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Riparian vegetation and sediment gradients determine invertebrate diversity in streams draining an agricultural landscape

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ABSTRACT

Separation of natural and anthropogenic influences is required to identify land-use impacts on stream ecosystems. We investigated the effects of water quality and riparian condition on invertebrate assemblages along streams draining agricultural land by partitioning out changes in geomorphological characteristics. There was a strong negative relationship between invertebrate richness and distance downstream, driven by a gradient of reducing stream power and substratum particle size along the streams. When substratum particle size was accounted for, richness was reduced by ~24% when there was limited availability of coarse particulate organic matter, resulting from lower riparian forest cover upstream. High concentrations of fertilizer-derived nitrate boosted invertebrate abundances, but only in mid sections of streams, where coarse substrata (>100 mm) and high insolation were available. Sampling of multiple sites along streams facilitated partitioning of land-use impacts from natural gradients. Invertebrate richness was a good indicator of stream biophysical condition (e.g. nature of the substratum, riparian condition) at the stream scale irrespective of taxonomic resolution (family or higher) or sample size (down to 50 individuals per site), and was therefore a useful monitoring tool. The finding that riparian vegetation is a key determinant of invertebrate diversity should encourage catchment-scale maintenance and rehabilitation of native riparian forest.

1. Introduction

Declines in stream health associated with land clearing and agricultural development are well- documented (e.g., Richards et al., 1996; Wang et al., 1997; Maloney and Weller, 2011). Streams draining agricultural lands have high concentrations of fertilizer-derived nutrients and other agrichemicals (Connolly et al., 2015), altered geomorphology and instream habitats (Mazeika et al., 2004; Dahm et al., 2013) and disturbed riparian zones (Pusey and Arthington, 2003; Waite, 2014), and they support fewer species of invertebrates and fish than streams draining forested catchments (Lenat and Crawford, 1994; Wang et al., 1997). However, land-use impacts are varied and complex, because of variation in hydrogeomorphic settings and composition of biotic assemblages, and because streams can be affected by multiple disturbances (Connolly and Pearson, 2004; Maloney and Weller, 2011; Clapcott et al., 2012; Clements et al., 2015). Anthropogenic land use is frequently superimposed on natural gradients, and better understanding of these interactions is a key challenge in assessing the ecological integrity of streams (Allan 2004).

Natural longitudinal gradients of physical conditions (e.g., slope, current velocity, substratum) are characteristic of streams and are typically accompanied by progressive changes in composition of biotic assemblages (Vannote et al., 1980; Grubaugh et al., 1996; Marchant et al., 1999). Catchment landform dictates patterns in human land use, with agriculture dominant on low-relief land and floodplains (e.g. Quinn, 2000). Natural longitudinal gradients are also typically accompanied by anthropogenic influences, which accumulate along streams as increasing areas of the catchment are used for agriculture. For example, in the Australian Wet Tropics bioregion, the lowland floodplains are largely developed for intensive sugar cane production, and stream water quality and riparian condition both negatively correlate with area of anthropogenic land use (Bainbridge et al., 2009; Mackay et al., 2010; Connolly et al., 2015). In determining the influence of land use on stream ecology it is therefore necessary to account for both natural and anthropogenic gradients. However, there are few studies that investigate the influence of agricultural land use on the natural longitudinal patterns in benthic community composition

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(exceptions are Grubaugh et al., 1996; Delongi and Brusven, 1998; Niyogi et al., 2007).

Connections between riparian vegetation and stream invertebrate assemblages have been reported over several decades, particularly with reference to the detrital pathway, but also in relation to shade and habitat (Kaushik and Hynes, 1971; Cummins and Lauff, 1969; Dudgeon, 1989, 1994; Cheshire et al., 2005). Riparian zones are vital to normal ecosystem function and stream health, but the links are variable and may be difficult to apply to riparian management to achieve ecological objectives (Greenwood et al., 2012; Hansen et al., 2015). Waite (2014) reported that the nature of the riparian zone was one of the most important variables influencing invertebrate assemblages in eastern US streams, while Lorenz and Feld (2013) found that sub-catchment variables influenced ecological status more than local variables in German streams. In south-eastern Australian streams, standing stocks of benthic detritus, an important resource for invertebrates, were very low where riparian canopy cover was <35%, indicating the need for restoration of canopy cover (Reid et al., 2008). However, responses of invertebrate assemblages to riparian restoration can be delayed—for example, >8 years in south-eastern Australian streams (Becker and Robson, 2009). Lorion and Kennedy (2009) found few studies on the effectiveness of riparian buffers on tropical stream ecosystems, but reported that buffers reduced the effects of deforestation on benthic communities. Likewise, Dudgeon (1994) demonstrated a positive influence of riparian shade and detritus on invertebrates in New Guinea streams, but found no such clear relationship in Hong Kong (Dudgeon, 1989).

We investigated land-use impacts and invertebrate distributions in lowland reaches of four streams in two adjacent Australian catchments (Mulgrave River and Russell River) with the aim of identifying the causes of any differences in invertebrate assemblages along and between streams. Both catchments have extensive areas of their floodplains used for sugarcane production, but there is contrasting stream management between them: on the Mulgrave floodplain, riparian forest has been maintained and improved through replanting schemes, while on the Russell floodplain, the riparian zone is severely degraded, with sugarcane grown up to the stream banks, as in much of the Wet Tropics region (Werren, 1998; Mackay et al., 2010; Connolly et al., 2015). We hypothesised that the invertebrate assemblages would not respond differentially to agricultural land-use gradients, water quality characteristics and condition of the riparian vegetation, but might respond to natural gradients in hydraulic habitat and water quality variables. Our approach was to sample along the floodplain reach of each stream and statistically separate the confounding effects of natural gradients and land use, so that we could compare streams with different degrees of anthropogenic disturbance, especially loss of riparian forest.

2. Methods

2.1. Study region and streams

The Australian Wet Tropics is a discrete bioregion (Commonwealth of Australia, 2005) with high rainfall and perennial (though seasonal) stream flow, in contrast to streams in most of tropical Australia (Pearson et al., 2015). Despite its small land area (0.26% of the continent), the Wet Tropics supports Australia's richest terrestrial and freshwater biodiversity (Connolly et al., 2008; Stork et al., 2008; Pearson et al., 2015). Streams connect the Wet Tropics and Great Barrier Reef World Heritage Areas and so are of exceptional scientific and conservation value. However, waterways on the floodplains are affected by loss of native riparian vegetation and contamination by agricultural chemicals (Bainbridge et al., 2009; Pearson et al., 2013; Connolly et al., 2015), and are not included in the World Heritage Areas or other protected areas (Januchowski-Hartley et al., 2011).

We sampled two streams in the Mulgrave catchment (1315 km^2) (Little Mulgrave R. and Behana Ck.) and two in the adjacent Russell catchment (668 km^2) (Woopen Ck. and Babinda Ck.) (see Connolly et al., 2015; Supplementary Fig. A1). The catchments are adjacent and drain the eastern escarpment of the Great Dividing Range, on the mountain massif of Mt Bartle Frere and Mt Bellenden Ker (~1600 m), and are therefore similar in topography and climate. The study streams descend through pristine montane rainforest (gradient 6–16%), then meet and cross narrow coastal floodplains (gradient 0.4–1%). They have perennial flows, frequent over-bank floods in the wet season (December to March), and occasional spates throughout the year. Both catchments have had their floodplains largely cleared of native forest for agriculture, and their land use is similar, including about 20% agriculture, with about 58% in conservation areas on the forested slopes (Great Barrier Reef Marine Park Authority, 2013). However, in the Mulgrave catchment native riparian forest is largely intact, whereas in the Russell catchment riparian vegetation has been extensively degraded, with native forest cover largely replaced by exotic herbs and grasses. Like all streams in the Wet Tropics, as the study streams cross the floodplain their geomorphological characteristics gradually change: for example, substratum particle size declines naturally with decreasing gradient (Supplementary Table A1). Fertilizer application practices were similar in the two catchments (approximately 160–170 kg N/ha—Rayment, 2003), and all streams draining them had high concentrations of nitrate (Connolly et al., 2015), which increased with increasing agricultural land area (Supplementary Table A1; Connolly et al., 2015).

To investigate how longitudinal changes affected the invertebrate assemblages and to account for them in subsequent analyses, water-quality, substratum and invertebrate samples were collected at 8–13 riffle sites along each stream, at intervals of approximately 1–1.5 km (Connolly et al., 2015; Supplementary Fig. A1). Sampling was undertaken in June and July 2005 during a period of stable base flow.

2.2. Hydraulic, water quality and riparian variables

At each site, mean water velocity and depth were estimated from multiple measures taken across transects in the study riffle (Mackay et al., 2010) and were used to calculate stream discharge and power, Reynolds number and Froude number, following Gordon et al. (2004). The stream gradient was measured as the change in height of the water surface from 50 m upstream to 50 m downstream of the riffle, using a staff and dumpy level. Sediment particle size was sampled using a zig-zag method (Bunte and Abt, 2001) and classified using the Wentworth Scale (Wentworth, 1922). A systematic bank-to-bank path was chosen to pick up and measure the intermediate axis of 100 clasts (substratum particles) spaced regularly across the stream bed. Fine-particle size distributions were determined by dry-sieving samples in the laboratory using mesh sizes from 4 ϕ (0.0625 mm) to −6 ϕ (64 mm). Finer material was separated hydrometrically, while particles larger than −6 ϕ were measured with vernier callipers (Rowell, 1994; Gordon et al., 2004). Particle-size statistics were calculated using *GRADISTAT* Version 4.0 (Blott and Pye, 2001).

Water quality samples were collected for determination of nutrients as total nitrogen, nitrogen oxides (NO_X), ammonia, total phosphorus and filterable reactive phosphorus and were analysed by the Australian Centre for Tropical Freshwater Research at James Cook University, Townsville (Connolly et al., 2015). Water temperature, pH, electrical conductivity and dissolved oxygen were measured at each site at the time of sampling (between 9 am and 4 pm each day) using a Hydrolab H20 meter.