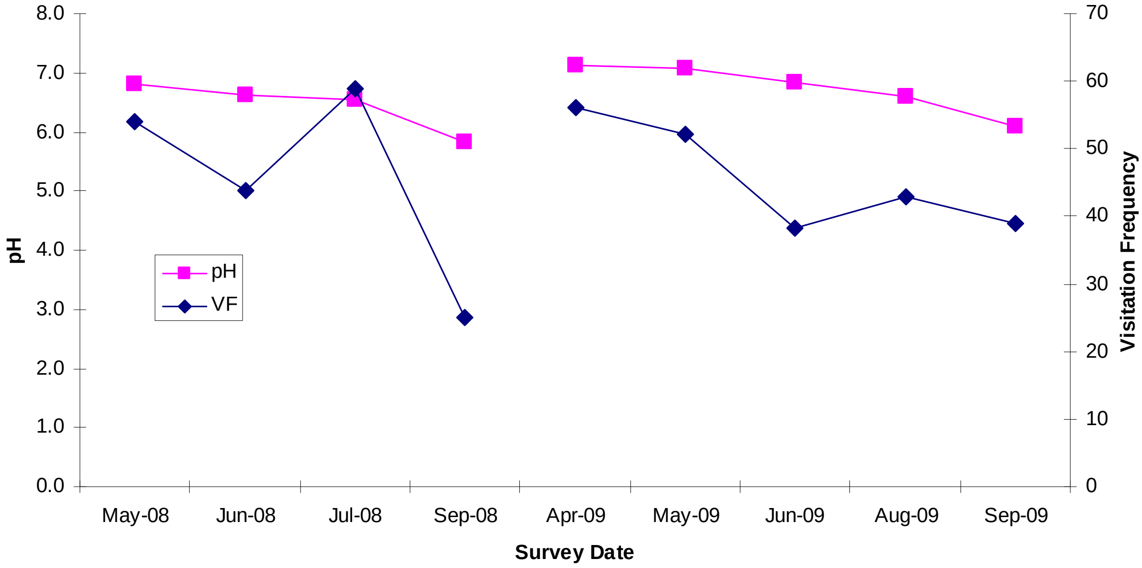

Figure 3.17. The mean pH and visitation frequency for all lake systems for all survey periods.

(7) SIGNAL

The average SIGNAL score for all lake systems over all survey periods was 1.86. There was no significant difference in SIGNAL score between the lake systems over all of the survey periods. There was a significant difference in the SIGNAL scores between the survey periods ($F_{8,213} = 4.78$, $p < 0.05$) and a significant lake by survey interaction ($F_{24,213} = 2.43$, $p < 0.05$). There was a significant difference in SIGNAL levels between the lake systems within the September 09 survey with Jacks Lakes and Broads Lagoons having significantly higher SIGNAL scores then North Kennedy Lakes and Caulders Lakes. There was no significant association observed between the SIGNAL Score and pig activity. For each year of the surveys, SIGNAL levels remained relatively level throughout the draw down process except for the September 09 survey where the SIGNAL score decreased to the lowest level of the project. A graph of the association of SIGNAL levels with the main abundance index (visitation frequency) is presented to illustrate trends in the association (Figure 3.18).

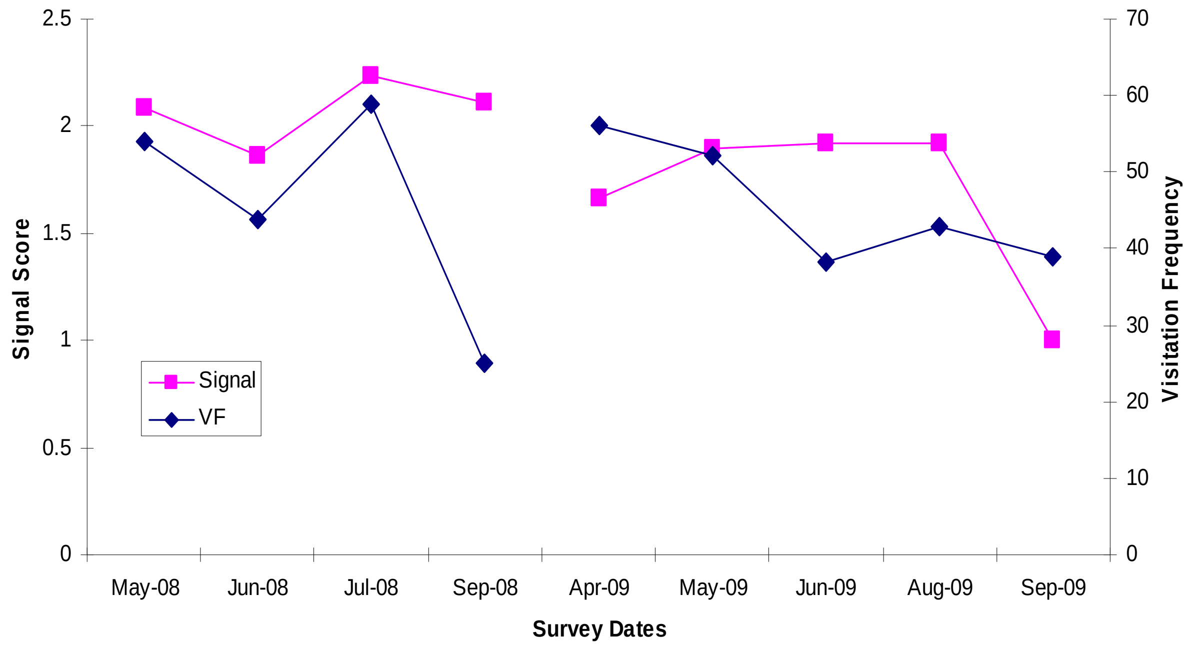

Figure 3.18. The mean SIGNAL score and digging frequency for all lake systems over all survey periods.

3.3.4. Associations between the ecological indicators

There were a number of significant correlations between the ecological indicators as presented in Table 3.2.

Table 3.2. Correlation analysis between the measured ecological indicators.
Note - * denotes a significant correlation ($p < 0.05$) and ** denotes a highly significant correlation ($p < 0.01$). Pos denote a positive correlation and Neg denote a negative correlation.

Ammonia (AM)							
Chlorophyll (CH)	-						
Turbidity (TU)	Neg**	Pos**					
Total Dissolved Solids (TDS)	Neg*	Pos**	Pos *				
pH	Pos*	Neg**	Neg**	Neg**			
Aquatic Veg Index (AVI)	-	Neg**	Neg**	Neg**	Pos**		
SIGNAL	-	-	-	-	-	-	
	AM	CH	TU	TDS	pH	AVI	Signal

The data for all of the ecological indicators for all survey periods within each lake system are presented in the following Figures 3.19 to 3.25. Note that for Caulders Lake, the September 08 survey results are strongly influenced by the combined shooting and poisoning implemented in August 08.

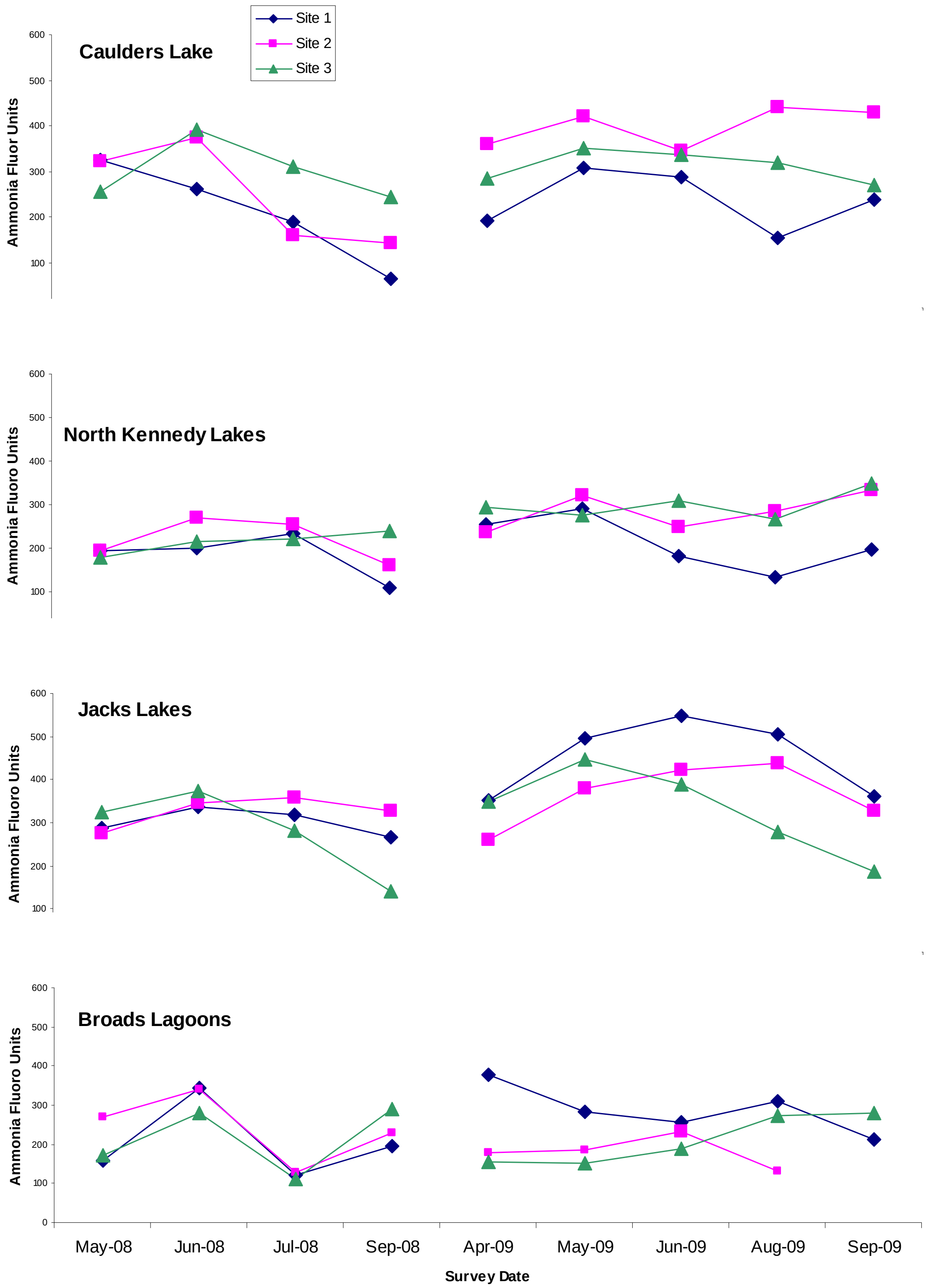

Figure 3.19. Ammonia levels for each site in each lake system for all survey periods.

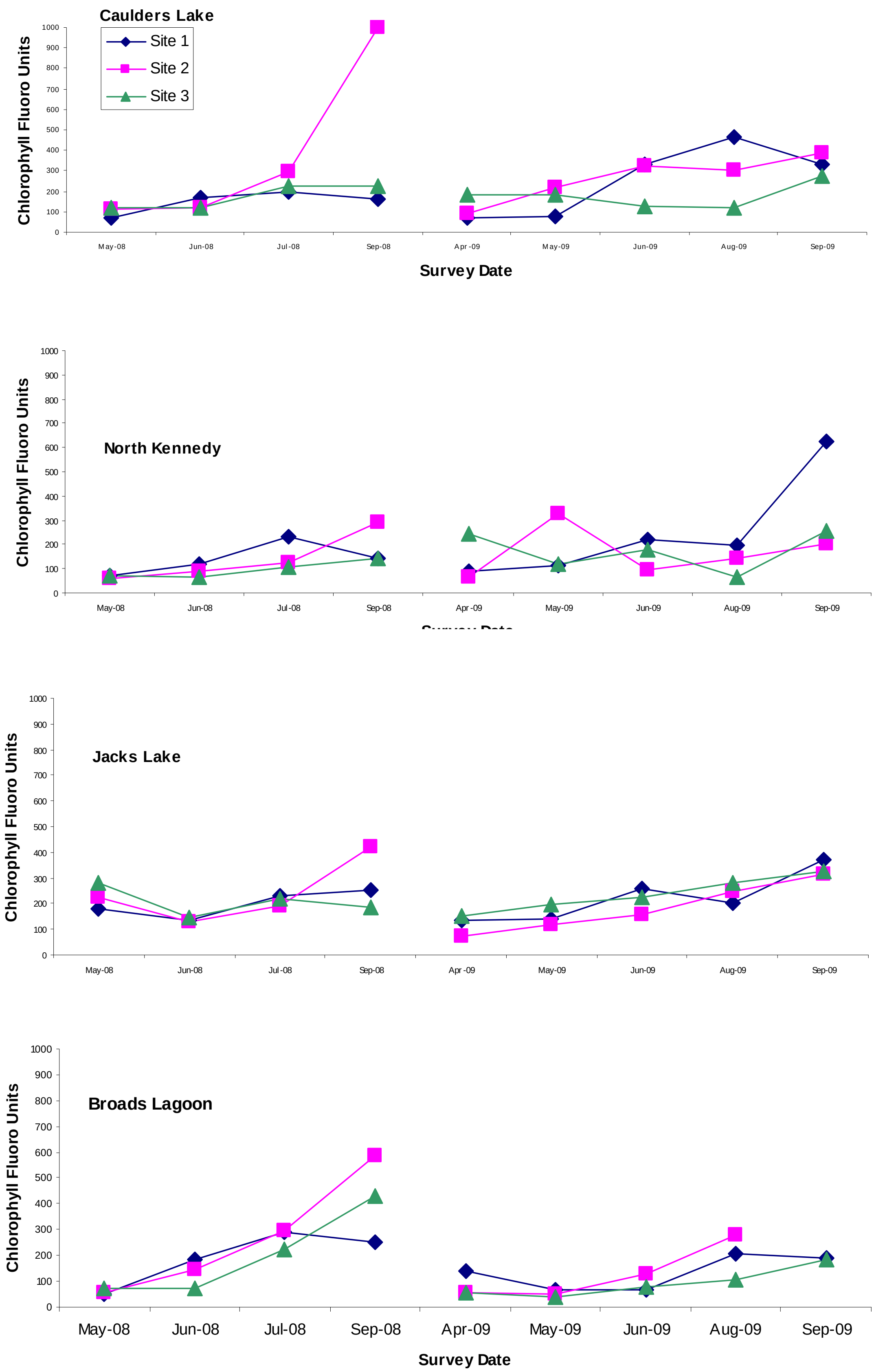

Figure 3.20. Chlorophyll levels for each site in each lake system for all surveys periods.

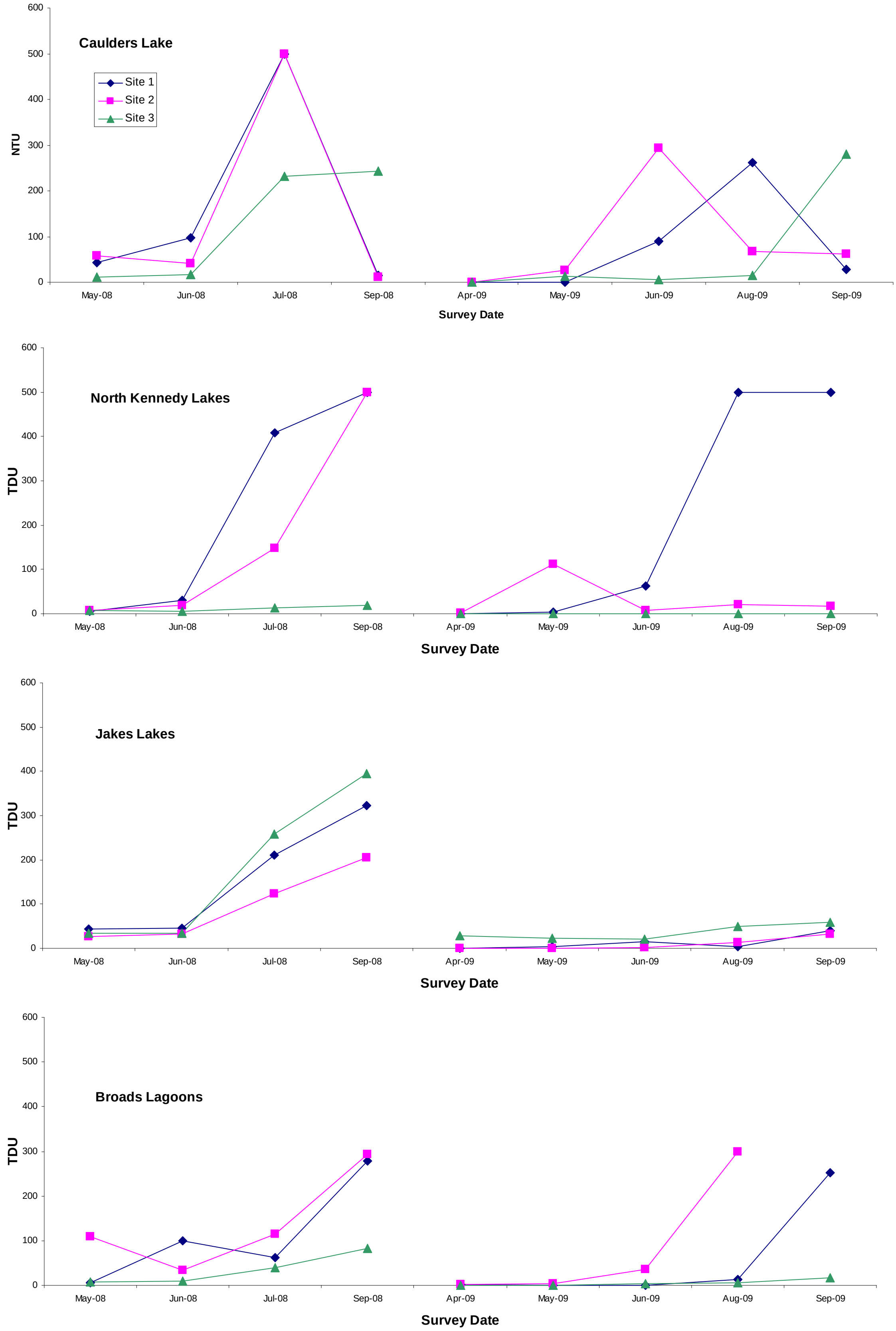

Figure 3.21. Turbidity levels for sites in each lake system for all surveys.

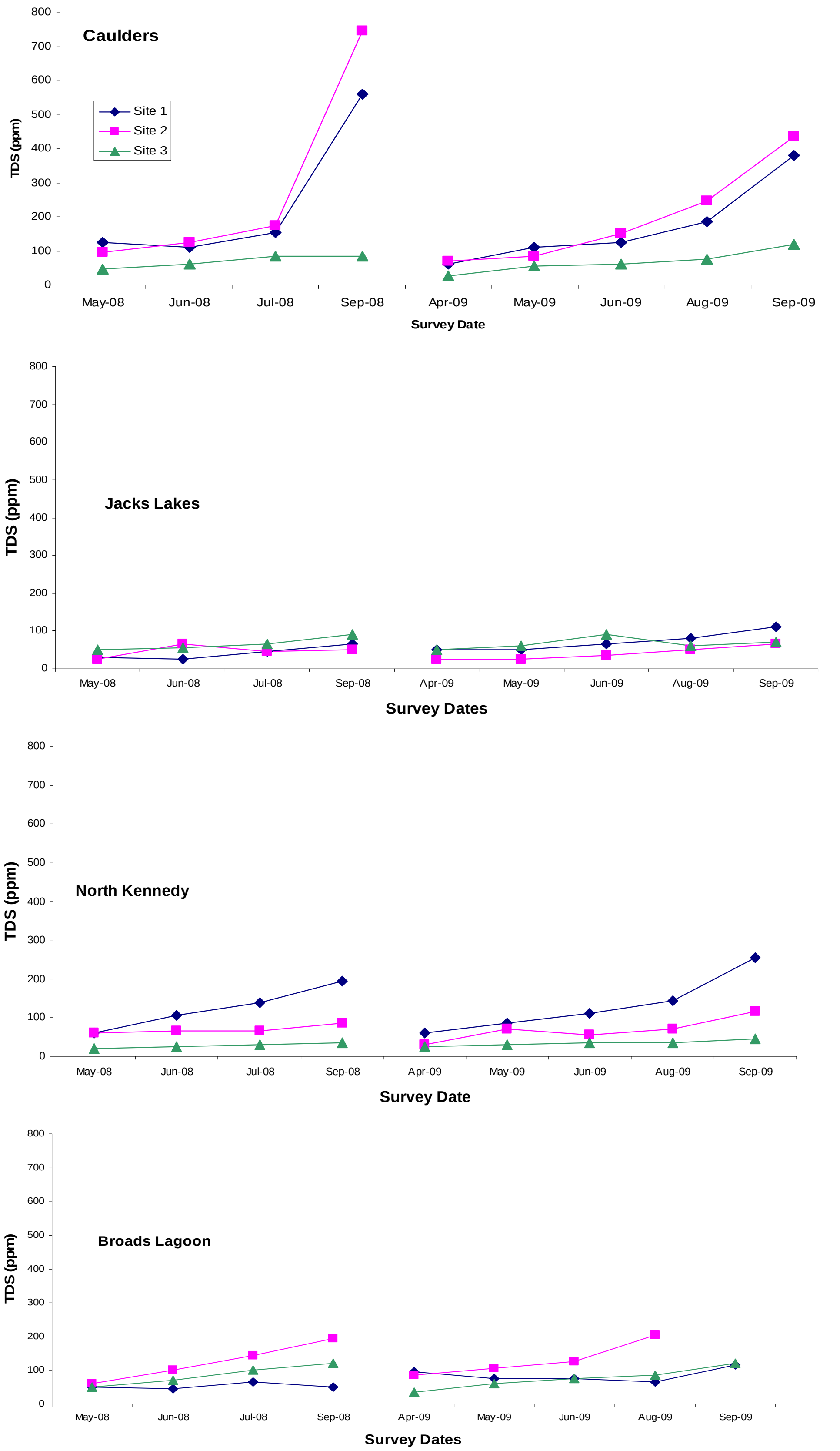

Figure 3.22. Total Dissolved Solids levels for site in each lake system for all surveys.

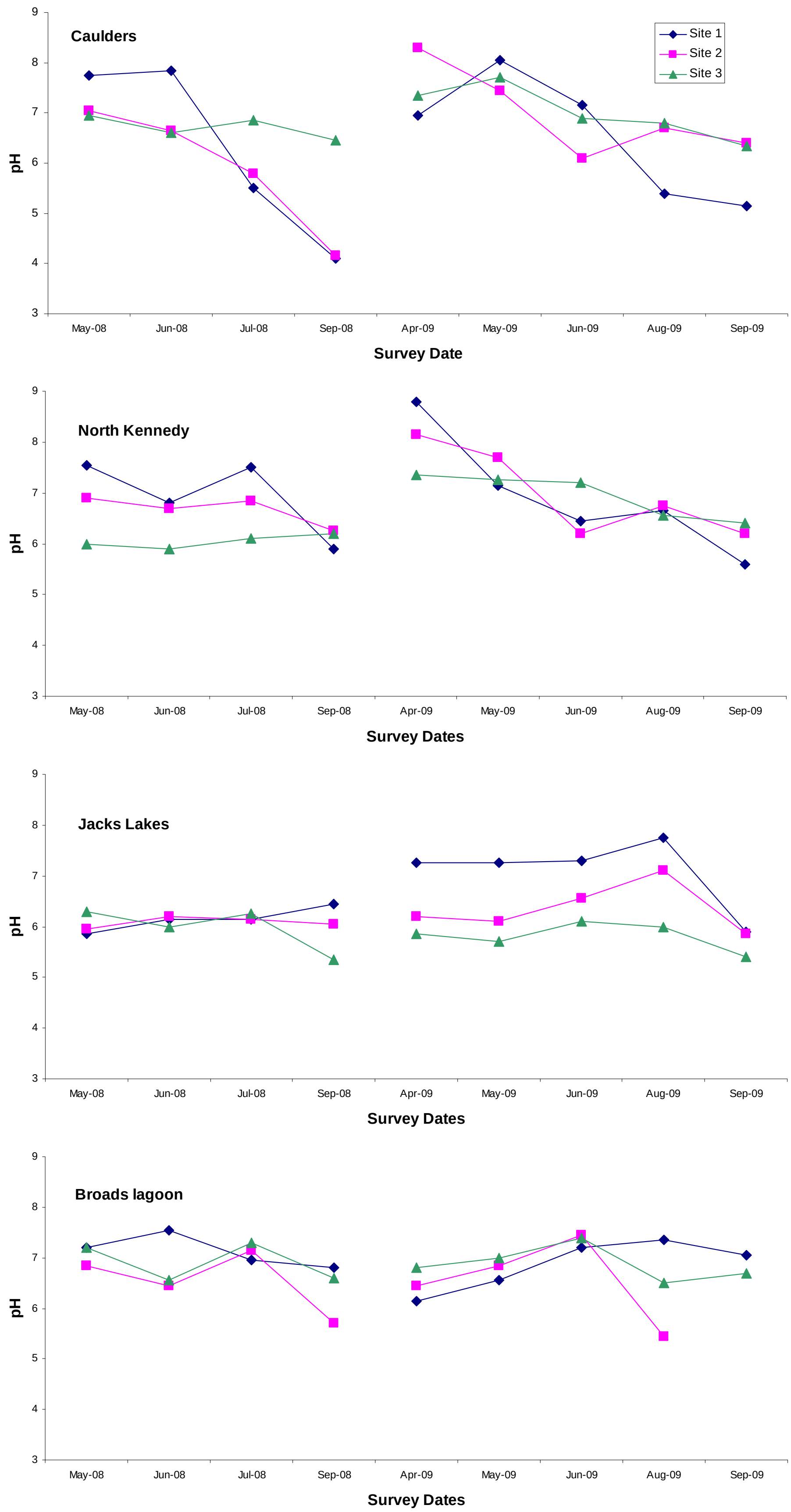

Figure 3.23. pH levels for each site in each lake system for all survey periods.

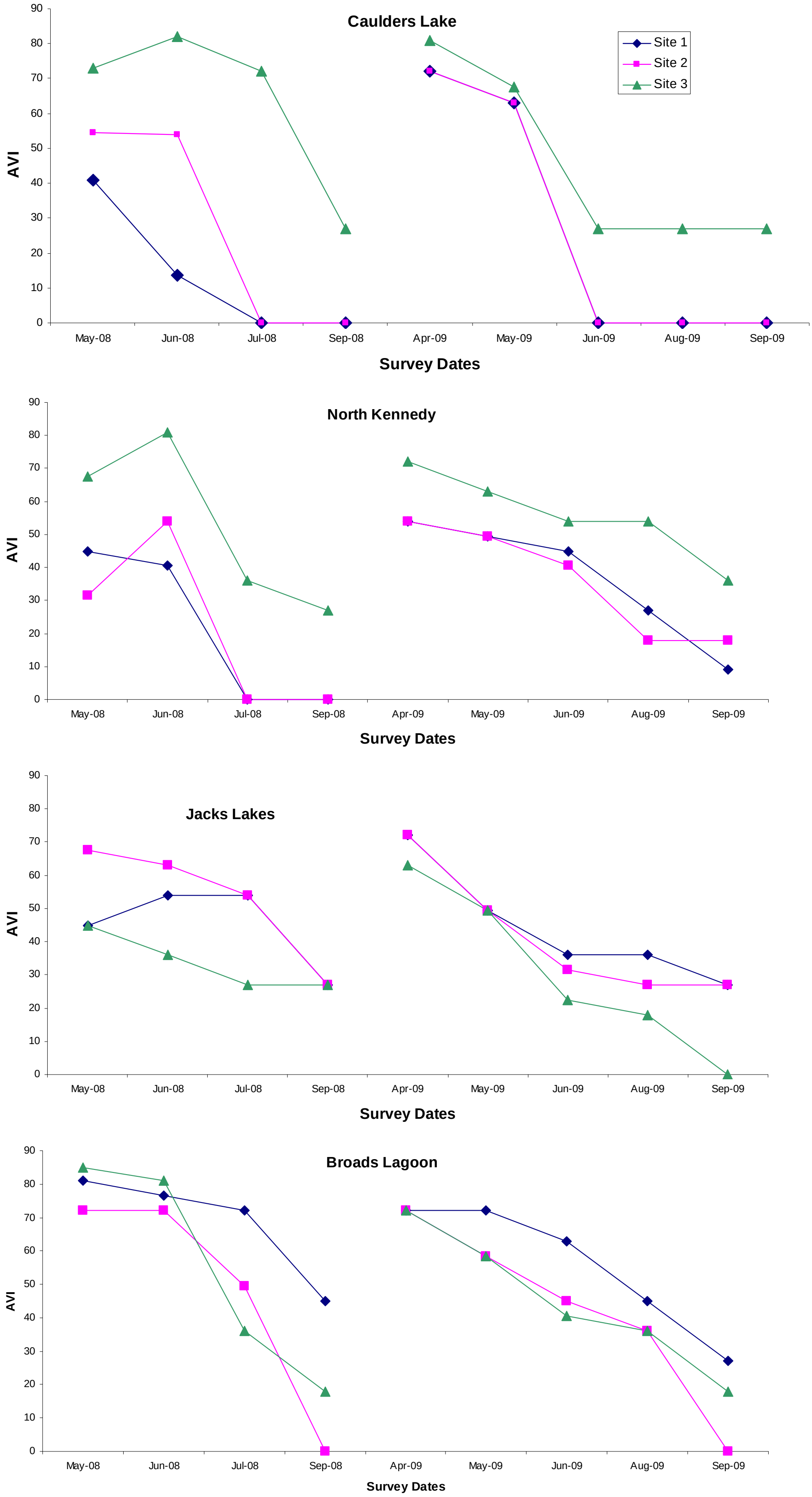

Figure 3.24. Aquatic Vegetative Index levels for each site in each lake system.

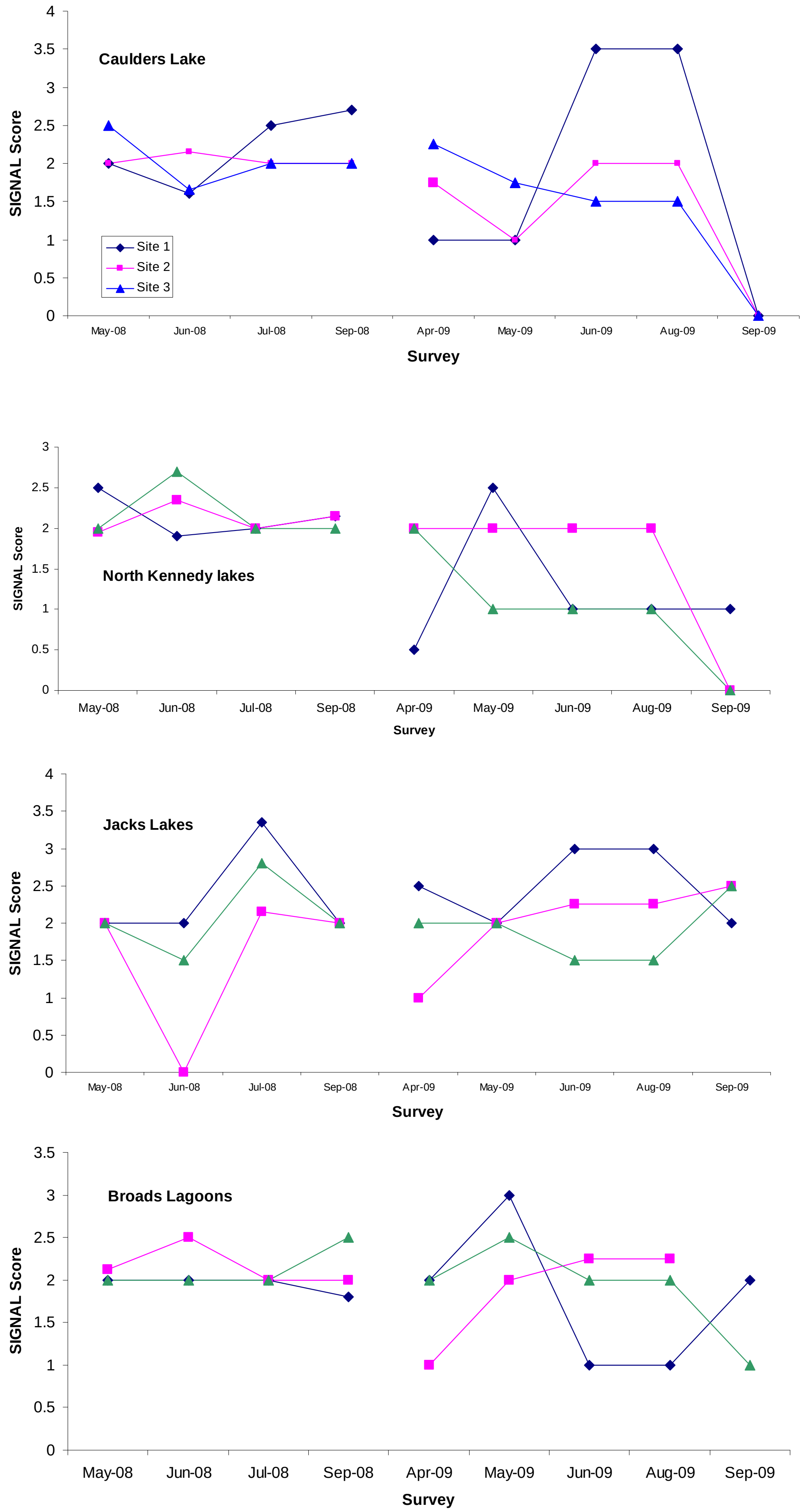

Figure 3.25. SIGNAL score for each site in each lake system for all survey periods.

3.4. DISCUSSION

Two associations of pig abundance with pig impacts were detected in this study. Firstly, there was a demonstrated positive association of pig abundance (as measured by the frequency of occurrence of sign and frequency of occurrence of diggings), with the digging area index (the impact indicator of the amount or extent of recent pig diggings that occurred). Thus the level of impact caused by pig diggings is positively associated with the level of pig abundance: more pigs, more diggings. Secondly there was an overall significant association of the Aquatic Vegetation Index with two abundance indices; although the associations were positive, pig abundance increased with increasing aquatic vegetation abundance.

The association of pig abundance with the extent of diggings impact indicator agrees with Hone's (2002) results. He also demonstrated a significant positive curved exponential relationship across years between pig abundance (as measured as the proportion of plots with dung in his study) and the frequency of occurrence of pig diggings and the extent of these pig diggings. Hone (2007) lists seven studies of the damage (diggings) caused by pigs to the environment; three studies demonstrate a positive linear relationship between pig abundance and environmental damage. Other published studies by Belden and Pelton (1975) and Giles (1980) assumed a significant positive relationship existed between the extent of pig diggings and pig abundance. Ralph and Maxwell (1984) also reported a significant positive correlation.

However, other published studies have shown conflicting views of this relationship. Hone (2002) listed two studies that demonstrated no relationship between impacts and abundance and two studies reported a concave down relationship. Cooray and Mueller-Dombois (1981) reported no correlation could be found. Choquenot *et al.* (1996) viewed these contrasting results and discussed the complexities that exist in this impact / abundance relationship. Hone (1987) suggested that impacts associated with diggings may persist beyond changes in the abundance of pigs complicating the nature of the relationship; diggings may also take a long time to regenerate. He suggested that new diggings, as used in our study, had a complex relationship with pig abundance.

The shape of the relationship curve is also important when considering how to evaluate impact on a per capita basis (Choquenot *et al.* 1996). The slope of the curve and the point of origin when graphing this relationship must be quantified to avoid inaccuracies in estimating the impact levels at differing pig abundance levels. Our study identified an exponential concave up curve to this relationship. Impacts increased exponentially with increasing pig abundance. Thus when pig abundance is high, a minimal level of population control is required to reduce impacts. When pig abundance is low a substantial level of pig control is required to decrease impact levels. The point on this relationship curve where the minimal level of population control required to maximise impact reduction appears to be in the range of 40% to 60% visitation frequency on plots. Thus a monitoring program could be instigated whereby when the frequency of visitation is higher then 50%, implementation of a control program would maximise impact reduction. Visitation frequency below 50% would only trigger a control program if resources were available. The level of control required to achieve a substantial impact reduction at this point may be financially prohibitive.

There was also a significant positive association demonstrated in this study between the extent of diggings and the percentage of plots with diggings which also occurred in the Hone (1988a) study. He stated that the frequency of occurrence of diggings is usually assumed to be curvilinear related to the extent of pig diggings. He also reported the percentage of plots with diggings was 18.1% to 26.6% in his study which is slightly higher then the 14.8% reported in this study.

The mean proportion of ground disturbed by diggings of 2.8 % was similar to 2.7% in the Hone (1988a) study. Mitchell and Mayer (1997) reported a digging extent of 4% in north Queensland rainforests. In contrast, Alexiou (1983) found 32% and Katahira *et al*. (1993) only found 0.7%. The ground disturbance at LNP represents the diggings that occurred over a three day period so this diggings rate could theoretically represent 0.93% of the ground surface disturbed on a daily basis. If we assume that pigs did not revisit old diggings, then in just over 100 days the total surface area 1 metre above the water mark surrounding the lakes could be disturbed by pig diggings. This was observed in all lake systems at the end of the dry season - the total surface area surrounding the lakes was dug up by pigs. Indeed on most surveys, especially as the water began to recede, the cumulative pig diggings on the water's edge were estimated to be close to 100% of the area being disturbed by pig diggings.

There appeared to be a strong seasonal effect with digging variations between 0.1% and 16.8% between the surveys compared to 3.6 % and 2.2% in the Hone (1988a) study. However the digging variability was comparable to the 6% to 11% and 14% to 38% found in Hawaiian forests (Cooray and Mueller-Dombois 1981; Ralph and Maxwell 1984). Digging patterns may be influenced by the presence of micro-variables which cause intense diggings in a small number of favourable microhabitats. Large areas of less favourable microhabitats tend to be associated with low levels of diggings. Hone (1988) also found this negative exponential frequency distribution of diggings; many sites had little or no activity while a few sites had high digging activity.

One aspect of this study was the frequency of pig sign on the activity transects were strongly associated with the number of pigs observed during the aerial shooting programs. As the aerial counts were assumed to represent the total number of pigs present at the lake systems during the shooting period then the frequency of visit index utilised in this study accurately represents the abundance of pigs at the lake systems over the shooting period.

The abundance and distribution of large herbivores such as feral pigs can be influenced by micro-variable factors such as food supply (Choquenot 1998). The intrinsic food hypothesis (Sinclair and Byrom 2006) proposed that density dependant mortality regulates population abundance through food shortages. Andrewartha and Birch (1984) argued that the limitation of food resources could influence abundance via intrinsic factors such as the animals foraging activity limiting the food availability or extrinsic factors such as the weather limiting food resources. They suggested that the relationship between the abundance of food resources and the survivability of the population will lead to density dependent population regulation limitations. Both intrinsic and extrinsic factors are apparent in LNP; pig diggings destroy a proportion of the very food resources they are utilising, and the weather or the natural hydrological cycle from the wet season to the dry season is a major component of the abundance of food resource availability.

The strong association of the extent of diggings with the availability of aquatic vegetation is seen as a positive response to increasing food resource availability and not as an indicator of pig impacts. The majority of diggings occur just after the wet season. There was a clear trend at all of the lake systems for pig activity to decrease during the dry season as the natural cycle of the amount of aquatic vegetation decreased as the water receded. This demonstrated a clear association of the availability of aquatic vegetation as a food resource with the level of pig abundance at the waters edge. This was also demonstrated in the diet study where macrophytes; especially waterlilies, was the dominate preferred food in their diets.

There was a considerable spike in the amount of diggings which occurred in the July 08 period, especially for Caulders Lake system and for the North Kennedy Lakes, and also to a smaller extent in the August 09 survey. This could be related to the decreasing depth of the water bodies. Although pigs have been observed with their heads underwater foraging for food (see Plate 4.2), it is logical to assume that pigs would not prefer this method of feeding. On many occasions pigs were observed to dig out water lily plants and take them to shore before consuming them (see Plate 4.1). Only mature boars were observed to forage in water deeper then 50 cm. The geomorphology of the lake systems is predominately saucer-shaped. As the water recedes there is a point where the depth of water does not deter pigs from foraging but is still deep enough for aquatic plants, especially water lilies, to survive. This point is where the majority of diggings occurred. Casual observations support this view, when the water is deep, pigs forage on the floating vegetation on the waters edge. When the depth of water is insufficient to deter pigs entering, then pigs venture out (sometimes into the middle of the lakes) to search for water lily bulbs, the dominate food resource. The presence of large crocodiles may also deter pigs digging under the water when the water is deep (Bowman and McDonough 1991). Predation of pigs by saltwater crocodiles was observed in this study.

There was no clear overall statistical association of pig abundance with the other ecological impact indicators demonstrated in this study although strong associations were evident when the results were presented graphically. Some statistically significant associations were detected for the individual lakes; for example, there was a significant association detected in the Caulders Lake system where the Total Dissolved Solids levels increased with increasing pig abundance. A strong trend of this association was also detected in all sites in the North Kennedy Lakes system. There was also a significant association of increasing turbidity with increasing digging activity detected in the Caulders lake system. All of the other lakes also demonstrated a strong association of turbidity with increasing pig abundance. A variety of significant associations were also demonstrated within some of the individual study sites (predominantly independent bodies of water), although there was no overall significant associations detected between or within the overall lake systems. The reasons for this are complex; however, I believe the natural variation in site ecology, the variability in landscape features between sites and the temporal changes caused by the normal hydrological cycle tend to obscure the influence that the abundance of pigs had on these impact indicators.

All of the water health parameters and biodiversity indicators in this study indicated that water health in the lake systems was strongly influenced by temporal factors principally the normal hydrological cycle for each year. All of the abundance and ecological indicators demonstrated significant differences between the survey periods and a strong temporal interaction effect, indicating a strong influence of the hydrological effect.

The influence that broader spatial scale factors such as landscape composition has on the population dynamics, and their associated impacts, of many animal populations has been the subject of study throughout the world. The composition of habitat types in the landscape and the spatial arrangements of these habitats are the two essential features in describing the suitability of the landscape to support animal populations (Dunning *et al*, 1992). A number of landscape processes are thought to influence the amount and scope of pig impacts in LNP especially their foraging or digging activity, or to influence why the pigs had a preference for a particular site or avoided a particular site.

A good example to illustrate this influence of landscape features is a site on LNP called "Devil-Devil lagoon". This lagoon is only 1 km from the North Kennedy lake system yet receives minimal pig impact compared to the lake system. The landscape features of Devil-Devil consist of steep banks which drop off into deep water, limited vegetation on the banks, no access to water lilies and the presence of large crocodiles. The limited numbers of small creeks that run into this lagoon receive severe pig impacts where shallow water and flat banks exist.

Analysis of the correlation of these landscape features with pig impacts revealed a strong significant association of pig abundance with the Aquatic Vegetation Index; pig abundance increased with increasing macrophyte abundance: more food more pigs. Preliminary analysis failed to detect any associations with other landscape features with pig abundance; however, further analysis is still to be conducted.

The distribution and abundance of feral pigs in this environment is dependant on a range of spatial and temporal factors. Choquenot *et al* (2003) described the landscape process that proximity of habitats containing essential resources has on feral pig abundance at large spatial scales as landscape complementation. They suggest that this effect can influence interactions between feral pigs and their food resources; the proximity of suitable habitats may constrain their foraging or demographic efficiency. Landscape complementation occurs when the required patch types occur in close proximity within a landscape and thus supports a larger population then do landscape where these habitat patches are far apart.

Lakefield NP can be described as a landscape composed of different types of habitat patches of preferred pig microhabitats, each containing different suitable landscape features. If the pigs require two or more different resources at some time in their life cycle then the pigs must travel between these patches, since the critical resources are found in patches of different types. This study suggests that landscape complementation is prevalent in LNP. The temporal availability of abundant food resources such as water lilies in the lake systems and the temporal need for access to shade for thermoregulation in the tropical summer creates this spatial patch effect.

Thus the impacts of pigs in this environment are also related to temporal and spatial influences.

Although not documented in this study the long term ecological effects of pig diggings have been documented in many other studies. Hone (2002) for example, demonstrated a significant negative curved relationship between plant species richness and the extent of pig diggings, with plant species richness declining to zero in the presence of extensive pig diggings. Alexiou (1983), in his study near Canberra, described significant changes in density and cover of a wide range of plant species following disturbance by pigs. Hone (1998) suggested that species richness is inversely related to the amount of pig digging disturbance. If diggings are less then 25% of the area, there will be a short-term effect. If pig diggings cover more than 25% of the areas, there will be a rapid reduction of species richness. This was seen in this study where 100% of the lake system edge was dug up by pigs. Species richness in these areas was zero.

A special case to demonstrate the impact of feral pig diggings was observed at Caulders Lake in 2008. Large groups of pigs (sixty plus) were observed feeding at this site. A combination of an aerial shooting exercise and a feral pig baiting program was conducted around this system on the 10th October 2008. A total of 400 kg of kangaroo meat (obtained from a licensed pet food abattoir) was used to prepare 850 baits which were laid around the periphery of the water bodies and 110 pigs were aerially shot. The average activity indices and water parameters values obtained from the previous July survey were compared to the September post baiting / shooting indices. On average, activity signs decreased by 91.6 % and sightings also decreased by 59% indicating a major decrease in the pig abundance levels. The level of ecological impact decreased dramatically with the amount of new diggings decreasing by 99.4 %. The water health of the lake also markedly improved after this control program with a drop in turbidity levels of 81%. The ammonia level also dropped by 29%, possibly due to less faeces and urine being deposited by pigs. Normally turbidity and ammonia levels would increase between these survey periods as shown in the 2009 surveys. The chlorophyll levels also changed dramatically with a major increase in chlorophyll at site 2 after the control program. This could be caused by the increased clarity of the water allowing more light to penetrate the water.

All of the lake systems were comparable for all of the environmental impact indicators with the exception of Jacks Lakes which showed a significantly higher ammonia level. However, all the lake systems also show significant lake system by survey period interaction due to the normal hydrological cycle over the dry season.

We found that the structure and function of the wetland ecosystems in this study are strongly influenced by seasonal changes in water level, and this drives the natural disturbance regime in these habitats. After the commencement of the wet season the wetland systems are "reset" back to their original condition. However, this study had no true control in the sense of an ecological "original" reference point without feral pig disturbance, since feral pigs are known to have been in the Lakefield region for a 100 years or more. It follows that because the wetlands in this region have been disturbed by feral pigs for very many years, then all may be altered to some extent by this prolonged disturbance and any truly pig-sensitive species may have been eliminated from the wetlands well before this study began.

CHAPTER 4.

Experimental Research to Quantify the Environmental Impact of Feral Pigs within Tropical Freshwater Ecosystems:

General Discussion

The primary aim of this study was to quantify the threat feral pigs pose to the health of tropical freshwater ecosystems. This study examined the ecological impacts feral pigs have on biodiversity and also the relationship of these ecological impacts with the level of feral pig abundance.

Pig foraging has a dramatic effect on the biodiversity within freshwater ecosystems. The major destruction of macrophyte communities and upheaval of wetland sediments significantly reduced water clarity and had subsequent effects upon key water quality parameters such as dissolved oxygen availability. However, despite these dramatic impacts, the effects on the composition and diversity of macrophyte, invertebrate and fish communities by feral pig foraging activities could not be statistically detected. Other water quality parameters such as nutrients were also strongly affected by seasonal water level declines and although the effects were not able to be statistically realised, it appeared that pigs contributed to an increase in nutrient levels.

Aquatic macrophytes are a key component of habitat in tropical wetlands, providing both an important food resource and structural complexity to the waterscape for associated biota (Thomaz *et al.* 2008). Since foraging feral pigs showed a strong preference for macrophytes in this study and indeed in moist habitats of northern Australia's wet-dry tropics (Hone 1990; Caley 1993, 1997), it implies that the ephemeral lagoons and their ecological communities in this region may be especially susceptible to significant disturbances by them (also see Bowman and Panton 1991; Mulrennan and Woodroffe 1998). Recently, Fordham *et al.* (2008) argued that protection of ephemeral wetland lagoons in northern Australia from "the destructive effects of feral pigs" was required (see also Bowman and McDonough 1991). The destruction of aquatic vegetation by feral pigs also significantly altered production/respiration regimes causing anoxic conditions and subsequent pH imbalances. Water samples frequently recorded dissolved oxygen levels between 30 - 70 % saturation, which provides at least chronic sub-lethal effects for the associated biota (Sprague 1985; Butler and Burrows 2007). We didn't find evidence for toxic effects on either fish or macroinvertebrates, although this may be because levels above 30 % are not thought to be lethal to freshwater fishes (Dean and Richardson 1999) and aquatic surface respiration by fishes may compensate for periodically anaerobic conditions (McNeil and Closs 2007). Little is known of how low dissolved oxygen levels affect macroinvertebrate faunas. We found evidence that macrophyte destruction due to pig foraging enhances lagoon acidity as pH levels of about 6 were recorded, and is most likely due to sustained respiration in what became consumption-driven environments. Psenner (1994) nominates pH levels of 6.5 as the trigger value for beginning sub-lethal effects on sensitive species.

This study demonstrated that the foraging activities of feral pigs in these floodplain lagoons disrupt their physical, chemical and biological environments. Pig disturbance dramatically increasing turbidity and pig foraging is clearly linked to the destruction of aquatic macrophytes however we found no evidence that feral pig diggings has affected either the number or diversity of plant species and nor have we found evidence for the invasion of exotic weeds as a consequence of pig disturbance.

The level of pig diggings is positively associated with the level of pig abundance: more pigs, more diggings and by association more ecological impacts from these diggings. Our study identified an exponential concave up curve relationship of the amount of diggings associated with the pig abundance; diggings increased exponentially with increasing pig abundance. It appears that when pig abundance is high, a minimal level of population control is required to reduce digging impacts. When pig abundance is low a substantial level of pig control is required to decrease digging impact levels. The point on this relationship curve where the minimal level of population control required to maximise impact reduction appears to be in the range of 40% to 60% visitation frequency on plots.

Thus a monitoring program could be instigated whereby when the frequency of visitation is higher then 50%, implementation of a control program would maximise impact reduction. Visitation frequency below 50% would only trigger a control program if resources were available. The level of control required to achieve a substantial impact reduction at this point may be financially prohibitive.

Graphical examination of the data concluded that there is a demonstrated effect of pig abundance on water quality. The specific example of improved water quality after pigs were shot out of Caulders is good evidence of a non-statistical water quality effect. However, there was no statistical association of pig abundance with ecological impacts as shown by the lack of a statistical association of pig abundance with the ecological impact indicators used in this study. Although a variety of significant associations between abundance and ecological impacts were demonstrated within some of the sites (predominantly independent bodies of water), there was no overall statistical associations detected. For example, two individual sites demonstrated a statistically significant positive association of increasing turbidity levels with increasing pig diggings while all of the other ten sites, although also demonstrating a positive association, none were statistically significant. Thus the overall effect was a strong trend of increasing turbidity levels (as also demonstrated in the exclosure study) with increasing pig abundance, but this effect was overwhelmed by the site variability. The reasons for this are complex; however, I believe the causes of this lack of statistical effect being detected are the variability in landscape features between sites and the temporal changes caused by the normal hydrological cycle. All of these factors tended to obscure the influence that the abundance of pigs had on these impact indicators.

The sites ecological variability is one of the principal causes for the lack of significant associations being detected. The variability of landscape features between the individual sites smothered a number of associations.

All of the water health parameters and biodiversity indicators in this study indicated that water health in the lake systems was strongly influenced by temporal factors, principally the normal hydrological cycle for each year. All of the abundance and ecological indicators demonstrated significant differences between the survey periods indicating a strong influence of the hydrological effect. Seasonal conditions significantly influenced diet composition and generally reflected how food availability changed with season. Macrophytes (water lilies) are the dominate food resource – when available. Pigs show a strong dependence on water sources for primary food items (e.g. water lily bulbs, snails, nardoo).

Population abundance is only one of many factors that influence the population dynamics of feral pigs and subsequent impacts in this environment. Dunning (1992) suggested that the composition of habitat types in the landscape and the spatial arrangements of landscape features are the two essential features in describing the suitability of the landscape to support animal populations. This suitability is reflected in the availability and quality of resources and the degree to which local constraints will influence the assimilation of these resources for survival and reproduction purposes.

The landscape process that proximity of habitats containing essential resources has on feral pig abundance at large spatial scales can be described as landscape complementation. Choquenot (2003) suggested that this effect can influence interactions between feral pigs and their food resources; the proximity of suitable habitats may constrain their foraging or demographic efficiency. Lakefield NP can be described as a landscape composed of different types of habitat patches of preferred pig microhabitats, each containing different resources. If the pigs require two or more different resources at some time in their life cycle then the pigs must travel between these patches, since the critical resources are found in patches of different types. Landscape complementation occurs when the required patch types occur in close proximity within a landscape and thus supports a larger population then do landscapes where these habitat patches are far apart.

Digging patterns appear to be influenced by the presence of micro-variables which cause intense diggings in a small number of favourable microhabitats. Pig diggings have a highly skewed frequency distribution and are concentrated into certain favourable areas. Large areas of less favourable microhabitats tend to be associated with low levels of diggings. Hone (1988; 2002) termed this negative exponential frequency distribution of diggings, many sites with little or no activity and few sites with high digging activity. A number of landscape processes are thought to influence the amount and scope of pig impacts in LNP especially their foraging or digging activity, or to influence why the pigs had a preference for a particular site or avoided a particular site. The determination of site (spatial) characteristics will also assist in determining whether ecosystems are prone to damage.

A number of studies have sound a close association of pig diggings with moist soils (Bratton *et al.* 1982; Kotanen 1995; Mitchell 1997; Mitchell 2002). Hone (1995) found that the proportion of plots with pig diggings were highest in wet locations in his study; moist soils attract pigs.

The soft soils within the lake systems and the predominance of bulb producing plant species, lend themselves to digging by pigs. The year round availability of soil moisture in the lake systems would also induce increased soil invertebrate populations particularly earthworms and insects. The diet study indicated insects were an important part of the pigs diets during the wet season with 100% occurrence in the diet. The combinations of soft soil and elevated invertebrate populations may explain the higher incidence of diggings. The elevated level of diggings should also be more pronounced during the dry season as pig activity would tend to concentrate within the preferred swamp areas.

Bowman and McDonough (1991) in their study in a tropical freshwater environment also found significantly more pig rooting activity in sedge lands and swamps and adjacent forests, then in monsoonal rainforest. They found significant spatial and temporal rooting activity and a positive relationship with the amount of ground cover available. They determined that less ground cover and subsequent lower availability of food resources in the rainforests lead to lower rooting activity.

Weather conditions, especially high temperatures, may cause a movement shift between habitats to avoid high temperatures. Dexter (1998) showed that when temperatures were high, pig populations concentrated in shady habitats. However food availability influenced habitat selection when temperature constraints were relaxed. Dexter (1999) found a sex difference in home range areas, females responded to changing environmental conditions while males did not. Females were influenced by two of the same variables that influence habitat usage; daily maximum air temperature and pasture biomass, suggesting that the factors governing habitat use and home range sizes are not independent.

High summer temperatures in the tropics would make pigs concentrate around the available wetlands for the water and shade in the riparian vegetation thus increasing impact from foraging behaviour in these favoured areas. The distribution of pigs during the tropical summer conditions represent a compromise between the need to forage more widely for the increasing limited food resources and the need to stay close to reliable water and cover (shade). Restricted movements may also be an adaptation to conserve energy during times of food shortage (Massei 1997).

The large resource availability in the larger lake systems would limit the need for feral pigs to move. The similarity of home range areas in suitable pig habitats throughout Australia suggest that pigs limit their movements to the smallest home range area where all resource requirements are met. During the wet season, all foreseeable resource requirements would be met at the lake systems, pigs would have no reason to move and consequently their impact would be targeted at the available resource – the freshwater ecosystem. Only during the dry season when resources are reduced or become totally absent will pigs need to move to survive. Resource availability is the dominant influence on the level of pig impact. Pigs concentrate their impact where adequate resources are present.

Movements or home range size is a predictable consequence of resource requirements. Larger body weight animals typically require more resource requirements with a consequential increased home range size. Dexter (1998) showed that a range of environmental constraints influence the distribution of feral pigs, however he suggested that if the major constraint of ambient temperature was relaxed then the distribution of pigs was influenced by the availability of food. High ambient temperatures during daylight in the summer months within this study area would significantly influence pig distribution - especially the availability of water on a daily basis.

We found that the structure and function of the wetland ecosystems in this study are strongly influenced by seasonal changes in water level, and this drives the natural disturbance regime in these habitats. The wetlands are "reset" back to their original condition after each wet season. However, a limitation of this study is the lack of a true ecological "original" reference point without feral pig disturbance. Since feral pigs have been in the LNP for over 100 years, then all may be altered to some extent by this prolonged disturbance and any truly pig-sensitive species may have been eliminated from the wetlands well before this study began.

There is no information on the long term ecological consequences of pig activity in this tropical freshwater habitat; however there is ample evidence in other environments from many previous studies. Studies have described significant changes in density and cover of a wide range of plant species following disturbance by pigs. Hone (1998) suggested that species richness is inversely related to the amount of pig digging disturbance. If pig diggings cover more then 25% of the areas there will be a rapid reduction of species richness which will decline to zero in the presence of extensive pig diggings. This was seen in this study where 100% of the lake system edge was dug up by pigs, species richness in these areas was zero. Other possible long term consequences of pig activity have been highlighted in Chapter 1 section 1.1.

The use of fencing to protect environmental resources from exotic ungulates has been used in Hawaii for several decades but it is expensive and requires ongoing maintenance (Loope and Scowcroft 1985; Stone *et al*. 1992; Katahira *et al.* 1993). However, given the finding that the wetlands are ecologically dissimilar to each other and the fact that most biological parameters are significantly affected by the temporal influence of season, this has implications for both the choice of wetlands targeted for protection and the practicality of this management tool. For example, how would we apply fencing on a wider scale in Lakefield given the ecological diversity of these wetlands and the effect of feral pigs on wetland biota may be dwarfed by seasonal climatic effects.

4.1 CONCLUSIONS

This study confirmed that the lagoons and lake systems that we surveyed harbour unique macrophyte, fish and macroinvertebrate communities. This suggests that these wetlands and probably those of the wider region have high conservation value since they each contribute to the biodiversity mosaic of Lakefield National Park.

At Lakefield National Park (LNP) and indeed across monsoonal northern Australia, there are two predictable primary disturbances that annually affect the ecological communities of these floodplains. One is the complete inundation of the floodplain for several months which is then followed by a seasonal desiccation of them. Natural disturbance plays an important role in maintaining ecosystem biodiversity (Grime 1973), and changes from it can vary successional pathways (Sousa 1984) and so alter ecological communities and processes (Hobbs and Huenneke 1992; Schmid *et al*. 2001). At LNP, juxtaposed on this natural and predictable disturbance regime is the secondary disturbance of feral pig intrusion which increases as the wetlands dry. Throughout the world, ecological systems are becoming increasingly affected by non-native species whose activities can either introduce new forms of disturbance, or enhance or suppress existing disturbance regimes (Mack *et al.* 2000). We need to disentangle these potentially confounding influences to accurately understand how feral pigs are affecting ecological diversity and ecosystem functioning in these habitats, or if the differences we have measured are simply reinforcing the role of seasonal climatic change as the major driver of diversity and function in this floodplain landscape. This would then allow a better understanding of the ecological consequences of pig disturbance in this region and of how to manage ecological systems that regularly experience disturbance such as these.

We have demonstrated that feral pigs do have significant impacts on the water health of wetlands in these tropical wetland environments. Macrophyte populations and water clarity / nutrients are strongly influenced by pig foraging. We have also demonstrated that the level of impacts is also related to the pig population abundance. However, we have also demonstrated that there are significant natural disturbances also operating in this ecosystem that should be taken into account when assessing pig impacts on these tropical wetlands.

4.2 RECOMMENDATIONS

It is recommended that future research in this environment include more permanent lagoons and wetlands. Permanent water bodies are arguably more ecologically important aquatic habitats and also have the advantage that the seasonal effects of lagoon drying are less extreme, thus making delineation of the effects of pigs more achievable.

Further research into the true ecological effects of feral pigs might be best measured in a landscape-specific framework. We also need to further understand the ecological drivers of pig ecology in this environment. We recommend that further research is required to understand factors such as the influence of landscape complementation, the landscape process that proximity of habitats containing essential resources has on feral pig abundance and subsequent feral pig impact. The interactions between feral pigs and their resources requirements and the proximity of suitable habitats have on their foraging or demographic efficiency needs to be further refined so a more complete model of pig ecology / impacts can be developed.

We recommend that research be undertaken to quantify the drivers of pig movements in relation to seasons and refuge areas characteristics to identify habitats to target for control. Identification of landscape characteristics that promote feral pig disturbance needs to be further defined.

There are a number of rare and endangered species within this wetland ecosystem. The effects of feral pigs on these species are unknown. Management and recovery plans for these species need to incorporate research into quantifying the ecological impacts of feral pigs on the survival of these species.

The management of Lakefield (Department of Environment and Resource Management) should consider establishing a monitoring program utilising the activity plots established in this study. Regular monitoring during the dry season will trigger a control program to be instigated when pig activity reaches a predetermined level. The results of this study indicated that a frequency of pig activity on plots of over 50% is the best time to maximise control effectiveness in terms of reducing impact levels. This level will also need to be set using resource priorities.

The results of this study have shown a very strong relationship between estimates of population levels using activity plots and estimates from aerial shooting surveys. It is recommended that activity plots can be used to accurate estimate the true population level without the expenses associated with aerial surveys.

The specific feral pig control programs used in this study at Caulders Lake were shown to significantly improve water quality. It is recommended that aerial shooting combined with strategic baiting program can be used to protect high priority wetland systems in this environment.

4.3. BACKGROUND PLATES

A series of photographs are presented below to illustrate some of the points raised in the discussion.

Plate 4.1.

Water Lilies are a preferred food source for feral pigs. However they are more prolific in deep water which is generally inaccessible to pigs. Pigs overcome this problem (and the threat of predation by crocodiles) by venturing as deep into the water as possible and then with their heads under water they root out the whole plant and carry it back to shore for consumption.

The most preferred part of the plant is the bulb (rhizome) and flowers, although all parts may be consumed.

Plate 4.2 Large boars have been observed feeding in deep water for water lily bulbs (photo a), although they generally prefer shallower water (photo b). Feral cattle also forage on the water lily flowers in deep water (photo c).

(a)

(b)

(c)

Plate 4.3. Pigs prefer to forage in shallow water. The maximum depth they forage is seen in the photographs as the line of water lilies in the background. As the water recedes, pigs continue to forage further into the lake until the water depth of the lake is sufficiently shallow to allow the pigs to venture into the middle of the lake.

Plate 4.4. Landscape features have a profound influence on the frequency and scope of pig activity. Some landscape features (photo a) are not suitable for pig foraging such as steep banks, hard soil, deep water, no accessibility to macrophytes and the high risk of crocodile attacks. Other landscape features (photo b) are more suitable for pig foraging such as flat banks, soft soil, shallow water, macrophytes accessible and the reduced risk of crocodile attack.

(a)

(b)

Plate 4.5. Pigs generally occur in family groups, (50 individuals was the largest group seen in this study.

Plate 4.6. Water Samples were taken 5 m from the waters edge using an extension pole.

Plate 4.7. Extensive pig diggings occur as the water recedes.

Plate 4.8. Exclusion fencing after the wet season. All exclosures were covered with flood water during each wet season. All fences remained pig proof throughout the study.

Plate 4.9. Feral pigs were aware of the predation threat of crocodiles which influenced their digging activity. Photo (a) shows pig diggings at the top and bottom of the photo, the centre section is where two crocodiles used to bask in the sun (Photo b). For the duration of the study no pig diggings were observed in this section – probably due to the scent of the crocodiles.

(a)

(b)

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WetlandCare Australia’s Wetland Assessment Technique Version 3.4 (March 2007)
Estuarine Wetland Assessment & Prioritisation Project Compiled by Che Golus, Cassie Burns, Megan Westlake and Adam Gosling WetlandCare Australia,

APPENDIX A
Raw Fish Data from 2008 and 2009

DATE	SITE	*Ambassis agrammus*	*Amniataba percoides*	*Glossamia aprion*	*Leiopotherapon unicolor*	*Melanotaenia splendida*	*Mogurnda adspersa*	*Nematalosa erebi*	*Neosilurus atar*	*Neosilurus hyrtlii*	*Oxyeleotris lineolata*	*Toxotes chatareus*	Total No. of Species	Total No. of Fish
May-08	12 Mile Fenced	92		1		1			3	56	1		6	154
May-08	12 Mile Unfenced	94						3	1	9	2		5	109
May-08	Anabranch Fenced	15		1	1	4					3		5	24
May-08	Anabranch Unfenced	22								1			2	23
May-08	Laura Fenced	178				1							2	179
May-08	Laura Unfenced	37			4	1							3	42
May-08	Welcome Fenced	21		1						4	1		4	27
May-08	Welcome Unfenced	18		1		2			1	25	5		6	52
Jul-08	12 Mile Fenced	8				3				12	2		4	25
Jul-08	12 Mile Unfenced	15								7	11		3	33
Jul-08	Anabranch Fenced	59				2							2	61
Jul-08	Anabranch Unfenced	40				3					6		3	49
Jul-08	Laura Fenced	56				1							2	57
Jul-08	Laura Unfenced	18			3	5							3	26
Jul-08	Welcome Fenced	2				6				29	2		4	39
Jul-08	Welcome Unfenced	3		1						1			3	5
Sep-08	12 Mile Fenced	8				3				12	2		4	25
Sep-08	12 Mile Unfenced	15								7	11		3	33
Sep-08	Anabranch Fenced	304		3	2	15					6		5	330
Sep-08	Anabranch Unfenced	24					1			3	7		4	35
Sep-08	Laura Fenced	135							1	1			3	137
Sep-08	Laura Unfenced	15			3	5							3	23
Sep-08	Welcome Fenced	2		1							2		3	5
Sep-08	Welcome Unfenced										8		1	8
May-09	12 Mile Fenced	16	1	2	1				1	71	1	6	8	92
May-09	12 Mile Unfenced	16			1			3	1	1		2	6	22
May-09	Anabranch Fenced	10		1		4							3	15
May-09	Anabranch Unfenced	57		5		1					4		4	67
May-09	Laura Fenced	59			2								2	61
May-09	Laura Unfenced	33			2		1						3	35
May-09	Welcome Fenced	30		4		1							3	35
May-09	Welcome Unfenced	20			1	3					3		4	27
Jul-09	12 Mile Fenced	68				6					1		3	75
Jul-09	12 Mile Unfenced	64			1	1					5		4	71
Jul-09	Anabranch Fenced	16			1	1							3	18
Jul-09	Anabranch Unfenced	2		3						1	3		4	9
Jul-09	Laura Fenced	30				3							2	33
Jul-09	Laura Unfenced	40			1								2	41
Jul-09	Welcome Fenced	13		2		8					3		4	26
Jul-09	Welcome Unfenced	8									11		2	19

APPENDIX B-1 2008 Aquatic Invertebrate Raw Data

	Twelve mile unfenced May 2008	Anabranch unfenced May 2008	Laura unfenced May 2008	Welcome unfenced May 2008	Twelve mile unfenced July 2008	Anabranch unfenced July 2008	Laura unfenced July 2008	Welcome unfenced July 2008	Twelve mile unfenced September 2008	Anabranch unfenced September 2008	Laura unfenced September 2008	Welcome unfenced September 2008	**Unfenced Sub-total**	Twelve mile fenced May 2008	Anabranch fenced May 2008	Laura fenced May 2008	Welcome fenced May 2008	Twelve mile fenced July 2008	Anabranch fenced July 2008	Laura fenced July 2008	Welcome fenced July 2008	Twelve mile fenced September 2008	Anabranch fenced September 2008	Laura fenced September 2008	Welcome fenced September 2008	**Fenced Sub-total**
Aeshnidae		11		1		2		1					15				1	1		2	1	1	3			9
Atyidae	1												1					1				1				2
Ancylidae														1												1
Baetidae	12	45	1	18	1	27	2	10	1	27	14		158	21	17	32	21		4	4	17		27	19	13	175
Belastomatidae											1		1						1							1
Caenidae	1							2					3	1	1		2				5				2	11
Carabidae				1									1													
Ceratopogonidae	1	6	9	15	6	28	20	21	6	3	2	66	183		11	15	7	2	5	9	16	2	9	28	94	198
Chaoborinae																										
Chironominae	38	37	153	41	210	99	150	144	210	434	133	1	1650	1	9	98	62	45	60	138	47	45	38	76	68	687
Chrysomelidae																		2				2			1	5
Cladocera		29		22	5		6	4	5	6	8	7	92		9	31	34	3		3	13	3	1	16	10	123
Cnidaria (Hydridae)			1										1			6										6
Coenagrionidae	7	66	8	27	1	51	14	4	1	18	5	3	205	15	22	21	29	19	8	3	17	19	3	8	8	172
Conchostraca			4	3		16				1	8		32			17		7	3	1		7		13		48
Copepoda	52	42	4	55	40	7	3	23	40	2	36	16	320	11	162	23	92	16	7	4	35	16	28	63	25	482
Corixidae	3	3		141	21	1	3	12	21	2		13	220	4		2	11	1			23	1	4	2	7	55
Culicidae	3	9		1		4	1						18	1			2		1		5		12		1	22
Curculionidae		1				1	1						3	18			3	1	5			1			1	29
Dytiscidae	839	80	53	469	750	51	153	124	750	8	310	142	3729	344	22	26	332	489	55	142	895	489	46	355	46	3241

Ecnomidae			1										1			1										1
Gerridae		1	1							4			6								2			1		3
Glossiphoniidae		1						1				6	8	1	4		2			1	4		6			18
Gyrinidae		1			2				2				5											1		1
Haliplidae		2		6		2		3		2		1	16			1	27			2	14				5	49
Hemicorduliidae	1												1													
Hydracarina	1	1	7	5		2	7	1		6	6	1	37		4	12	5	2	1	1		2	66	4	2	99
Hydraenidae							2						2								3					3
Hydrochidae		4	4	8	1	3	5	11	1		8	5	50	8	10		24	11	15	2	61	11	11	17	11	181
Hydrometridae		1								1			2		1		1	1			2	1				6
Hydrophilidae	11	11	22	25	5	21	25	51	5	17	18	22	233	10	9	9	10	27	22	20	43	27	14	23	10	224
Hydroptilidae		1									1		2							4				2	1	7
Hygrobiidae	1		2	1									4			4										4
Hyridae								2					2													
Isostictidae																										
Leptoceridae		1	1	4						1			7							1	5			1		7
Libellulidae	10	19	9	11	1	15	24	19	1	47	6	3	165		24	5	58	4	22	12	81	4	20	39	108	377
Limnichidae																										
Lindeniidae			3	2	1		1	1	1				9			3										3
Lymnaeidae	2	3		1		1	7	1		1	3		19	1	13	1	1	2	11	1	1	2		8	2	43
Mesoveliidae				1									1		1		1						1			3
Naucoridae	14	22	5	2	7	9	22	11	7	20	50	1	170	22	37	3	1	20	17	3	9	20	25	14	12	183
Nematoda		2		4									6												1	1
Nepidae		3	2	2		4	3	3		4	11	1	33		2	3	5		3	3	3		4	3	1	27
Noteridae			2				3				88		93			2			1	10				5		18
Notonectidae	1	6			2	5	9	3	2	7	2	5	42	3	4		1	1	1	5	2	1	11	5	7	41
Oligochaeta	3	1	4	6	1		2	3	1			3	24	8		2	3	9		2	4	9			1	38
Orthocladiinae	5	4	3	2		1							15		3		2			2			2			9
Ostracoda	3	14	2	5	1	1		2	1	8	1	1	39		116	5	28	1	8	4	13	1	460	6	46	688
Planorbidae	72	22	113	40	6	23	49	17	6	22	17	5	392	103	21	100	98	58	16	26	17	58	7	17	57	578
Pleidae	2	28	32	26		4	165	13		8	44	9	331	3	11	74	9	9	24	42	17	9	12	120	35	365
Protoneuridae						3							3													

Pyralidae				1			1						2			1	2	1		1		1				6
Richardsonianidae		1	1	2						3			7	2	9			1	2		1	1	1	1	1	19
Sciomyzidae							2						2					1				1				2
Scirtidae											3		3													
Sphaeriidae																										
Staphylinidae	1	2						2					5													
Stratiomyidae			1		1	1	2		1	6	1		13		2			2	20		4	2	36	4	4	74
Sundathelphusidae	1	2	2	3	3								11	3	1		2	1								7
Syrphidae	1	2				7							10		3		1									4
Tabanidae		4		1		6		1				1	13	2	9		7	4	9		9	4	2		1	47
Tanypodinae	1	41	17	18	4	14	35	27	4		13	2	176	3	17	22	27	21	18	20	22	21	8	24	8	211
Tipulidae																										
Undiff HUL	1	3		1				1					6		12	1	7		1		1					22
Undiff. Zygoptera		12					1						13	1	3		1									5
Undiff. chironomid						1					1		2						2							2
Urothemistidae																										
Veliidae	4		1	1									6	1		1					10					12
Viviparidae	5	67		146		13		77		1		4	313	12			250	4			95	4			19	384
No. Individuals	1097	611	468	1118	1069	423	718	595	1066	659	790	318	8932	600	569	521	1169	767	342	468	1497	766	857	875	608	9039
No. Taxa	29	39	30	37	21	30	28	30	20	26	25	23	59	25	29	28	35	32	26	29	34	31	27	28	32	55

APPENDIX B-2 2009 Aquatic Invertebrate Raw Data

	Twelve mile unfenced May 2009	Anabranch unfenced May 2009	Laura unfenced May 2009	Welcome unfenced May 2009	Twelve mile unfenced July 2009	Anabranch unfenced July 2009	Laura unfenced July 2009	Welcome unfenced July 2009	**Unfenced Sub-total**	Twelve mile fenced May 2009	Anabranch fenced May 2009	Laura fenced May 2009	Welcome fenced May 2009	Twelve mile fenced July 2009	Anabranch fenced July 2009	Laura fenced July 2009	Welcome fenced July 2009	**Fenced Sub-total**
Aeshnidae																		
Atyidae																		
Ancylidae	3				1				4	2				2	1			5
Baetidae	2	10	14	16	2	12	14	74	148	2	4	2	15	20	39	5	6	98
Belastomatidae																		
Caenidae						5			5	6				5			2	13
Carabidae																		
Ceratopogonidae	3		5	14	17	5	10	19	73	4	8		21	24	19	25	12	113
Chaoborinae		4	10	2			5		94			5				9		127
Chironominae	88	103	89	40	38	54	112	218	763	19	282	48	87	97	193	136	56	932
Chrysomelidae						1			1									918
Cladocera	3	1	6		12	6	25	1000	1054		3	3	7	53	5	22	4	97
Cnidaria (Hydridae)				1					1		1							1
Coenagrionidae	1	9		38	1	33	5	1	89	9	12		13	4	11	1	36	87
Conchostraca							3		91	1	1	2	2	1	2	5		100
Copepoda	10	21	15	11	65	95	28	50	298	9	9	4	19	66	21	30	5	177
Corixidae					292	18		7	612				2	31	2		4	202
Culicidae	1			3	2			15	338		1	1		6	1			48
Curculionidae	1	1							23	5				10				24
Dytiscidae	63	16	19	61	197	53	169	140	720	8	8	26	452	18	31	21	4	583
Ecnomidae									718						1			569
Gerridae	1				1	1			3									1

Glossiphoniidae						2		11	16	2	1		3	4	2	1		13
Gyrinidae		1						1	15						4			17
Haliplidae	6	1		5		5		2	21	1	1				1			7
Hemicorduliidae									19									3
Hydracarina	25	6	36	4	111	41	61	14	298	41	10	25	4	224	32	36	2	374
Hydraenidae									298									374
Hydrochidae	1	1	4	12			1	2	21	2	1	35	4	3	7	6	22	80
Hydrometridae				3					24						1			81
Hydrophilidae	6	16	8	27	20	21	5	17	123	6	28	30	4	54	23	5	12	163
Hydroptilidae						1	7		128						4	2		168
Hygrobiidae			1						9		1							7
Hyridae									1									1
Isostictidae													1					1
Leptoceridae			1	6	1	14			22	1		1	2	2				7
Libellulidae		13	7	11	3	38	4	3	101		20	3	9	2	23	4	21	88
Limnichidae									79						2			84
Lindeniidae					1			1	2		1							3
Lymnaeidae	1	2	6	9	3	17	5	4	49		9	2	9	25	36	6	10	98
Mesoveliidae			1	2	2	3	1	2	58	6		4	1	9	1			118
Naucoridae	6	9	9	19	10	15	34	7	120	8	20	4		28	25	11	1	118
Nematoda					1	2		2	114				2	2	1		16	118
Nepidae		1		2		1	1	6	16			2	2		4	1	2	32
Noteridae							1		12		1	1						13
Notonectidae		3		3	4	5	1	7	24				4		17		1	24
Oligochaeta	6		7	3	1			1	41	1	1	3	2	7	1	10	1	48
Orthocladiinae			1	1				1	21		1	1			2			30
Ostracoda		15		3	2	57	5	2	87		7	2	7	8	43	16	3	90
Planorbidae	15	8	32	224	16	50	131	145	705	31	138	15	53	92	80	9	105	609
Pleidae	5	33	17	64	5	18	15	14	792	9	11	6	12	13	14	42	51	681
Protoneuridae									171									158
Pyralidae												1		1				2
Richardsonianidae	1				1	3			5		17	6	2	1	1	1		30

Sciomyzidae								2	7								2	30
Scirtidae								2	4		1	1						4
Spaeriidae									2			2						4
Staphylinidae																		2
Stratiomyidae				1	7	2	3	3	16		2	2		7	1		4	16
Sundathelphusidae		2							18									16
Syrphidae				4	1				7									
Tabanidae	3		2	4		2	2	1	19		6	8	1	1	1		5	22
Tanypodinae	3	3	7	6	10	23	16	6	88	7	12	7	15	54	31	36	30	214
Tipulidae								1	75								1	193
Undiff HUL				1		2			4									1
Undiff. Zygoptera				3		1			7		1		1		1			3
Undiff. chironomid									4									3
Urothemistidae							2	8	10							1	1	2
Veliidae									10	8		1				1		12
Viviparidae	9	62		87		10		7	175	63			90	7			5	175
No. Individuals	263	341	297	690	827	616	666	1796	5496	251	619	253	846	881	684	442	424	4400
No. Taxa	24	24	22	33	29	34	27	36	52	24	31	31	29	33	37	26	29	54

Appendix C

Report
Caulders Lake – Feral Pig Baiting Trial

By
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Biosecurity Queensland
Dept Primary Industries and Fisheries

2nd October 2008

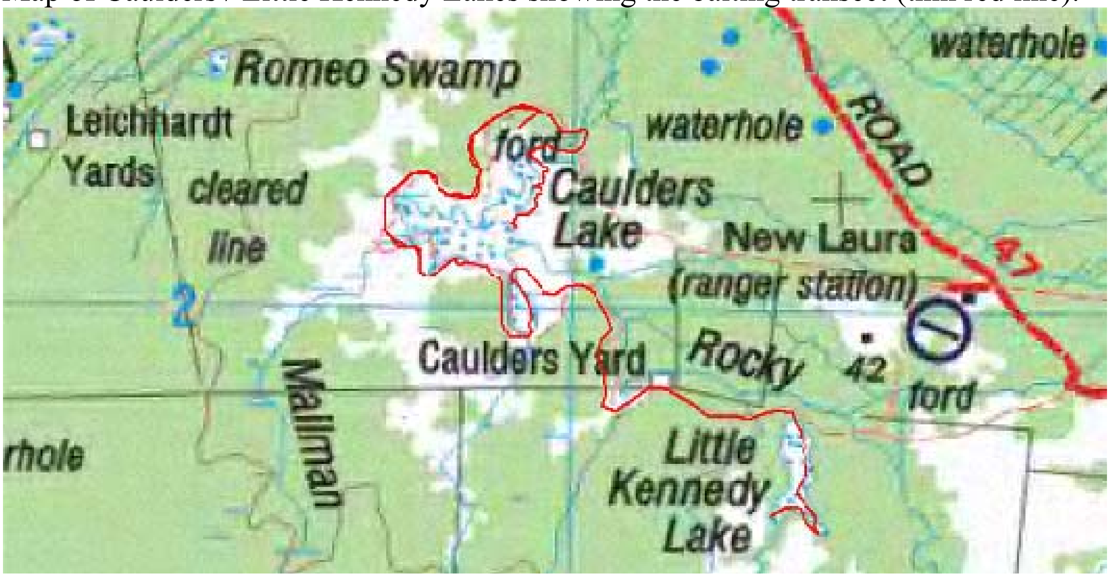
Introduction

As part of the DPI&F research project, "Ecological Impacts of Feral Pigs on Freshwater Ecosystems"; the Caulders Lake / Little Kennedy Lake complex are a designated pig population manipulation site where a relationship between pig impact and pig population level is being quantified. The pig population at this site is being manipulated by aerial shooting to try and maintain a relatively "low" pig population level. However the unanticipated very high pig population density around these lakes has meant that the maintenance of the population at a low level through aerial shooting alone has been difficult. An analysis of the aerial shooting data has calculated a population of 2179 pigs exist around these lakes. To reduce the pig population down to a more manageable level, to suit the aims of the research project, a combination of aerial shooting and ground baiting was considered the best strategy.

A feral pig baiting program was conducted around Caulders Lake – Little Kennedy Lake on the 10th October 2008. In total 400 kg of kangaroo meat (obtained from a licensed pet food abattoir) was used to prepare 850 baits (approx 500g each) which were injected with the standard 72 mg of 1080 (sodium fluoroacetate) by a DPI&F Land Protection Officer (Shane Ross).

Figure 1 shows the transect line where baits were deposited. Baits were hand placed every 50m to 100 m. along the available tracks. ATV's were also used to place baits around the lake perimeter and where pig pads were encountered, a line of baits were also distributed along the pad line. No baits were laid more then 1 km from the waters edge. No baits were deposited on the track between Caulders Lake and Little Kennedy Lake. The majority of baits were placed under trees, logs or in long grass to decrease the baits visibility to birds.

Figure 1.
Map of Caulders / Little Kennedy Lakes showing the baiting transect (thin red line).



An additional twenty two baits (placed 500 m apart) were also monitored with motion sensor cameras to detect and photograph visiting species. These baits were also impregnated with radio transmitters to enable baits that had been moved to be found (Figure 2). Baits were monitored daily for 5 days. These baits were not covered but exposed to the sky to simulate baits that had been dropped from an aircraft.

Figure 2.
Impregnation baits with radio transmitters (circled)

A pig population index was obtained prior to and 3 days after the baiting. The standard activity transect method was used in this survey – identical to the methods employed in the DPI&F project – "Ecological Impacts of Feral Pigs on Freshwater Ecosystems".

Forty five activity plots –10 m x 1 m, separated by at least 30 m were established on the lakes perimeter and cleared of pig sign. New pig activity (digging, tracks etc) on each plot were then assessed three days later. A presence or absence of pig sign on each plot derived a "Visitation rate" index. Digging on the plots was recorded to derive a "digging frequency" (proportion of available plots with diggings) and the length of digging on the plots 10m centre line derived the digging index (proportion of length of plot disturbed by diggings).

Figure 3.
Activity transect showing pig activity – diggings and tracks.

Results.

The results were a combination of a successful aerial shooting program conducted some 2 weeks prior to the baiting and the ground baiting program. The average activity indices and water parameters values obtained from the previous July survey were compared to the September post baiting indices. On average, activity signs decreased 91.6 %. Sightings also decreased by 59%.

Activity Index	July 08	September 08	% Change
Visitation rate (%)	86.7	17.8	- 79.5
Digging Frequency	53.3	2.2	- 95.8
Digging %	16.8	0.1	- 99.4

Water parameters

Turbidity (NTU) *	409.7	74.4

* Note - Turbidity instrument was only calibrated to a maximum of 500 NTU

Ammonia (fluoro index)	220.1	158.5
Chlorophyll (fluoro index)	403.9	421.8
TDS	139.2	461.7
pH	6.0	4.9

Turbidity was markedly improved after the control programs. Ammonia was also lower possibly due to less faeces and urine deposited by pigs. Chlorophyll was higher due to increased light penetration and Total Dissolved Solids (TDS) was higher due to the normal evaporation of water concentrating the salt solids.

Eight fresh feral pig carcasses were found 2 days after baiting. Nine live pigs were also seen with 2 showing obvious signs of poisoning. On day 5 post- baiting only 2 live pigs was observed during the survey. A dead hawk was also found 2 days post baiting. The carcass has been sent to Brisbane for analysis. No other non-target carcasses were found although an extensive search was conducted each day.

For the monitored baits, only 1 was taken after 5 days post baiting. Unfortunately no photo of the visiting animal was available; however from prior experience this suggest a raptor was probably responsible as they sometimes take the bait on the wing. The transmitter was found 100m away – no sign of the bait material was found.

Photographed visits from other species included: (Figure 4)
Hawk – 3 visits with only 1 bait pecked at
Crows – 2 visits no bait material eaten
Goanna – 1 visit with 1/8 of bait material eaten.
Dingo – 1 visit – bait not taken but regurgitated bait material was found at the site which suggests poisoning.
Cat - 1 visit, no bait material eaten
Stork or Ibis – curiosity visit
Horse and Cattle – curiosity visits.

Figure 4
Non-target species at bait stations

Summary

The combination of aerial shooting followed up by a baiting program has had a major effect on the feral pig population around Caulders / Little Kennedy Lake. A 92% decrease in pig activity was recorded since the July survey period. The lake is visually very changed from the July survey; the remaining water is clear without the turbid plumes of mud from pig activity. This is in stark contrast from the October survey in 2007. Pig diggings on the waters edge are greatly reduced and green grass is now growing throughout the lake bottom (Figure 5). There is evidence that the past aerial shooting programs have had a major effect throughout the year with 10m wide strips of relatively low diggings surrounding the lake (Figure 6). I believe these strips occur as the water level receded in a period of low pig activity following a shooting.

The baiting program appeared to be very target selective. Only one non-target carcass was found – the cause of death is yet to be determined although poisoning is suspected. A rough count of carnivorous birds observed each survey day was kept. No decrease in bird populations were observed – raptors, crows, butcher birds etc were still plentiful 5 days after the baiting. Two hawks were observed eating baits on the second day post baiting, no carcasses were subsequently found.

The method of semi hiding the baits reduced the visibility to birds. More selective techniques such as burying baits or rolling them in the dirt to obscure the meat baits red colour would be a more target selective technique employed if bird take was considered a problem. I consider this baiting program had a negligible impact on non-target species.

After 5 days the remaining baits were dehydrated, hard and black. This will make the baits increasingly more target specific as black baits are difficult for birds to find. Past research has shown a marked drop off of bird take after 2 days as the baits turn black. The remaining baits will still be attractive to pigs so a continuation of the poisoning success is possible as immigration of pigs from surrounding areas occur. The coming wet season will remove the 1080 toxin (which is biodegradable) from the bait material.

The monitored baits had a zero pig uptake. However, I believe this was mainly due to these baits being placed some distance from the water on the vehicle track that surrounded the lakes. The baits placed closer to the water would have a significantly larger pig encounter rate and I believe this reduced the pig population to such a very low level that no pigs survived to encounter the monitored baits placed some distance from the water. No pig tracks or signs were encounter on the vehicle tracks; simply no pigs were present away from the water.

Overall I believe the combination of aerial shooting and ground baiting has been very successful in this situation. I recommend that further aerial shooting and ground baiting be implemented immediately that access to Lake Caulders is possible in 2009. Past surveys indicate that pig activity at these lakes at this time is very high due to the abundance of food (water lilies) which are readily accessible due to the shallow water around the lake perimeter when the lake is full.

Figure 5.
Illustrates the lack of recent diggings occurring on the waters edge. Green grass is growing throughout the lake and the water is clear and blue.

Figure 6.
Strips of little digging activity occur around the lakes due to past aerial shooting programs