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Aerial Herbicide Spray to Control Invasive Water Hyacinth (*Eichhornia crassipes*): Water Quality Concerns Fronting Fish Occupying a Tropical Floodplain Wetland

N. J. Waltham[1] and S. Fixler[2]

Abstract
Water hyacinth (*Eichhornia crassipes*) is an aquatic weed degrading tropical floodplains everywhere. On the Burdekin floodplain, northern Australia, it is widespread and contributes to poor water quality, specifically hypoxia which contributes to voluminous wetland fish kills each summer. Removing weeds have focused on applying herbicides using aerial spraying, though restoration success is not monitored. Here, we investigated four aerial spray applications scheduled between November 2013 (Year 1, November 2013 to November 2014) and November 2015 (Year 2, November 2014 to November 2015) in Lochinvah wetland (35 ha wetland, Burdekin floodplain). Using high-frequency (20 min) loggers, dissolved oxygen (DO%) was tracked, which revealed that concentrations were similar before and several weeks after a spray application (independent *t* test, $p > 0.01$, except spray application 2, $p = 0.06$). More interestingly, aquatic weed coverage was low (5% of wetland) during Year 1 and DO had a typical diurnal cycle (20% to 130%). In contrast, low wetland flushing in Year 2 and high weed coverage (80% coverage) combined to increase DO hypoxia exposure risks for fish, with nearly 100% of the logging time failing acute and chronic values known for local fish. The Year 2 weed cover also increased water temperature exposure risk (twofold increase), which was unexpected and which means that fish probably could access cool, deeper, water refugia more frequently compared with Year 1. Controlling aquatic weeds using aerial spraying seems to have minimal risk for fish when cover is low; however, the proliferation of aquatic weeds and spraying has deleterious impact on available oxygen for fish.

Keywords
wetland restoration, freshwater, tropical ecology, dissolved oxygen, critical trigger values

Introduction

We are losing in the order of 95 km^2 of global floodplain wetlands each year (Mitsch & Gosselink, 2016), and this is not likely to slow with continuing urban and industrial expansion in many coastal locations (Davidson, 2014). Because coastal wetlands provide essential habitat for many aquatic species, managers are moving toward implementing large-scale programs to repair and restore them (Barbier, 2013). Restoration examples exist in North America (e.g., Repair America's Estuaries, www.estuaries.org), and plans are underway in other places such as China where $1 billion will be invested toward more than 50 large programs by 2030—intended to restore and recreate wetlands to mitigate poor water quality and provide habitat for local wildlife species (An et al., 2007). While wetland restoration is vital (Creighton, Hobday, Lockwood, & Pecl, 2016), access to relevant and appropriate scientific data demonstrating biodiversity and conservation return for the investment is lacking (Zedler, 2016).

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Fish are commonly studied animal in coastal wetlands with particular attention on distribution and composition in response to altered water quality and habitat conditions (Arthington, Godfrey, Pearson, Karim, & Wallace, 2015; Baran, Van Zalinge, & Bun, 2001). Dissolved oxygen (DO%) and temperature are the most studied determinants given both are directly linked to acute and chronic effects on fish growth, reproduction, and overall fitness (Heath, 1995). This is particularly pertinent for wetlands, which are increasingly hindered by excessive aquatic plant growth because of excessive nutrients and increased hydrology residents time (Villamagna & Murphy, 2010), and where low DO events are linked to thick beds of submerged aquatic vegetation cover, particularly at dawn, but also with plant decomposition (Kaenel, Buehrer, & Uehlinger, 2000; Miranda & Hodges, 2000). The densely packed canopy of floating leaves will not only lower light by shading and reduce water temperature (Greenfield et al., 2007) but will also prohibit gas exchange, thereby suppressing DO further and increasing asphyxiation risks to local wetland fauna (Butler & Burrows, 2007). Controlling excessive aquatic weeds is challenging, and the range of possible control methods (e.g., harvesting, spraying with chemicals, and biological control) can become expensive both for the initial control treatment and on-going maintenance treatment (Villamagna & Murphy, 2010; Perna & Burrows, 2005). Management costs in the United States each year are close to \$100 million (USD; Pimentel, Lach, Zuniga, & Morrison, 2000), which raises questions relating the responsible parties to pay for these restoration works (Holl & Howard, 2000). Rising evidence suggests that prevention of aquatic weed spread is cheaper than control (Hussner et al., 2017).

Australia faces a legacy of degraded coastal wetland habitats succeeding more than 200 years of urban or industrial development and agricultural intensification (Creighton et al., 2016). The most critical region is the Great Barrier Reef (GBR) lagoon, a World Heritage Area (GBRWHA) and National Marine Park, protected under an assortment of international agreements and national and state legislation and policies. On-going poor water quality subsequent of catchment agricultural runoff and intensification (Brodie & Waterhouse, 2012), and the loss of coastal habitats concomitant with coastal development (Waltham & Sheaves, 2015), has placed major pressure on reef resilience (Department of Environment Heritage and Protection [DEHP], 2016). Approximately 60% of the pre-European (1770) GBR coastal wetlands have been modified or drained completely for agricultural intensification or urban and industrial expansion (Great Barrier Reef Marine Park Authority, Commonwealth of Australia [GBRMPA], 2009). Conservation and repair of the GBR coastal wetland ecosystems' have come into focus following media converging on the point that the reef health and resilience has been compromised, particularly around major agricultural centers (DEHP, 2016), with ecosystem protection and restoration a key performance measure in the development of long-term strategic planning policies (GBRMPA, 2015).

The Burdekin catchment is Australia's largest sugarcane production district. Concomitant with on-going expansion of sugarcane since the 1980s, the Queensland Government regional ecosystem mapping estimates that 23% of freshwater wetland coastal ecosystems remain mostly in a fragmented state (GBRMPA, 2015), while those remaining have altered hydrology, poor water quality (O'Brien et al., 2016), and extensive invasive macrophytes and have barriers impeding fish migration (Burrows, Sheaves, Johnston, Dowe, & Schaffer, 2012; Perna & Burrows, 2005; Waltham & Davis, 2015). The Governments of Queensland and Commonwealth of Australia recognize the need to reverse the decline in water quality reaching the GBR (GBRMPA, 2009). As part of this response, funding for the "Delivering biodiversity dividends to the Barratta Creek, Burdekin Catchment" project was granted in 2013 with the aim of restoring ecological function into the creek or floodplain and the offshore Bowling Green Bay Ramsar wetland. While the broader project covered riparian tree planting, fire management, feral animal management, and extension training in land management for farmers, this study examined water quality and fish community occupying Lochinvah wetland, which was treated with herbicides to control invasive aquatic plants (particularly species listed under Australian legislation as Weeds of National Significance; WONS)—most notably the water hyacinth (*Eichhornia crassipes*) which is one of the world's most prevalent invasive aquatic plants (Montiel-Martínez, Ciros-Pérez, & Corkidi, 2015; Toft, Simenstad, Cordell, & Grimaldo, 2003; Villamagna & Murphy, 2010). This aquatic plant occurs in wetlands across northern Queensland, particularly the Burdekin floodplain because of permanent water and high nutrients (Perna & Burrows, 2005). Clearing this declared WONS has become a burgeoning obligation for local government agencies and natural resource management groups (Burrows et al., 2012), particularly in the lower Burdekin floodplain (Figure 1) where it has an extensive water distribution network (over 1,500 km linear of channels) delivering irrigation water to sugarcane farmers through the landscape, with the tailwater (rich in nutrients, sediments, and herbicides) then flowing via coastal wetlands to Bowling Green Bay (Davis et al., 2014). As such, wetlands (such as Lochinvah wetland) suffer poor water quality, including low dissolved oxygen, fluctuating water clarity, and are heavily infested with invasive aquatic weeds. Restoration and protection of the Barratta wetland complex and downstream RAMSAR wetland is

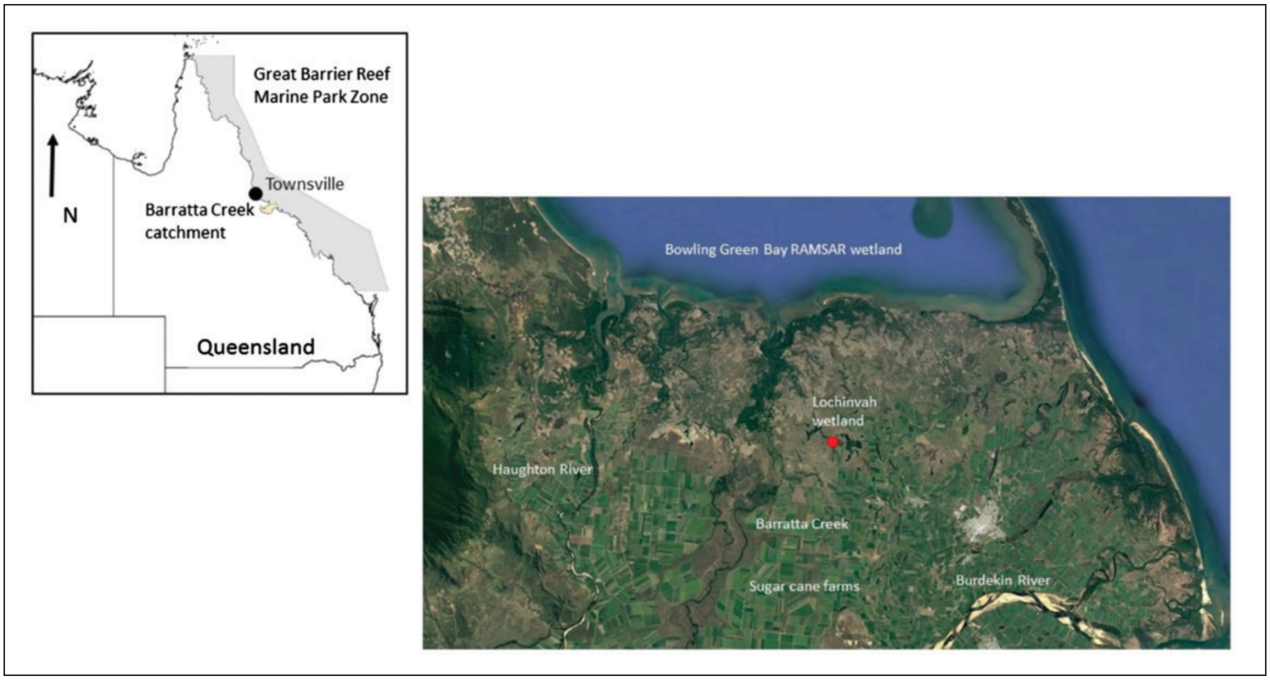

Figure 1. Location map of Barratta wetland, Burdekin River delta, north Queensland, Australia.

critical for floodplain protection (Burrows et al., 2012). In this study, we examine wetland water quality conditions before and after aerial spraying, and in doing so determine the exposure risk to fish, particularly DO conditions and water temperature. This study provides data for managers to evaluate wetland system repair projects in the GBR catchments, but these data have broader relevance given the global infestation of invasive aquatic weeds in coastal wetlands (Villamagna & Murphy, 2010).

Methods

Wetland Location, Climate, and Weed Treatments

The Lochinvah wetland is approximately 30 ha in size, located on the Burdekin floodplain complex, which is an extensive network of permanent and ephemeral palustrine wetlands (Figure 1). Land use across the floodplain is dominated by sugarcane, with some cattle grazing and tree crops (Davis et al., 2014). Rainfall is strongly seasonal with 90% of total annual rainfall occurring during the wet season (November and March). Mean annual rainfall on the floodplain (1887 to 2016) is 1,047 mm (Burdekin Shire Council gauge station, station number 33001). The wet seasons during this study were low compared with long-term data (386 mm, 2014/15; 386 mm; 133 mm 2015/16), within the 5% percentile of historical records for the region. Daily flow recorded at the Northcote gauge station (station number 119101A) on Barratta Creek reveals that this study coincided with an extended period of low rainfall, well below antecedent years prior to this study (Supplementary Material S1).

Four aerial spray applications were completed to target water hyacinth (*Eichhornia crassipes*): Spray 1—6 December 2013 (30 ha of wetland); Spray 2—30 June 2014 (35 ha); Spray 3—20 January 2015 (30 ha); Spray 3—17 May 2015 (30 ha); and Spray 4—25 September 2015 (30 ha). Each spray was completed in 4 hr using a helicopter (<10 m above the water surface), where glyphosate (and freshwater mixture—10 mg/L ratio) was sprayed as a mist across invasive aquatic plants in the wetland. Sprays 1 and 2 had low aquatic plant coverage (~5% wetland coverage), restricted to the wetland edges, while Sprays 3 and 4 aquatic plant growth was high (approximately 80% and 60%, respectively).

Water Quality Measurement

A calibrated multiprobe data logger (Hydrolab DS5, OTT Hydromet, CO, USA) was deployed in the near-surface water layer (0.2 m below the surface, attached to a surface float) to monitor diel cycling of temperature, pH, electrical conductivity, and dissolved oxygen (%) measured at 20 min intervals in the middle of the wetland (though only DO and temperature are examined here). The Hydrolab was deployed for approximately 2 to 3 weeks prior to each spray and remained in place until after (several weeks) spraying, or was deployed shortly before (between 1 and 2 weeks) and shortly after (1 to 4 weeks) spraying to record post spray water quality conditions. A Hydrolab was deployed before the spray application in January 2015; however, the logger was damaged by an estuarine crocodile (*Crocodylus porosus*) with all data (before and after spray) lost.

A second logger configuration (Pendant Temperature, Onset Corporation, Bourne, MA) was deployed to measure water temperature every 20 min with a logger position at a depth of 0.2 m below water surface (surface), and a second hobo logger deployed at 0.1 m above the bottom of the waterhole (bottom). Following Waltham and Sheaves (2017), the surface logger was attached on the underside of a 0.15-m buoy to shield the logger from direct sunlight, while the bottom logger was attached to a concrete block (so the sensor remained at approximately 1.5 m depth during the logging period). This rig was positioned adjacent to the Hydrolab and remained in place between 1 November 2014 and 28 February 2015.

Flow data recorded on the gauge in Barratta creek (recorded as part of the Queensland Department of Natural Resources program), adjacent to the wetland, highlight that the study coincided with a period of low flow (Supplementary Material S1). This low flow period has been shown to reduce the extent and duration of connection across the Burdekin floodplain more broadly (Waltham, 2017). The wetland depth at the logger site was 1.25 m at the beginning of the study, and although depth increased following small rain events, there was a net loss in wetland water level (0.95 m) on completing the study.

Data Analysis

The change in dissolved oxygen saturation in the wetland before and after each spray was examined separately using an independent t test. To accommodate for diel differences in water temperature, the daily average was calculated before and after each spray. These analyses were completed using SPSS (v23.0) using a significant probability of $p < .01$, and data were checked for normality (qq plots) and homoscedasticity (residual vs. fitted values) and required no transformation (Underwood, 1997).

Thermal exposure risk to aquatic fauna was examined using frequency distribution plots (Wallace, Waltham, Burrows, & McJannet, 2015; Wallace et al., 2017). We used the surface Hydrolab temperature data between 25 November 2013 and 25 January 2014 (<5% coverage), while hobo loggers (surface and bottom data) examined whether the exposure risk changed during a period of high coverage (80% coverage; 25 November 2014 and 25 January 2015). To ensure the two loggers were comparable, a regression was fitted for a separate logging period (8 August 2014 and 3 September 2014, $R^2 = 0.96$; $p < .01$). Based on the regression equation ($y = 1.00029x + 0.1723$; y is the hydrolab logger adjusted value and x is the hobo logger recorded temperature), hydrolab logger was adjusted accordingly, which helped to correct for the slight logger difference. Climate data were sourced from the Scientific Information for Land Owners database (https://www.longspaddock.qld.gov.au/silo/) where the mean of the daily ambient air temperature for the region was 31.2°C (±0.3 SE) and the mean humidity was 53% (±6.9 SE) during 22 November 2013 and 25 January 2014, while these were slightly higher between 22 November 2014 and 25 January 2015, that is, 32.2°C (±0.2 SE) and 54% (±10.2 SE), respectively (Supplementary Material S2).

Results

Dissolved Oxygen

DO% saturation during the four aerial spray periods is presented in Figure 2, with summary statistics in Table 1. Cycling in the first spray (December 2013) ranged between 0% and 160% (Figure 2), with critically low levels occurring in the early morning hours (4 a.m. to 6 a.m.) though (at approximately 10% saturation/hr) and increased during the day reaching maxima between 3 p.m. and 4 p.m. There was no significant difference in DO before and after Spray 1 (independent t test, $p = .187$, $F = 1.793$). During the second spray (June 2014), DO reached overnight lows toward 20% and, however, recovered again by mid-afternoon. There was a significant difference in daily average DO before and after Spray 2 (independent t test, $p = .01$, $F = 6.848$). Spray 3 (May 2015) occurred during the period when aquatic plant cover was highest, ~80% surface coverage, DO cycling was less pronounced with daily maxima reaching ~30% (again between 3 p.m. and 4 p.m.) with nighttime minima of 0% (Table 2). There was no significant difference in daily average DO before and after Spray 3 (independent t test, $p = .06$, $F = 3.433$). By the final spray, DO was persistently less than 1% saturation and remained so before and after spraying, though there was no significant difference in daily average DO (independent t test, $p = .02$, $F = 8.287$).

To provide context field, DO data were compared with Chronic Trigger Value and Acute Trigger Value thresholds (long-term exposure is called chronic and short-term exposure is called acute—both may cause health effects that are immediate or health effects that occur days or years later) determined for barramundi (*Lates calcarifer*; Butler & Burrows, 2007). In accordance with these thresholds, barramundi would have a lower risk to Acute Trigger Value and Chronic Trigger Value exposure during the first and second sprays, though exposure increased during the third and final spray with effectively 100% of the logger period below these trigger values (Table 1).

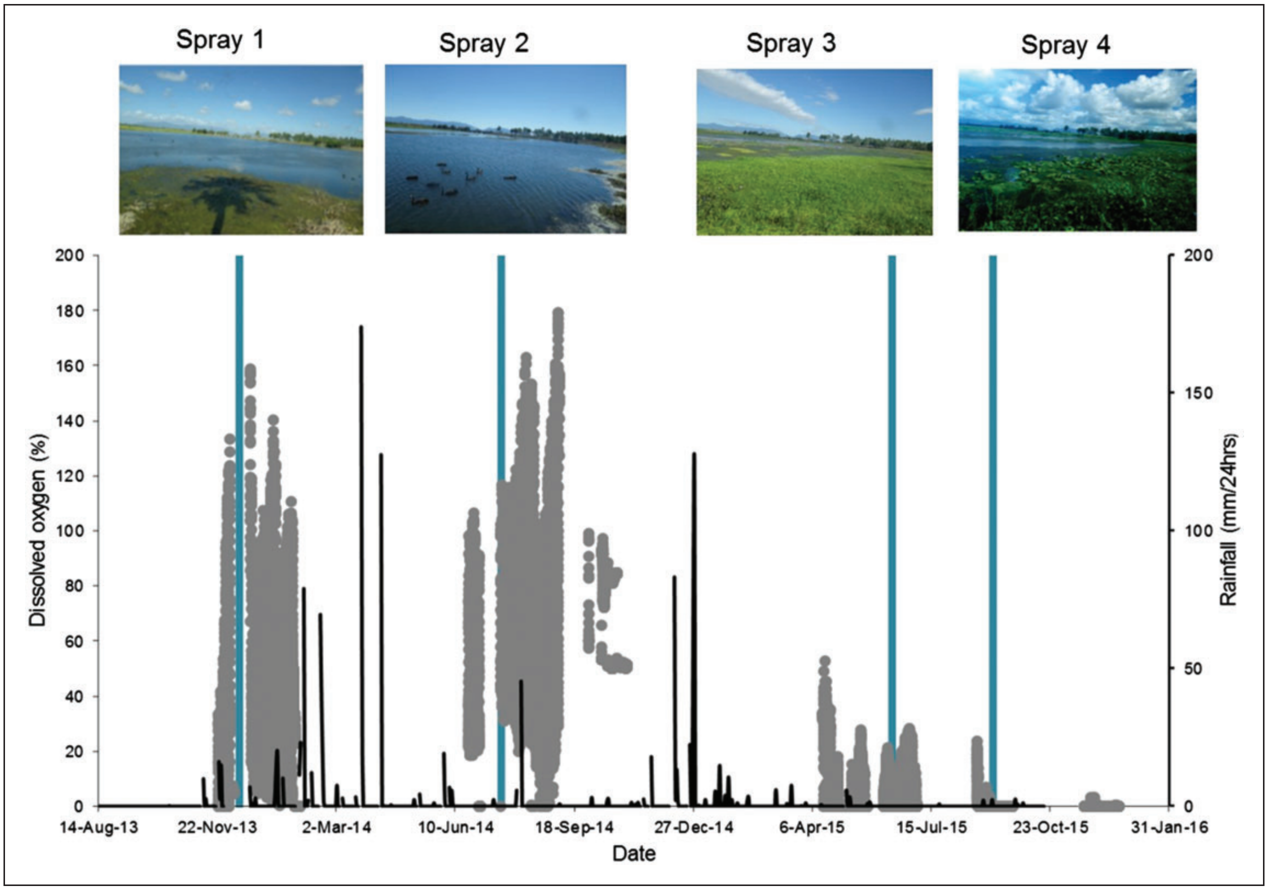

Figure 2. Time series plot (20 min logging period) for surface (0.2 m) dissolved oxygen (%) in wetland before and after spray events. Blue vertical lines show spray events, and black bars are daily total rainfall recorded at Burdekin Shire Council station. Photos illustrate each spray application.

Table 1. Summary Statistics of Dissolved Oxygen Concentrations Recorded During the Four Survey Periods in the Wetland (From Figure 2).

Spray	Period	Logging dates	Mean	5% percentile	95% percentile	DO% ATV	DO% CTV
1	**Before**	**22/11/13 to 5/12/13**	**23.7**	**0**	**97.9**	**10**	**80**
	After	19/12/13 to 25/1/14	46.7	5	102.8	15	65
2	Before	20/6/14 to 29/6/14	48.7	21.2	92.9	0	50
	After	17/7/14 to 4/9/14	62.7	0.4	140.2	0	35
3	Before	4/5/15 to 30/6/15	6.4	<1	22.1	90	100
	After	21/8/15 to 1/9/15	2.0	<1	14.2	100	100
4	Before	2/9/15 to 10/9/15	<1	<1	<1	100	100
	After	11/9/15 to 20/9/15	<1	<1	<1	100	100

Note. Percentage of time in each logging series where dissolved oxygen saturation (%) was below Acute Trigger Value (ATV, 16%) and Chronic Trigger Value (CTV, 62.5%) for barramundi (*Lates calcarifer*) reported in Butler and Burrows (2007).

Water Temperature

The water temperature logging series (64 days) is presented in Figure 3. There is a difference in the temperature conditions between the first logging period (where plant coverage was low; <5%), compared with the same logging period a year later when aquatic plant coverage across the wetland was high (~80%). Unfortunately, the first year of logging included surface water temperature (0.3 m below surface), while the second year included both surface (0.3 m) and bottom (0.15 m above wetland sediments) loggers. Notably, there were very little differences between surface and bottom temperature in the second year indicating that the water column was vertically well mixed. The difference in surface temperature between the first year and second year highlights that the thermal risk for fish is in a state of continual fluctuation. For bony bream (*Nematolosa erebi*, a widely distributed freshwater fish species across the northern Australia, the T_{pref} threshold, i.e., the preferred temperature point that which beyond fish display acute hypothermic responses for *N. erebi* is 31°C; Pusey, Kennard, & Arthington, 2004), the surface water temperature exceeded the threshold 16% of the logging period in the first year, increasing in the second year to 33% at the surface and 32% at the bottom. By comparison, the fly-specked hardhead (*Craterochephalus stercusmuscarum*), another common wetland species in the region (Waltham, 2017), has a critical thermal maximum (T_{max} is the maximum temperature end point before death) of 33.4°C (Burrows & Butler, 2007) which was exceeded only in the second logging period at both surface (8%) and bottom (3%) waters.

Table 2. Daytime (6 a.m. to 6 p.m.) Statistics of Dissolved Oxygen Concentrations (DO%) Recorded During the Four Survey Periods in the Wetland.

	Mean	Median	5%	95%	Min	Max
1 Before	32.14	26.56	6.14	60.87	5.99	67.66
After	61.79	61.20	37.52	80.88	29.73	132.62
2 Before	64.49	65.89	54.24	69.74	51.69	70.27
After	76.41	74.95	52.40	102.53	41.39	120.76
3 Before	5.17	3.77	0.00	14.42	0.00	16.16
After	1.89	0.95	0.04	7.24	0.00	16.16
4 Before	0.00	0.00	0.00	0.01	0.00	0.01
After	0.00	0.00	0.00	0.00	0.00	0.00

Discussion

Water Quality Considerations in Wetland System Repair

These data outline that spraying using a helicopter to control invasive aquatic plants does not dramatically

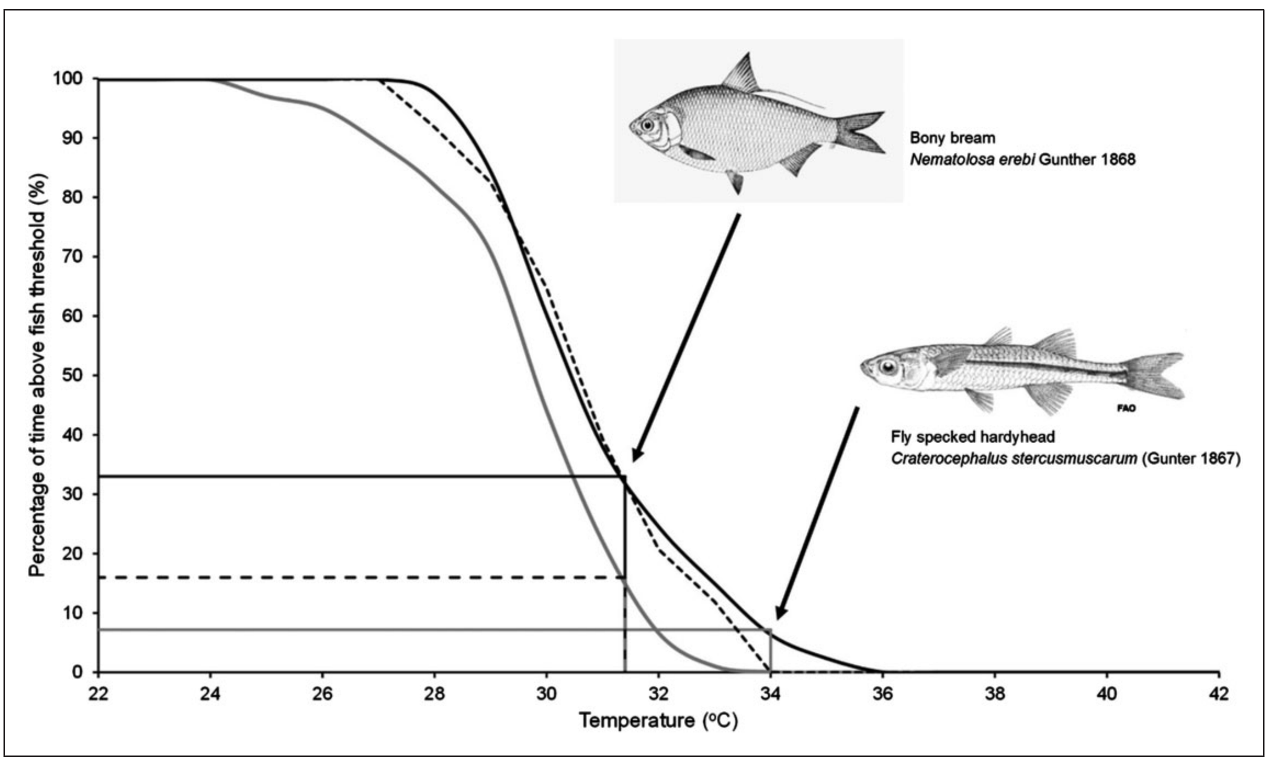

Figure 3. The percentage of time water temperature exceeded a given fish temperature threshold in the wetland. The frequency curves were compiled using all 20-min logging data between 22 November and 25 January. Gray line is 2014 (low [~5%] coverage of invasive aquatic weeds) and black line is 80% weed coverage (black line surface logger, broken black line is bottom logger). The exceedance of thresholds T_{pref} 31.8°C and T_{max} 34.8°C are shown for bony bream (*Nematolosa erebi*; black full line surface, broken line bottom logger) and fly-speck hardyhead (*Craterocephalus stercusmuscarum*; gray full line surface, broken line bottom logger).

reduce water quality conditions, here dissolved oxygen, over the following weeks after mitigation. In fact, this treatment is appropriate when cover is minor, and restricted to the wetland margins because it had little consequences on DO availability for fish. This was evident during Sprays 1 and 2, where the diurnal DO amplitude was proximal before and after treatment. In both surveys, the early morning oxygen deficit recovered over the course of the day, meaning that the net amount of aquatic plant photosynthesis compensates the wetland nocturnal respiration deficit. Even during the early morning hours when DO was low, fish were still observed active in the wetland, presumably the ability to deal with the availability of DO is supplemented via surface respiration (Richards, 2011; Yang, Cao, & Fu, 2013), or because of increased ventilation rates that balance oxygen pressure in the arterial blood with that found under normoxic conditions (Heath, 1995).

Problems occurring with spraying aquatic weeds in wetlands are in situations where a higher coverage of aquatic plant material exists (up to 80% recorded at the peak here), in combination with ongoing routine spraying and low rainfall, does the risk of hypoxia become relevant and concerning for coastal wetland function and productivity. These data show under such extreme conditions that overnight DO deficits are not able to recover with daytime photosynthesis. In fact, DO during Sprays 3 and 4 were obstinately below thresholds for fish common in the region (Butler & Burrows 2007). The prolonged anoxic conditions measured during Sprays 3 and 4 are probably resultant of accumulating bioavailable dissolved organic carbon in the wetland, where the rate of oxygen consumption necessary for decomposition exceeds the rate of oxygen diffusion in the water (even including atmospheric diffusion; O'connell, Baldwin, Robertson, & Rees, 2000). Under anoxic conditions, aquatic animals, such as fish, are more susceptible to asphyxiation (Breitburg, 2002), as in the example here for the barramundi (*Lates calcarifer*), a coastal iconic species (a popular sports fish species) widely distributed across northern Australia (James et al., 2017), and with a diadromous movement ecology where it needs to migrate between fresh and tidal waters to complete lifecycle stages. This fish is affected by poor water quality, particularly dissolved oxygen (Collins, Clark, Rummer, & Carton, 2013; Flint, Crossland, & Pearson, 2015), with fish kills regularly occurring across the floodplain in summer with the first rainfall flow which delivers oxygen deficient "black water" (Butler & Burrows, 2007; Butler, Loong, & Davis, 2009). The challenge of controlling invasive aquatic plants without contributing to effects on DO highlights the enormity of the invasive aquatic plant problem for land managers in the GBR catchments (Arthington et al., 2015; Dubuc,Waltham, Malerba, & Sheaves, 2017), but also apparently elsewhere across the distribution of this invasive aquatic weed (Masifwa, Twongo, & Denny, 2001).

In tropical northern Australia, freshwater fish face aquatic thermal exposure risks and particularly so leading up to and during summer months (Wallace et al., 2015). Temperature data here revealed that wetland fish are exposed to thermal regimes exceeding conditions ideal for optimal growth, even reaching lethal conditions. Similar thermal risks were reported by Wallace et al. (2017) in arid river waters of northern Queensland and also in a coastal freshwater tropical wetland on the Herbert River floodplain (approximately 150-km north of Lochinvah wetland; Waltham, 2017). In our wetland site, the risk was highest at the surface water layer ($\sim$0.2 m) with bottom waters ($\sim$2 m) rarely, if ever, exceeding thresholds. This presents the model that the thermal risk for fish at the surface waters could be overcome by accessing, cooler, deeper wetland waters. However, the thermal refugia afforded in bottom waters would be compromised by the fact that it exposes fish to critically low DO conditions in the bottom waters (Waltham et al., 2013), especially in weed-choked wetlands where the surface vegetation mat (including water hyacinth) averts vertical mixing of oxygenated surface waters with bottom waters (Villamagna & Murphy, 2010). The warmer surface water conditions during the second years' monitoring contrasts Waltham (2017), where surface water temperature was cooler under excessive aquatic plant coverage. A likely reason here for higher temperature with high plant coverage could be explained by the warmer ambient air temperature experienced in the region (see Supplementary Material S2). In addition to fish, freshwater crustacean species also have thermal exposure thresholds (Stewart, Close, Cook, & Davies, 2013), many of which are an important food source for these type of wetlands.

Controlling Invasive Aquatic Wetland Plants

Attempts to repair and restore wetland ecological function with respect to removal of invasive aquatic vegetation in GBR catchments has concerted on three approaches. The common is aerial spraying (Burrows et al., 2012), while data accompanying this treatment option are not generally available. The most important point for manages administering weed removal based on the data provided here is that when restricted to edges, aerial spraying for invasive plant maintenance presents a low risk of hypoxia exposure to fish. Under a situation where rainfall flow has been low combined with high plant coverage, continuing to implement spraying contributes to water quality implications. Indeed, spraying has the added risk of herbicide and pesticide toxicity to local receiving waters, and also coastal ecosystems some distance away (e.g., coral reef water quality monitoring has detected trace levels of herbicide from land farming; Lewis et al., 2009). Another strategy is mechanical removal where an aquatic harvester progressively removes floating weed species by collecting it using a floating conveyor belt fixed to a barge, with the material collected and dumped in piles on the bank for drying and disposal. While this method has been used in the region, barge operation and maintenance is expensive, in addition to the ongoing need for follow-up spot spraying where the barge is not able to access. The third strategy, which has been recently examined on the Herbert River floodplain (150 km north of Lochinvah wetland), is to remove bund earth walls which reinstates tidal exchange with the upstream freshwater wetland (Waltham, 2017). In that trial, those authors first developed a hydrodynamic model to define the section of the wall to remove that would maximize saltwater ingress to destroy invasive freshwater aquatic plants. After the bund wall was breached, ingress of saltwater quickly killed most of the WONS, which contributed to improvements in DO concentrations (Waltham, 2017). Not only did the saltwater ingress destroy the WONS (in the case of that study: water hyacinth, *Eichhornia crasspies*, and Hymenachne, *Hymenachne amplexicauli*), but the reconnection had the dual benefit of permitting estuarine fish species (including barramundi, *Lates calcarifer*) to again access to the wetland (Waltham, 2017). The cost to remove the bund wall (cost to hire excavator machinery) is a one-time expense, compared with on-going costs in aerial spraying and weed harvesting.

Implications for Conservation

The prevalence of the invasive water hyacinth (*Eichhornia crassipes*) and its impact on water and habitat quality has directed management control programs toward not only removing the vegetation but contributing to broader protection and repair of the ecology and socioeconomic values of coastal wetlands. Explicit data examining specific management control techniques for freshwater aquatic weeds are limited; this project provides a repeated assessment of water quality conditions before and after aerial spray treatment, a commonly used method by authorities. Although this project occurred on a single wetland, it is similar to many others in the region, meaning these results have broader application, and advocate that aerial spraying when coverage is restricted to edges probably has only minor impacts on water quality conditions. Under an extended dry period and when coverage is higher, aerial spraying delivers visible effect though the overall organic loading on the wetland biogeochemistry, as the material decomposes, presents a critical and persistent oxygen demand. Fish therefore have a higher chronic exposure risk than when compared with minor sprays that focus along wetland water edge. The advice here for managers is that scheduling aerial sprays routinely should be avoided, with decisions made after site inspections and a review of antecedent weather conditions.

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Declaration of Conflicting Interests

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Supplementary material

Supplementary material is available for this article online.

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