“Within-lake detection of the effects of tourist activities in the littoral zone of oligotrophic dune lakes.”

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Abstract

Intensive recreational use of oligotrophic lakes can lead to increases in epilimnetic nutrient concentrations (through direct inputs from urine or re-suspension of sediments) and the development of undesirable algal blooms. Despite these adverse ecological responses to tourist activities, many lake monitoring programs do not address tourist nutrient inputs at appropriate spatial and temporal scales. This paper presents results of investigations aimed at detecting the effects of nutrient inputs to perched dune lakes on Fraser Island, principally through within-lake comparisons of nutrient and algal variables. Nutrient concentrations and algal biomass were measured in heavily visited (disturbed) and inaccessible (reference) sites within five perched dune lakes on Fraser Island, Australia, during the 1999-2000 summer. Whilst nutrient and phytoplankton chlorophyll *a* concentrations did not differ between sites, periphyton chlorophyll *a* concentrations was occasionally significantly higher in disturbed sites than in reference sites, particularly in the very popular clear lakes, suggesting that algal growth may be enhanced by tourist activities.

Experimental manipulations of nitrogen and phosphorus concentrations in algal (phytoplankton and periphyton) bioassays were undertaken in each lake over the 2000-2001 summer, to assess algal responses to nutrient additions. The response of phytoplankton communities to nutrient additions varied greatly between lakes, with evidence of limitation or co-limitation by nitrogen and phosphorus in all systems. Periphyton biomass showed similar trends to phytoplankton in some lakes, but these were not significant.

Nutrients added to lakes by tourists are likely to be rapidly assimilated by littoral zone periphyton communities in these oligotrophic lakes. As a result, impacts of tourism are not likely to be detected by traditional measurements of open water nutrient and phytoplankton chlorophyll *a* concentrations. Instead, measurement of periphyton growth and/or biomass (chlorophyll *a*) in the littoral zone might be the most spatially and temporally relevant indicator of tourist impacts in these lakes.

Keywords: periphyton, phytoplankton, nutrients, tourism, impacts, monitoring

Introduction

Oligotrophic waterbodies in wilderness areas are increasingly threatened by small-scale nutrient inputs from tourists (Goodrich 1997, Wilkinson 1999). Nutrients can be delivered either via the physical re-suspension of sediments (Liddle & Scorgie 1980) or through direct addition of nutrients in the form of urine and other wastes (King & Mace 1974, Butler et al. 1996). Regardless of the source and/or mechanism of delivery, nutrient inputs from tourists typically represent relatively minor additions relative to the total nutrient load for the waterbody (King & Mace 1974). Nevertheless, the well-documented responsiveness of oligotrophic systems to nutrient additions suggests that ongoing additions may have considerable environmental consequences over time (Shortreed & Stockner 1990). Significantly, increases in ambient nutrient concentrations have been shown to inflate primary production beyond the regulatory control of grazer communities in numerous oligotrophic systems (Cyr & Pace 1992, Hansen et al. 1997, Blomqvist 2001). Furthermore, excessive nutrient inputs can also facilitate the proliferation of ecologically and aesthetically undesirable algal communities (Ganf & Oliver 1982, Smith 1986).

Fraser Island is a World Heritage listed wilderness area off the coast of South East Queensland, Australia (Anon 1999). The island's nomination for World Heritage status focussed on the unique series of oligotrophic perched dune lakes (UNESCO 2001) and these systems have since become the focus of swimming and recreation activities on the island (Hadwen et al. 2003, Hadwen & Arthington, 2003). With in excess of 300 000 tourists visiting Fraser Island each year, it has been suggested that the most popular lakes receive more than 1000 visitors per day in peak tour times (Hadwen & Arthington 2003).

Despite the rapid growth of tourism on Fraser Island (Sinclair 2000, UNESCO 2001, Anon 2002), only two studies have attempted to measure the effects of tourists on the ecology of these lakes (Arthington et al. 1990a, Hockings 1999). However, in the late 1980's, Arthington et al. (1989) and Outridge et al. (1989) documented changes in the trophic status of dune lakes

in the nearby Cooloolo Region and on Moreton Island and suggested that increased tourist visitation was probably the single most important factor driving these changes. Furthermore, Outridge et al. (1989) postulated that the typical phytoplankton communities of dune lakes in south-eastern Queensland (dominated by atmospheric nitrogen-fixing cyanobacteria), might be particularly responsive to phosphorus additions.

These early studies and others relied heavily on the comparison of trophic status between disturbed (visited) and reference (not visited) lakes (Outridge et al. 1989, Arthington et al. 1990a). However, examination of the consequences of small-scale nutrient additions from tourist sources is undoubtedly more appropriate at the within-lake scale (Underwood & Kennelly 1990, Underwood 1996). Whilst this seems obvious given the spatially and temporally restricted nature of tourist activities in wilderness areas (Liddle & Scorgie 1980, Buckley & Pannell 1990), particularly in lakes (King & Mace 1974), within-lake comparisons are often overlooked when monitoring programs are developed (Underwood 1996).

Tourist impacts on oligotrophic lakes can be viewed at a range of temporal and spatial scales. Direct nutrient additions from tourists can be regarded as temporally variable acute disturbances (pulse of inputs at peak tour times, with high visitation in a particular area), or long-term chronic disturbances (consistent and/or increasing visitation levels across sites) (Underwood & Kennelly 1990, Underwood 1996, Lake 2000). Given this broad range of spatial and temporal impacts, it is likely that traditional monitoring tools will not adequately measure impacts at all of the important temporal and spatial scales. Significantly, the most common monitoring measures (analyses of water column nutrient and phytoplankton chlorophyll *a* concentrations) are not likely to facilitate the early detection of tourist impacts in aquatic environments. This is particularly important in wilderness areas, as resource managers are typically charged with maintaining the current status of aquatic environments and isolating threats to their ecological health, rather than just reporting on changes to lake trophic status that have already occurred.

The relevance of epilimnetic measures of nutrient and chlorophyll *a* concentrations in detecting tourist impacts is further questioned when we consider the spatial scales at which tourists are active. For example, Liddle and Scorgie (1980) noted that in aquatic ecosystems, most human impacts occur within the littoral zone. Furthermore, since beds of aquatic macrophytes are often more productive than phytoplankton communities in the open water of shallow lakes (Loeb et al. 1983), nutrient additions are likely to elicit strong growth responses from primary producers within these shallow shoreline reaches (Havens et al. 1999).

Therefore, in shallow lake systems where nutrient additions from tourists are likely to be spatially focussed within each lake, a more sensitive monitoring approach might be to measure nutrient and chlorophyll *a* concentrations at disturbed (tourist access points) and reference (inaccessible) points within the littoral zone. This approach promises to provide useful information relating to system responses to tourist nutrient inputs, before irreversible, lake-wide responses to nutrient additions are detected (Henderson-Sellers & Markland 1987, Underwood & Kennelly 1990, Underwood 1996).

The aim of this study was to determine whether tourist activities influence nutrient status and algal growth (phytoplankton and periphyton) on small spatial (within lake) and temporal (over the course of one summer) scales in the littoral zones of perched dune lakes. In addition, we aimed to determine which nutrients, if any limit the production of littoral zone phytoplankton and periphyton communities, as this may guide future management plans regarding the delivery of nutrients to these systems.

Methodology

Study Area

Fraser Island is situated off the Queensland coast between 24° 35' - 26° 20'S and 152° 45' - 153° 30'E and is the largest sand island in the world (over 160 000 hectares: Anon 1999, UNESCO 2001, Fig. 1). Annual rainfall on the island exceeds 1800 mm and the subtropical

climate is strongly influenced by the Pacific Ocean to the east, with mean daily air temperatures ranging from 14.1°C in winter, to 28.8°C in summer (Anon 1999).

Lake Selection

In 1999, five of the most popular (for recreation) perched dune lakes on Fraser Island were selected for detailed study. Lakes were chosen to ensure representation of both clear and tannin-stained (humic) systems, since dissolved organic carbon (and subsequent light attenuation) is likely to influence system responses to nutrient additions (Vinebrooke & Leavitt 1998, Williamson et al. 1999, Klug & Cottingham 2001). Three of the lakes chosen (Basin, McKenzie and Birrabeen – Fig. 1) were clear (with very low tannin concentrations) and two (Jennings and Boomanjin – Fig. 1) were tannin-stained (Table 1, Hadwen et al. 2003). The lakes also differed in their ambient water chemistry and fish species compositions (Table 1).

For all of the lakes on Fraser Island, tourist access points are discrete and are often demarcated by heavy trampling both along tracks and within the littoral zone (Table 2, Hadwen et al. 2003, Hadwen & Arthington 2003). Access is often impossible in other areas around the lake and as a result, tourist activities both in and around the lakes are concentrated at the existing access points (Hadwen & Arthington 2003). In contrast, areas where tourists cannot gain access to the lake are generally heavily forested. In these reaches, the littoral zones are characterised by dense monospecific stands of the reed, *Lepironia articulata* (Cyperaceae, Table 2). As a result, we were able to identify a single disturbed (access point) site and a comparable reference site (rarely visited by tourists) in each of the study lakes. The reference site in each lake was chosen to be as physically, chemically and biologically similar as possible to the disturbed site within the lake (see Table 2), to ensure that statistical noise associated with the location of sites (e.g. aspect and exposure to prevailing winds) was minimised. Furthermore, the disturbed and reference sites were chosen to be geographically

distant, with reference sites often only accessible by boat. Samples were collected over summer, in December 1999, February 2000 and March 2000.

Ambient Nutrient Concentrations

Littoral zone water samples were collected in reverse osmosis washed polyethylene bottles for nutrient analyses. Unfiltered samples were collected for quantification of total nitrogen (TN) and total phosphorus (TP) concentrations, whilst samples for the quantification of soluble reactive phosphorus (SRP), ammonium (NH_4^+) and nitrogen oxides (NO_x^-) were filtered through a 0.45 μm glass-fibre filter to remove suspended solids. Samples were immediately placed on ice in the dark and were frozen within 5 hours of collection. All nutrient samples were analysed using standard methods (American Public Health Association 1995) at the Scientific Services Division of Queensland Health in Brisbane, Australia.

Phytoplankton Chlorophyll *a*

A hand pump and filter apparatus was used to filter triplicate water samples (collected from the littoral zone sites of each of the lakes) through 0.7 μm glass-fibre filter papers. Each filter paper was rolled up and stored in a centrifuge tube wrapped in aluminium foil. All samples were stored on ice in the dark before being frozen.

Chlorophyll *a* analyses followed the standard methods of Parsons et al. (1984). Chlorophyll *a* was extracted overnight at 4°C in 90% v/v acetone, before being placed in an ultra-sonic bath for 1 minute and centrifuged for 3 minutes at 3000g. Sample absorbances were measured using a 'Shimadzu UV-1601' spectrophotometer with acidification for phaeo-pigment corrections. Chlorophyll *a* concentrations were standardised according to the volume of water filtered to attain the samples.

Periphyton Chlorophyll a

At each sampling site, three reeds (*Lepironia articulata* - Cyperaceae) were collected from the littoral zone growing at a depth of 1 metre. Reeds were carefully cut near their base and the 30 – 70 cm long submerged sections of the reeds (complete with attached algae) were placed in individually labeled zip-lock bags for storage. Samples were stored in the dark on ice, before being frozen for transportation back to the laboratory.

In the laboratory, periphyton was gently scraped from individual reed stems into glass beakers using a fine-toothed brush. The resultant algal samples were made up to 200 ml of distilled water and then filtered onto 0.7 µm glass fibre filter papers. Samples were thereafter processed according to the methods outlined for phytoplankton chlorophyll *a*. Preliminary measurements found very little variation in *Lepironia articulata* stem diameters, so chlorophyll *a* concentrations were scaled according to the length of reed scrubbed.

Artificial Reeds

In November 1999, 10 artificial ‘reeds’ were deployed in the disturbed and reference sites of each lake to assess the accrual of periphyton biomass over the course of the 1999-2000 summer. The one metre long artificial reeds were constructed from inert Ultra-High Molecular Weight (UHMW) polyethylene rods and covered with 100 µm mesh sleeves (fastened using cable ties) to provide a substrate for algal attachment. In all cases, rods were deployed in approximately 1.5 metres of water in the littoral zone and spaced at least 5 metres apart along a stretch of 50 metres of shoreline. Wherever possible, the artificial reeds were concealed amongst beds of *Lepironia articulata* (Cyperaceae), to ensure that they were not visible to tourists.

After six weeks of *in situ* incubation, mesh sleeves were carefully removed from the artificial reeds. Collected sleeves were placed in individually labeled zip-lock bags and stored in the

dark on ice, before being frozen for transportation. In the laboratory, attached algal material was carefully removed from the mesh sleeves using a scalpel blade and fine toothed brush. Analysis of chlorophyll *a* concentrations, calculated per unit area of mesh sleeve, followed the methods described for periphyton samples.

Algal Bioassays of Nutrient Limitation

During the 2000-2001 summer, experimental algal bioassays were conducted in reference sites of each lake (to minimise potential disturbance from tourists) to determine which (if any) nutrients limited algal growth. Bioassays (for both phytoplankton and periphyton) presented algal communities with nutrient additions in four experimental treatments. Nutrients were added at concentrations determined on the basis of known maximum concentrations in perched lake systems (Outridge et al. 1989, Arthington et al. 1990a) and with regard to the likely nutrient sources from tourists (e.g. urine – Strasinger 1994). The control treatment (C) had no nutrients added, the nitrogen treatment (N) had 50 $\mu\text{g L}^{-1}$ of ammonium nitrate (NH_4NO_3) added, the phosphorus treatment (P) had 10 $\mu\text{g L}^{-1}$ of sodium phosphate (Na_2HPO_4) added and the nitrogen + phosphorus treatment (N+P) had additions of both the ammonium nitrate and sodium phosphate ($\text{NH}_4\text{NO}_3 + \text{Na}_2\text{HPO}_4$).

Phytoplankton bioassays were conducted in early summer (November 2000). In each lake, twenty 6-litre carboys (clear plastic containers) were filled with 5 litres of unfiltered lake water and randomly assigned to one of the four nutrient treatments. Once treated and sealed, carboys were floated in approximately 1 metre of water in the littoral zone. After seven days of *in situ* incubation, carboys were collected and samples were obtained for chlorophyll *a* analyses by filtering measured quantities of carboy contents onto 0.7 μm glass fibre filter papers using a hand pump and filter apparatus. Due to a storm event in November 2000, all of the Lake Boomanjin carboy samples were lost.

Periphyton bioassays were undertaken using nutrient diffusing substrates. Small (300 ml) polyvinyl chloride pots were used to house nutrient-enriched agar. Five replicate pots were randomly assigned to each of the four experimental treatments (C, N, P and N+P). Pots were filled with a mixture of 2% bacteriological agar (Oxoid # 1) and nutrients at the same concentrations used in the phytoplankton bioassays (see above). The aperture of each pot was covered with fine (100 μ m) mesh to facilitate algal attachment and sampling (Mosisch et al. 1999). Each mesh-covered agar pot was individually wrapped in plastic wrap and refrigerated prior to transportation and all pots were kept on ice until they were deployed.

Periphyton algal bioassays were conducted for two reasons. First, control pots were deployed in disturbed and reference sites, to identify differences in algal biomass attributable to tourist recreational use of the study lakes. Second, all treatments were deployed in the reference site in each lake to examine which nutrients, if any, were limiting to periphyton growth. As a result, 25 agar pots were deployed in each lake, with 5 control pots in the disturbed site and 20 [5 Control, 5 Nitrogen added, 5 Phosphorus added and 5 Nitrogen + Phosphorus added] in the reference site). Pots were placed at least 1 metre apart and were deployed in five blocks of four, with one replicate from each of the four treatments randomly assigned in each block. Each block ran parallel to the shoreline of the lake and pots were placed in approximately 30 cm of water. Blocks were spaced 5 m apart and the five blocks consequently spanned in excess of 45 metres of the shoreline. Pots were secured to the lake bottom using large (30 cm) plastic pegs and were deployed in November 2000 and retrieved in December 2000.

Significant losses of agar from pots in Lake McKenzie and Lake Birrabeen, apparently as a consequence of feeding by turtles (personal observation), meant that the experiment had to be re-run in these systems in January 2001. This second experiment used small turtle exclusion cages (20 cm x 20 cm x 20 cm) made out of PVC gardening mesh (mesh diameter 1cm), to inhibit interference by turtles. The cages were secured around the pots using large (30 cm) plastic pegs.

In both runs, collected mesh lids were placed in individually labeled zip-lock bags and stored on ice in the dark, before being frozen for transportation back to the laboratory. In the laboratory, mesh lids were trimmed to 45 cm², to ensure that only the surface that was in direct contact with the agar-nutrient mix was analysed for chlorophyll *a* concentrations (Mosisch et al. 1999). Samples were thereafter processed for chlorophyll *a* using the methods outlined above.

Statistical Analyses

Ambient Nutrients and Chlorophyll *a* Concentrations

Statistical analyses were conducted using the SAS computer package (SAS Institute 1989). All data (ambient nutrient and phytoplankton and periphyton chlorophyll *a* concentrations as well as phytoplankton and periphyton bioassays) were analysed using the ANOVA design of McKone and Lively (1993) to examine differences between sites/treatments. This ANOVA model was specifically designed to enable the detection of within-site differences across multiple sites (McKone & Lively 1993). Site (disturbed versus reference) effects within each sampling period (December 1999, February 2000 and March 2000) were consequently analysed separately for each lake, since the primary goal of the analysis was to detect differences at this within-lake spatial scale rather than to determine differences between lakes.

Results

Ambient Nutrients

There was very little variation in ambient nutrient concentrations between disturbed and reference sites in each of the five lakes (Fig. 2) and with one exception, there were no significant differences. In that instance, total phosphorus concentrations in the disturbed site of Basin Lake were significantly higher than those in the reference site in February 2000 ($p < 0.01$, Fig. 2D).

Ammonium concentrations in Basin Lake increased with time to be in excess of $200 \mu\text{g L}^{-1}$ (an order of magnitude greater than the concentrations recorded in all other lakes) in March 2000 (Fig. 2A). Lake McKenzie ammonium concentrations were stable over the course of the summer, whilst small declines were observed in Lake Birrabeen, Lake Jennings and Lake Boomanjin (Fig. 2A).

Nitrogen oxide concentrations in Lake Boomanjin were up to four times greater than those measured in any of the other lakes sampled, with concentrations in excess of $40 \mu\text{g L}^{-1}$ in some samples (Fig. 2B). In Lake Birrabeen, a significant trend of decline in nitrogen oxide concentrations was observed across the course of the summer ($p < 0.01$, Fig. 2B). Significant differences in temporal concentrations were also found for Basin Lake ($p = 0.01$) and Lake Boomanjin ($p = 0.03$), although these results appear to reflect stochastic temporal fluctuations rather than directional trends over summer (Fig. 2B).

Total nitrogen concentrations in the heavily stained perched lakes, Jennings and Boomanjin, were generally higher than those in the clear perched lakes, McKenzie and Birrabeen (Fig. 2C). However, the relatively clear Basin Lake had elevated total nitrogen concentrations due to the extremely high ammonium concentrations (Fig. 2A). There were significant temporal differences in total nitrogen concentrations in Lake Jennings ($p < 0.01$) and Lake Boomanjin ($p < 0.01$), with both lakes having lower concentrations at the end of summer than at the beginning (Fig. 2C).

Total phosphorus concentrations in each lake were always low, highlighting their oligotrophic status (Fig. 2D). Temporal variation in total phosphorus concentrations was similarly small, with a significant difference only in Lake Boomanjin ($p < 0.01$) where total phosphorus concentrations fell across the course of the summer (Fig. 2D).

Phytoplankton Chlorophyll *a*

Phytoplankton chlorophyll *a* concentrations were found to be highly variable (Fig. 3A) and as a consequence, only one significant difference between disturbed and reference site chlorophyll *a* concentrations was recorded, (Lake McKenzie in March 2000 $p=0.02$). In all but Lake McKenzie ($p=0.11$), there was significant temporal variation in phytoplankton chlorophyll *a* concentrations (Boomanjin $p=0.01$, Birrabeen $p=0.01$, Basin $p=0.01$, Jennings $p=0.01$, $df=2$). In Basin Lake and Lake Jennings, this temporal variability was largely due to a trend of decreasing chlorophyll *a* concentrations over the summer (Fig. 3A). In contrast, the significant temporal variation in phytoplankton chlorophyll *a* concentrations for Lake Birrabeen and Lake Boomanjin were presumably due to mid-summer (February) declines prior to increases again at the end of summer (March) (Fig. 3A).

Periphyton Chlorophyll *a*

With few exceptions, periphyton chlorophyll *a* concentrations in all lakes were lowest in the December 1999 samples and highest in the March 2000 samples (Fig. 3B). In Basin Lake, Lake Jennings and Lake Boomanjin, this trend of increasing periphyton chlorophyll *a* concentrations over the course of the summer was consistent across both the disturbed and reference sites (Fig. 3B). However, for the clear lakes, McKenzie and Birrabeen, periphyton chlorophyll *a* concentrations increased in disturbed sites over the course of the summer, whilst remaining relatively low in reference sites (Fig. 3B).

Across the course of the 1999-2000 summer, significant differences in periphyton chlorophyll *a* concentrations in disturbed and reference sites were found in all but Lake McKenzie (Basin: December $p=0.03$, February $p=0.01$, Birrabeen: February $p=0.01$, March $p=0.01$, Jennings: February $p=0.05$, Boomanjin: February $p=0.05$, $df = 1$). Whilst differences in periphyton chlorophyll *a* concentrations in Lake Birrabeen and Lake Jennings reflected the expected trend of higher concentrations in disturbed sites, all of the significant differences in Basin

Lake and Lake Boomanjin were the result of higher periphyton chlorophyll *a* concentrations in the reference site than in disturbed site (Fig. 3B).

Artificial Reeds - Disturbed versus Reference Sites

Across all lakes, measurably quantities of periphyton had accrued on artificial reeds during their four week incubation (Fig. 4A). However, the variable quantity of periphyton resulting in a significant difference in chlorophyll *a* concentrations between lakes in the ANOVA model (Fig. 4A, $p=0.01$, $df=4$). However, as this variability was typically lake-wide in its extent, Lake McKenzie was the only system to display a significant difference between disturbed and reference site chlorophyll *a* concentrations. As predicted, the artificial reeds in Lake McKenzie were found to be more heavily colonised by algae in the disturbed site than in the reference site ($p=0.01$, $df=1$).

Agar Control Pots

Presumably as a consequence of their visibility to tourists, the retrieval of control pots deployed in disturbed sites in November 2000 was incomplete, particularly in the most popular clear lakes, Lake McKenzie and Lake Birrabeen. These losses inhibited statistically powerful evaluation of chlorophyll *a* concentrations between reference and disturbed sites. Nevertheless, there was a tendency for periphyton chlorophyll *a* concentrations to be higher in disturbed sites than in reference sites in Basin Lake, Lake McKenzie, Lake Birrabeen and Lake Boomanjin (Fig. 4B). This pattern was not observed in the less frequently visited Lake Jennings, where disturbed and reference site chlorophyll *a* concentrations were similar (Fig. 4B).

Nutrient Limitation of Phytoplankton

There was considerable variability in phytoplankton community responses to nutrient additions (Fig. 5A). In Basin Lake, treatment responses differed significantly ($p=0.01$, $df=3$), with higher chlorophyll *a* concentrations in the P and N+P treatments, relative to C and N

treatments (Fig. 5A). In Lake McKenzie, a significant treatment effect ($p=0.01$, $df=3$) was generated by the higher chlorophyll *a* concentration recorded in the N+P treatment relative to that recorded in the C treatment. However, chlorophyll *a* concentrations in the N treatment were intermediate to those in the C and N+P treatments (Fig. 5A). Furthermore, since the P treatment did not differ from the C treatment, most of the response in the N+P treatment can presumably be attributed to the added nitrogen rather than the added phosphorus. Lake Birrabeen chlorophyll *a* concentrations showed little or no response (relative to C) to the N or P treatments (Fig. 5A). However, a significant ($p<0.01$, $df=3$) response in the N+P was observed. Chlorophyll *a* concentrations in the Lake Jennings phytoplankton bioassays exhibited substantial, though not significant, responses to both the N and P treatments (Fig. 5A). For this lake the significant treatment effect ($p=0.01$, $df=3$) was due to the extremely high chlorophyll *a* concentrations in the N+P treatment (Fig. 5A).

Nutrient Limitation of Periphyton

Periphyton chlorophyll *a* responses to the four treatments differed between the lakes (Fig. 5B, $p=0.01$, $df=4$), but there were no detectable differences between control and nutrient addition treatments in any of the lakes (Fig. 5B). Despite the absence of significant results, there were some trends in chlorophyll *a* responses to nutrient treatments (Fig. 5B). In Basin Lake, chlorophyll *a* concentrations were highest in the P treatment and in Lake McKenzie, all treatments elicited a substantial algal growth response with uniformly high concentrations of chlorophyll *a* being recorded (Fig. 5B). Periphyton chlorophyll *a* concentrations in Lake Birrabeen, Lake Jennings and Lake Boomanjin were low and consistent across all treatments.

Discussion

Tourists are unlikely to have measurable effects on ambient nutrient concentrations in oligotrophic perched dune lakes in the short term, unless measurements are taken immediately following the departure of large groups at disturbed sites (e.g. the significant result for phosphorus in Basin Lake in February 2000). This failure to detect changes in open water

ambient nutrient concentrations is likely due to the rapid assimilation of nutrients by algal communities in the littoral zone (Loeb et al. 1983, McCormick & Stevenson 1998). As a result, historical monitoring programs focussing on detecting changes in ambient open water nutrient concentrations may not be appropriately scaled to assess the effects of tourist activities on oligotrophic lakes.

Whilst ambient nutrient concentrations are of limited use in detecting within lake impacts of tourist activities, recent evidence suggests that they may be useful in detecting long-term responses to tourist impacts (Hadwen et al. 2003). However, system-wide responses to impacts detected over a period of years are likely to represent irreversible trends (Henderson-Sellers & Markland 1987, Larson 1996) and are therefore, of limited use in proactive management programs in natural wilderness areas like Fraser Island.

Traditional monitoring programs on Fraser Island have not detected recent changes in ambient nutrient concentrations (Hockings 1999). Furthermore, if the data presented in this paper were to be analysed as a series of between-lake comparisons, it is likely that the conclusions would be that all of the lakes examined are presently oligotrophic and as such, tourist impacts are (and have been) minimal. Such an interpretation of water quality data highlights the limitations of using water column measurements of ambient nutrient and phytoplankton chlorophyll *a* concentrations for monitoring changes in oligotrophic systems (Havens et al. 1996, McCormick & Stevenson 1998). In addition, despite possible statistical concerns with comparisons between samples collected from just one disturbed and one reference site in heavily visited lakes, it is obvious that impacts from tourist activities are most likely to occur (and be detected) at this within-lake scale of examination. As a result, it is more important to ensure that representative samples are collected from each site to ensure the most logical spatial and temporal comparisons, rather than to attempt to select multiple sites within each system.

This proposed within-lake focus for monitoring on Fraser Island is further supported by the results from the phytoplankton bioassays, which suggest that whilst all lakes exhibited significant responses to at least one of the nutrient addition treatments (N, P or N+P), the magnitude and direction of responses were lake-specific. This variability in algal response to nutrient additions is particularly interesting given that the most northern and southern lakes in the series were separated by less than 30 kilometres (Fig. 1). Such drastically differential algal responses to nutrient additions have been rarely reported for lakes within the same region (White 1983, Elser et al. 1990). Furthermore, since the catchment characteristics of Fraser Island lakes are very similar (Arthington et al. 1990b, Sinclair 2000, UNESCO 2001), there is no clear *a priori* reason to expect this degree of lake-specific response to nutrient additions. However, since perched lake systems represent discrete hydrological units (Bayly 1964, James 1984, Timms 1986), it is possible that differences in algal responses to nutrient additions may be governed by differences in algal species composition (Arthington *et al.* 1990a) and/or food web structure (Hansson 1992, Flecker et al. 2002, Hillebrand 2002).

Despite the lack of statistically significant responses to nutrient treatments in the periphyton bioassays, similar trends to those reported for the phytoplankton bioassays were apparent in some of the lakes. For example, the P treatment elicited a strong algal growth response in Basin Lake and periphyton chlorophyll *a* responses tended to be greater in the N+P treatment in Lake McKenzie. In contrast, the periphyton communities in Lake Jennings, Lake Boomanjin and Lake Birrabeen did not follow the patterns observed in phytoplankton bioassays. As such, it is likely that other variables, such as light and/or grazing pressure, may regulate periphyton algal biomass in these systems (Rhee & Gotham 1980, Gresens 1995, Hillebrand & Kahlert 2001).

In Lakes Jennings and Boomanjin, the treatment-independent and comparatively low periphyton chlorophyll *a* concentrations suggest that nutrients are not primarily responsible for limiting periphyton production in these lakes (Rhee & Gotham 1980, Mosisch et al. 1999,

Hillebrand & Kahlert 2001). Instead, since both of these systems are heavily stained by tannins (Table 1, Bowling 1988, Arthington et al. 1990a, Hadwen et al. 2003), it is likely that light attenuation through the water column (and subsequent reduction of the photic zone) strongly regulates periphyton production (van Dijk 1993, Marks & Lowe 1993). In contrast, the comparatively high periphyton biomass across all treatments in the clear Lake McKenzie suggests that light does not limit periphyton production in this system. In fact, the results from monitoring, artificial reed and agar pot experiments suggest that in Lake McKenzie, periphyton grows wherever a surface is available for colonisation. Periphyton biomass may, therefore, be partially substrate limited in this system (as in other systems – e.g. Cattaneo & Amireault 1992, Smoot et al. 1998). In contrast to the high periphyton biomass in Lake McKenzie, Lake Birrabeen chlorophyll *a* concentrations were consistently low across all treatments. Interestingly, a recent food web study suggests that periphyton is a more significant component of consumer diets in Lake Birrabeen than it is in the other lakes examined in this study (Hadwen unpublished data). As a result, the low periphyton chlorophyll *a* concentrations in Lake Birrabeen may reflect comparatively high rates of top-down regulation by consumers (Hunter 1980, Mazumder et al. 1989, France et al. 1991).

Periphyton has increasingly been advocated as a useful indicator of ecosystem health in a wide variety of aquatic ecosystems (Marks & Lowe 1993, Havens et al. 1996, McCormick & Stevenson 1998, Bourassa & Cattaneo 2000). Similarly in this study, it seems likely that the within-lake comparison (disturbed site versus reference site) of ambient periphyton chlorophyll *a* concentrations is a potentially useful tool for detection of tourist impacts in oligotrophic perched dune lakes on Fraser Island. Given the ease of sampling this material in the littoral zone, the inclusion of this component in a monitoring program would be comparatively inexpensive and potentially more informative than analyses of nutrient concentrations. In addition, assessment of chlorophyll *a* concentrations on inert artificial substrates deployed throughout a lake will likely enable the calculation of an index of tourist pressure, upon which lake-specific management plans could be developed to ensure that

ongoing tourist activities do not threaten the trophic status of the system. Given that excessive periphyton biomass can adversely influence tourist perceptions and motivations (Buckley & Pannell 1990), ongoing monitoring (and management) of this biological component of perched dune lakes may also ensure the sustainability of tourist activities in perched dune lakes. In this regard, long-term trends in lake-wide increases in periphyton chlorophyll *a* concentrations are likely to be useful in alerting resource managers to irreversible increases in lake trophic status and sustainability of tourist use of these systems for recreational purposes.

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Table 1. Summary of mean ($\pm$ S.E.) water quality characteristics and fish species compositions in the five study lakes. Chemical data from Hadwen et al. (2003) and fish species data from personal observations (Hadwen, unpublished data).

LAKE	pH	Tannins ($\mu\text{g/L}$)	TN ($\mu\text{g/L}$)	TP ($\mu\text{g/L}$)	NH_4^+ ($\mu\text{g/L}$)	NO_x^- ($\mu\text{g/L}$)	Chl a ($\mu\text{g/L}$)	Fish species present
Basin	5.00 (0.01)	200 (0)	475 (5)	2.5 (0.5)	305.0 (5)	6.0 (0)	0.25 (0.03)	None
McKenzie	4.82 (0.02)	100 (0)	70 (0)	2.0 (0.0)	7.0 (0)	32.0 (0)	0.09 (0.03)	<i>Mogurnda adspersa</i> (Gobiidae)
Birrabeen	4.64 (0.02)	100 (0)	165 (5)	2.0 (0.0)	25.0 (0)	79.0 (0)	0.12 (0.04)	<i>Hypseleotris galii</i> (Gobiidae)
Jennings	4.44 (0.01)	1450 (50)	200 (0)	2.0 (0.0)	2.5 (5)	2.0 (0)	0.10 (0.00)	<i>Hypseleotris galii</i> (Gobiidae)
Boomanjin	4.60 (0.05)	1650 (50)	185 (25)	2.5 (0.5)	3.0 (0)	2.0 (0)	0.14 (0.01)	<i>Hypseleotris galii</i> (Gobiidae) <i>Melanotaenia duboulayi</i> (Melanotaeniidae) <i>Tandanus tandanus</i> (Plotosidae)

Table 2. Summary of aspect and botanical characteristics of Disturbed and Reference sites in five Fraser Island dune lakes.

Lake	Basin		McKenzie		Birrabeen		Jennings		Boomanjin	
Site	Disturbed	Reference	Disturbed	Reference	Disturbed	Reference	Disturbed	Reference	Disturbed	Reference
Aspect	E	ENE	S.E.	S.E.	ESE	E	S.E.	S.S.E.	E	S.E.
Macrophyte Species	<i>Lepironia articulata</i>	<i>Lepironia articulata</i>	<i>Lepironia articulata</i>	<i>Lepironia articulata</i>	<i>Lepironia articulata</i>	<i>Lepironia articulata</i>	<i>Lepironia articulata</i>	<i>Lepironia articulata</i>	<i>Lepironia articulata</i>	<i>Lepironia articulata</i>
Macrophyte Percent Cover	50%	95%	20%	98%	60%	95%	85%	100%	45%	80%
Dominant Riparian Vegetation Species	<i>Eucalyptus</i>	<i>Eucalyptus</i>	<i>Eucalyptus</i> <i>Banksia</i> <i>Melaleuca</i>	<i>Eucalyptus</i> <i>Banksia</i> <i>Melaleuca</i>	<i>Casuarina</i> <i>Melaleuca</i>	<i>Casuarina</i> <i>Melaleuca</i>	<i>Eucalyptus</i> <i>Casuarina</i>	<i>Eucalyptus</i> <i>Casuarina</i>	<i>Eucalyptus</i> <i>Melaleuca</i>	<i>Melaleuca</i>

Figure 1. Map of Fraser Island, Australia. Inset shows detailed map of study area and the five lakes chosen for the study.

Figure 2. Mean ($\pm$ s.e.) ambient nutrient concentrations in Disturbed and Reference Sites in five Fraser Island perched dune lakes in December 1999 (D), February 2000 (F) and March 2000 (M). A – Ammonium, B – Nitrogen Oxides, C – Total Nitrogen and D – Total Phosphorus.

Figure 3. Mean ($\pm$ s.e.) Chlorophyll *a* concentrations in Disturbed and Reference sites in five Fraser Island perched dune lakes in December 1999 (D), February 2000 (F) and March 2000 (M). A – Phytoplankton Chlorophyll *a*, B – Periphyton Chlorophyll *a* collected from the *Lepironia articulata* reeds within the littoral zone.

Figure 4. Mean ($\pm$ s.e.) Chlorophyll *a* concentrations from periphyton growing on artificial substrates in Disturbed and Reference sites in five perched dune lakes. A – artificial reeds deployed in December 1999 and retrieved in February 2000, B – control agar pots deployed in November 2000 and retrieved in December 2000 (Basin, Jennings and Boomanjin) and deployed in January 2001 and retrieved in February 2001 (McKenzie and Birrabeen).

Figure 5. Mean ($\pm$ s.e.) Chlorophyll *a* concentrations from algal bioassay nutrient addition treatments in five perched dune lakes. A – Phytoplankton Bioassays, November 2000, B – Periphyton Bioassays, December 2000 (Basin, Jennings and Boomanjin) and February 2001 (McKenzie and Birrabeen).