Habitat and biodiversity of On-Farm Water Storages: A Case Study in Southeast Queensland, Australia

Author

Markwell, Kim, Fellows, Christy

Published

2008

Journal Title

Environmental Management

DOI

10.1007/s00267-007-9037-7

Rights statement

Downloaded from

Griffith Research Online

Habitat and biodiversity of on-farm water storages: A case study in South-East Queensland, Australia

(Shortened title: Habitat and biodiversity of on-farm water storages)

Kim A. Markwell, Christine S. Fellows*

Australian Rivers Institute, Griffith School of Environment, Nathan, QLD 4111, Australia.

*Corresponding author: email. *c.fellows@griffith.edu.au*, Ph. +61 7 3735 3840, Fax. +61 7 3735 7615

Abstract

On-farm water storages (locally known as farm dams or farm ponds) are an important part of many agricultural landscapes as they provide a reliable source of water for irrigation and stock. Although these waterbodies are artificially constructed and morphologically simple, there is increasing interest in their potential role as habitat for native flora and fauna. In this paper, we present results from a case study which examined the habitat characteristics (such as water physical and chemical parameters, benthic metabolism, and macrophyte cover) and the macrophyte and macroinvertebrate biodiversity of eight farm ponds on four properties in the Stanley Catchment, South-East Queensland, Australia. Each landowner was interviewed to allow a comparison of the management of the ponds with measured habitat and biodiversity characteristics, and to understand landowners' motivations in making farm pond management decisions.

The physical and chemical water characteristics of the study ponds were comparable to the limited number of Australian farm ponds described in published literature. Littoral zones supported forty-five macroinvertebrate families, with most belonging to the orders Hemiptera, Coleoptera, Odonata and Diptera. Invertebrate community composition was strongly influenced by littoral zone macrophyte structure, with significant differences between ponds with high macrophyte cover compared to those with bare littoral zones. The importance of littoral zone macrophytes was also suggested by a significant positive relationship between invertebrate taxonomic richness and macrophyte cover.

The landowners in this study demonstrated sound ecological knowledge of their farm ponds, but many had not previously acknowledged them as having high habitat value for native flora and fauna. If managed for aquatic organisms as well as reliable water sources, these artificial habitats may help to maintain regional biodiversity, particularly given the large number of farm ponds across the landscape.

Key words: Farm pond, aquatic invertebrates, macrophytes, management, agriculture

Much of the land available for terrestrial habitat is now used for agriculture, contributing to biodiversity loss through its conversion of complex natural systems to simplified managed systems (Altieri 1999; Tscharntke and others 2005; Reidsma and others 2006). Agricultural activities have therefore usually conflicted with nature conservation (Tress 2002). However, recent research suggests that appropriate agricultural management can enhance biodiversity in these modified environments, contributing to conservation efforts (Tscharntke and others 2005). Most of this research has been based in terrestrial environments, although waterbodies (dams, ponds and wetlands) in agricultural areas have been found to have the potential to sustain both local and regional biodiversity in the few studies that have focused on these environments (Williams and others 2003; Hazell and others 2004; Robson and Clay 2005).

Farm ponds are now part of many agricultural landscapes, especially where there is a need for reliable water supply, such as Australia (Casanova and others 1997; Brock and others 1999; Hazell and others 2004). Construction of ponds with the goal of water reliability creates water bodies that are usually fairly small, deep and steep-sided and therefore contain less shallow edge habitat than natural water bodies (Brock and others 1999; Hazell and others 2004). Nevertheless, a range of aquatic organisms have been found in ponds, especially those with greater habitat complexity due to the presence of littoral zone plants (Timms 1980; Frankenberg 1998; Hazell and others 2001; Nicolet and others

2004). In addition to morphological characteristics, the ability of ponds to support aquatic organisms depends on water quality and ecosystem processes that support food webs (i.e. primary production) (Platt and Corrick 1994; Brainwood and others 2004). If farm ponds are able to provide suitable habitat for native flora and fauna, they may play an important role in the conservation of native biodiversity in agricultural landscapes in which natural water bodies, such as wetlands, typically have been degraded or lost (Casanova and others 1997; Hazell and others 2004).

In the agricultural landscape, it is also important to look at the influence of property management on the biodiversity and potential conservation value of farm ponds. There have been a limited number of studies addressing these issues. Hazell and others (2001; 2004) acknowledged that farm ponds might help conserve frogs across the landscape, and that landowners had the capacity to increase the conservation value of ponds through management. It is important to recognise that farm management decisions are complex, with many underlying determinants influencing farmers' attitudes and motivations towards farm management decisions (Rickson and others 1987; Beedell and Rehman 1999; Lisson and others 2003). Rickson and others (1987) also documented that the landowner's perception of the conservation value of their land is also important when making these management decisions and have evaluated farmers' perceptions of erosion on their properties, comparing measured erosion to perceived erosion as estimated by landowners. To-date, this has not been done to measure landowners perceptions of the conservation value of their farm ponds.

There is currently a lack of information on the potential of farm ponds to support native biodiversity and how landowner's farm management strategies may influence this biodiversity. This case study aimed to discover if farm ponds in South-East Queensland could support native flora and fauna and to uncover if management of these storages could increase their biodiversity value and how landowners might be motivated to undertake these strategies. The first objective of the study was to assess the biodiversity and habitat value of on-farm water storages on cattle properties in South-East Queensland. We also sought to examine the factors that influence habitat, biodiversity, and the links between them, such that we could make recommendations on how these systems might be managed to increase their ecological value. Finally, we interviewed landowners to examine their motivations to manage their farm ponds as well as their perception of their water storages as habitat for native species. Combining ecological science with social science in this case study was undertaken with the aim of understanding the scope for change in management practices towards increasing the conservation value of farm ponds.

Methods

Study area

The potential conservation value of farm ponds in South-East Queensland, Australia was evaluated by sampling a number of ponds in the Stanley River catchment (Figure 1). The catchments total area is 1,527 km^2, which includes 1,045 km of stream length. Water supply is the dominant land use as two major water supply dams, Wivenhoe Dam and Somerset Dam, are located in this catchment. The other predominant land use is rural, with beef and dairying encompassing most of the central and lower sections of the catchment (Waterways 2002; QNRM 2003). The upper catchment is relatively undisturbed, dominated by natural forests with some forestry plantations (Waterways 2002; QNRM 2003).

The South-East Queensland region has a sub-tropical climate with annual rainfall dominated by the summer months. In the Stanley River catchment, summer months (December to February) have an average maximum temperature of 30^{o}C, while during winter (June to August), minimum temperatures may drop under 9^{o}C (Loi and Malcolm 1998; BOM 2005b). Average rainfall for the catchment is around 1350 mm/year. In Kilcoy (situated in the western plains of the catchment), the summer rainfall (Dec to Feb) averages 115-140 mm/month, while in the drier winter months (May-Sept), monthly rainfall was less than 60 mm for the period 1890-1993 (Loi and Malcolm 1998). This variability, combined with high evaporation rates, increases demand for water supply in winter and has led to the development of 323 water storages in the catchment that capture overland flow (QNRM 2003). These storages have a combined volume of 793 megalitres with most (95%) having a surface area of $\leq$ 5000 m^2 and a volume of less than 5 megalitres.

Farm pond sampling approach

Landowners interested in participating in this case study were found via the Brisbane Valley-Kilcoy Landcare group. Four landowners (A, B, C and D) in the Stanley River catchment were selected for this study due to their willingness to partake in interviews and the presence of multiple farm ponds on their property. A preliminary visit was carried out to each property to select 2 ponds to use in this study, based on size and surrounding land use. Two ponds on each property were chosen (1 and 2) and all eight ponds were fairly similar in size and were surrounded by land used for cattle grazing. Sampling was carried out during July 2004.

Habitat characteristics included measurement of pond morphology, water physical and chemical characteristics, benthic metabolism and littoral zone structural complexity (as measured by macrophyte cover). Ponds were surveyed for perimeter, surface area and littoral zone slope (as measured by the change in depth (cm)/distance from the shore (cm)). Dissolved oxygen and temperature were recorded for 24 hours by a TPS logger (WP-82Y) at 2 depths (<5cm and >70cm). Photic depth was defined as the depth where one percent incident light remained (Wetzel and Likens 1991) and was measured by a depth profile taken at midday with a LICOR quantum sensor (LI-192) and data logger (LI-1400). Spot measures of conductivity, pH, and turbidity were also recorded. Depth integrated water samples were collected at each of the ponds at the time of sampling and analysed for nutrient concentrations (total nitrogen (TN), total phosphorus (TP), filterable reactive phosporus (FRP), ammonium nitrogen (NH_4^+), nitrogen oxides (NO_3^-)). Soluble nutrients were measured on an automated LACHAT 8000QC flow injection system using the following methods: ascorbic acid reduction of phosphomolybdate for FRP; cadmium reduction of nitrate to nitrite by diazotizing the nitrite with sulfanilamide and coupling with N- (1-naphthyl) ethylenediamine dihydrochloride for NO_3^- and NO_2^-; production of the indophenol blue colour complex for NH_4^+ (Greenberg and others 1992). Total nutrient samples were digested using a persulfate digestion procedure, after which analyses were performed as described above for soluble nutrients (Greenberg and others, 1992). Surface waters typically have very little nitrite, and therefore nitrogen oxides will be referred to as nitrate (NO_3^-). Chlorophyll *a* was measured as a relative measure of phytoplankton biomass. Water column samples of known volume were filtered using glass fibre filters (0.7µm), and absorbance was measured on a spectrophotometer (Shimadzu Model 1601) following extraction of the filter in a 90% alkaline acetone solution. Samples were acidified for phaeophytin correction (Wetzel and Likens 1991; Goldsborough 2001). Additional information about the ponds (for example, pond age) was obtained from each of the landowners.

Sampling within the ponds was focused on the littoral zones for benthic metabolism, vegetation and biota, due to sampling accessibility and the known high diversity and productivity of these areas (Brock and others 1999). Littoral zones were defined in this study as the submerged area from the water's edge to one metre horizontally out from the shoreline. Benthic metabolism was measured with in-situ recirculating clear perspex chambers with an attached oxygen probe (Fellows and others 2006). Chambers were deployed in the littoral zone sediment between the hours of 9:30 am and 5:00 pm. A data logger (TPS model WP-82Y) recorded measurements of the dissolved oxygen and temperature of the water in the chamber every ten minutes. After at least two hours of midday sunlight, the chambers were covered with opaque, reflective material to make them completely dark to measure the rate of respiration (Fellows and others 2001). The rate of change of DO over time was used to calculate the combined rates of oxygen consumption and production (net ecosystem production (NEP)) during the daylight period, and the rate of oxygen uptake (respiration (R)) during the darkened period. Gross primary production (GPP) was calculated as the difference between the two rates. The hourly rates were extrapolated to represent daily rates in order to calculate GPP/R ratios. GPP rates were multiplied by the average daylight hours for July in South-East Queensland (seven hours; BOM 2005a), and respiration rates were multiplied by 24 hours.

For vegetation and macroinvertebrate sampling, the major habitats present in the pond littoral zones were identified (eg. dense macrophytes, bare) and composite samples were taken from three 20 m x 1 m transects. Along each transect, all macrophyte species were identified and densities visually estimated as percent cover of emergent, floating and submerged macrophytes. These were recorded separately and then added to obtain an overall percentage cover. Transects were also sampled for macroinvertebrates with a D-framed dip net. The net contents were drained of excess water and 100% ethanol was added resulting in a final concentration of approximately 70% (Cummins and Merritt 2001). Sorting was conducted in the laboratory after washing through three sieves – 1 cm, 1000 μm and 250 μm. The three fractions retained on the sieves were sorted by eye in the laboratory and then identified to the greatest resolution possible with the use of a dissecting microscope. Invertebrate data is presented in this paper at the family level so comparisons could be made with data collected from nearby streams by the South-East Queensland environmental health monitoring program (EHMP). Invertebrate data collected 2002-2003 from 10 streams located near the study ponds was used for this comparison. For all taxonomic groups, introduced (non-native) taxa were identified and recorded.

Landowner interviews

Interviews were organised with each of the landowners to gain knowledge about the use and management of the ponds, as well as the ecological knowledge the landowner had regarding their farm ponds. These interviews were structured, consisting of a set of pre-established questions in a questionnaire format, enabling comparisons between landowner responses. Landowners were asked to rank factors that may influence their decisions on farm pond management on a scale of importance from one to ten (10 = most important). Landowner knowledge of habitat and biodiversity values of their farm ponds was examined by asking them to predict the biodiversity of their storages given the following categories: poor, fair, good, very good (ranked from 1 = poor to 4 = very good). These closed questions which asked the participant to rank the strength of their opinion were usually followed by an open question to provide more information on the strength of these attitudes and to allow the participant to address alternatives that were not included in the previous closed question (Schuman and Presser 1981; Babbie 1990, Nichols 1991). All questionnaires were conducted by a common interviewer (KM), ensuring a standardised approach to data collection. Because the interviewer was present while the questionnaire was completed, any ambiguous questions were carefully explained and all questions were answered. The interviewer was also able to further explore answers that lacked precision (Bentham 1982). In summary, using a questionnaire containing mostly closed questions and administering it personally reduced the potential error and variability of the data collected in these interviews (Sudman and Bradburn 1974; Schuman and Presser 1981; Denzin and Lincoln 2003).

Data Analysis

Taxonomic richness is presented both as the total taxonomic richness of the pond (the total number of taxa found in all transects) and as a transect mean value with standard error. Richness and diversity measures are presented and analysed at the species level for macrophytes and at the family level for invertebrates. Shannon diversity indices have been used as biodiversity measures, and are reported as the mean value with associated standard error. Tests for normality revealed that much of the data did not have normal distributions, even after transformations. Therefore nonparametric univariate statistics were chosen and the analyses were carried out in SAS Version 8 (SAS 1999), using a significance level of $p<0.05$. Nonparametric ANOVAs (Kruskal Wallis tests using Wilcoxon scores) were used to investigate differences among the ponds in benthic metabolism, and taxomonic richness and biodiversity of macroinvertebrates (family level) and macrophytes (species level) as well as macrophyte % cover. Site (pond) was used as the main factor (eight levels) and multiple measurements within each site were used as replicates. Bonferroni (Dunn) t-tests were further used to identify statistical differences between ponds ($p<0.05$). Spearman’s rank correlation was used to investigate relationships between habitat characteristics, such as turbidity, photic depth, macrophyte species richness and macrophyte abundance, as well as between habitat characteristics and measured aquatic biodiversity, such as littoral zone % cover and invertebrate richness. The relationship between the total aquatic diversity of the pond (macrophyte + invertebrate richness) and the age and size of the pond was also investigated.

Differences among invertebrate community composition of the ponds were explored using multivariate analyses based on abundance data. A stress of 0.2 or less was considered low stress, and therefore supportive of a strong pattern. Analysis of Similarity (ANOSIM), using the Bray Curtis association measure, was performed on abundance data using PRIMER. SIMPER analyses were used to identify which taxonomic groups were responsible for observed differences in community structure.

Landowner interview questions which required the landowner to rank their responses provided quantitative results which could be analysed. These values were averaged across all landowners and are presented as the mean value with its associated standard error. The relationship between the landowners' predictions of the biodiversity of their ponds and the measured total aquatic biodiversity was tested using Spearman's rank correlation.

Results

Habitat characteristics

The ponds differed in morphology and history, but had physical and chemical characteristics which could support aquatic organisms. The study ponds ranged from 282 m^2 to 1259 m^2 in surface area with perimeters between 70 m and 180 m (Table 1). Littoral zone slope also varied across the study ponds. The age of the ponds ranged from 2 to 65 years since construction. Only 3 of the ponds had any structural farm pond management strategies in place. A1 and B1 both had vegetation planted near them and D2 had fencing around it. Dissolved oxygen concentrations in the farm ponds did not drop below critically low levels at night, as demonstrated by 3.05 mg L^{-1} being the lowest dissolved oxygen reading recorded (deep water, pond A1). There were generally moderate swings in both temperature and dissolved oxygen over 24 hours. Shallow water diel temperature swings were larger in magnitude than the deeper water due to warming of the shallow water layer during the day. The changes in dissolved oxygen over 24 hours were generally similar in the shallow and deep water. All ponds had circumneutral pH (5.73 in C2 to 7.75 in B2) and had quite high light penetration with photic depths over 1m, except for pond C1.

Although nutrient levels were reasonably low across the ponds, they did not seem to be at levels which would limit primary production. C1 had much a much higher concentration of TP (0.14 mg P L^{-1}) than the other ponds (0.014 to 0.056 mg P L^{-1}) and dissolved inorganic nitrogen levels in D1 (NH_4^+ =

0.8, NO_3^- = 0.064 mg N L^{-1}) were also substantially higher than those measured in the other ponds (NH_4^+ = <0.002 to 0.029, NO_3^- = <0.002 to 0.019 mg N L^{-1}).

Gross primary production (GPP) rates ranged from 39 mg O_2 m^{-2} h^{-1} to 247 mg O_2 m^{-2} h^{-1}. Hourly respiration rates (R) were between 50 mg O_2 m^{-2} h^{-1} and 252 mg O_2 m^{-2} h^{-1}. There were significant differences in benthic GPP, NEP and R hourly rates among the study ponds (Kruskal Wallis, p = 0.013, 0.023 and 0.021, respectively). Pairwise comparisons showed that D1 had significantly higher GPP than A1, A2 and B2. C1 and C2 also had significantly higher GPP than A1 and B2. B1 had significantly higher benthic respiration than all of the other study ponds. All ponds were heterotrophic, with GPP/R ratios well below one (0.12 to 0.53) and these ratios did not differ significantly across the study ponds (Kruskal Wallis, p =0.053).

There was a total of 16 macrophyte species found in the farm pond littoral zones with individual ponds having between 1 to 10 species present (Table 2, Appendix 1). The abundance of macrophytes (measured as total % cover) ranged from 2 to 108 %. C1, D1 and D2 had significantly lower macrophyte cover than A1, B1, B2 and C2 (Kruskal Wallis, p = 0.004). The dominance of each structural type (emergent, floating and submerged) also differed across the waterbodies, resulting in the 8 study ponds encompassing a large range of habitat complexity. Photic depth and turbidity did not have a significant relationship with total macrophyte species richness (Spearman's rank coefficients 0.386 and -0.422 respectively) or macrophyte abundance (Spearman's rank coefficients 0.476 and -0.595 respectively).

Littoral Zone Biodiversity

Of the sixteen macrophyte species recorded in the study ponds, only four of these were introduced species (*Nymphaea caerulea, Callitriche stagnalis, Cabomba caroliniana* and *Urochloa mutica*). The most common plants (found in at least half of the ponds) were the native species *Juncus usitatus, Schoenoplectus mucronatus* and *Ludwigia pepliodes* ssp. *montevidensis* and the introduced species *Urochloa mutica*. There was a significant difference for both macrophyte species richness and diversity among the eight study ponds (Kruskal Wallis, p = 0.006 and p = 0.005, respectively). The highest species richness recorded in the farm ponds was ten in pond B1 (Table 2). Pairwise comparisons showed that this pond contained significantly more macrophyte species than C1, D1 and D2 which contained only one or two different species. Pairwise comparisons also grouped the ponds into two separate groups based on their macrophyte diversity. A1, A2, B1, B2 and C2 were all more diverse than C1, D1 and D2. During the sampling, it was also visibly noticeable that A1, A2, B1, B2 and C2 had higher % littoral zone macrophyte cover (>45%) while C1, D1 and D2 all had very low percentage cover (<12%). These two distinct groupings of ponds are referred to as well vegetated and poorly vegetated in subsequent analyses which consider % macrophyte cover as being a measure of littoral zone habitat structure.

Fourteen invertebrate orders and forty-five families were found across the eight study ponds (Table 3; Appendix 1). Most families were representatives of the orders Hemiptera, Coleoptera, Odonata and Diptera. There was a significant difference among ponds when comparing both invertebrate family richness and family diversity (Kruskal Wallis, p = 0.004 and p = 0.007, respectively). B1 had the highest total family richness and average family richness (27 and 19.7 respectively) while D1 had the lowest total richness value of 15 and A2 had the lowest average family richness of 10.3. Pairwise comparisons (based on replicate data) showed that B1 had significantly greater invertebrate family taxonomic groups than D1, C1 and A2. Pairwise comparisons also showed that C2 and A1 were significantly more diverse than A2, D1 and B2. The number of aquatic invertebrate families occurring in the littoral zones of the study ponds was significantly related to the percentage macrophyte cover (Spearman’s rank coefficient 0.671, p<0.001) (Figure 2a).

The community structure of invertebrates among the eight study ponds was significantly different based on multivariate analysis of abundance data (ANOSIM, Global R = 0.951, p = 0.001; Figure 3a). Invertebrate community structure was significantly different between ponds with poorly and well vegetated littoral zones (ANOSIM, Global R = 0.492, p = 0.001; Figure 3b). SIMPER analysis revealed that this grouping was due to the presence of two Chironomid subfamilies, chironominae and tanypodina, and the Corixididae family in poorly vegetated ponds and the Atyidae family in well vegetated ponds. These four family groups cumulatively contributed 73% to the dissimilarity in invertebrate community structure in the differently vegetated ponds.

The highest total number of taxa (macrophytes + invertebrates) was recorded at B1 (37 different taxa). D1 supported the lowest total number of taxa, with only 18 different taxa recorded. Total species richness did not exhibit significant relationships with pond age or area (Spearman's rank coefficients 0.188 and -0.048 respectively) (Figures 2b and c).

Landowner knowledge and farm pond management

Cattle have regular access (ranging from unrestricted to access one week/month) to all but one of the study ponds (D2), which is permanently fenced due to its steep slopes. Two other ponds (B1 and A1) have had riparian vegetation successfully planted around or near the pond. These are the only major management strategies in place for any of the study ponds (Table 1).

Not surprisingly, all of the landowners recognised the importance of the farm ponds as reliable water sources, with this role being ranked as the most important factor (average rank of 9/10) to consider when making management decisions. Habitat and nature conservation were the next highest ranked factors (ranked 7 and 5 out of 10 respectively on average). Other factors such as financial and policy requirements were not ranked by any of the landowners as important when making management decisions regarding the ponds.

The landowners were able to correctly identify which of their ponds was most diverse, and also fairly accurately predicted how diverse their ponds were on a scale of fair to very good. For the three ponds with the lowest measured biodiversity (C1, D1 and D2), the landowners correctly predicted that these ponds only had 'fair' biodiversity. The three ponds that were predicted to have 'very good' biodiversity were the three ponds for which the highest biodiversity was measured. There was a highly significant correlation between the biodiversity predicted by the landowners and the biodiversity measured in the ponds (Spearman correlation coefficient = 0.856; p = 0.007) (Figure 4).

Discussion

Farm ponds as aquatic habitat

This case study clearly shows that farm ponds in the Stanley River catchment, South-East Queensland can support native macrophytes and invertebrates. Our results are similar to findings from temporary ponds located in grazed areas of Great Britain that support both rare and common macrophytes and invertebrates (Nicolet and others 2004). Further, farm ponds in Japan have been shown to be important habitats for rare aquatic plants and Odonates (Takasaki 1994 and Kadono 1998, as cited in Maezono and Miyashita 2004). Together these studies and our results clearly demonstrate the potential importance of managed waterbodies in nature conservation.

Although no rare or uncommon species were found in the farm ponds, many of the invertebrate families sampled in the farm ponds have been recorded in nearby streams by the South-East Queensland Environmental Health Monitoring Program (EHMP) in 2002/2003 (Table 3). Even though the farm ponds would be assumed to support habitat conditions very different to streams, highly variable rainfall in South-East Queensland results in many streams have very low or even no flow during the dry season, resulting in dominance of pool environments (Smith and others 2001). This could explain why many of the following orders had almost all of the same families recorded in the farm ponds and the EHMP sites: Acarina, Hirudinea, Bivalvia, Decapoda, Ephemoptera, Hemiptera, and Lepidoptera. The average family richness (number of families recorded at each site) across all ten EHMP sites was 28 (ranging from 18 to 34 over the two sampling years) while the farm ponds averaged 21 invertebrate families/pond (values between 15 and 27). Considering only three of the farm ponds had any management strategies in place, these values are quite promising when considering farm ponds as possible habitats for fauna.

It is generally argued that aquatic richness and biodiversity are positively related to either the size of the water body (MacArthur and Wilson 1967; Allen and others 1999; Hansson and others 2005) or its habitat complexity (Zimmerman and Bierregaard 1986, as cited in Hazell and others 2001; Bunn and Arthington 2002; Statzner and Moss 2004). The results from this study suggest that the presence and complexity of littoral zone macrophyte habitats within the study ponds was more important than area in influencing aquatic invertebrate richness. This is supported by other studies that have found that the presence of invertebrates in lentic aquatic systems is often closely linked with habitat complexity, such as the presence of macrophytes (Timms 1980; Murkin and Ross 2000; Nicolet and others 2004). Further tests of the role of pond size would require both a broader range in sizes and some control over pond age as this can also influence the species richness of benthic invertebrates (Hansson and others 2005). Age was not significantly related to aquatic taxonomic richness in this study due to ponds like B1, which was one of the youngest ponds, but supported the highest number of invertebrate families.

The presence of aquatic vegetation also had an effect on the composition of the invertebrate communities. Ponds with poorly vegetated littoral zones had communities that were dominated by Corixidae and two Chironomid subfamilies (chironominae and tanypodina), which are common in artificial freshwater habitats such as farm and aquaculture ponds (Timms 1980; Ingram and others 1997). The presence of Atyidae also contributed to the difference between the invertebrate communities of the well vegetated and poorly vegetated ponds. These are detritus feeders which are commonly found in the littoral environments of farm ponds and aquaculture ponds (Ingram and others 1997). Although there was a difference in the community compositions between ponds with heavily vegetated ponds and those with bare littoral zones, most of families present in both types were common and tolerant organisms.

The factors responsible for the establishment of macrophytes in farm ponds are not well known (Casanova and others 1997), but the environmental conditions needed to support their growth have been studied in natural lakes and wetlands (Perkins and Underwood 2002; Herb and Stefan 2003, Hietala and others 2004). These include relatively clear water and adequate nutrient concentrations;

although if the threshold nutrient levels are exceeded, high phytoplankton and epiphyte density can shade out macrophytes (Scheffer and others 1993). Inorganic/sediment turbidity can also serve to limit light penetration (Scheffer and others 1993; Declerck and others 2006). Although high turbidity and associated shallow photic depths are known to influence macrophyte growth due to limitation of light for photosynthesis, there was no significant relationship found between turbidity and macrophyte richness or abundance in this study. This may be due to all but one pond (C1) having photic depths greater than one metre (which was generally the greatest depth recorded in the littoral zones of any of the ponds), thereby not limiting plant growth in these areas.

The abundance and richness of aquatic vegetation in farm ponds is also likely to be affected by the pond's age, stock access and substrate type (Timms 1980; Hansson and others 2005; Declerck and others 2006). Older ponds are likely to have more complex littoral zone habitats, although this was not observed in this study, as one of the newest ponds supported the most diverse macrophyte community (B1). Stock access was only restricted to one of the study ponds (D2), which actually had one of the least rich macrophyte communities. These factors alone could not explain the absence of complex macrophyte habitats in some of the study ponds. It is likely that a complex interaction of these factors affects the establishment and survival of macrophytes in farm pond littoral zones (Herb and Stefan 2003), and looking across the 8 study ponds, these interactions were not able to be resolved definitively. However, in terms of suggestions for management practices, it is generally recognised that likelihood of macrophyte occurrence in farm ponds can be enhanced by managing stock access and reducing nutrient inputs (Declerck and others 2006). Direct planting of macrophytes in the littoral zone may also be useful method if the water quality of the ponds are adequate to support them (Frankenberg 1998).

Water quality

Water quality can exert a strong influence on habitats, ecosystem functioning and the biota of aquatic systems (Platt and Corrick 1994; Brainwood and others 2004; EHMP 2004). Overall, the physical and chemical characteristics of the water in the study ponds are of better quality than the limited number of studies examining Australian farm ponds (Hazell and others 2001; Brainwood and others 2004) (Table 4). The study ponds should provide better submerged macrophyte habitat than the three farm ponds in the Central NSW Tablelands studied by Brainwood and others (2004) which had higher levels of nutrients, turbidity and chlorophyll *a*. The ranges of values across the eight study ponds for temperature, dissolved oxygen and pH were smaller than those observed in other farm ponds studied in Australia. This is likely due to the smaller sample size of this study when compared to Hazell and others (2001) and the fact that the study ponds were only sampled once compared to Brainwood and others (2004) in which sampling was conducted monthly over a year.

Since all of the properties were in agricultural areas, and the properties themselves all had cattle, turbidity and nutrient levels were expected to be high (Timms 1980; Garnier and others 2000). The water clarity in the study ponds was quite surprising, as stock usage is likely to increase turbidity due to trampling. Overall, the ponds had fairly similar nutrient concentrations except D1, which had high levels of dissolved forms of nitrogen and C1 which had the highest TP concentration. Both of these ponds have unrestricted stock access, low macrophyte cover and limited riparian zone vegetation (only short grass and bare ground), which may explain these high nutrient levels since there are no buffers in place (Declerck and others 2006). However, even these high concentrations are low compared the findings of Brainwood and others (2004) for three ponds surrounded by more intensive land use including pasture, a pet food production factory, and several smaller industrial plants and a grain store.

GPP rates in this study are high enough to suggest benthic algae could serve as an important food source. However, the GPP/R ratios in all of the farm ponds were well below 1, suggesting that algal production alone is not sufficient to fuel the observed levels of respiration (Hanson and others 2003). This supports statements made by Robinson and others (2000) who state that the decomposition of macrophyte detritus may primarily fuel food webs in wetland environments. However, several studies using stable isotopes have shown that even in littoral zones dominated by macropytes, attached algae can still represent the major carbon source in the aquatic food web (Hecky and Hesslein 1995; Hadwen and Bunn 2005). Additional research characterising both potential food sources such as benthic production, and foodweb structure in farm pond littoral zones over time would provide a clearer picture how these ecosystems operate.

Managing farm ponds as habitats

In areas of cattle grazing, managing stock access to farm ponds should increase aquatic habitat value. Cattle utilising farm ponds are likely to degrade the quality of littoral habitats due to their weight, frequent return to the same site to drink, and by grazing on emergent plants (Lloyd and others 1998). They also cause erosion by trampling and removing palatable riparian plant species, and their excreta can cause nutrient enrichment (Platt and Corrick 1994; Frankenberg 1998; Declerck and others 2006). Fencing off ponds to restrict cattle access is one suitable management strategy. Vegetated buffer zones near the ponds are also beneficial to water bodies, protecting the system from pollutants as well as providing terrestrial habitat (Platt and Corrick 1994; Nicolet and others 2004; Declerck and others 2006). Appropriate management may minimise impacts to water quality and macrophytes and ensure that farm ponds function as a suitable aquatic habitats (Brainwood and others 2004).

Even though the landowners in this study had a good understanding of the biodiversity of their ponds and rated conservation as a high factor in determining decisions regarding on-farm management, only three of the eight study ponds had management strategies in place. Only two of these ponds have actually had management strategies implemented to increase biodiversity (A1 and B1). Both of these landowners have strong conservation values as shown by the fact that they both ranked habitat and/or nature conservation factors “ten” (out of ten). Landowner B stated that “restoring (the) environment to its original condition and to see the effect of riparian restoration on the health of the pond” is important, and landowner A recognised the “need for more wetland habitats”.

Results from this study suggest that strong conservation values may not necessarily lead to the implementation of farm pond management strategies. Even though landowner C places high value on habitat (ranked nine out of ten), he did not recognise farm ponds as important habitats or the potential influence of management strategies could have on them.

> “Most ponds won't be like natural habitat. They are based on clay, not topsoil [therefore the water is not clear]…..They eventually get silted up and then plants can grow in them.”

This landowner has instead concentrated on restoration efforts along the creek that runs through his property. Perhaps increasing this landowners awareness of the potential of farm ponds to provide habitat would result in the implementation of management strategies on these waterbodies, as Tenge and others (2004) found that increased awareness about soil erosion positively influenced landowners adoption of conservation measures.

Findings in this study also suggest that factors other than conservation values, such as livestock care, can lead to management decisions that may inadvertently promote farm pond biodiversity. Landowner D did not rank habitat or nature conservation as important factors influencing management decisions for farm ponds and placed fencing around D2 due to its steep slopes that may injure his cattle.

Overall, the results from this case study suggest that while farmers with pro-conservation values may introduce farm pond management strategies, such as tree-planting in the riparian zone, other factors such as livestock care will also influence management decisions which may inadvertently improve farm pond biodiversity. Increasing landowners’ awareness of the potential of these artificial systems to provide habitat for many native aquatic macrophytes and invertebrates may also increase the likelihood of farm pond management strategies being put in place (Tenge and others 2004). Potential limitations of the study were the small sample size of the survey, as well as the fact that the participants in this case study were all environmentally aware, having connections with the local Landcare group which aims to increase awareness and knowledge of landowners with regard to the sustainable resource management (Curtis and others 2000). It is therefore important to focus more research on a range of farmers’ attitudes towards conservation to determine how to encourage landowners who do not have pro-conservation values to uptake farm pond management strategies to increase habitat values (Beedell and Rehman 1999). Pyrovetsi and Daoutopoulos (1997) recognise that providing landowners with relevant information and education about on-farm conservation is a key strategy to help landowners form a more pro-conservation attitude and encourage them to adopt pro-environmental on-farm management strategies.

Conclusion

Farm ponds are able to support a wide range of native aquatic flora and fauna. The invertebrate communities recorded in these artificial habitats are comparable to surrounding natural streams, although somewhat less rich in at the family level. The potential habitat value of these ponds may be even greater, as most of the study ponds currently have no management strategies in place to enhance biodiversity. The close relationship observed between macrophyte cover and invertebrates suggest management strategies that result in establishment and maintenance of littoral macrophytes will improve habitat value. Landowners should be informed on the potential importance of farm ponds as habitats for native flora and fauna. This increased knowledge may lead to more landowners implementing farm pond management strategies. Overall, with natural wetlands diminishing in the landscape, and farm ponds proliferating due to the need for reliable water, it seems that these artificial habitats can provide an important alternative habitat for many native floral and faunal species.

To best determine how to manage these artificial systems for increased biodiversity, we suggest a larger study should be undertaken to uncover the cause and effect of management strategies on pond littoral zone habitat and biota. This case study gains its strength by combining social science techniques with ecological research. We recommend that future studies also attempt to combine both sciences to optimise proposed management strategies by ensuring that they are practical and are designed to achieve landowners goals as well as biodiversity goals.

Acknowledgements

We are indebted to all of the landowners who allowed us access to their properties and provided additional information about the management of their farm ponds. Thanks also to the Ecosystem Health Monitoring Program (EHMP) of the Moreton Bay Waterways and Catchments Partnership for making data available to be used in this paper. We also thank Fran Sheldon for her advice in sampling techniques and data analysis; Dan Wruck for analysing our nutrient samples; Wade Hadwen for providing helpful comments to help develop this manuscript; and the many field volunteers who helped to collect this data. This project was funded by the Griffith School of Environment and the Australian Rivers Institute, Griffith University.

Literature Cited

Allen, A.P., T.R. Whittier, P.R. Kaufmann, D.P. Larsen, R.J. O'Connor, R.M. Hughes, R.S. Stemberger, S.S. Dixit, R.O. Brinkhurst, A.T. Herlihy and S.G. Paulsen. 1999. Concordance of taxonomic richness patterns across multiple assemblages in lakes of the northeastern United States. *Canadian Journal of Fisheries and Aquatic Sciences* 56: 739-747.

Altieri, M.A. 1999. The ecological role of biodiversity in agroecosystems. *Agriculture Ecosystems and Environment* 74:19-31.

Babbie, E. 1990. Survey Research Methods. Wadsworth Publishing Company, California.

Beedell, J.D.C. and T. Rehman. 1999. Explaining farmers' conservation behaviour: Why do farmers behave the way they do? *Journal of Environmental Management* 57: 165-176.

Bentham, C.G.M., M.J. 1982. Social surveys. In: R. Haynes (ed.) Environmental Science Methods, Chapman and Hall, London.

BOM. 2005a. Average daily sunshine hours - January to December: Brisbane, Commonwealth Bureau of Meteorology. (online) Available from: http://www.bom.gov.au/cgi-bin/climate/cgi_bin_scripts/idl_sunh_graphs/sunh_script.cgi?40214 (Accessed 27 February 2005).

BOM. 2005b. Climate Maps - Temperature Percentiles, Commonwealth of Australia, Bureau of Meteorology. (online) Available from: http://www.bom.gov.au/climate/map/anual_rainfall/index_tmp.shtml (Accessed 27 February 2005).

Brainwood, M.A., S. Burgin and B. Maheshwari. 2004. Temporal variations in water quality of farm dams: impacts of landuse and water sources. *Agricultural Water Management* 70: 151-175.

Brock, M.A., R.G.B. Smith and P.J. Jarman. 1999. Drain it, dam it: alteration of water regime in shallow wetlands on the New England Tableland of New South Wales, Australia. *Wetlands Ecology and Management* 7: 37-46.

Bunn, S.E. and A.H. Arthington. 2002. Basic principles and ecological consequences of altered flow regimes for aquatic biodiversity. *Environmental Management* 30: 492-507.

Casanova, M.T., A. Douglas-Hill, M.A. Brock, M. Muschal and M. Bales. 1997. Farm Ponds in New South Wales, Australia: relationship between macrophyte and phytoplankton abundances. *Marine and Freshwater Research* 48: 353-360.

Cummins, K.W. and R.W. Merritt. 2001. Application of Invertebrate Functional Groups to Wetland Ecosystem Function and Biomonitoring. In: R.B. Rader, D.P. Batzer and S.A. Wissinger (eds.) Bioassessment and Management of North American Freshwater Wetlands, Wiley, New York.

Curtis, A., Nouhuys, M.V., Robinson, W. and Mackay, J. 2000. Exploring Landcare effectiveness using organisational theory. *Australian Geographer* 31: 349-366.

Declerck, S., T. De Bie, D. Ercken, H. Hampel, S. Schrijvers, J. Van Wichelen, V. Gillard, R. Mandiki, B. Losson, D. Bauwens, S. Keijers, W. Vyverman, B. Goddeeris, L. De meester, L. Brendonck, and K. Martens. 2006. Ecological characteristics of small farmland ponds: Associations with land use practices at multiple spatial scales. *Biological Conservation* 131:523-532.

Denzin, H.K. and Y.S. Lincoln. 2003. The Discipline and Practice of Qualitative Research. Sage Publishers, Thousand Oaks, California, London.

EHMP. 2004. Ecosystem Health Monitoring Program 2002-2003 Annual Technical Report, Moreton Bay Waterways and Catchments Partnership, Brisbane.

Fellows, C.S., J.E. Clapcott, J.W. Udy, S.E. Bunn, B.D. Harch, M.J. Smith, and P.M. Davies. 2006. Benthic metabolism as an indicator of stream ecosystem health. *Hydrobiologia* 572: 71-80.

Fellows, C.S., Valett, H.M. and Dahm, C.N. 2001. Whole-stream metabolism in two montane streams: contribution of the hyporheic zone. *Limnology and Oceanography* 46: 523-531

Frankenberg, J. 1998. Enhancing Farm Dams. In: F. Stelling (ed.) From Little Things Big Things Grow..... Revegetation Guide for the Riverina Highlands, Murray Catchment Management Committee, Albury, NSW.

Garnier, J., G. Billen, N. Sanchez and B. Leporcq. 2000. Ecological functioning of the Marne reservoir (upper Seine basin, France). *Regulated Rivers-Research and Management* 16: 51-71.

Goldsborough, G. 2001. Sampling algae in wetlands. In: R.B. Rader, D.P. Batzer & S.A. Wissinger (eds.) Bioassessment and Management of North American Freshwater Wetlands, Wiley, New York.

Greenberg, A.E., Clesceri, L.S. and Eaton, A.D. (eds). 1992. Standard methods for the examination of water and wastewater, 18th edition. American Public Health Association, American Water Works Association and Water Environment Foundation, Washington.

Hadwen, W.L. and Bunn, S.E. 2005. Food web responses to low-level nutrient and ^{15}N-tracer additions in the littoral zone of an oligotrophic dune lake. *Limnology and Oceanography* 50: 1096-1105.

Hanson, P.C., D.L. Bade, S.R. Carpenter and T.K. Kratz. 2003. Lake metabolism: Relationships with dissolved organic carbon and phosphorus. *Limnology and Oceanography* 48: 1112-1119.

Hansson, L., C. Bronmark, P.A. Nilsson and K. Abjornsson. 2005. Conflicting demands on wetland ecosystem services: nutrient retention, biodiversity or both? *Freshwater Biology* 50: 705-714.

Hazell, D., R. Cunnningham, D. Lindenmayer, B. Mackey and W. Osborne. 2001. Use of farm dams as frog habitat in an Australian agricultural landscape: factors affecting species richness and distribution. *Biological Conservation* 102: 155-169.

Hazell, D., J.M. Hero, D. Lindenmayer and R. Cunningham. 2004. A comparison of constructed and natural habitat for frog conservation in an Australian agricultural landscape. *Biological Conservation* 119: 61-71.

Hecky, R.E. and Hesslein, R.H. 1995. Contributions of benthic algae to lake food webs as revealed by stable isotope analysis. *Journal of the North American Benthological Society* 14 (4): 631-653.

Herb, W. R., and H. G. Stefan. 2003. Integral growth of submersed macrophytes in varying light regimes. *Ecological Modelling* 168:77-100.

Hietala, J., K. Vakkilainen, and T. Kairesalo. 2004. Community resistance and change to nutrient enrichment and fish manipulation in a vegetated lake littoral. *Freshwater Biology* 49:1525-1537.

Ingram, B.A., J.H. Hawking and R.J. Shiel. 1997. Aquatic Life in Freshwater Ponds: A Guide to the Identification and Ecology of Life in Aquaculture Ponds and Farm Dams in South Eastern Australia. Co-operative Research Centre for Freshwater Ecology, Albury, NSW.

Kadono, Y. 1998. Aquatic plants community in farm pond. In: Y. Ezaki and T. Tanaka (eds) Preservation of Waterside Environment: from viewpoint of Bio-Community. Asakura Shoten, Tokyo, Japan (in Japanese).

Lisson, S.N., L.E. Brennan, K.L. Bristow, B.A. Keating and D.A. Hughes. 2003. DAM EA$Y - software for assessing the costs and benefits of on-farm water storage based production systems. *Agricultural Systems* 76: 19-38.

Lloyd, S.D., P. Bishop and I. Reinfelds. 1998. Shoreline erosion: a cautionary note in using small farm dams to determine catchment erosion rates. *Earth Surface Processes and Landforms* 23: 905-912.

Loi, J.K. and D.T. Malcolm. 1998. Land Resource Assessment SEQ 2001 Report 6: Soils and Land Suitability The Kilcoy-Woodford Area, Department of Natural Resources, Brisbane.

MacArthur, R.H. and E.O. Wilson. 1967. The Theory of Island Biogeography. Priceton University Press, Princeton.

Maezono, Y. and T. Miyashita. 2004. Impact of exotic fish removal on native communities in farm ponds. *Ecological Research* 19: 263-267.

Murkin, H. R., and L. C. M. Ross. 2000. Invertebrates in Prairie Wetlands. In: H. R. Murkin, A. G. Van Der Valk, and W. R. Clark, (eds.). Prairie Wetland Ecology: The Contribution of the Marsh Ecology Research Program. Iowa State University Press, Ames, Iowa.

Nichols, P. 1991. Social Survey Methods: A Fieldguide for Development Workers. Oxfam, Oxford.

Nicolet, P., J. Biggs, G. Fox, M.J. Hodson, C. Reynolds, M. Whitfield and P. Williams. 2004. The wetland plant and macroinvertebrate assemblages of temporary ponds in England and Wales. *Biological Conservation* 120: 265-282.

Perkins, R. G., and G. J. C. Underwood. 2002. Partial recovery of a eutrophic reservoir through managed phosphorus limitation and unmanaged macrophyte growth. *Hydrobiologia* 481:75-87.

Platt, S. and A. Corrick. 1994. Management of shallow freshwater wetlands for wildlife. *Trees and Natural Resources*, September 2004; 12-18.

Pyrovetsi, M. and Daoutopoulos, G. 1997. Contrasts in conservation attitudes and agricultural practices between farmers operating in wetlands and a plain in Macedonia, Greece. *Environmental Conservation* 24: 76-82.

QNRM. 2003. Preliminary Assessment of Overland Flow Significance in the Moreton Region, Department of Natural Resources and Mines, Queensland Government.

Rickson, R., P. Saffigna, F. Vanclay and G. McTainsh. 1987. Social bases of farmers' responses to land degradation. In: A. Chisholm and R. Dumsday (eds.) Land Degradation: Problems and Policies, Cambridge University Press, Cambridge.

Reidsma, P., T. Tekelenburg, M. van den Berg, and R. Alkemade. 2006. Impacts of land-use change on biodiversity: An assessment of agricultural biodiversity in the European Union. *Agriculture Ecosystems and Environment* 114:86-102.

Robinson, G.G.C., S.E. Gurney and L.G. Goldsborough. 2000. Algae in Prairie Wetlands. In: H.R. Murkin, A.G. Van Der Valk and W.R. Clark (eds.) Prairie Wetland Ecology. Iowa State University Press, Ames, Iowa.

Robson, B. J., and C. J. Clay. 2005. Local and regional macroinvertebrate diversity in the wetlands of a cleared agricultural landscape in south-western Victoria, Australia. *Aquatic Conservation: Marine and Freshwater Ecosystems* 15:403-414.

Scheffer, M., S. H. Hosper, M.-L. Meijer, B. Moss, and E. Jeppesen. 1993. Alternative equilibria in shallow lakes. *Trends in Ecology and Evolution* 8:275-279.

Schuman, H. and S. Presser. 1981. Questions and Answers in Attitude Surveys : Experiments on Question Form, Wording, and Context. Academic Press, New York.

Smith, M.J., Bunn, S.E., Storey, A.W., Harch, B.D. and Redfern, F.M. 2001. Process undertaken to develop and EHMP for rivers and streams in South East Queensland. In: M.J. Smith and A.W. Storey (eds.) Design and Implementation of Baseline Monitoring (DIBM3): Developing an ecosystem health monitoring program for rivers and streams in Southeast Queensland. Centre for Catchment and In-Stream Research (CCISR), Griffith University and South East Queensland Regional Water Quality Management Stragegy (SEQRWQMS), Brisbane.

Statzner, B. and B. Moss. 2004. Linking ecological function, biodiversity and habitat: a mini-review focusing on older ecological literature. *Basic and Applied Ecology* 5: 97-106.

Sudman, S. and N.M. Bradburn. 1974. Response Effects in Surveys: A Review and Synthesis. Aldine Publishers, Chicago.

Takasaki, Y. 1994. Odonata of irrigation pond. In The research group of natural irrigation pond (eds). Introduction to Natural History of Irrigation Pond. Gogo Shuppan, Tokyo, Japan (in Japanese)

Tenge, A.J., De Graaff, J. and Hella, J.P. 2004. Social and economical factors affecting the adoption of soil and water conservation in West Usambara Highlands, Tanzania. *Land Degradation and Development* 15:99-114.

Timms, B.V. 1980. Farm Dams. In: W.D. Williams (ed.) An Ecological Basis for Water Resource Management, Australian National University Press, Canberra.

Tress, B. 2002. Application of Landscape Ecology. In: O. Bastian and U. Steinhardt (eds.) Development and Perspectives of Landscape Ecology, Kluwer Academic Publishers, Dordrecht, The Netherlands.

Tscharntke, T., A. M. Klein, A. Kruess, I. Steffan-Dewenter, and C. Thies. 2005. Landscape perspectives on agricultural intensification and biodiversity - ecosystem service management. *Ecology Letters* 8:857-874.

Waterways. 2002. Stanley Catchment, Healthy Waterways. (online) Available from: http://www.healthywaterways.org/PAGE120417PM5Y31FI.html (Accessed 10 August 2004).

Wetzel, R.G. and G.E. Likens. 1991. Limnological Analyses, 2nd edition. Springer-Verlag, New York.

Williams, P., M. Whitfield, J. Biggs, S. Bray, G. Fox, P. Nicolet, and D. Sear. 2003. Comparative biodiversity of rivers, streams, ditches and ponds in an agricultural landscape in Southern England. *Biological Conservation* 115:329-341.

Zimmerman, B.L. and R.O. Bierregaard. 1986. Relevance of the equilibrium theory of island biogeography and species-area relations to conservation with a case from Amazonia. *Journal of Biogeography* 13: 133-143.

Appendices

Appendix 1: List of all macrophyte and invertebrate taxa recorded across all 8 ponds. * = subfamily

Class	Order	Family	Genus/Species	Location (Pond ID)
CHAROPHYCEAE				
	Charales	Characeae	*Charophyte sp.*	B1, B2
PTEROPSIDA				
	Marsileales	Azollaceae	*Azolla sp.*	A1, B1, C2
LILIOPSIDA				
	Arales	Lemnaceae	*Wolffia sp.*	B1, B2
	Commelinales	Philydraceae	*Philydrum languginosum*	A1, A2
	Hydrocharitales	Hydrocharitaceae	*Ottelia ovalifolia*	B1, B2, D1
	Typhales	Typhaceae	*Typha orientalis*	A1, B1, B2
	Cyperales	Cyperaceae	*Eleocharis acuta*	C2
			Schoenoplectus mucronatus	A1, A2, B1, C2
			Cyperus difformis	A1
		Gramineae	*Urochloa mutica*	B1, B2, C1, C2, D2
	Juncales	Juncaceae	*Juncus usitatus*	A2, B1, B2, C1
MAGNOLIOPSIDA				
	Asterales	Menyanthaceae	*Nymphoides indica*	C2
	Lamiales	Callitrichaceae	*Callitriche stagnalis*	B1, B2
	Myrtales	Onagraceae	*Ludwigia peploides ssp. montevidensis*	A1, B1, B2, C2
	Nymphaeales	Cabombaceae	*Cabomba caroliniana*	A2
		Nymphaeaceae	*Nymphaea caerulea*	A1
HIRUNDINEA				
	Hirudinidae			B1, B2, C1, C2, D1, D2
MOLLUSCA				
	Bivalvia	Sphaeriidae		B1
	Gastropoda	Lymnaeidae	*Austropeplea sp.*	B1, B2, C2, D2
		Physidae	*Physa acuta*	B1, C2, D1, D2
			unknown	A1, D1
CRUSTACEA				
	Cladocera	Daphniidae (in part)		B1, C1
	Decapoda	Parastacidae	*Cherax sp.*	A1, A2
		Atyidae	*Caridina mccullochi*	A2, C2
		Atyidae	*Paratya australiensis*	B1, B2
ARACHNIDA				
	Acariformes	Acarina		B1, C1, C2
COLLEMBOLA				
	Collembola	Isotomidae		A1
INSECTA				
	Ephemeroptera	Baetidae		A1, A2, B1, B2
		Caenidae	*Tasmanocoenis sp.*	B1, B2, C1
	Odonata	Aeshnidae		A1, A2, B1, B2, D1, D2
		Coenagrionidae		A1, A2, B1, B2, C1, C2, D1, D2
		Corduliidae		D1
		Hemicorduliidae	*Hemicorduliidae*	A1, A2, B1, B2, D2
		Libellulidae	*Nannophya sp.*	A1, B1, B2, C1, C2, D2

	Lindeniidae		B1
Hemiptera	Belostomatidae		A1, B1, C2, D2
	Corixidae	*Agroptoerixa*	B1, B2, C1, C2, D1, D2
	Corixidae	*Micronecta*	A2, B1, B2, C1, D1
	Gerridae		A2
	Hebridae		A1, B1, B2
	Mesoveliidae	*Mesovelia sp.*	A1, A2, B2, C2
	Naucoridae		A1, A2, B1, C2
	Nepidae	*subfamily: Ranatrinae*	A2, C2
	Notonectidae	*Anisops sp.*	A1, A2, B1, B2, C1, C2, D1, D2
	Pleidae	*Paraplea sp.*	A1, B1, B2, C2
	Veliidae		B2
Coleoptera	Dytiscidae	*Antiporus*	A1
	Dytiscidae	*Bidessodes*	C2
	Dytiscidae	*Cybister*	D1, D2
	Dytiscidae	*Eretes*	B1, D2
	Dytiscidae	*Hydacticus*	C2
	Dytiscidae	*Hyphydrus*	A1, A2, B1, B2, C1, D2
	Dytiscidae	*Laccophilus*	B1, C2, D1, D2
	Dytiscidae	*larvae*	A1, A2, B1, B2, C1, C2,
	Dytiscidae	*Necterosoma*	A2, B1, B2, C1, C2
	Dytiscidae	*unknown*	A1, C2
	Gyrinidae	*adult*	A2, B1, D2
	Gyrinidae	*larvae*	C1
	Haliplidae	*Haplilus sp.*	B1
	Hydraenidae	*Hydraena*	A1, B1, B2, C1, C2, D2
	Hydrophilidae	*adult*	A2, C2
	Hydrophilidae	*Berosus sp. larvae*	B1, C1, D1, D2
	Hydrophilidae	*Subfamily Hydrohpilinae*	A1, A2, C1, C2, D2
	Hygrobiidae		A2, B2, C2
	Psephenidae	*Sclerocyphon sp.*	B2
	Scirtidae	*larvae*	A1, B2, C2, D2
Diptera	Ceratopogonidae		A1, A2, C1, C2, D1, D2
	Chironominae *		A1, A2, B1, B2, C1, C2, D1, D2
	Culicidae		A1, C2
	Ephydridae		D2
	Stratiomyidae		A1
	Tanypodinae *		A1, A2, B1, B2, C1, C2, D1, D2
	unknown diptera larvae		A1, D1
Trichoptera	Leptoceridae		A1, A2, B1, B2, C1, C2, D1, D2
Lepidoptera	Pyralidae		A1

Figure captions

Figure 1: Location of the study sites, Stanley River catchment, South-East Queensland. Circles represent the locations of the towns in which the four properties (A, B, C and D) are located. Two ponds (1 and 2) were sampled from each property.

Figure 2: Relationship between taxonomic richness and farm pond morphological and historical characteristics. (a) Replicate values of invertebrate family richness and littoral zone habitat complexity (measured as total macrophyte % cover). Total taxonomic richness (invertebrate + macrophyte) with (b) age and (c) pond area for each study pond.

Figure 3: MDS ordination of the invertebrate communities (abundance data) for all eight ponds. Each point is one of three replicate samples from the littoral zones in each pond. Symbols indicate samples from (a) all eight ponds separately and (b) ponds grouped based on vegetation status. Stress = 0.18.

Figure 4: Relationship between total biodiversity predicted by landowners (as ranked from 1 to 4; "fair" to "very good") and the measured total number of taxa in the ponds (macrophytes + invertebrates). Different landowners are represented as: Landowner A = ◆, Landowner B = ☐, Landowner C = ▲, Landowner D = ○

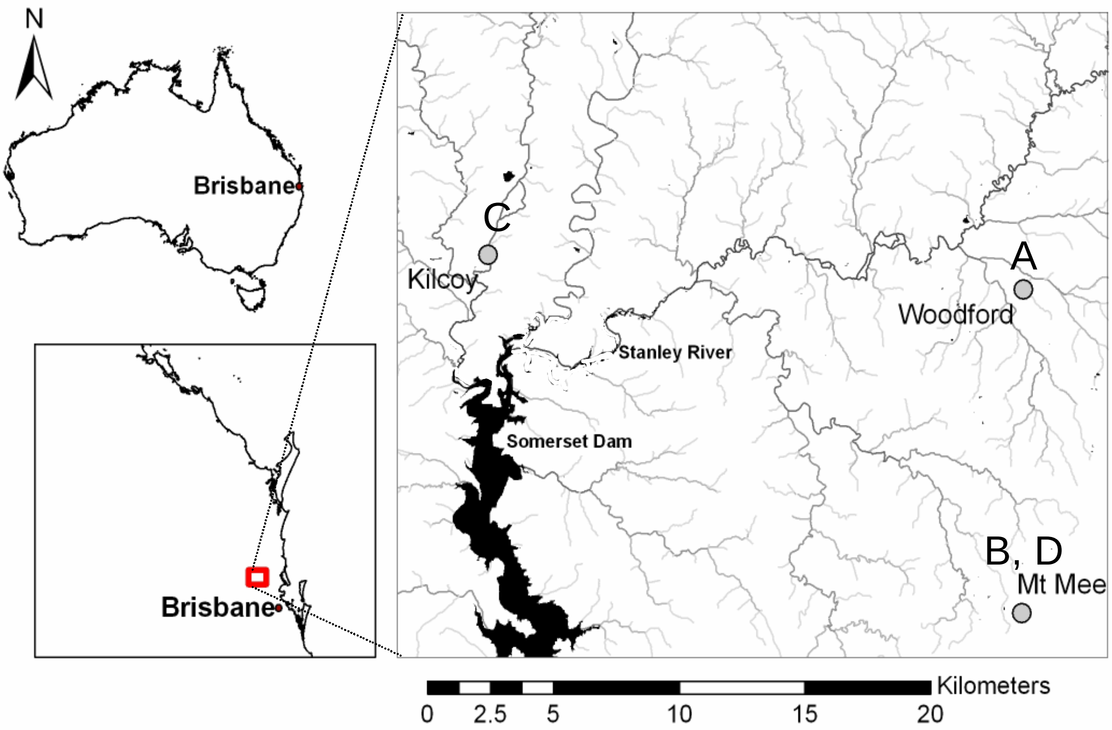
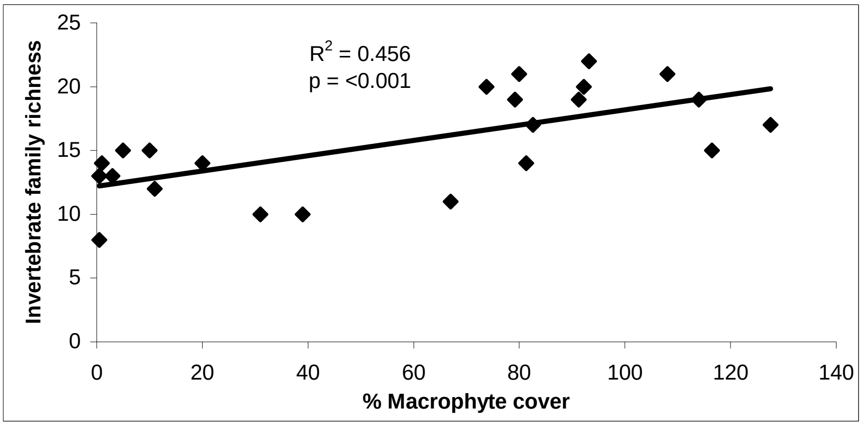
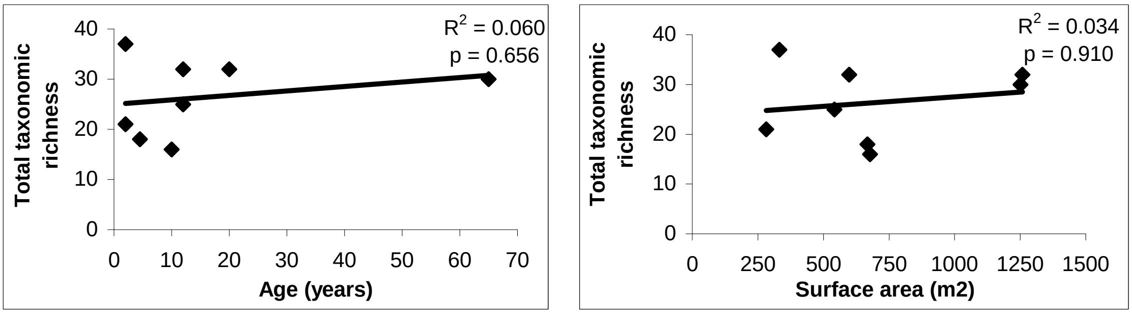

N
Brisbane
Brisbane
C
Kilcoy
A
Woodford
Stanley River
Somerset Dam
B, D
Mt Mee
0
2.5
5
10
15
20
Kilometers

Figure 1

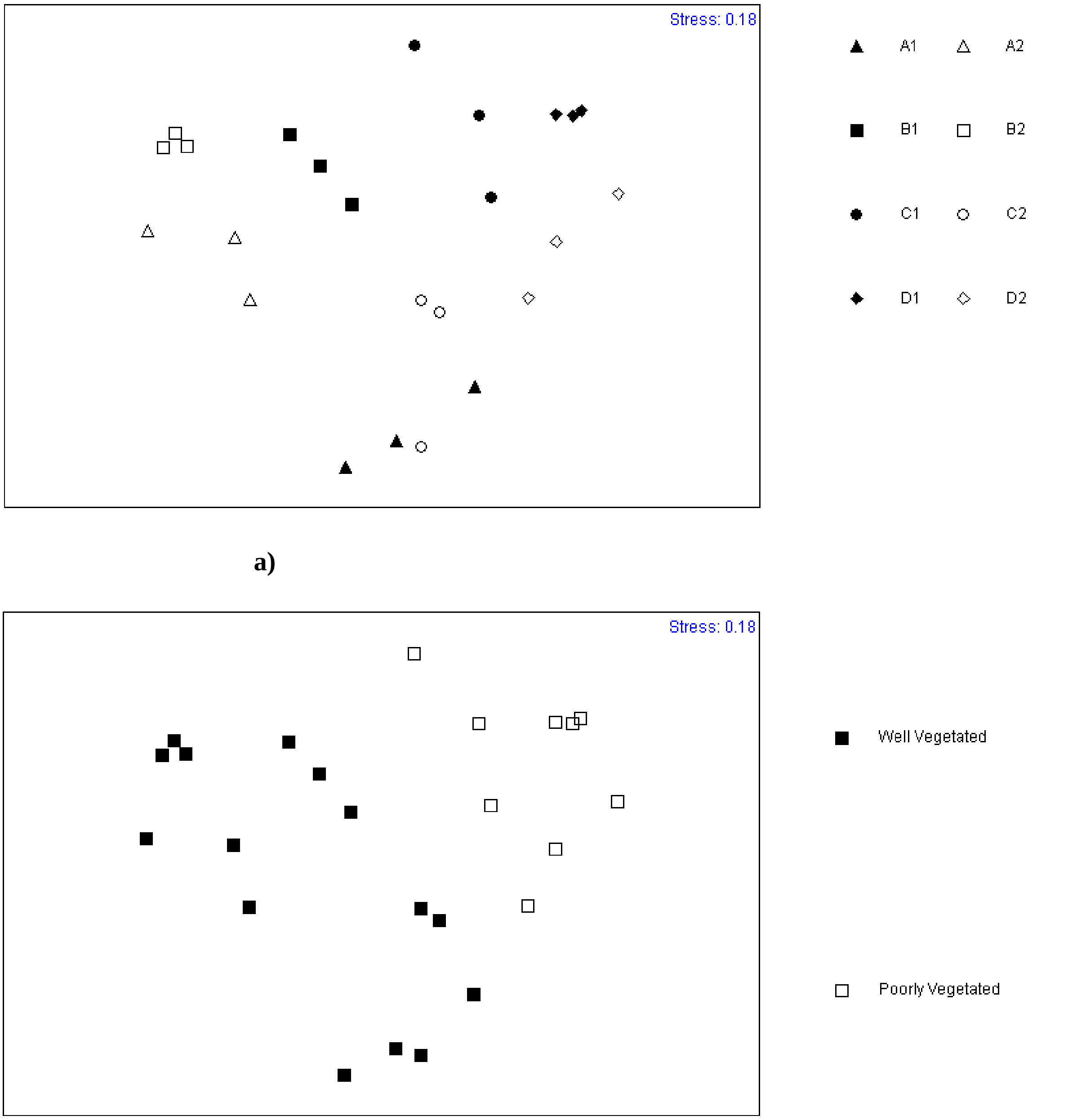

25
20
15
10
5
0
0
20
40
60
80
100
120
140
Invertebrate family richness
% Macrophyte cover
$R^2 = 0.456$
p = <0.001

a)


Total taxonomic richness
40
30
20
10
0
0
10
20
30
40
50
60
70
Age (years)
$R^2 = 0.060$
p = 0.656
Total taxonomic richness
40
30
20
10
0
0
250
500
750
1000
1250
1500
Surface area (m2)
$R^2 = 0.034$
p = 0.910

b)

Figure 2


Stress: 0.18
▲ A1
△ A2
■ B1
□ B2
● C1
○ C2
◆ D1
◇ D2
Stress: 0.18
■ Well Vegetated
□ Poorly Vegetated

a)

b)

Figure 3

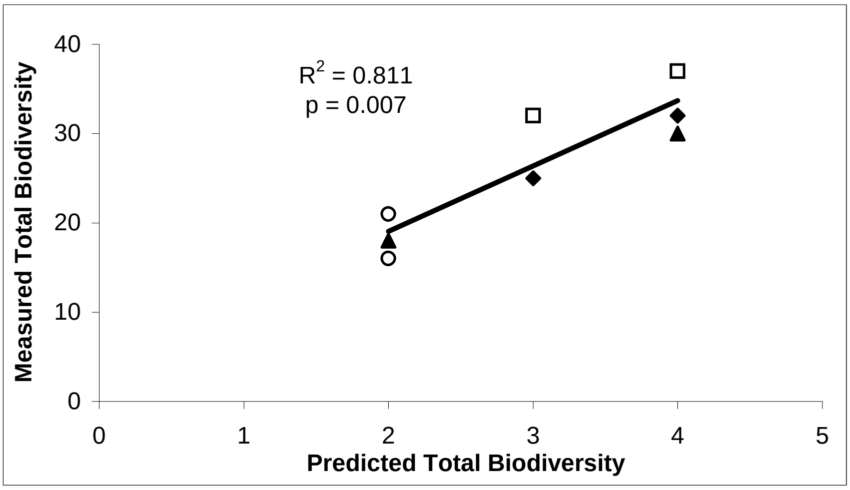

$R^2 = 0.811$
$p = 0.007$
Measured Total Biodiversity
Predicted Total Biodiversity
0
10
20
30
40
0
1
2
3
4
5

Figure 4

Tables

Table 1: Summary of historical, morphological, water physical and chemical characteristics and benthic metabolism data for each pond. Mean values are presented with associated standard errors.

POND ID		**A1**	**A2**	**B1**	**B2**	**C1**	**C2**	**D1**	**D2**
POND HISTORY									
Age (years)		12	12	2	20	4.5	65	10	2
Current management strategy		Riparian vegetation	None	Riparian vegetation	None	None	None	None	Fenced
MORPHOLOGICAL CHARACTERISTICS									
Perimeter (m)		179.87	90.45	85.78	166.84	104.82	136.81	118.10	70.07
Surface area (m)		1258.56	542.05	332.11	597.48	667.35	1251.56	677.17	281.56
Littoral zone slope (Δdepth/Δdistance from shore)	Mean	0.28	0.19	0.39	0.39	0.22	0.25	0.24	0.55
	SE	0.14	0.02	0.05	0.08	0.03	0.04	0.03	0.11
Deepest littoral zone depth (cm)	Mean	34.25	19.75	40.80	49.20	20.88	31.75	33.60	53.00
	SE	12.35	2.39	5.89	9.06	0.97	5.81	6.05	11.02
PHYSICAL CHARACTERISTICS									
Temperature									
Shallow	Min	9.9	11.8	14.1	14.1	12.4	13.0	10.8	11.0
	Max	16.2	18.1	17.9	18.6	19.0	17.1	13.6	16.0
	Diel swing	6.30	6.30	3.80	4.50	6.59	4.07	2.85	5.00
Deep	Min	10.0	11.7	13.4	13.5	12.6	12.9	11.3	11.9
	Max	10.3	12.7	14.3	14.3	14.2	14.8	13.4	13.3
	Diel swing	0.30	0.98	0.86	0.80	1.60	1.90	2.16	1.40
Photic depth (cm)		136	122	295	455	69	223	192	331
Turbidity (NTU)		8.8	15.5	4.4	1.5	41.1	9.7	13.6	3.4
Chlorophyll *a* (mg m^{-2})	Mean	0.002	0.014	0.001	0.001	0.022	0.006	0.004	0.005
	SE	0.001	0.004	<0.001	<0.001	<0.001	0.001	0.001	0.001
CHEMICAL CHARACTERSTICS									
Dissolved oxygen (mg L^{-1})									
Shallow	Min	3.59	8.83	10.81	9.75	7.02	4.82	5.79	4.93
	Max	5.53	10.09	12.81	13.62	8.97	7.89	7.03	7.57
	Diel swing	1.94	1.26	1.99	3.87	1.95	3.07	1.24	2.64
Deep	Min	3.05	8.96	6.20	11.70	6.97	4.30	5.56	5.74

	Max	4.01	10.53	14.61	15.58	9.33	6.41	6.61	7.62
	Diel swing	0.96	1.57	8.41	3.88	2.35	2.11	1.05	1.88
pH		6.32	6.02	7.75	7.95	6.25	5.73	7.23	6.59
Conductivity (µS cm^{-1})		518	125	614	680	116	135	623	740
Total phosphorus (mg L^{-1})		0.019	0.041	0.018	0.014	0.14	0.020	0.056	0.022
Filterable reactive rhosphorus (mg L^{-1})		< 0.002	< 0.002	0.002	0.003	0.002	< 0.002	0.006	0.002
Ammonium-N (mg L^{-1})		0.006	0.010	< 0.002	0.003	0.015	0.002	0.80	0.029
Nitrate-N (mg L^{-1})		0.003	< 0.002	< 0.002	0.009	0.002	< 0.002	0.064	0.019
Total nitrogen (mg L^{-1})		0.46	1.2	0.31	0.29	1.5	0.45	1.8	0.38
BENTHIC METABOLISM									
R (mg O_2 m^{-2} h^{-1})	Mean	49.94	79.27	251.61	103.96	112.22	125.54	155.92	134.96
	SE	7.10	23.68	10.52	7.86	3.87	4.40	8.39	34.70
NEP (mg O_2 m^{-2} h^{-1})	Mean	-10.46	14.44	-73.37	-63.76	91.69	83.03	91.40	30.37
	SE	28.11	21.54	99.11	19.86	9.37	29.62	9.10	6.94
GPP (mg O_2 m^{-2} h^{-1})	Mean	39.48	93.71	178.24	40.21	203.91	208.58	247.32	165.34
	SE	21.85	16.25	88.59	14.39	11.18	32.36	17.35	41.64
GPP/R	Mean	0.28	0.44	0.21	0.12	0.53	0.48	0.46	0.36
	SE	0.13	0.12	0.11	0.04	0.02	0.07	0.01	0.00

Table 2: Summary of macrophyte structure in farm pond littoral zones. Percent cover, species richness and diversity values for each pond are the mean of three transects (SE). Total species richness and introduced species are presented as the total number located in each pond. Ponds with significantly ($p<0.05$) lower Total % cover, mean species richness and mean species diversity are indicated with * and are subsequently classified as poorly vegetated.

POND ID	**A1**	**A2**	**B1**	**B2**	**C1**	**C2**	**D1**	**D2**
% Emergent Cover	65 (1.7)	37 (9.3)	4 (1.9)	1 (0.8)	0.5 (0.3)	54 (7.8)	0	0
% Floating Cover	17 (4.2)	0	20 (3.4)	38 (9.5)	0	42 (3.3)	2 (0.8)	0
% Submerged Cover	0	9 (4.8)	64 (4.5)	69 (8.5)	3.5 (3.2)	6 (4.1)	0	12 (4.4)
Total % Cover	82 (5.5)	46 (10.9)	88 (4.1)	108 (14.0)	4 (3.5) *	102 (9.6)	2 (0.8) *	12 (4.4) *
Total Species Richness	6	4	10	8	2	6	1	1
Introduced Species	1	1	2	2	1	1	0	1
Mean Species Richness	6 .0 (1.0)	3.0 (0.6)	7.7 (0.9)	5.3 (0.3)	1.3 (0.3) *	5.7 (0.3)	1.0 (0.0) *	1.0 (0.0) *
Mean Diversity	0.76 (0.15)	0.78 (0.18)	1.24 (0.13)	1.04 (0.16)	0.10 (0.10) *	1.27 (0.07)	0.00 (0.00) *	0.00 (0.00) *

Table 3: Invertebrate family richness and diversity in littoral zones of the study ponds. Values shown represent the total number of families identified from each order in each of the ponds. Total family richness is given for each of the ponds as well as mean taxonomic richness and diversity based on three replicate transect samples (SE). The last two columns show the total number of families identified for each order across the eight study ponds and ten streams monitored by the Ecosystem Health Monitoring Program (EHMP) sites in the Stanley River catchment. The total number of families identified from all of the ponds and all of the streams are also presented.

	A1	**A2**	**B1**	**B2**	**C1**	**C2**	**D1**	**D2**	**Total # families in study ponds**	**Total # families in stream (EHMP) sites**
Acariformes				1	1	1			1	1
Hirudinidae			1	1	1	1	1	1	1	1
Bivalvia			1						1	2
Gastropoda	1		2	1		2	2	2	3	6
Cladocera			1		1				1	0
Decapoda	1	2	1	1		1			2	3
Collembola	1								1	0
Ephemeroptera	1	1	2	2	1				2	3
Odonata	4	3	5	4	2	2	3	4	6	10
Hemiptera	6	6	6	6	2	7	2	3	10	11
Coleoptera	4	4	5	5	4	5	2	5	8	11
Diptera	6	3	2	2	3	4	4	4	7	11
Trichoptera	1	1	1	1	1	1	1	1	1	7
Lepidoptera	1								1	1
Total Family Richness	26	20	27	24	16	24	15	20	**45**	**67**
Mean Family Richness	19.7 (0.3)	10.3 (0.3)	20.7 (0.9)	15.3 (0.9)	11.0 (1.5)	19.0 (1.2)	13.3 (0.3)	14.7 (0.3)		
Mean Diversity	2.4 (0.1)	1.0 (0.2)	1.9 (0.2)	0.9 (0.1)	1.6 (0.0)	2.5 (0.1)	1.0 (0.1)	1.5 (0.3)		

Table 4: Chemical and physical parameters measured in the study ponds compared to those measured in other farm ponds in Australia. Values are shown as the minimum and maximum measurement of each parameter.

Chemical/Physical Parameter	**Study Farm Ponds**	**Brainwood and others (2004)**	**Hazell and others (2001)**
Water column chlorophyll *a* ($\mu g\ L^{-1}$)	1-22	0-390	
TP ($\mu g\ P\ L^{-1}$)	14-140	10-2800	
FRP ($\mu g\ P\ L^{-1}$)	<2-6		
TN ($\mu g\ N\ L^{-1}$)	290-1800	0-9900	
NO_3^- ($\mu g\ N\ L^{-1}$)	<2-64	0-1600	
NH_4^+ ($\mu g\ N\ L^{-1}$)	<2-800		
DO ($mg\ L^{-1}$)	3.05-15.58	0.06-27	
pH	5.7-7.95	6.6-9.3	6.05-9.85
Turbidity (NTU)	1.5-41	9.0-210	10-400
Temperature	9.9-18.99	5.4-28	
Conductivity ($\mu S\ cm^{-1}$)	115.5-740	250-870	25-964