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Extent estimates and land cover relationships for functional indicators in non-wadeable rivers

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ABSTRACT

Functional indicators are being increasingly used to assess waterway health but their responses to pressure in non-wadeable rivers have not been widely documented or applied in modern survey designs that provide unbiased estimates of extent. This study tests the response of river metabolism and loss in cotton strip tensile strength across a land use pressure gradient in non-wadeable rivers of northern New Zealand, and reports extent estimates for river metabolism and decomposition rates. Following adjustment for probability of selection, ecosystem respiration (ER) and gross primary production (GPP) for the target population of order 5–7 non-wadeable rivers averaged −7.3 and 4.8 g O_2 m^{-2} d^{-1}, respectively, with average $P/R < 1$ indicating dominance by heterotrophic processes. Ecosystem respiration was <−3.3 g O_2 m^{-2} d^{-1} for 75% of non-wadeable river length with around 20% of length between −10 and −20 g O_2 m^{-2} d^{-1}. Cumulative distribution functions of cotton strength loss estimates indicated a more-or-less linear relationship with river km reflecting an even spread of decay rates (range in k 0.0007–0.2875 d^{-1}) across non-wadeable rivers regionally. A non-linear relationship with land cover was detected for GPP which was typically <5 g O_2 m^{-2} d^{-1} where natural vegetation cover was below 20% and greater than 80% of upstream catchment area. For cotton strength loss, the relationship with land cover was wedge-shaped such that sites with >60% natural cover had low decay rates (<0.02 d^{-1}) with variability below this increasing as natural cover declined. Using published criteria for assessing waterway health based on ER and GPP, 232–298 km (20–29%) of non-wadeable river length was considered to have severely impaired ecosystem functioning, and 436–530 km (42–50%) had no evidence of impact on river metabolism.

1. Introduction

Lotic habitats in lowland landscapes are often dominated by non-wadeable rivers that integrate a complex range of upstream activities, and provide goods and services such as provision of fisheries and recreational opportunities (Tockner et al., 2011; Palmer and Febria, 2012). Assessing the ecological health of these larger waterways using biological indicators is hampered by the physical difficulties associated with accessing deep flowing water, and the complexity and scale of habitats that may require sampling (Flotemersch et al., 2000). Nevertheless, frameworks and field procedures for bioassessment programmes in mainstem rivers have recently been developed for biotic and abiotic characteristics (Flotemersch et al., 2006a,b; Angradi et al., 2009), and several studies have tested biocriteria for non-wadeable river bioassessment (Dolédec and Statzner, 2008; Jackson et al., 2010).

Recently, there has been increased interest in the use of functional indicators for assessing ecological health of waterways as integrators of biological processes (e.g., Gessner and Chauvet, 2002; Fellows et al., 2006; Uehlinger, 2006; Young et al., 2008; Woodward et al., 2012), and as direct measures of valuable ecosystem services (Rapport et al., 1998). Functional indicators can include rates of nutrient uptake (Sabater et al., 2000; Hall and Tank, 2003), benthic microbial respiration (Niyogi et al., 2001; Hill et al., 2002), denitrification (Bernhardt et al., 2002; Udy et al., 2006), fine particulate organic matter export (Wallace et al., 1997) and organic matter retention (Speaker et al., 1984; Quinn et al., 2007). Rates of organic matter decomposition and ecosystem metabolism appear particularly suited as indicators for non-wadeable river biomonitoring since they respond to a range of physical and chemical stressors (Pascoal et al., 2003; Mulholland et al., 2001), are relatively inexpensive and straight-forward to deploy in deep water, and metabolism measurements at least are amenable to automation (Izagirre et al., 2008; Young et al., 2008).

Measurements of functional indicators have the added benefit of providing insights into ecological processes that underpin life-supporting capacity. Rates of organic matter decay and

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ecosystem metabolism, for example, help determine the trophic basis underlying secondary production. All river ecosystems are fuelled to varying degrees by combinations of terrestrially-derived organic material and autochthonous carbon, and a number of competing hypotheses have been developed to account for carbon flow in larger rivers where organic matter derived from floodplain interactions and/or internal processes can underpin food-webs (see Pingram et al., 2012 for a review). Furthermore, the data underlying functional indicator calculations indicate whether biological limits are being exceeded for some fundamental physicochemical parameters (e.g., dissolved oxygen for metabolism, water temperature for decay rates).

Developing defensible inferences of ecological condition from functional indicators used in large scale monitoring programs is dependent on (i) implementing an appropriate survey design to achieve the desired objectives, and (ii) developing an understanding of pressure–response relationships with key regional stressors. Linear responses are easiest to interpret since ecosystem health is simply proportional to indicator value. However, these simple relationships are not necessarily expected for many ecosystem process measurements (Rama Rao et al., 1979; Niyogi et al., 2003; Hagen et al., 2006), and little is known about response trajectories in non-wadeable rivers. In terms of survey design, a key objective of environmental monitoring programmes is to draw quantitative conclusions, with a known level of certainty, about the extent and quality of a target population. Probability-based site selection yields spatially balanced sampling networks that provide unbiased estimates of resource condition (Collier and Olsen, 2013). This monitoring design is now used in the USA for national scale assessments of aquatic resources following acknowledgement that previous designs did not adequately describe the condition of waterways (Paulsen et al., 1998; Hughes and Peck, 2008), or provide unbiased estimates of their features and extent (Olsen and Peck, 2008; Paulsen et al., 2008), but few if any studies have applied this approach to functional indicators.

In the present study, a probability survey design is used to develop regional extent estimates of non-wadeable river (order 5–7) metabolism and organic matter decay rates. The aims were to (i) demonstrate how quantification of functional indicator extent in large scale monitoring programs can provide insights into regional non-wadeable river condition, and (ii) determine the nature of functional indicator response relationships with the dominant regional land cover stressor. This work addresses an important information gap on functional responses of non-wadeable rivers to environmental stress, and reinforces the value of probability sampling to quantify the extent of waterway condition in biomonitoring programmes.

2. Methods

2.1. Study area

The Waikato Region covers 25,000 km^2 across latitudes 36° and 39° S in New Zealand's central North Island. Average annual rainfall is variable, but lower in the north (1000–1500 mm p.a.) than in the eastern and western ranges (up to 2500 mm p.a.) and high (5000 mm p.a.) on southern mountaintops. Mean annual air temperatures in most of this region are in the range 12.5–15.0 °C, but decline to <8.0 °C on southern mountains (Kilpatrick, 1999). Most of the Waikato Region has been developed for pastoral agriculture or pine forestry, with extensive remnants of original forest persisting only in upland parts of the region (28% of pre-European extent). A substantial portion of the region (57%) contributes drainage to the 7th order Waikato River. Other significant river systems in the region include the Mokau River (order 7), the Waipa River (order 6), and the Waihou and Piako rivers which are both 6th order and drain extensive lowland plains.

2.2. Sampling site selection

A probability, rotating panel design was used to select 30 non-tidal, non-wadeable river sites ≥5th order which were sampled during summer over 2009–2011. Non-wadeable sites were defined as those where water depth precluded use of wadeable stream protocols designed to sample from a representative range of lotic habitats (Stark et al., 2001). Equal numbers of 5th, 6th and 7th order sites, based on the River Environment Classification (Snelder and Biggs, 2002) river network layer, were selected using the R software library spsurvey (http://www.epa.gov/nheerl/arm) (i.e., balanced unequal probability design). The Generalised Random Tessellation Stratified (GRTS) approach was used to achieve spatial balance across the REC network (Stevens and Olsen, 2004). Candidate sites were screened initially using aerial photos to determine whether they could form part of the target population, and potential sites were then visited to verify suitability. Riparian vegetation composition was variable among sites and ranged from open pasture to dense cover by introduced willows (*Salix* spp.).

2.3. Environmental and pressure variables

Dissolved oxygen concentrations and water temperatures were recorded during December in non-wadeable sites every 15 min over 1 day (year 1) or 7 consecutive days (years 2 and 3 to obtain a more representative measure) using calibrated data loggers (D-Opto, Zebra-Tech Ltd., Nelson, New Zealand) suspended at approximately mid-water on weighted buoys or attached to metal stakes in flowing water. Average depth upstream of each site was calculated from hand-held depth sounder (Speedtech Depthmate) measurements at five points across five cross-sections spaced at regular intervals over several hundred meters, and widths at these cross-sections were determined from high resolution aerial photos. Measurements were made once at each site and were phased over three years.

2.4. Ecosystem metabolism

Metabolism values were calculated using a spreadsheet model described in Young and Collier (2009). Briefly, mean daily ecosystem respiration (ER) and the reaeration coefficient (k) were determined using the night-time regression method (Owens, 1974). The reaeration coefficient and ecosystem respiration rate obtained were then used to determine gross photosynthetic rate over the sampling interval using:

$$GPP_t = \frac{dO}{dt} + ER - kD$$

where GPP_t is the gross photosynthetic rate (g m^{-3} s^{-1}) over time interval t (s), and D is the oxygen deficit (i.e. the difference between the observed concentration and the concentration at 100% saturation). Daily gross primary production (GPP, g m^{-3} d^{-1}) was estimated as the integral of all temperature-corrected photosynthetic rates during daylight (Wiley et al., 1990). Areal estimates were obtained by multiplying the volume based estimates by average reach depth (m) which allowed comparison among sites with different depths. Where more than one full day of DO data was collected, metabolism values were calculated for each day and the average used in subsequent analyses.