RESILIENT SHORELINES

Earthquake effects on sea levels and their implications for conservation and climate change

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Thesis submitted in partial fulfilment of the degree of

Doctor of Philosophy

May 2020

UC UNIVERSITY OF CANTERBURY Te Whare Wānanga o Waitaha CHRISTCHURCH NEW ZEALAND

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Mō āpōpō, mō ake, ake tonu rā

For tomorrow, for the future, through the eons of time

CONTENTS

ACKNOWLEDGEMENTS

There are a long list of people who have assisted with the journey of this thesis from its beginnings of conceptualisation in post-earthquake Christchurch and through many explorations, interesting findings, and decisions to be made along the way. A big thanks to everyone who has been involved! Foremost I'd like to thank my supervisory team of Prof. David Schiel, Prof. Ken Hughey and Dr. Mike Hickford. I'm sure none of the team, myself included, knew how the investigation would unfold. Additional support was provided by Prof. Bryan Jenkins in the early stages of the project, and later by Dr. Ed Challies at the Waterways Centre for Freshwater Management. Support from many other staff at the University of Canterbury was also greatly appreciated. Particular thanks to Suellen Knopick, Prof. Jenny Webster-Brown and John Revell at the Waterways Centre for Freshwater Management, Dr. Matthew Hughes at Civil and Natural Resources Engineering, Jan McKenzie, Jesse Burns, Zoe Smeele, Thomas Falconer, Genny Schiel and Dr. Russell Taylor at the Marine Ecology Research Group, and Associate Professor Te Maire Tau and Kirsty Ameriks at the Ngāi Tahu Research Centre. Discussions with many others involved in post-earthquake research and wider resource management were also greatly encouraging. Thanks to Dr. Will Allen, Prof. Islay Marsden, Dr. Hamish Rennie, Dr. Leanne Morgan, Prof. Wolfgang Rack, Dr, Christian Wild, Dr. Phil Clunies-Ross, Prof. Eric Pawson, and Dr. Markus Pahlow for helpful discussions along the way.

Field work was supported by many fellow students and also by volunteers from the community. The level of support made many of the ambitious surveys much more possible! A big thanks to Duncan Keenan, Jason Telford, Ryan Taylor, Nicole Wehner, Lorena Vigoya, Claire Macintyre, Lilian Clark, Dr. Eimear Egan, Dr. Greg Burrell, Zoe Smeele, Shawn Gerrity and Irene Setiawan for volunteering their time at different stages. Community connections and collaboration were also an important aspect of this research and included collaborative projects that made use of the research findings and created additional interest and opportunities to assist in post-disaster recovery processes. One of the most notable was Whaka Inaka: Causing Whitebait, a joint project between EOS Ecology, Ngāi Tahu, and the University of Canterbury with input from many other community groups and supporting organisations and with funding from the Department of Conservation (DOC) Community Fund. A big thanks to all who were involved but particularly to the core team of Shelley McMurtrie, Kirsty Brennan, Te Marino Lenihan and Dr. Mike Hickford. Outreach projects included the Te Tiaki Īnanga brochure produced with the DOC Freshwater Team and Helen Kettles from the DOC Marine Ecosystem Team, and a simple dune survey method with David Bergin and Jim Dahm from the Coastal Restoration Trust of New Zealand.

Collaboration with a wider network of colleagues was also much appreciated and included many others involved with post-earthquake research and related recovery work in Canterbury. Thanks to Richard Measures, Jochen Bind and David Plew at the National Institute for Water and Atmospheric Research (NIWA), Justin Cope, Bruce Gabites, Philip Grove and Mark Parker at Environment Canterbury, and Rodney Chambers, Jason Roberts, Pieter Borcherds and Andrew Crossland from Christchurch City Council. Conservations with many people involved in local community initiatives were also greatly appreciated and always stimulating! Particular thanks to members of the Avon Ōtākaro Network, Ōpāwaho-Heathcote River Network, Greening the Red Zone, Avon Ōtākaro Forest Park, and Avon Heathcote Estuary Ihutai Trust.

Funding is acknowledged and greatly appreciated from the Ngāi Tahu Research Centre, Brian Mason Scientific and Technical Trust, Coastal Restoration Trust of New Zealand, Engineering New Zealand / Water NZ Rivers Group, and with additional support from the Waterways Centre for Freshwater Management and NIWA.

Last but not least, thanks to my wonderful partner Judith for your endless support throughout this project.

~ ***Ehara taku toa i te toa takitahi**, **ēngari he toa takimano** e* ~

ABSTRACT

Coastal margins are exposed to rising sea levels that present challenging circumstances for natural resource management. This study investigates a rare example of tectonic displacement caused by earthquakes that generated rapid sea-level change in a tidal lagoon system typical of many worldwide. This thesis begins by evaluating the coastal squeeze effects caused by interactions between relative sea-level (RSL) rise and the built environment of Christchurch, New Zealand, and also examples of release from similar effects in areas of uplift where land reclamations were already present. Quantification of area gains and losses demonstrated the importance of natural lagoon expansion into areas of suitable elevation under conditions of RSL rise and showed that they may be necessary to offset coastal squeeze losses experienced elsewhere. Implications of these spatial effects include the need to provide accommodation space for natural ecosystems under RSL rise, yet other land-uses are likely to be present in the areas required. Consequently, the resilience of these environments depends on facilitating transitions between human land-uses either proactively or in response to disaster events. Principles illustrated by co-seismic sea-level change are generally applicable to climate change adaptation due to the similarity of inundation effects. Furthermore, they highlight the potential role of non-climatic factors in determining the overall trajectory of change.

Chapter 2 quantifies impacts on riparian wetland ecosystems over an eight year period post-quake. Coastal wetlands were overwhelmed by RSL rise and recovery trajectories were surprisingly slow. Four risk factors were identified from the observed changes: 1) the encroachment of anthropogenic land-uses, 2) connectivity losses between areas of suitable elevation, 3) the disproportionate effect of larger wetland vulnerabilities, and 4) the need to protect new areas to address the future movement of ecosystems. Chapter 3 evaluates the unique context of shoreline management on a barrier sandspit under sea-level rise. A linked scenario approach was used to evaluate changes on the open coast and estuarine shorelines simultaneously and consider combined effects. The results show dune loss from a third of the study area using a sea-level rise scenario of 1 m over 100 years and with continuation of current land-uses. Increased exposure to natural hazards and accompanying demand for seawalls is a likely consequence unless natural alternatives can be progressed. In contrast, an example of managed retreat following earthquake-induced subsidence of the backshore presents a new opportunity to restart saltmarsh accretion processes seaward of coastal defences with the potential to reverse decades of degradation and build sea-level rise resilience.

Considering both shorelines simultaneously highlights the existence of pinch-points from opposing forces that result in small land volumes above the tidal range. Societal adaptation is delicately poised between the paradigms of resisting or accommodating nature and challenged by the long perimeter and confined nature of the sandspit feature.

The remaining chapters address the potential for salinity effects caused by tidal prism changes with a focus on the conservation of īnanga (*Galaxias maculatus*), a culturally important fish that supports New Zealand's whitebait fishery. Methodologies were developed to test the hypothesis that RSL changes would drive a shift in the distribution of spawning sites with implications for their management. Chapter 4 describes a new practical methodology for quantifying the total productivity and spatiotemporal variability of spawning sites at catchment scale. Chapter 5 describes the novel use of artificial habitats as a detection tools to help overcome field survey limitations in degraded environments where egg mortality can be high. The results showed that RSL changes resulted in major shifts in spawning locations and these were associated with new patterns of vulnerability due to the continuation of pre-disturbance land-uses. Unexpected findings includes an improved understanding of the spatial relationship between salinity and spawning habitat, and identification of an invasive plant species as important spawning habitat, both with practical management implications.

To conclude, the design of legal protection mechanisms was evaluated in relation to the observed habitat shifts and with a focus on two new planning initiatives that identified relatively large protected areas (PAs) in the lower river corridors. Although the larger PAs were better able to accommodate the observed habitat shifts inefficiencies were also apparent due to spatial disparities between PA boundaries and the values requiring protection. To reduce unnecessary trade-offs with other land-uses, PAs of sufficient size to cover the observable spatiotemporal variability and coupled with adaptive capacity to address future change may offer a high effectiveness from a network of smaller PAs. The latter may be informed by both monitoring and modelling of future shifts and these are expected to include upstream habitat migration driven by the identified salinity relationships and eustatic sea-level rise.

The thesis concludes with a summary of the knowledge gained from this research that can assist the development of a new paradigm of environmental sustainability incorporating conservation and climate change adaptation. Several promising directions for future research identified within this project are also discussed.

THESIS STRUCTURE AND CONTRIBUTIONS

The thesis consists of seven data chapters presented in two parts followed by a synthesis of new contributions and suggestions for further research. Part 1 addresses physical changes resulting from the CES with a focus on the quantification of ground level movements and associated relative sea-level changes across the wider study area and with particular attention to spatiotemporal variance using high resolution techniques. Part 2 addresses salinity effects associated with tidal prism changes with a focus on the conservation of īnanga. All chapters, with the exception of the synthesis, were written with the intention of being published as peer reviewed articles. Each of these seven articles is co-authored in recognition of inputs from supervisors and colleagues. Contributions to each are summarised in the co-authorship declarations that follow.

Due to the timeline of the various studies, the investigations presented in Part 2 were completed first and the research publications submitted for peer review. Three of the four articles have been published in scientific journals and the 4th has been accepted for publication in a forthcoming book. The three articles comprising Part 1 have been recently prepared and are in the early stages of submission. Additional peer reviewed articles that include information from the research are also listed below.

Peer reviewed publications

Orchard, S., Hickford, M. J. H., & Schiel, D. R. (2018). Earthquake-induced habitat migration in a riparian spawning fish has implications for conservation management. *Aquatic Conservation: Marine and Freshwater Ecosystems, 28*(3), 702-712. doi:10.1002/aqc.2898

Orchard, S., Hickford, M. J. H., & Schiel, D. R. (2018). Use of artificial habitats to detect spawning sites for the conservation of *Galaxias maculatus*, a riparian-spawning fish. *Ecological Indicators, 91,* 617-625. doi:10.1016/j.ecolind.2018.03.061

Orchard, S., & Hickford, M. J. H. (2018). Census survey approach to quantifying īnanga spawning habitat for conservation and management. *New Zealand Journal of Marine and Freshwater Research, 52*(2), 284-294. doi:10.1080/00288330.2017.1392990.

In press (accepted)

Orchard, S., & Hickford, M. J. H. (in press). Protected area effectiveness for fish spawning habitat in relation to earthquake-induced landscape change. In R. Maiti, H. G. Rodríguez, C. A. Kumari, D. Mandal, & N. C. Sarkar (Eds.), *Sustainable Bioresource Management: Climate Change Mitigation and Natural Resource Conservation*. Apple Academic Press.

Articles for submission

Orchard, S., Hughey, K. F. D, Measures, R., & Schiel, D. R. (2018). Coastal tectonics and tipping points: response of a tidal lagoon to co-seismic sea-level change.

Orchard, S., Hughey, K. F. D., & Schiel, D. R. (2018). Risk factors for coastal habitat and blue carbon loss revealed by earthquake-induced sea-level rise.

Orchard, S., & Schiel, D. R. (2018). Nature-based solutions for climate change on a peri-urban sandspit.

Additional peer reviewed publications

Published articles

Orchard, S. (2014). Potential roles for coastal protected areas in disaster risk reduction and climate change adaptation: a case study of dune management in Christchurch, New Zealand. In: Murti, R. and Buyck, C. (ed.) (2014). *Safe Havens: Protected Areas for Disaster Risk Reduction and Climate Change Adaptation*. Gland, Switzerland: IUCN. pp 83-93.

Orchard, S. (2016). Community-led approaches and climate change: Perspectives from coastal restoration projects. In: *Adapting to the consequences of climate change: engaging with communities*. NZ Coastal Society Special Publication. Wellington: IPENZ. pp 23-27.

Orchard (2018). Community-based approaches for restoring biodiversity in coastal parks. In M. Fishborn & Z. Levitina (Eds.), *Solutions in Focus: Community-led successes in marine conservation*. Gland, Switzerland: IUCN. pp 22-23.

In press (accepted)

Orchard, S., & Challies, E. (in press). Managing peri-urban floodplains and urban-rural connectivity: A case study in ecosystems governance following a natural disaster. In L. Vasseur (Ed.), *Ecosystem Governance: Urban & Rural Linkages*. Gland, Switzerland: IUCN.

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Please indicate the chapter/section/pages of this thesis that are extracted from co-authored work and provide details of the publication or submission from the extract comes:

Chapter 4

Census survey approach to quantifying īnanga spawning habitat for conservation and management

Published in *New Zealand Journal of Marine and Freshwater Research*

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SO contribution 95%

Conceived and designed the experiments: SO. Performed the experiments: SO. Analyzed the data: SO. Wrote the manuscript: SO with contributions from co-authors

Certification by Co-authors:

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The undersigned certifies that:

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- In cases where the candidate was the lead author of the co-authored work he or she wrote the text

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Signature:

M J H Hickford

Date: ***20/12/19***

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Chapter 5

Use of artificial habitats to detect spawning sites for the conservation of *Galaxias maculatus*, a riparian-spawning fish

Published in *Ecological Indicators*

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SO contribution 90%

Conceived and designed the experiments: SO. Performed the experiments: SO and collaborators. Analyzed the data: SO. Wrote the manuscript: SO with contributions from co-authors

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Chapter 6

Earthquake-induced habitat migration in a riparian spawning fish has implications for conservation management

Published in *Aquatic Conservation: Marine and Freshwater Ecosystems*

Please detail the nature and extent (%) of contribution by the candidate:

SO contribution 95%

Conceived and designed the experiments: SO. Performed the experiments: SO. Analyzed the data: SO. Wrote the manuscript: SO with contributions from co-authors

Certification by Co-authors:

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Chapter 7

Protected area effectiveness for fish spawning habitat in relation to earthquake-induced landscape change

In press *Sustainable Bioresource Management: Climate Change Mitigation and Natural Resource Conservation (Chapter 22).*

Please detail the nature and extent (%) of contribution by the candidate:

SO contribution 95%

Conceived and designed the assessment SO. Performed the experiments: SO. Analyzed the data: SO. Wrote the manuscript: SO with contributions from co-authors

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STUDY DESIGN AND CONTEXT

Over recent decades the conservation of coastal margins has become increasingly difficult alongside human population growth and land-use intensification. In part, these trends reflect the popularity of coastal areas for people though also illustrate a need for improved attention to the sustainability of modern development. Climate change poses an additional and pervasive threat that will be difficult to accommodate in many low-lying areas due to the effects of sea-level rise. Likely changes include new patterns of coastal erosion, tidal inundation and exposure to water-borne extreme events with widespread potential for negative effects on both built and natural environments. The concept of ecosystem-based adaptation is one of the so-called nature-based solutions that involve working with natural ecosystems to overcome societal challenges. In contrast to hard engineering paradigms with a focus on taming nature these approaches promote the maintenance and services of natural ecosystems including their potential roles as natural defences.

Despite these promising holistic frameworks, the severity of climate change suggests that solutions will lie predominately outside of the bounds of direct experience. As a consequence, society faces an urgent need for innovation to improve its capacity to manage climate-related risks. To help inform these challenges, I undertook a programme of observations, experiments, and modelling to investigate an extreme disturbance event that offered novel empirical insights for both disaster recovery and climate change adaptation. It involved a rare situation of relative sea-level change (ca. 0.5 m) associated with vertical displacement of the coastal environment during the Canterbury Earthquake Sequence (CES) in the city of Christchurch, New Zealand (Fig. A1). There have been few studies of tectonic displacement in contemporary aquatic environments and similarly few studies of rapid sea-level rise events due to their limited occurrence. From these perspectives the CES represented a unique opportunity to improve the understanding of earthquake effects on coastlines and identify linkages between disaster recovery and climate change adaptation in relation to relative sea-level rise.

Impacts were assessed from a socio-ecological standpoint in which physical changes and the role of anthropogenic contributing factors were each evaluated. The process of disaster recovery was investigated as a context for climate change adaptation, adding an additional dimension to the evaluation of impacts and potential responses. In this case disaster recovery initiatives included government acquisition of previously residential land in two areas of very different scale and environmental setting, and the reinstatement of pre-disaster land-uses elsewhere. These contemporary management contexts are characterised by diverse stakeholder interests typical of many communities facing natural hazards and climate change.

Two major impact themes were followed, shoreline position changes and salt water intrusion effects, both of which are pervasive drivers of change. The research was organised around a primary case study area (the Avon Heathcote Estuary Ihutai) and subsidiary cases defined by characteristic natural environments and current conservation priorities. These encompassed a variety of conceptual and geographical scales to include: the estuary and barrier sandspit landforms at the natural feature and landscape scale; riparian wetlands and intertidal zonation at the characteristic ecosystem scale; and spawning habitat for a culturally important fish (*Galaxias maculatus*) at the species-specific scale. Each of these environments was expected to be impacted by relative sea-level changes induced by tectonic displacement, though the magnitude and detectability of these changes was initially unknown.

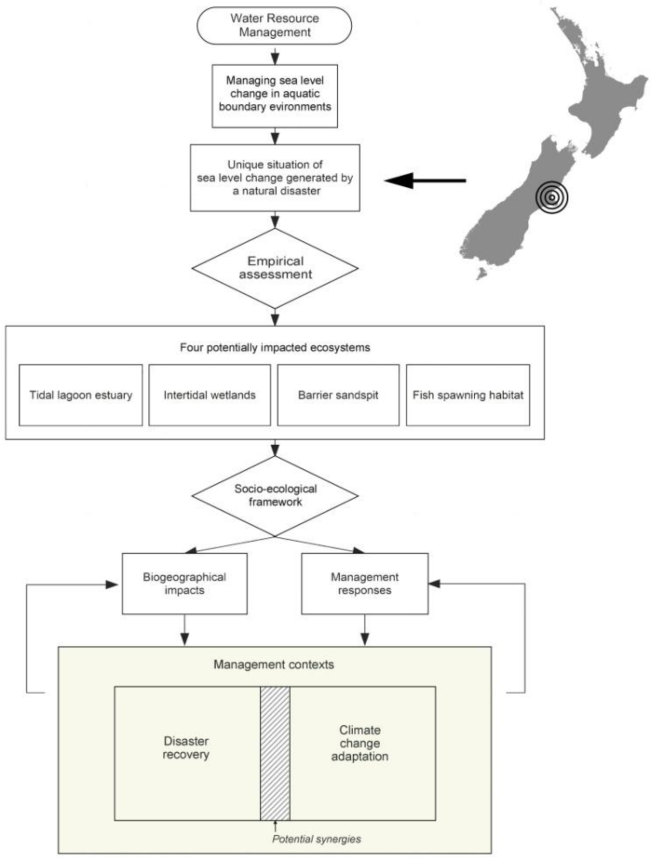

Fig. A1 Overview of the research programme for evaluating relative sea-level change effects of tectonic displacement caused by earthquakes in Christchurch, New Zealand.

PART 1

SHORELINE CHANGE ANALYSES

Part 1 of the thesis investigates the phenomenon and consequences of shoreline position changes that arose from tectonic displacement, and similarly, are an important aspect of adaptation to climate change. The three studies are each exploratory in nature and deal with different thematic and geographical scales within the overall context of a linked socio-ecological system responding to a pervasive disturbance. The studies were conceived and designed sequentially building on the results of the initial investigations..

Chapter 1 presents empirical observations of environmental change pertaining to the Avon Heathcote Estuary Ihutai as a natural feature and ecosystem that is the subject of protection under current policy and law. The key themes in this paper are the quantification of ground movements and shoreline position changes in relation to sea levels together with the characterisation of spatio-temporal variability in the observed patterns of change. The methodologies employed are underpinned by the availability of high resolution spatial datasets that cover a considerable period post-disaster and include a comparable dataset representative of pre-earthquake conditions.

Chapter 2 extends the theme of relative sea-level and shoreline position changes to examine consequences for coastal wetlands and the considerable ecosystem services they provide. The methodologies employed focus on landscape-scale aspects through the use of geospatial analyses to detect and quantify spatiotemporal variation across the study system as a whole before exploring specific mechanisms of change at the scale of individual wetlands. To address the potential for lag effects in the vegetation response the temporal aspects of this study were extended to as late as possible in the research programme (resulting in the 2019 data point).

Chapter 3 builds on these considerations to consider the unique challenge of shoreline management on an estuarine barrier sandpit. This study is one of few we are aware of that have addressed the conservation of peri-urban sandspits despite these being relatively common worldwide. The methodologies employed extend the baseline and transect analyses used in the above studies to simultaneously consider both open coast and estuarine shoreline scenarios under sea-level rise.

Chapter 1

Coastal tectonics and tipping points: response of a tidal lagoon to co-seismic sea-level change

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Abstract

We investigated the response of a tidal lagoon system to a unique situation of relative sea-level change induced by powerful earthquakes (up to M_w 7.1) on the east coast of New Zealand in 2010-2011. Spatiotemporal impacts were quantified over five years post-earthquake using airborne light detection and ranging (LiDAR) datasets complemented by hydrodynamic modelling and evaluation of anthropogenic factors. Ground level changes included examples of uplift and extensive areas of subsidence (ca. 0.5 m). Consequences included a reduction of total intertidal area and compression of the supratidal zone, both associated with the loss of important habitat. Uplift reversed the effects of coastal squeeze caused by previous land reclamations in the intertidal zone. Subsidence in other areas resulted in new coastal squeeze examples. Quantification of area gains and losses demonstrated the importance of natural lagoon expansion into areas of suitable elevation and showed that they may be necessary to offset losses from coastal squeeze elsewhere. The combination of intertidal area reductions and shallowing suggests tidal prism changes that may alter salinity and drive further habitat shifts. These empirical observations highlight the influence of anthropogenic factors on the outcomes of rapid sea-level changes and the importance of the disaster recovery process as a context for climate change adaptation.

Keywords
Resilience, risk, impact assessment, coastal squeeze, disaster recovery, climate change adaptation.

1.1 Introduction

Coastal river mouths and estuaries are characteristic natural features supporting highly productive ecosystems, important biodiversity and a wealth of natural resources (Kennish 1986). Their benefits include food, coastal protection, recreational opportunities, water filtration and many other ecosystem services (Pendleton 2008; Thrush et al. 2013). Economic evaluations have shown their high value to society (Barbier et al. 2011; Costanza et al. 1998), underpinned by the popularity of coastal floodplains for human uses and settlement (Lichter et al. 2011; McGranahan et al. 2007). Unfortunately, the integrity of estuarine environments has also suffered from anthropogenic effects (Kennish 2002). Pervasive issues include hydrological and morphological changes associated with nearby land-uses and the cumulative effects of land reclamation within the intertidal zone (Duarte et al. 2015; Perkins et al. 2015). These aspects illustrate a need for conservation measures that address both local spatial planning and catchment-wide land-use trends.

Whilst accommodating dynamic natural environments is already difficult in heavily populated areas, climate change introduces a further considerable threat (Martínez et al. 2007). Under sea-level rise low-lying landscapes may be eliminated if the landward retreat of erodable shorelines becomes constrained by anthropogenic infrastructure such as engineered coastal defences (Berry et al. 2013; Chapman 2012; Robins et al. 2016). Specific threats of ‘coastal squeeze’ situations include the erosion of substrate-dependent ecosystems seaward of such defences, and drowning as a result of exposure to inundation. In both cases, constraints on natural system movement underpin ecosystem risk and are often anthropogenic in origin (Martinez et al. 2014; Schleupner 2008). Despite this, built infrastructure is amenable to improvement through design (Macreadie et al. 2017) and may include the use of nature-based solutions to help overcome societal challenges (Cohen-Shacham et al. 2019; Kabisch et al. 2016). Such solutions include the concept of ecosystem-based adaptation, defined as “adaptation that integrates ecosystem services and biodiversity into a strategy to limit the adverse impacts of climate change” (Renaud et al. 2016; UNEP 2010). Challenges, however, include the need for strategies to begin the process of adaptation planning and the identification of solutions are both workable for affected communities and effective in accommodating natural ecosystems (Bardsley & Sweeney 2010; Füssel 2007).

In this study we investigated a unique situation of sea-level change associated with a tectonic displacement event. A defining feature of this case involved large areas of subsidence with an accompanying sea-level rise (ca. 0.5 m) that affected coastal communities and generated long term environmental change. The context provided a novel opportunity for the empirical assessment of impacts and evaluation of contributing factors in a low-lying socio-ecological system typical of many facing the global challenge of rising sea levels.

1.1.1 Tectonic displacement context

The Canterbury Earthquake Sequence (CES), involved a series of strong earthquakes (up to M_w 7.1) beginning 2010 on the east coast of New Zealand's South Island. The most serious earthquake occurred on 22 February 2011 beneath the city of Christchurch (Fig. 1.1). As one of New Zealand's worst natural disasters, it caused 185 fatalities and damages estimated at NZ$40 billion, or approximately 20% of the Gross Domestic Product (Kaiser et al. 2012; Potter et al. 2015). Three other earthquakes exceeded M_W 6.0, all on previously unrecognised fault lines (Beavan et al. 2012; Bradley et al. 2014). Along with catastrophic effects on built infrastructure, the CES caused severe impacts on the natural environment. Many of these were associated with surface deformation phenomena including liquefaction, lateral spread, subsidence and landslides (Quigley et al. 2016; Robinson et al. 2012; Zeldis et al. 2011). Many residential areas were affected by increased flood risk associated with subsidence and coastal defence breaches, particularly in the east of the city (Hughes et al. 2015). Societal responses included central government acquisition of thousands of residential properties along the estuary shoreline and lower river corridors, creating a rare opportunity for reconfiguring the relationship between people and the aquatic environment (Orchard 2017a).

This study investigates disturbance and resilience aspects of the earthquake-induced change. Our particular focus was the identification of long-term effects on the aquatic margins and footprint of the Avon Heathcote Estuary Ihutai, a tidal lagoon typical of many worldwide (Hume et al. 2007). Complications for gaining a comprehensive picture arise from the lag times of responses and the potential for further time-varying effects associated with physical change. Initially, the latter were significant due to the high frequency of aftershocks and associated further land movements and erosion effects (Beavan et al. 2012; Quigley et al. 2013). To address this, we collected data over a considerable period as conditions stabilised. The magnitude and frequency of aftershocks have generally declined since 2011, with the exception of a 5.7 M_W earthquake on 14 February 2016.

In this paper, we provide a comprehensive overview of relative sea-level changes, shoreline movements, and impacts on the extent of intertidal areas associated with tectonic displacement. We draw conclusions for the management of sea-level rise derived from direct empirical analysis, and discuss natural disaster recovery and climate change adaptation (CCA) principles that may be identified from this case.

1.2 Methods

1.2.1 Study area

The study area is within the city of Christchurch on the east coast of New Zealand’s South Island (Fig. 1.1). The estuarine system includes two river mouth environments (Avon Ōtākaro and Heathcote Ōpāwaho) and several smaller tributaries connected to a tidal basin of ca. 8 km^2. The lagoon is a barrier-built system enclosed by a 10 km long beach and sand-spit formation. In recent history it has been permanently open to the Pacific Ocean via an entrance channel is located in the southern corner of Pegasus Bay, a shallow embayment 54 km long extending north from Banks Peninsula (Hicks 1998; Kirk 1979). The estuary supports a wide variety of native birds, fish and invertebrate species, and indigenous plant communities including seagrass meadows, saltmarsh and other coastal wetland types (Jones & Marsden 2007). It is an important site for shorebirds and migratory waders, supporting aggregations of at least 13 species exceeding the 1% international importance threshold defined by Wetlands International (Crossland 2013; Delaney & Scott 2006). We have adopted a bilingual naming for major aquatic features because the estuary is of high significance to Māori for *mahinga kai* (food gathering), and other traditional practices (Jolly & Ngā Papatipu Rūnanga Working Group 2013; Tau et al. 1990). However, residential and industrial development has had adverse impacts on cultural values, especially those dependant on the maintenance of natural ecosystems and traditional resources (Lang et al. 2012; Pauling et al. 2007). A large proportion of the estuarine shoreline has been modified by seawalls and stopbanks, some of which are associated with a sewage treatment facility on the western shore (Fig. 1.1).

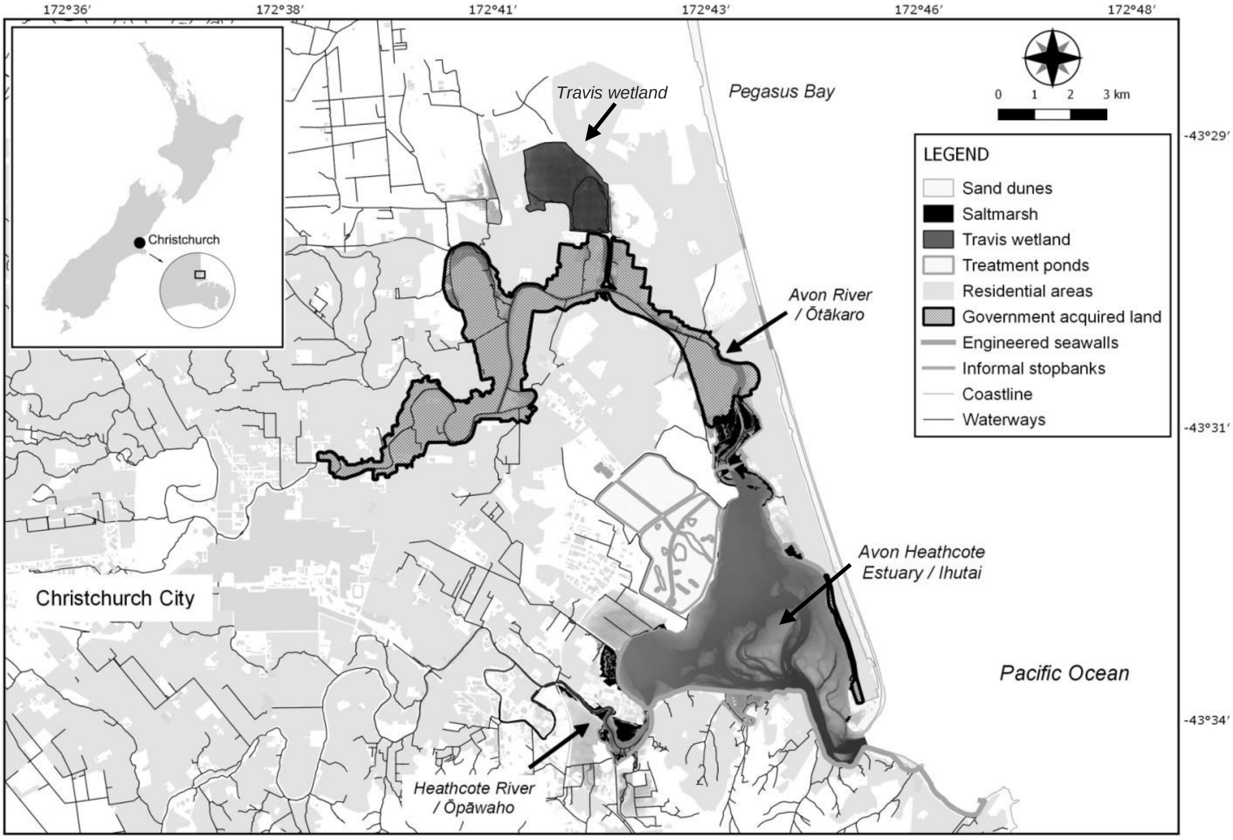

Fig. 1.1 Configuration of the Avon-Heathcote Estuary Ihutai and surrounding area in Christchurch, New Zealand, showing the position of key natural features, coastal defences, and earthquake-impacted land that was acquired by the New Zealand government after the Canterbury earthquakes of 2010 – 2011.

1.2.2 LiDAR data and digital elevation models

Shoreline change was investigated using geographic information system (GIS) analyses of digital elevation models (DEM) derived from LiDAR datasets. Four datasets were available with complete coverage of the study area. These include a pre-quake (2003) dataset and others captured after key events in the CES (Table 1.1a). Bare earth DEMs representing averaged ground-return elevations were included in the LiDAR products at 1 x 1 m resolution for the 2015 survey, and 5 x 5 m resolution for all others (Canterbury Geotechnical Database 2014; LINZ 2017). Identical DEM configurations were developed by reprocessing the 2015 DEM to 5 m resolution. Elevation errors have at least three components that include locality-dependent interpolation errors, potential geoid errors, and measurement errors in the underlying LiDAR point cloud. However, the 5 m DEMs have relatively high accuracy due to the quantity and geographic spread of point elevations captured in the source data. Table 1.1 shows the estimated horizontal and vertical accuracy for each dataset excluding GPS network error and approximations within the New Zealand Quasigeoid 2009 reference surface which have a vertical accuracy of ±0.06 m (Canterbury Geotechnical Database 2014).

To investigate inundation patterns in the lower intertidal zone we used DEMs developed from echosounder surveys covering areas of the estuary and river channels that were submerged during the LiDAR surveys and therefore not captured. Although only two such DEMs were available due to the limited collection of bathymetric data over the CES, they are representative of pre- and post-earthquake conditions. Each DEM was generated using triangular irregular network (TIN) interpolation constrained with manually digitised break-lines following the main estuary channels to preserve channel connectivity and minimise interpolation artefacts (Measures & Bind 2013). Data sources are provided in Supplementary Material (Table S1.1).

1.2.3 Shoreline change

Shoreline sampling transects were developed using the AMBUR package ("Analyzing Moving Boundaries Using R") for detecting movement and trend changes relative to a baseline position (Jackson et al. 2012). A baseline was developed from the Land Information New Zealand (LINZ) 1:50,000 coastline polyline and smoothing to improve fit with 0.075 m aerial imagery (LINZ 2016). A set of perpendicular transects (n=1428) were cast at 10 m spacing from a start point at the southern estuary entrance (Lat. 43° 56' S, Long. 172° 75' E). Transect lengths were adjusted to cover all areas of potential tidal inundation (Fig. 1.2a). At river mouths, the sampling area was confined to the confluence with the main tidal lagoon basin. This breakpoint approximates the Coastal Marine Area boundary, an important jurisdictional division within environmental legislation (Orchard 2011). Ground-level changes were assessed by point sampling of the DEMs at 1 m spacing on the sampling transects followed by differencing. Spatial variation was investigated by grouping transects within five contiguous zones (Southshore, South Brighton, Bromley, Ferrymead and Redcliffs), reflecting changes in shoreline aspect and proximity to river mouths (Fig. 1.2a).

Shoreline position changes were calculated for two tidal heights of particular interest: Highest Astronomical Tide (HAT) and Mean High Water Springs (MHWS). These were delineated as orthographic heights (Table 1.1) obtained from the average predicted values over a full 18.6 year tidal cycle (2000-2018) at Port Lyttelton (LINZ 2018a). For MHWS the yearly variation is 0.1– 0.15 metres using current predictions (1 July 2018 – 30 June 2019). The current MHWS height is 2.6 m above chart datum, or 0.11 m above the mean value for the full tidal cycle (LINZ 2018b). These considerations do not affect HAT which is based on the full cycle. Shoreline position changes were quantified by extraction of the MHWS and HAT heights from each DEM followed by contour and intersection analysis on the sampling transects. Changes were measured as seaward or landward movement relative to the baseline.

1.2.4 Tidal inundation

Supratidal area changes were assessed for the four DEMs using the elevation band bounded by MHWS and HAT (Table 1.1b). Upstream freshwater regions were removed from the analysis by clipping at the limit of salt water intrusion as measured in field surveys on spring high tides. All DEM analyses assumed full connectivity between adjacent hydrological basins, regardless of engineered modifications such as tidal gates and seawalls. Due to extensive earthquake damage to such infrastructure, this assumption approximates the actual post-disaster context and provides an assessment of potential inundation through connectivity improvements.

For mid-lower intertidal ranges where water surface slope and hydrological connectivity can strongly influence the inundation regime, we used a calibrated Delft3D hydrodynamic model based on the bathymetric DEMs (Measures & Bind 2013). The model extends 15 km into the open ocean and has a curvilinear grid with horizontal resolution of ca. 20 m within the rivers and estuary basin (Fig. 1.2). Each grid cell is split into five vertical layers, with layer thickness proportional to water depth. Pre- and post-earthquake versions are identical apart from the DEM used to assign the bed level within each cell (Measures & Bind 2013). Month long simulations were computed using the pre- and post-earthquake models to quantify changes in inundation. Both simulations modelled identical astronomic tidal conditions and median river flows (Avon Ōtākaro = 1.65 m^3/s, Heathcote Ōpāwaho = 0.77 m^3/s). It is important to note that the model extent does not include all of the floodable intertidal areas in the estuary catchment. This results in an underestimation of potential inundation in the upper intertidal range, particularly above MHWS. The static DEM analyses are therefore more reliable for the investigation of changes at these higher elevations.

1.2.5 Data analyses

Raster analysis was used to quantify spatiotemporal changes in the elevation bands of interest with differencing between rasters to quantify ground level changes over time. Summary statistics were calculated for shoreline position changes on the sampling transects. Differences were identified using Kruskal-Wallis rank sum tests for independent variables of time and locality followed by post hoc pairwise tests where there was a significant movement relative to the pre-earthquake (2003) baseline. Hydrodynamic modelling outputs were post-processed to calculate the bed area inundated for different proportions of time under pre- and post-earthquake conditions, and summarised for the five main basin localities and additional upstream portions of the two major river catchments (Fig. 1.2a). Geospatial analyses were conducted in NZTM 2000: ESPG 2193 projection using QGIS v 2.18 (QGIS Development Team 2017). Statistical analyses were conducted in R v3.3.3 (R Core Team 2017). Model post-processing was conducted in MATLAB using functions from the OpenEarthTools repository (https://publicwiki.deltares.nl/display/OET/Tools).

Table 1.1 Data sources.

(a) LiDAR datasets and specifications.

Timing in earthquake sequence	Major earthquake events in period†	LiDAR acquisition dates	Supplier	Commissioning agencies	Accuracy specification (m) vertical	Accuracy specification (m) horizontal
Pre-earthquake	n/a	6-9 Jul 2003	AAM Brisbane	Christchurch City Council	±0.15	±0.55
Post-February 2011	Darfield 7.1 M_w Christchurch 6.4 M_v	20-30 May 2011	AAM Brisbane	Christchurch City Council	±0.07	±0.55
Post-June 2011	Christchurch 6.2 M_v	2-3 Sep 2011	New Zealand Aerial Mapping	Earthquake Commission	±0.07	±0.55
Post-December 2011	Christchurch 6.1 M_v	5 Oct-7 Nov 2015	AAM Brisbane	Canterbury Regional Council	±0.20	±1.00

† Beavan et al. (2012). December 2011 magnitude represents combined moments of the two largest tremors.

(b) Comparison of tidal and benchmark heights at the Standard Port of Lyttelton (Lat. 43° 36´S., Long. 172° 43´E) against chart datum and two vertical datums in current use. LVD37 = Lyttelton Vertical Datum 1937, NZVD = New Zealand Vertical Datum 2016.

Tidal level or feature	Height above Chart Datum† (m)	Orthographic heights (m) LVD37	Orthographic heights (m) NZVD2016
HAT	2.72	1.479	1.072
MHWS	2.49	1.249	0.842
B40V geodetic reference mark	4.478	3.237	2.83

† LINZ (2018a, 2018b).

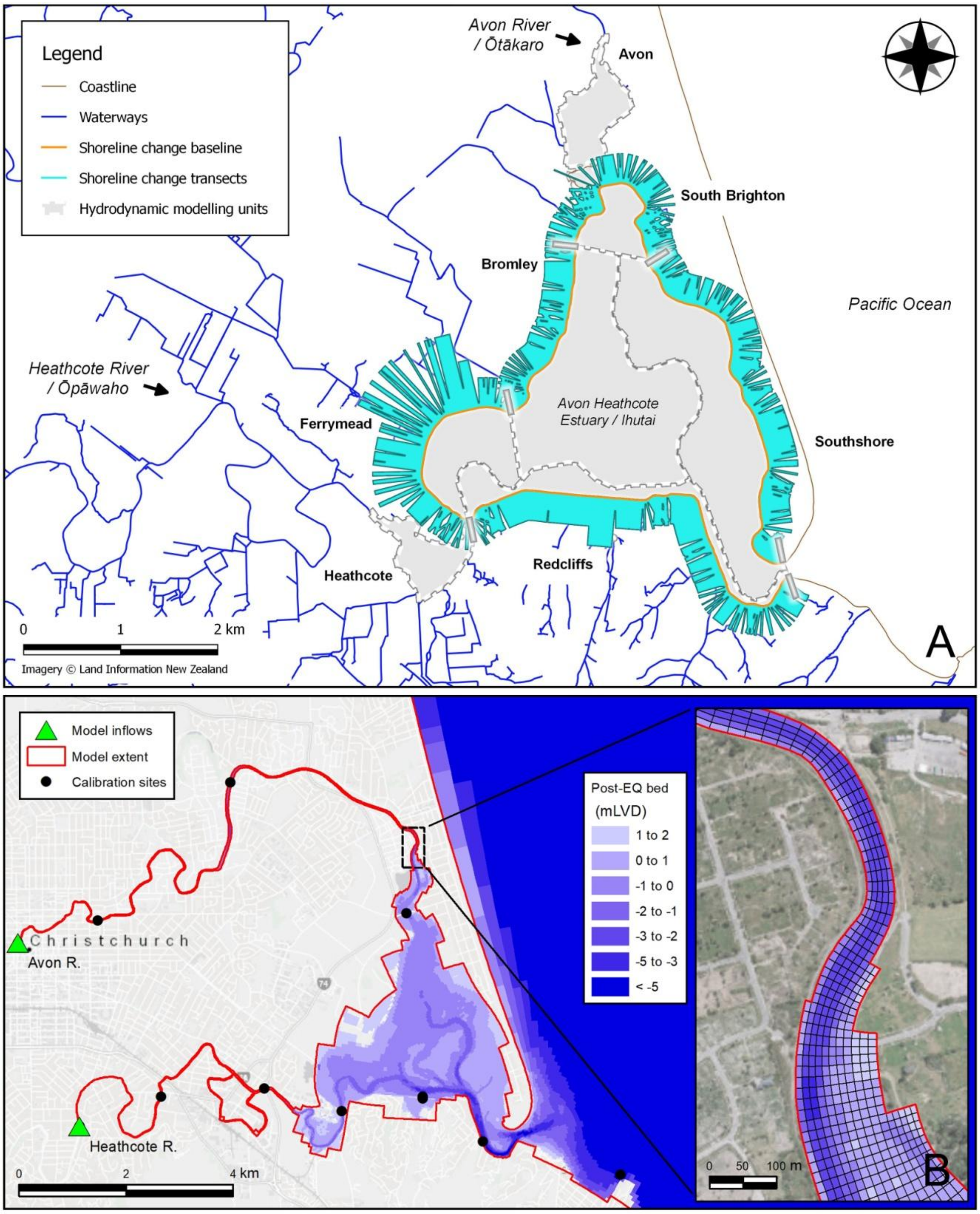

Fig. 1.2 Sampling design. (A) configuration of sampling transects for shoreline change analysis and estuarine localities used for spatial comparisons including boundaries of hydrodynamic modelling units. (B) Hydrodynamic model extent and example of model cell fit in relation to the estuary shoreline.

1.3 Results

1.3.1 Ground level changes

Relative to pre-quake (2003) conditions, results from individual sampling points (n = 475,000) showed more uplift than subsidence and marked differences between localities (Fig. 1.3). Subsidence occurred in the South Brighton area near the Avon Ōtākaro river mouth throughout 2011, though this had reduced by 2015. Here, and in other areas, the results also illustrated the ongoing nature of change in relation to key time periods in the CES, highlighting the difficulty of drawing conclusions from singular before-after comparisons. Large variations in the measured changes were seen in some areas, particularly in Redcliffs which is located at the foot of prominent hill-slopes (Fig. 1.3). This reflects the elevation signature of horizontal displacements on sloping ground that cannot be separated from the assessment of vertical movement at point coordinates. However, these effects are unlikely to affect shoreline change analyses due to the relatively flat topography of intertidal areas.

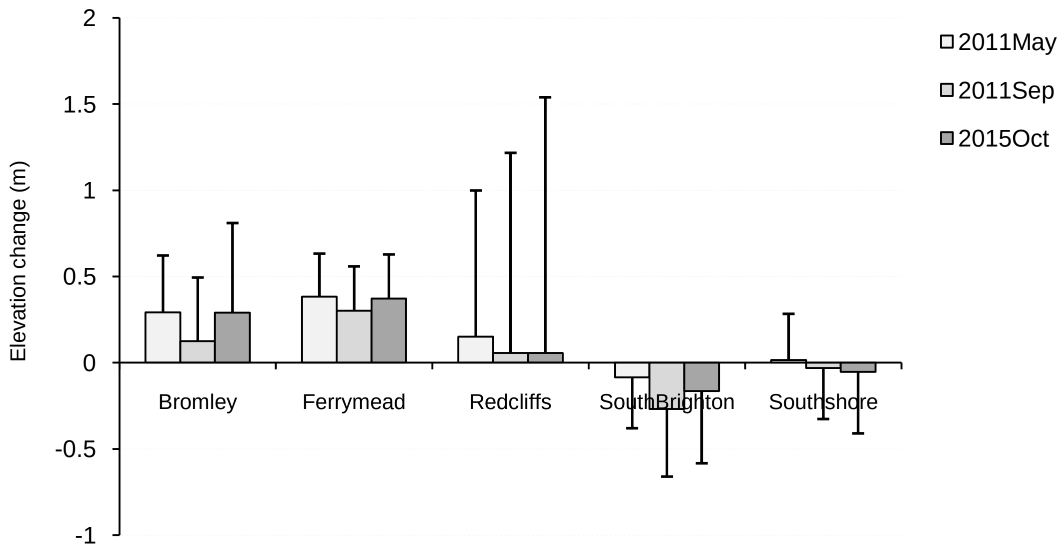

Fig. 1.3 Mean ground surface elevation changes relative to 2003 in the Avon Heathcote Estuary Ihutai for five contiguous localities on the shoreline of the tidal lagoon basin. Error bars represent one standard deviation. These results were obtained by differencing of individual sampling points (n = 475,000) located on shore-perpendicular transect lines around the estuary perimeter.

1.3.2 Shoreline movement

Large movements were detected in the position of post-quake versus pre-quake shorelines. Changes of over 500 m seaward were recorded for MHWS and HAT on individual transects. Landward shifts were also recorded to a maximum of 269 m for MHWS and 182 m for HAT (Fig. 1.4a). Shoreline change was significantly different between localities for both MHWS (Kruskal-Wallis $\chi^2 = 107.86$, $df = 2$, $p < 2.2e\text{-}16$) and HAT (Kruskal-Wallis $\chi^2 = 88.67$, $df = 2$, $p < 2.2e\text{-}16$), with several pronounced trends being evident. At Ferrymead, large seaward shifts were recorded for MHWS and HAT (Fig. 1.4a), with the majority of movement having occurred by May 2011 as reflected by mean shifts of 161 m (HAT) and 154 m (MHWS) relative to 2003. At Southshore and Redcliffs the post-quake positions were consistently seaward of the 2003 shoreline but the movement was less than at Ferrymead (10 – 50 m). South Brighton and Bromley shorelines experienced little change on average relative to 2003. At South Brighton this was associated with large variances in the magnitude and direction of shifts on individual transect lines, whereas at Bromley little change was recorded on most transects due to the influence of shoreline armouring which is extensive in this area (Fig. 1.4a).

Mean shoreline change for the estuary as a whole was in a seaward direction for both MHWS and HAT shorelines. At all three post-quake time points, seaward movement in the HAT line was greater than for MHWS (Fig. 1.5). Shoreline position differences after the major earthquake of February 2011 (represented by May 2011 data) were significant for MHWS and HAT ($p < 0.001$). Wilcoxon pairwise comparisons for MHWS also showed that the end-point position (October 2015) was significantly different from the May 2011 position ($p < 0.001$). HAT changes conformed to a similar pattern, although statistical analysis showed no significant difference ($p = 0.81$) due to greater variation on individual transect lines (Fig. 1.5). These temporal effects are interpreted as a modest expansion of the tidal lagoon basin between 2011 and 2015 that has reduced the initial contraction caused by the February 2011 earthquake. The same general trend can be seen in temporal pattern at most of the individual locations (Fig. 1.4b). However, these effects were not always directly proportional to the ground level changes shown in Fig. 1.3. For example, mean ground levels in 2015 at Southshore followed a trend of continuing subsidence whereas shoreline movement was in a seaward direction at the same time. This is potentially explained by the weathering of erodable surfaces accompanied by accretion at lower elevations.

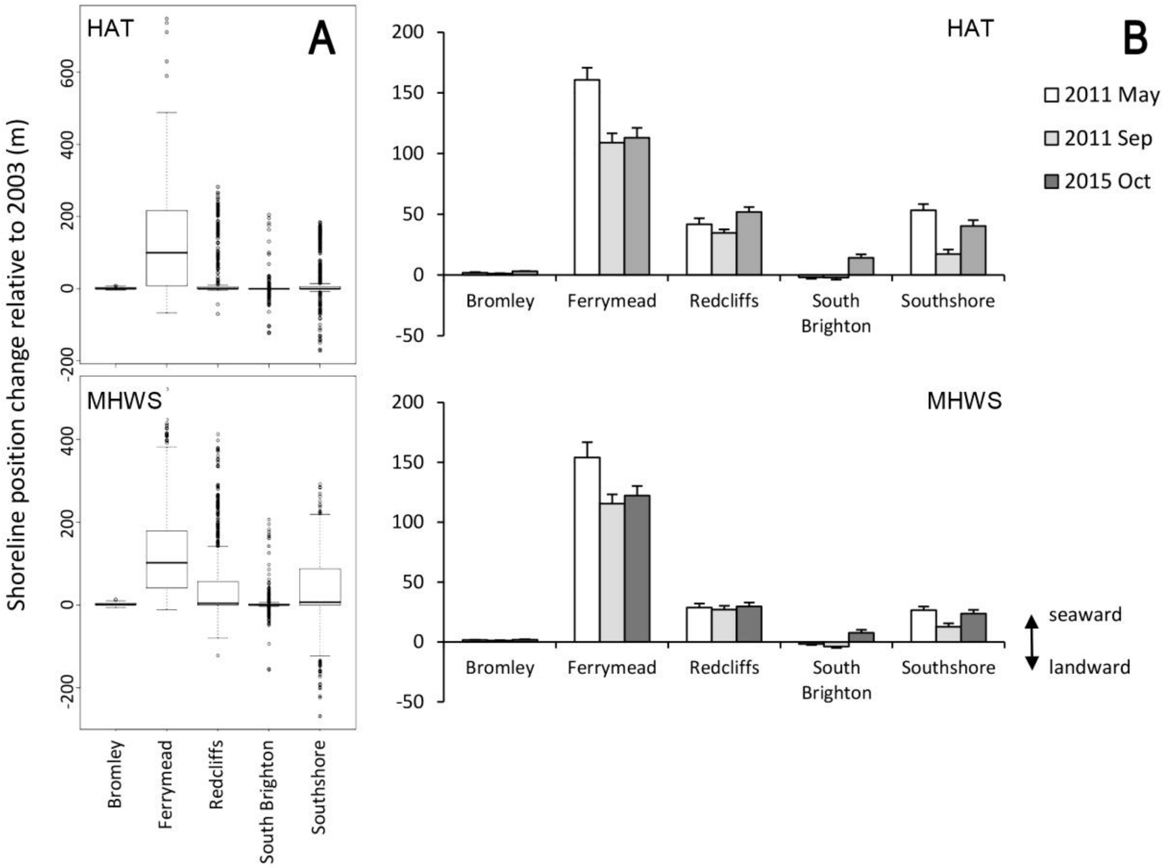

Fig. 1.4 Pattern of movement of two shorelines (HAT and MHWS) during the Canterbury Earthquake Sequence (CES) at the Avon Heathcote Estuary Ihutai. (A) Box plots showing the total range of shoreline changes recorded at three post-quake time points for five estuarine localities relative to the July 2003 (pre-quake) position. Boxes show the median and interquartile range. (B) Mean shoreline position changes relative to 2003 for each of the three points in time. Error bars are one standard error of the mean. See Table 1 for relationship to major tectonic events. Note different scales on the Y axis between (a) and (b). HAT = Highest Astronomical Tide. MHWS = Mean High Water Springs.

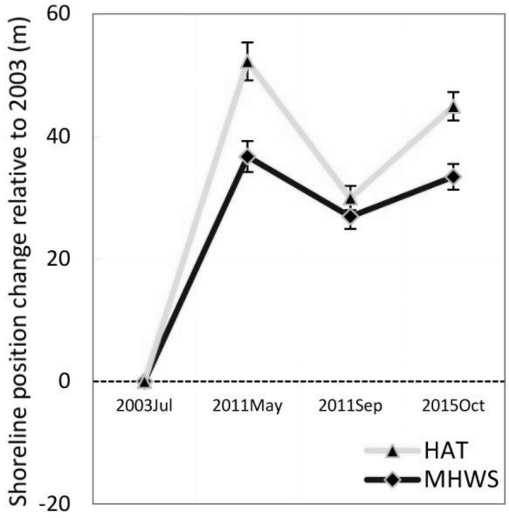

Fig. 1.5 Mean shoreline position change since 2003 for the Avon Heathcote Estuary Ihutai as a whole, as recorded at each of three points in time during the Canterbury Earthquake Sequence. Error bars are one standard error of the mean. HAT = Highest Astronomical Tide. MHWS = Mean High Water Springs.

1.3.3 Intertidal area changes

Based on the most recent time point (2015), the total estuarine area below HAT has reduced by 54.7 ha, and 33.4 ha for MHWS (Table 1.2). The HAT-MWHS difference implies compression of the supratidal zone of around 21.4 ha (represented by the area bounded by HAT and MHWS). However, there were pronounced differences between time periods over the course of the CES with expansion evident between May and September 2011 but contraction at other times. In comparison to other localities, changes in the Avon area make a disproportionate contribution to the net overall impact (Fig. 1.6).

Impacts of the major February 2011 earthquake included reductions in the area below the elevation of both HAT and MHWS in Ferrymead and Heathcote, consistent with the dominance of uplift effects towards the southwest (Fig. 1.6). At the same time there were increases in the Avon area, consistent with subsidence effects further north. System-wide impacts are explained by a combination of tilting and a dominance of uplift in overall ground surface displacements, leading to reductions of 44.5 ha in the area below HAT, and 22.6 ha for MHWS (Table 1.3). In the next time period (May – September 2011), large increases were observed in the Avon Ōtākaro area (144.5 ha below HAT, 117.6 ha for MHWS), and small increases elsewhere, consistent with widespread subsidence. Relative to pre-quake (2003), the intertidal area was 139 ha larger and included a modest increase (16.9 ha) in the supratidal zone. However, this estuarine expansion was relatively short-lived due to a dramatic reversal in the Avon area in the next time period (to 2015). The overall results are illustrative of complex spatiotemporal patterns that are important to resolve in determining longer term trends (Fig. 1.6). The most recent measurements showed the estuarine area was similar to May 2011 and smaller than in 2003 (Table 1.2). See Supplementary Material Fig. S1.1 for a map of baseline (2003) and endpoint (2015) conditions.

Table 1.2 Summary of key changes in the areal extent of the Avon Heathcote Estuary Ihutai over the period 2003 – 2015. The three later dates mark important time periods in the Canterbury Earthquake Sequence whilst the 2003 baseline is representative of pre-earthquake conditions. HAT = Highest Astronomical Tide. MHWS = Mean High Water Springs.

Estuarine areas	**Areal extent (ha)†**			
	Jul 2003	**May 2011**	**Sep 2011**	**Oct 2015**
area below HAT	1190.5	1146.0	1329.5	1135.8
area below MHWS	1032.3	1009.7	1154.4	998.9
supratidal area bounded by HAT and MHWS	158.2	136.3	175.2	136.9

† calculations assume full hydrological connectivity between adjacent basins within the elevation range of interest.

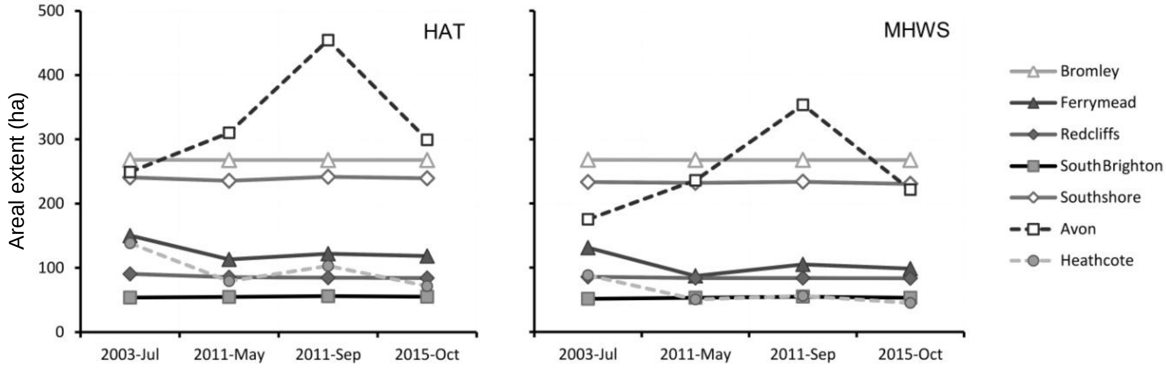

Fig. 1.6 Changes in the extent of estuarine areas below the elevation of Highest Astronomical Tide (HAT) and Mean High Water Spring tide (MHWS) for seven areas within the Avon Heathcote Estuary Ihutai catchment over 2003 – 2015.

Hydrodynamic modelling showed that subtidal area losses contributed additional intertidal area due to shallowing of the main estuary basin (Fig. 1.7). The biggest changes occurred in the uplifted southern parts of the estuary: Heathcote, Ferrymead, Redcliffs, and the southern parts of Bromley and Southshore (Fig. 1.7). In these areas the total intertidal area has generally increased due to the exposure of channels which were previously permanently submerged at low tide, and an accompanying reduction in the subtidal area. Areas which were already intertidal are now exposed for a longer duration on each tidal cycle. However, there are few areas which were previously intertidal and that are now above the modelled reach of the tide. This counterintuitive result can be explained by the observation of only small areas that were shallowly submerged at high tide in the pre-earthquake state. This is particularly evident for areas inundated for less than 30% of the time (Fig. 1.7) and is indicative of upper intertidal reclamations having already occupied those areas. The combination of both shallowing and an overall decrease in intertidal area at high tide suggests a reduced tidal prism with the potential to drive further habitat shifts through salinity effects.

At Bromley, uplift was insufficient to move either HAT or MHWS shorelines. This shows that the ‘coastal squeeze’ impacts of seawalls had extended well into the intertidal range and exceeded the tipping point for persistence of a high tide beach, even with the benefit of uplift. Similar results indicative of pre-earthquake degradation were also evident in Southshore and Ferrymead where pre-quake upper intertidal zones were much smaller than lower intertidal zones, but expanded markedly following uplift (Fig. 1.7). The CES both illuminated and reversed the pre-quake situation where land-uses were occupying areas that would otherwise be regularly inundated on moderate-sized tides. Moreover, these results demonstrate that the post-quake state remains vulnerable to sea-level rise impacts due to the current position of seawalls.

In areas that subsided (Avon and South Brighton), the hydrodynamic modelling is less reliable as an indicator of upper intertidal change due to limitations of model domain which excluded land outside of the estuary that is now subject to tidal inundation. However, these areas were captured within other assessments using the static DEMs.

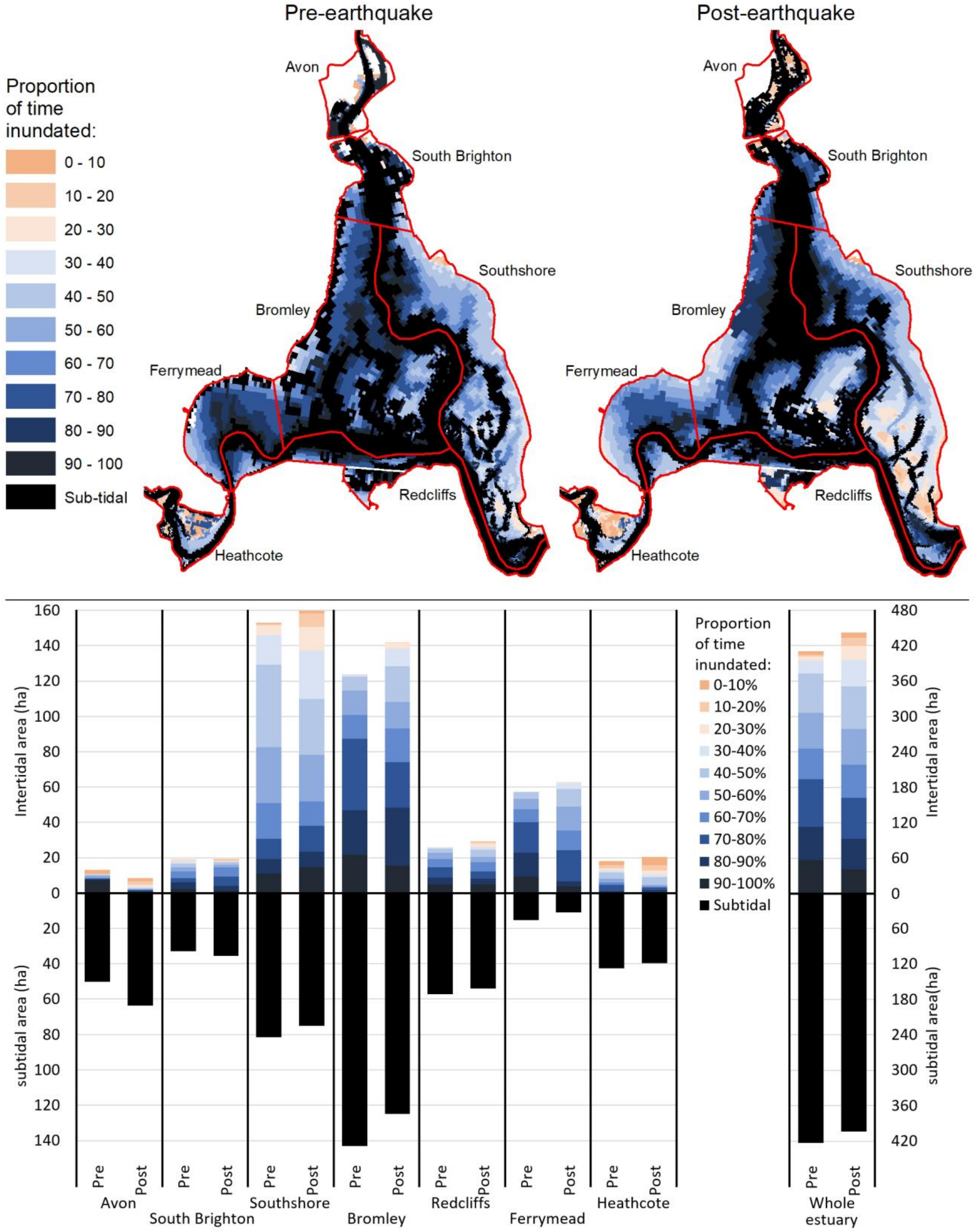

Fig. 1.7 Hydrodynamic model results for pre- and post-earthquake bed topographies representative of the Canterbury Earthquake Sequence and showing changes in the intertidal area inundated over a typical monthly tidal cycle. Both simulations used identical astronomic tidal conditions and median river flows (Avon Ōtākaro = 1.65 m^3/s, Heathcote Ōpāwaho = 0.77 m^3/s).

1.3.4 Impacts of sea-level rise

Appreciable subsidence occurred only in the Avon Ōtākaro catchment and adjacent South Brighton portion of the main lagoon basin. However, these areas provide an excellent opportunity to assess the actual effects of higher sea levels on a pre-disturbance landscape. At South Brighton, the measured sea-level rise was greatest in September 2011 with subsidence of 27 cm on the sampling transects compared to 2003 (Table 1.3). Despite this, shoreline change analysis showed only small landward movements in the position of HAT and MHWS (means of 2.2 m and 4.1 m respectively). The increase in area below HAT (2.1 ha) was much less than for MHWS (3.4 ha), leading to a 1.3 ha (58%) reduction in the land available between HAT and MHWS. The 2015 results showed a general reversal of these effects consistent with the raising of ground levels. Relative to 2003, the end result was an intertidal area loss of 1.3 ha, and a 0.3 ha compression of the supratidal zone (Table 1.3).

Table 1.3 Effects of higher sea levels on mean estuarine shoreline position and intertidal area at South Brighton for three time points during the Canterbury Earthquake Sequence relative to pre-quake (2003) conditions. HAT = Highest Astronomical Tide. MHWS = Mean High Water Springs.

Key changes since 2003†	**Assessment dates**		
	May 2011	**Sep 2011**	**Oct 2015**
Mean ground level elevation (m)	-0.09	-0.27	-0.16
Shoreline retreat (landward movement) (m)			
HAT	2.0	2.2	-14.2
MHWS	1.8	4.0	-7.5
Areal extent of intertidal areas† (ha)			
area below HAT	0.9	2.1	1.3
area below MHWS	1.7	3.4	1.6
supratidal area between HAT and MHWS	-0.8	-1.3	-0.3

† calculations assume full hydrological connectivity between adjacent basins within the elevation range of interest.

Figure 1.8 illustrates the mechanisms of change in supratidal zones as observed in South Brighton and the lower Avon Ōtākaro catchment under conditions of reltive sea-level rise. This area has extensive anthropogenic shoreline modifications. In the Bexley wetlands (arrowed) impacts included a large loss of supratidal area (Fig. 1.8a, b). Contributing factors included the raising of nearby ground levels to facilitate a housing development that had the effect of truncating landward movement of the supratidal zone under conditions of sea-level rise. On the opposite (eastern) shoreline, land-fills are not prominent in the development pattern despite the close proximity of residential property to the estuary. Some of these properties are now exposed to inundation at water heights of HAT (and less). However, these areas were not subject to the government land acquisition. As a result, these areas are less likely to be candidates for managed retreat strategies that could include the creation of future estuarine space despite that ground levels are much more favourable than in areas that were modified by land-fill (Fig. 1.1). On this eastern shoreline, the 2015 bounce-back effect (Table 1.3) is also notable as illustrated by the expansion of supratidal areas seaward of the shoreline armouring line (Fig. 1.8c, d). As yet, however, these changes are insufficent to restore the major losses inccurred earlier in the CES.

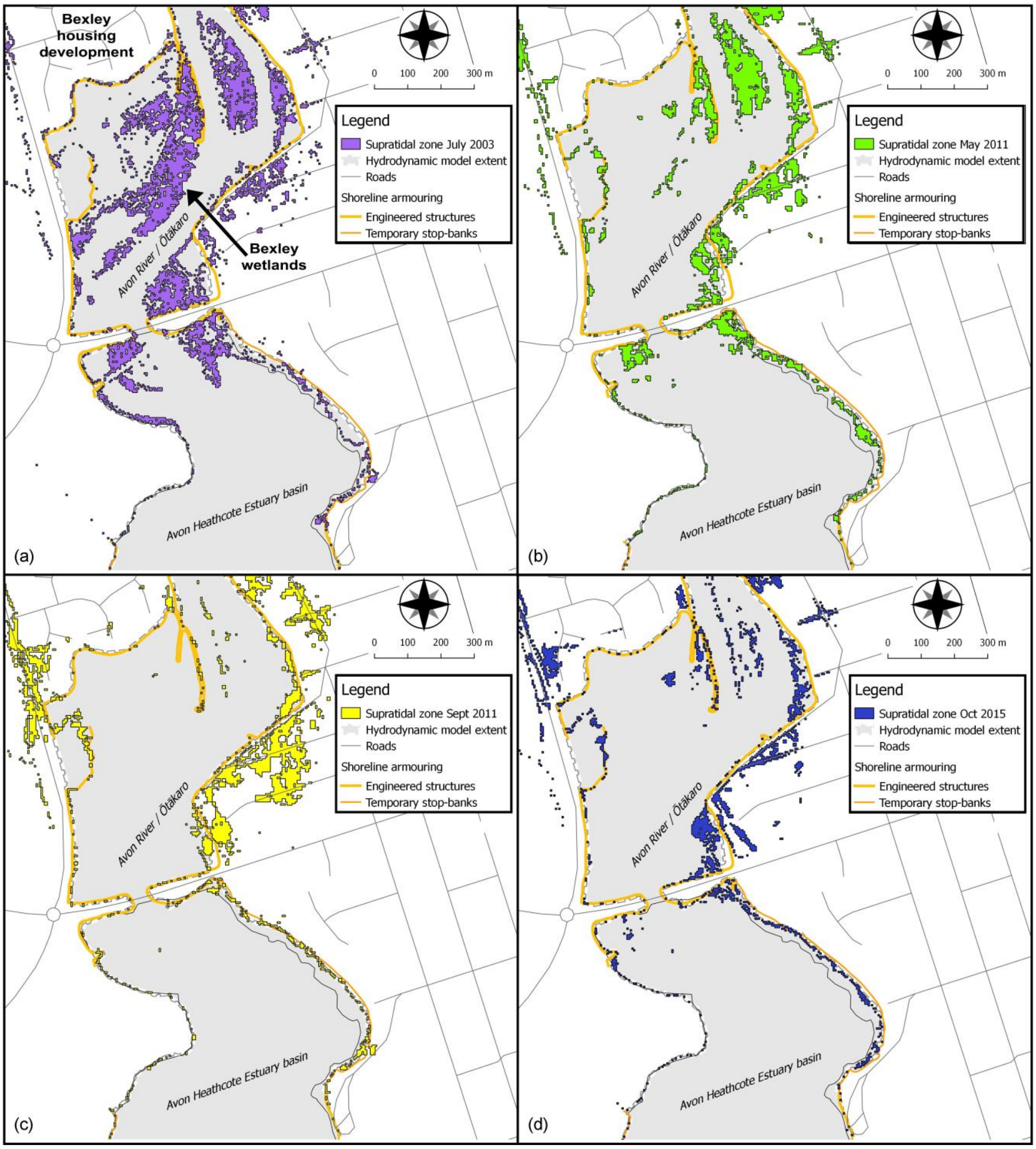

Fig. 1.8 Changes in the areal extent of the supratidal zone modelled as the elevation band between Highest Astronomical Tide and Mean High Water Springs over the period 2003 – 2015. The area shown is the lower Avon Ōtākaro catchment and northern portion of the main tidal lagoon basin of the Avon Heathcote Estuary Ihutai which experienced ground level subsidence during the Canterbury Earthquake Sequence.

1.4 Discussion

These findings describe the landscape-scale reconfiguration of an extensive coastal landscape. Quantification of changes in the overall intertidal area showed that the losses and gains had almost balanced out at the most recent assessment date. Therefore, the overall impact of the CES has primarily been movement of the lagoon system in a north-easterly direction as the consequence of a tilting effect along a broadly north-south axis. Here we discuss the key patterns of change with emphasis on the contribution of anthropogenic influences and site specific effects. We then focus on the areas experiencing subsidence to identify transferable learning for other situations of relative sea-level rise.

1.4.1 Patterns of change

Impacts identified in 2015 included seaward movement of HAT and MHWS shorelines and an overall 4.6% reduction in estuary space. Differential movement of HAT and MHWS shorelines resulted in compression of the supratidal zone (a 14% reduction). Area losses were highly variable between sites but often driven by the position of shoreline armouring in relation to the post-disturbance intertidal range. Specific attention to the availability of space within critical elevations bands is therefore a key principle for the design of natural solutions to flood defence and mitigation. For example, our results indicate negative impacts on the availability of high tide roosting habitat for shorebirds, an already well-established conservation concern in New Zealand (Woodley 2012), and elsewhere (Green et al. 2015; Zharikov & Milton 2009). Another important site-specific effect was the role of estuarine expansion driven by subsidence in the Avon Ōtākaro catchment for offsetting estuarine contraction driven by uplift elsewhere.

These effects present a compelling case for assisting the migration of important ecosystems to areas where they would be expected to move if unhindered by anthropogenic barriers, and the same principles are applicable to CCA (Hällfors et al. 2014). Conservation and natural resource objectives will be more readily achieved if these movements can be facilitated in upcoming decisions on land-use change, illustrating a key role for disaster recovery planning in this process. The government acquisition of riparian and floodplain land greatly facilitates such possibilities, and includes the potential for rewilding in formerly urbanised areas. In this case, the decision context demonstrates a clear linkage between the implementation of disaster recovery and the potential for making progress towards CCA due to similarities in the threat profile with regards to sea-level rise. Although the time-varying effects of eustatic sea-level rise indicate that a range of non-climatic factors will ameliorate the associated patterns of sea-level change (Nicholls et al. 2008), the earthquake-induced analogy provides an empirical scenario that exemplifies plausible outcomes. To build resilience against the negative aspects of such outcomes, key decisions involve spatial planning for anthropogenic infrastructure and attention to the temporal progression of changes and associated vulnerabilities.

1.4.2 Quantification of ongoing change

There was a substantial amount of ongoing change in the estuary and environs over the post-quake study period. Intertidal area changes were not always proportional to elevation changes with major fluctuations observed due to relatively flat topography (e.g., in the Avon Ōtākaro catchment) in the critical elevation band. These aspects suggest the need for further and relatively fine-scale monitoring to quantify ongoing spatiotemporal change, as needed to assess vulnerability to hydrological alterations and the role of future accretion as a potential modulator of sea-level rise (Gedan et al. 2011). A related theme, highlighted by the CES, concerns the role of tectonic displacement as a landscape shaping force. In seismically active regions, the movement of land masses can strongly and unpredictably influence sea levels and associated patterns of inundation and erosion.

Limitations of the present study include the lack of matched bathymetric and topographic datasets covering the full intertidal range. We recommend greater emphasis on the capture of seamless DEMs to help quantify the dynamics of inundation and accretion processes and support the implementation of adaptive management. Incremental changes can be just as important as extreme events and both involve interactions between many different socio-ecological dimensions at various rates and times (Gunderson & Holling 2001). As shown here, the natural topography, pre-existing land-use pattern, and human responses to disturbance events are all vitally important influences on the actual trajectory of change.

1.4.3 Responses to sea-level rise

At locations experiencing subsidence, seawalls constrained shoreline movement exemplifying the problem of coastal squeeze under conditions of sea-level rise and illustrating its actual impacts on natural environments. The affected areas also provide opportunities to identify resilience-building principles by considering the space now available in the intertidal elevation range. At Bexley, the infilling of land for a housing development limits the opportunities for habitat migration despite being within the area of government acquired land (Fig. 1.1). Unless major earthworks are undertaken, strategies for conserving intertidal areas must be implemented elsewhere. On the opposite shoreline, the landward migration of natural ecosystems could be assisted using relatively simple breaches of existing shoreline defences based on our modelled results. However, residential properties remain present in these areas since they were not included in the government land acquisition initiative. These examples illustrate how past and recent land-use decisions have each contributed to resiliency. Their consequences become more obvious once conditions change and risks become manifested as losses, yet it is important that they are identified proactively in advance of tipping points being reached if system resiliency is to be maintained (Folke 2006; Gunderson et al. 2010). Key principles identifiable from this case include the legacy effect of land-filling activities which dramatically alter the ‘rewildability’ of the underlying landscape as conditions change, and the need to consider both built and natural environments in the design of adaptation initiatives such as managed retreat.

1.4.4 Implications for climate change adaptation

There are widely transferable principles of importance in this study and close analogies with the seminal work of Turner (1978) on the man-made aspects of natural disasters. In this paradigm, risk reduction decisions are highlighted as key influences on outcomes. These are challenged by the need for agreement on the future scenarios for which effective responses are required. As applied to natural environments, decisions are required to prevent the reaching of tipping points that result in the loss of natural features and resources. To achieve this, we highlight the influence of anthropogenic factors on the impacts of rapid sea-level changes, and the importance of disaster recovery processes as a context for adaptation to climate change.

1.5 Acknowledgements

We thank the Ngāi Tahu Research Centre, Brian Mason Scientific and Technical Trust, and National Institute for Water and Atmospheric Research (NIWA) for funding support. We thank Jochen Bind for assisting the development of bathymetric datasets, and the Ministry of Business, Innovation and Employment (MBIE) and Environment Canterbury for funding the original hydrodynamic model.

1.6 References

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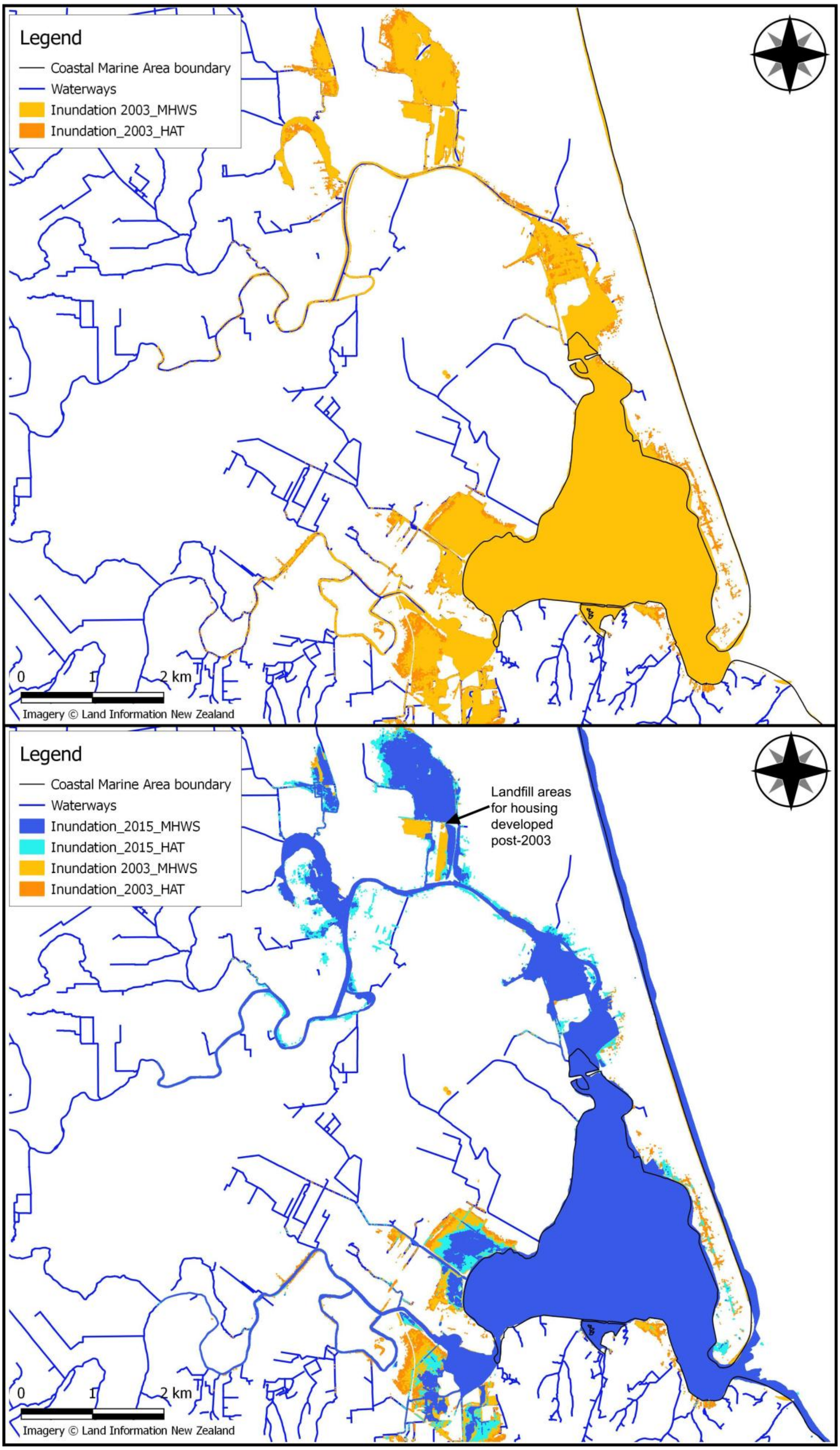
Supplementary Material

Table S1.1 Bathymetry data sources.

Timing in earthquake sequence	Survey date	Survey extent	Organisation
Pre-earthquake	1990's and earlier	Cross-section survey of the lower Heathcote River	Christchurch City Council
	November 1998 & April 1999	Bed level survey of the estuary and nearshore region between North Brighton and Sumner.	Eliot Sinclair
	2008	Cross-section survey of the lower Avon River	Christchurch City Council
Post-earthquake	March/April 2011	Echosounder survey of the estuary including the river mouths	Patterson Pitts Partners
	Late 2011	Cross-section survey of the lower Heathcote River	Christchurch City Council
	2012	Cross-section survey of the lower Avon River	Christchurch City Council
	June 2012	Topographic survey of the McCormacks Bay causeway (Redcliffs) and adjacent estuary	Stronger Christchurch Infrastructure Rebuild Team
	January 2013	Echosounder survey of the southern part of the estuary including the mouth.	National Institute of Water and Atmospheric Research

Fig. S1.1. Changes in the areal extent of land below the elevation of Mean High Water Springs (MHWS) and Highest Astronomical Tide (HAT) in the Avon Heathcote Estuary / Ihutai between 2003 and 2015.



Chapter 2

Risk factors for coastal habitat and blue carbon loss revealed by earthquake-induced sea-level rise

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Abstract

Vegetated coastal ecosystems (VCEs) are in global decline and sensitive to climate change; yet may also assist its mitigation through high rates of carbon sequestration and storage. Their persistence is a function of the tolerance limits of characteristic vegetation and the nature of environmental change. Alterations of relative sea-level (RSL) are pervasive drivers of ecohydrological dynamics that reflect the interaction between tidal inundation regimes and ground surface elevation. Although many studies have investigated sediment accretion within VCEs, relatively few have addressed spatiotemporal patterns of resilience in response to RSL change despite their relevance to the conservation of habitat and stored carbon. In this study, we used high resolution elevation models to quantify RSL changes in a New Zealand tidal lagoon system following tectonic displacement caused by powerful earthquakes. Concurrently, we quantified socio-ecological aspects of the disaster response. VCE losses were recorded in all areas in response to high rates of RSL rise (up to 41 mm yr $^{-1}$) over an 8 year period post-disturbance. Interactions with anthropogenic factors influenced observed losses and illustrated transferable principles for the management of other VCEs facing RSL rise. Four key principles emerged: i) anthropogenic encroachment results in resilience loss due to the need for landward migration when changes exceed the tolerance thresholds of VCEs at their lower elevational limits; ii) connectivity losses exacerbate encroachment effects, and conversely, are a practical focus for improving resilience; iii) risk exposure is disproportionately influenced by the largest wetland remnants illustrating the importance of site-specific vulnerabilities and their assessment; iv) the need to establish new protected areas to accommodate the movement of

ecosystems is an essential aspect of resilience and requires a combination of land tenure rearrangements and connectivity conservation measures. Embracing these concepts offers promise for new resilience-based solutions to halt riparian degradation, improve disaster recovery outcomes and respond to climate change.

Keywords
Ecohydrology, natural disasters, resilience, protected areas, connectivity conservation, socio-ecological system.

2.1 Introduction

Specialist coastal vegetation types include macroalgal forests, seagrass meadows, saltmarshes and mangroves. These vegetated coastal ecosystems (VCEs) typically occupy a narrow coastal fringe and yet are highly productive areas playing important roles as buffers between the land and sea. Benefits for humanity include filtration, waste assimilation, coastal erosion and flood protection, habitat for wildlife, and as nursery areas for fisheries (Gedan et al. 2011; McGlathery et al. 2007; Nicholls 2004). From an ecosystem services perspective VCEs have been assessed as among the most valuable of ecosystems worldwide (Barbier et al. 2011; Costanza et al. 1998). They are vulnerable to human impacts resulting in loss and degradation (Brisson et al. 2014; Coverdale et al. 2014; Duke et al. 2007), from stressors including land-use change, coastal reclamation, and nutrient pollution (Adam 2002; Orth et al. 2006; Pendleton et al. 2012). Global losses in the past 50 years range from 25-50% for key ecosystem types (Duarte et al. 2013), indicating an urgent need for conservation (Lotze et al. 2006).

Climate change presents additional challenges from effects on water depths, salinity, and range shifts along climatic gradients (Chmura 2013; Crosby et al. 2016; Krause-Jensen & Duarte 2014). Although VCEs are sensitive to these changes, they also play an important role in global efforts to reduce atmospheric CO_2 concentrations (Chmura et al. 2003; McLeod et al. 2011). In particular, the sequestration of 'blue carbon' by VCEs has generated renewed interest in their restoration and management (Chmura 2013; Howard et al. 2017). Important attributes include high primary production rates combined with the physical trapping and binding of particulate material from both autochthonous and allochthonous sources (Saintilan et al. 2013; Woodroffe et al. 2016), and the influence of saline conditions on microbial activity that promotes organic matter preservation within near-surface sediments (Fourqurean et al. 2012; Koho et al. 2013).

These properties result in carbon accumulation rates that are typically much higher than in terrestrial forests (Donato et al. 2011; McLeod et al. 2011; Pendleton et al. 2012).

In addition to their future roles in carbon sequestration, VCEs that have been stable over recent geological time are conservation priorities due to their accumulated carbon value (Irving et al. 2011; Iacono et al. 2008; Mateo t al. 1997). For comparison, global accumulation rates for below-ground carbon in terrestrial forests are in the order of 4 - 5 g m^{-2} $year^{-1}$, whereas rates of between 138 g m^{-2} $year^{-1}$ (seagrass) and 244 g m^{-2} $year^{-1}$ (saltmarsh) have been reported for VCEs (Ouyang & Lee 2014). There is a considerable risk of new emissions from these long-term carbon sinks if these habitats are degraded (Pendleton et al. 2012). Even small changes in the relative rates of important biogeophysical processes have the potential to convert long term sinks to new emission sources (Coverdale et al. 2014; Macreadie et al. 2013).

There is a high level of uncertainty regarding the fate of VCEs under climate change (Osland et al. 2016; Schuerch et al. 2018). While some studies report a bleak outlook (Blankespoor et al. 2014; Craft et al. 2009; Crosby et al. 2016), others point to feedbacks that may assist VCEs to self-maintain under changing conditions (Anisfeld et al. 2017; Kirwan & Megonigal 2013; Kirwan et al. 2016). It is becoming increasingly important to understand the mechanisms that promote VCE resilience, and conversely, the risks to their survival (Macreadie et al. 2019; Schuerch et al. 2018). Rising sea levels create the potential for VCE expansion and further carbon accumulation through landward migration and upward vertical accretion (Chmura et al. 2003; Kirwan & Mudd 2012; Kirwan et al. 2016). The self-engineering aspects of *in situ* accretion have undoubtedly contributed to past resilience in periods of RSL rise (McKee et al. 2007; McLeod et al. 2011). At the same time, however, losses may be expected where vegetation communities are overwhelmed by changes and are unable to survive or migrate inland (Duarte et al. 2013; Morris et al. 2002). This has led many authors to link climate change with threats to VCE persistence and the potential loss of accumulated carbon (Chmura 2013; Craft et al. 2009; Crosby et al. 2016).

There is a particular need to improve the understanding of how different VCEs respond to RSL changes and the range of potential responses (McKee & Vervaeke 2018). However, the lack of empirical data to demonstrate such relationships is a significant impediment to answering these questions (Voss et al. 2013). This is compounded by high levels of spatiotemporal variability that reduces the ability to generalise from historical patterns and small sample sizes (Cahoon

2015). Although historical changes studies such as those based on sediment sampling are useful indicators of vertical accretion rates (Chmura & Hung 2004; Kelleway et al. 2016; Woodroffe et al. 2016), they have seldom generated concurrent data on the horizontal dynamics of the underlying VCEs due to the sampling effort required. In addressing the prospects for VCE conservation it is clear that both horizontal and vertical dynamics are important (DeLaune & White 2012; Kirwan & Megonigal 2013; Schuerch et al. 2018). These aspects interact with the high level of anthropogenic modification typical of coastal zones worldwide creating new interactions and complexities (Anisfeld et al. 2017; Doody 2004; Phan et al. 2015; Spencer et al. 2016).

In investigating the potential impacts of accelerating eustatic sea-level rise (Church et al. 2013) the study of contemporary subsidence events can be particularly informative (Cahoon 2015). These include shallow subsidence resulting from surface elevation loss caused by autocompaction, organic matter decomposition, groundwater or hydrocarbon extraction (Cahoon et al. 1995; Rybczyk & Cahoon 2002), and deep subsidence referring to tectonic displacements and isostatic adjustments of land masses in relation to sea level (Woodroffe et al. 2016). Examples of appreciable RSL rise that have been the subject of empirical studies include deep subsidence caused by a mine collapse (Rogers et al. 2019) and earthquake-induced subsidence in the Solomon Islands (Albert et al. 2017). Some of the most insightful studies on mechanisms of wetland loss come from the southern USA where dramatic examples of shallow subsidence have occurred in deltaic wetlands (Day et al. 2000; Morton et al. 2010). These include several studies that have investigated spatiotemporal variability in relation to RSL effects through concurrent measurements of surface elevation changes and VCE responses (Cahoon 2015; Rybczyk & Cahoon 2002).

In this study, we assessed the impacts of powerful earthquakes in the Canterbury region of New Zealand that caused ground level displacements exceeding 0.5 m in the coastal aquatic environment (Quigley et al. 2016). We took advantage of a series of high resolution elevation datasets that were captured during the earthquake recovery process that provided an opportunity to assess RSL changes, VCE responses, and the role of anthropogenic influences. The objectives for this paper are to: a) provide an overview of the impacts on intertidal and supratidal VCEs in the Avon Heathcote Estuary Ihutai, b) identify relationships between vegetation change and RSL changes associated with ground level displacement, and c) identify anthropogenic risk factors for future VCE management with emphasis on the impacts of RSL rise in areas of subsidence.

2.2 Methods

2.2.1 Study site and survey design

The city of Christchurch on the east coast of the South Island was severely impacted by a sequence of large earthquakes in 2010-11 (Fig. 2.1). Four earthquakes exceeding M_W 6.0 were responsible for massive infrastructure damage (estimated at NZ$40 billion) and the loss of 185 lives (Beavan et al. 2012; Bradley et al. 2014; Potter et al. 2015). Complex decisions on property damage, infrastructure repair, and future hazard risk have required a prolonged period of disaster recovery. Lasting changes in the natural environment have interacted with the recovery process and created additional considerations. Widespread surface deformation changed the post-disaster landscape affecting many aspects of the socio-ecological system. Vertical displacements occurred in both directions resulting in new ground levels and hydrology. Smaller-scale effects included lateral spread, liquefaction, bank collapses and landslides (Quigley et al. 2016; Robinson et al. 2012; Zeldis et al. 2011). Ground-level subsidence on lowland floodplains increased flood risk hazard in many residential areas (Hughes et al. 2015). Societal responses have featured policy innovations, including the government acquisition of thousands of residential properties on low-lying land in the vicinity of the estuary and waterways (Fig. 2.2). As the recovery process has matured, attention is turning to the long-term uses of this land alongside strategic planning for climate change (Orchard 2017a; Regenerate Christchurch 2017).

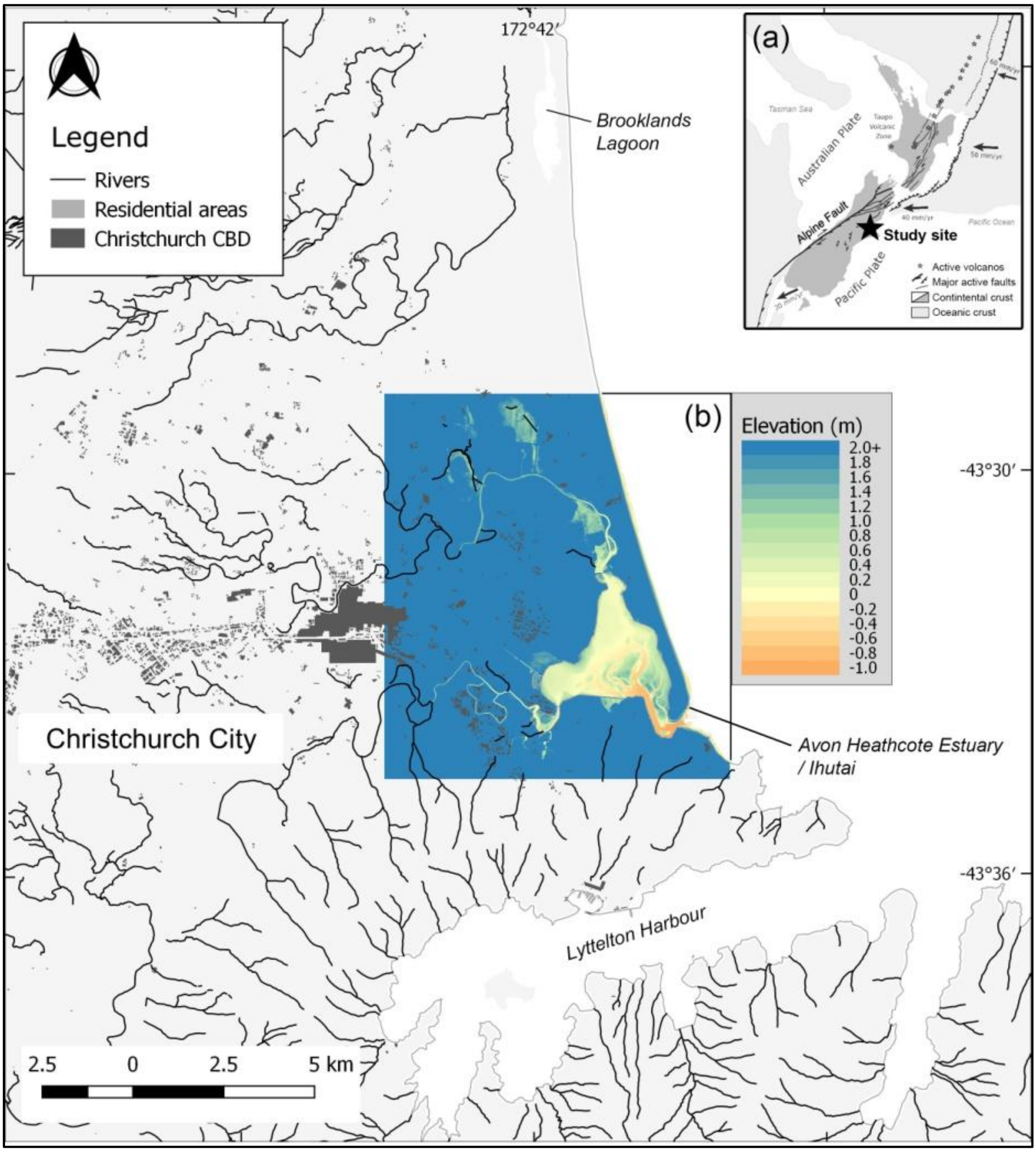

Fig. 2.1 Study area overview. (a) Location on the east coast of New Zealand's South Island, showing configuration of the main Alpine Fault-line and Australian – Pacific plate boundary. Source data: Waikato Regional Council. (b) Configuration of the Avon Heathcote Estuary Ihutai in the city of Christchurch. Inset shows ground elevations in the tidal lagoon and lower river floodplains as measured in October 2015. Datum is LVD37. Source data: Christchurch City Council.

Despite being partially urbanised, the underlying natural environment is characterised by an extensive tidal lagoon and network of lowland waterways. The Avon Heathcote Estuary Ihutai is a tidal basin of ca. 8 km^2 characterised by fine sediments and connected to the Pacific Ocean via a single entrance channel in the southern corner of a shallow embayment (Hicks 1998; Kirk 1979). Tides are semi-diurnal with a spring range of 2.22 m using average predicted values over the 18.6 year tidal cycle at nearby Port Lyttelton (Land Information New Zealand 2019). The background rate of sea-level rise is in the order of 1.9 mm yr^{-1} over the past century (Hannah & Bell 2012).

Coastal wetland vegetation types of the estuary were described by Jones & Marsden (2007) within three tidal zones as are typical of other temperate marsh systems (Engels et al. 2011; Odum 1988). Characteristic species include sea rush (*Juncus kraussii*) and oioi (*Apodasmia similis*) in the low marsh; saltmarsh herbfield species such as glasswort (*Sarcocornia quinqueflora*), buck's horn plantain (*Plantago coronopus*), remuremu (*Selliera radicans*) and suaeda (*Suaeda novae-zelandiae*) in the mid marsh; and saltmarsh ribbonwood (*Plagianthus divaricatus*) shrubland and introduced grasses in the high marsh (Jones & Marsden 2005). Previous studies have reported relatively stable vegetation patterns in the South Brighton area that is characterised by extensive rushland (Jupp et al. 2007). In contrast, several saltmarsh sites previously recorded by McCombs & Partridge (1992) had disappeared in the Ferrymead area due to estuary in-filling effects attributed to fluvial sediment sources (Jupp et al. 2007).

Coastal wetland vegetation was sampled using shore-perpendicular transects within each of three focus areas (South Brighton, Southshore, and Ferrymead) that account for the majority of all wetland remnants in the estuary (Fig. 2.2). The sampling transects (n = 30) comprised of two transects at the five largest wetlands in each of the three focus areas (Fig. 2.2). Transects were cast perpendicular to a smoothed baseline at the approximate position of Mean High Water Springs (MHWS) using the *ambur* package ("Analyzing Moving Boundaries Using R") following Jackson et al. (2012) and ground-truthed using real-time kinematic (RTK) GPS in the field.

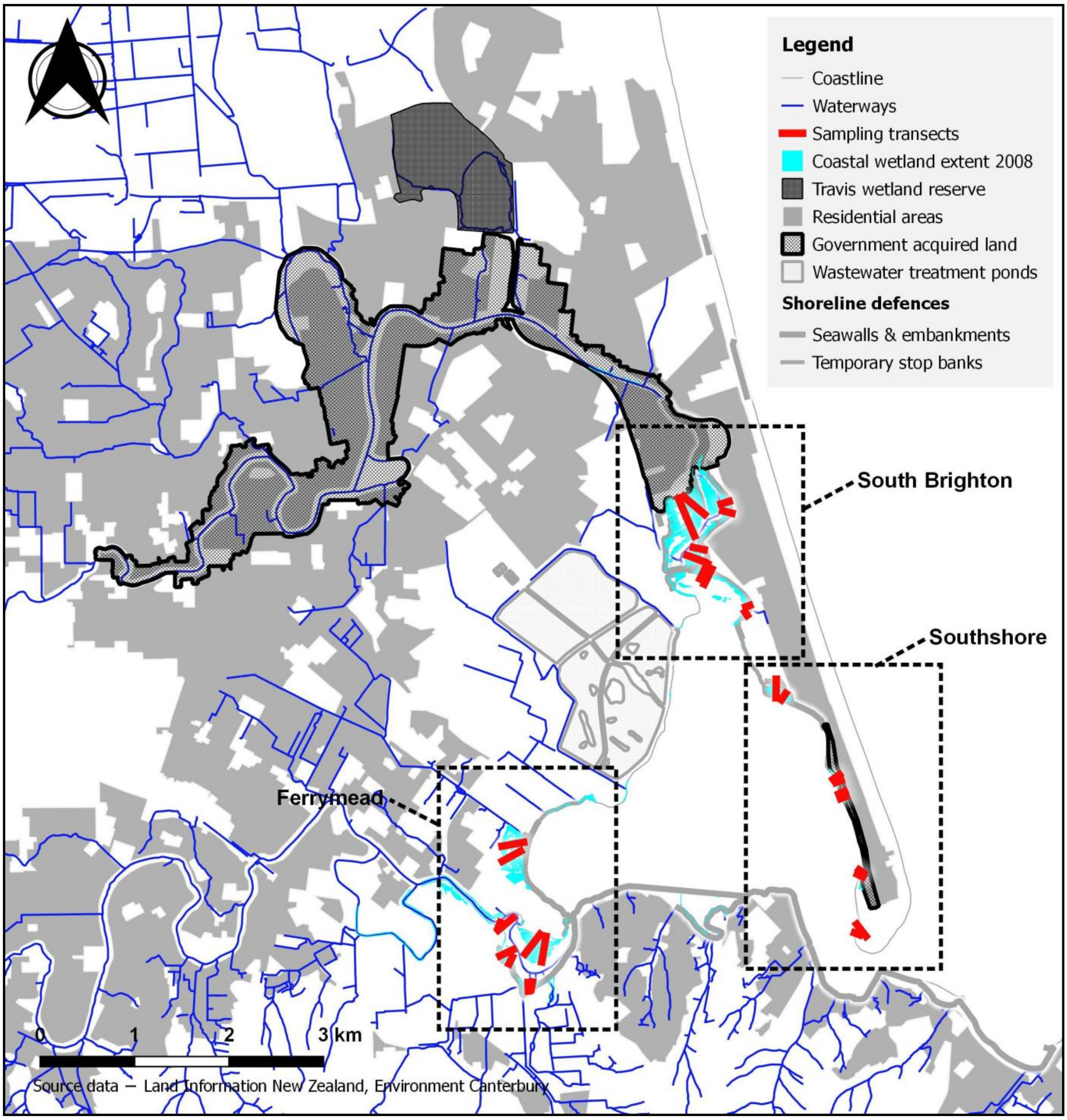

Fig. 2.2 Location of the three focus areas (South Brighton, Southshore and Ferrymead), shore-perpendicular sampling transects (n = 30) at 15 individual wetlands, and coastal wetland vegetation as recorded in 2008 (Grove et al. 2012).

2.2.2 Ground level change

Ground level changes were assessed using point sampling of digital elevation models (DEM) prepared from light detection and ranging (LiDAR) datasets within areas of interest. Additionally, we assessed recent changes using RTK-GPS and laser surveys in early 2019 to enable the consideration of further changes since the most recent airborne LiDAR survey. Ground levels were calculated at 1 m spacing on the transect lines producing a dataset of 4240 points that were re-sampled over the time. LiDAR data sets selected for this study were acquired in October 2015 (most recent survey), May 2011 (soon after the major February 2011 earthquake), and July 2003 (pre-quake baseline) and provide complete coverage of the study (Table 2.1). The data set included bare earth DEM at 5 x 5 m resolution for the 2003 and 2011 surveys and 1 x 1 m for 2015 (Canterbury Geotechnical Database 2014; LINZ 2017). Identical DEM configurations were developed by reprocessing the 2015 DEM to 5 m resolution. Table 2.1 shows the horizontal and vertical accuracy of the DEMs excluding potential GPS network errors (ca. ±0.06 m), and New Zealand Quasigeoid 2009 approximations. Profiles produced from the RTK-GPS surveys have a vertical accuracy of ±0.12 m or better. Positional errors on the transect lines are in the range ±0.05 – 0.10 m range and depend on the gradient of the site. Additional errors up to ±0.02 m are associated with ground surface capture (e.g., in soft sediments where the measuring staff may sink).

Table 2.1 LiDAR data sources and major tectonic events in the Canterbury Earthquake Sequence (CES).

LiDAR acquisition dates	Timing in relation to CES	Supplier	Commissioning agencies	Accuracy (m) horizontal	Accuracy (m) vertical
6-9 Jul 2003	Pre- CES baseline	AAM Brisbane	Christchurch City Council	±0.55	±0.15
20-30 May 2011	after Feb 2011 earthquake	AAM Brisbane	Christchurch City Council	±0.55	±0.07
5 Oct-7 Nov 2015	Post- CES	AAM Brisbane	Canterbury Regional Council	±1.00	±0.20

Major tectonic events

Location	Date	Magnitude[†] (M_w)	Max slip depth[‡] (km)
Darfield	4 September 2011	7.1	2-6
Christchurch	22 February 2011	6.4	4-6
Christchurch	13 June 2011	6.2	>1
Christchurch	23 December 2011	6.1	2-5

[†] Beavan et al. (2012). December 2011 magnitude represents combined moments of the two largest tremors.
[‡] Quigley et al. (2016).

2.2.3 Vegetation data and zonation model

Vegetation data were available from coastal wetland surveys by Environment Canterbury in 2008 and 2015 (Grove et al. 2012), and a further survey was completed in early 2019. The spatial extent of the 2008 and 2015 data sets included all coastal wetland vegetation in the wider estuary catchment. The scope of the 2019 survey was limited to the location of the sampling transects. The method used for all surveys was an adaptation of the Atkinson (1985) vegetation mapping system in which the delineation of mapped units is based on differences in the dominant vegetation observed. Boundaries between units are assigned in the field based on direct observations assisted with aerial imagery, and percentage cover of the dominant species is estimated for each unit mapped. Each unit was subsequently digitized in a GIS and the vegetation categorised using the hierarchical classification of Grove et al. (2012) for Canterbury coastal wetlands. We used the finest scale of the classification (‘vegetation type’) for which 50 classes are found in the wider study area, 28 of which were recorded on the transect lines. We also calculated changes for five ‘structural’ classes (rushland, reedland, herbfield, grassland, shrubland, and sparsely-vegetated) reflecting dominant habitat types as defined within the same classification (Grove et al. 2012). Additionally, we modelled the broader-scale intertidal zonation pattern using a combination of literature review and zonal statistics analysis. We sampled the 2003 DEM with the 2008 vegetation type polygons to test candidate indicators for major zonation boundaries as identified in the literature and field observations. These data provide the best available representation of pre-earthquake zonation with complete coverage of the study area.

2.2.4 Data analyses

Intersection analyses were used to sample the geospatial position of mapped vegetation polygons on the transects and differencing used to calculate ground level changes at each survey date. Measurement errors include horizontal discrepancies between the DEMs or sampled points and inaccuracies in the estimation or digitisation of the vegetation unit boundaries during surveys. The latter was reduced by applying a negative buffer to mapped vegetation polygons to reduce edge effects, with the buffer set at 1 m to account for the horizontal accuracy of the 2015 LiDAR (Table 2.1).

Two-way ANOVAs were used to evaluate within group variance for location (focus areas) and time. Data were transformed where necessary to meet variance homogeneity and normality assumptions, and post-hoc Tukey’s (HSD) tests were used to identify significant effects. In the case of heterogeneity of variances, Kruskal–Wallis rank sum tests were used to compare means followed by Wilcoxon post-hoc comparisons for significant treatment effects. All analyses were conducted in NZTM 2000: ESPG 2193 projection using QGIS v 2.18 (QGIS Development Team 2017). Statistical analyses were conducted in R v3.3.3 (R Core Team 2017). Orthographic heights are in Lyttelton Vertical Datum (LVD) that is defined by a geodetic benchmarking system. MHWS at the Port Lyttelton tide gauge is currently 1.479 m LVD (Land Information New Zealand 2019).

2.3 Results

2.3.1 Ground level change

Considerable ground level changes were detected throughout the study period within the pre-quake intertidal vegetation footprint (Fig. 2.3). Two distinct phases can be identified: vertical displacements in both directions resulting from the February 2011 earthquake, and a subsidence trend since. South Brighton experienced the greatest overall change due to progressive subsidence, with a mean change of 0.56 m over the study period. Ferrymead and Southshore experienced initial uplift (0.18 m and 0.12 m, respectively) associated with the February 2011 earthquake. For both, these uplift effects were reduced to only 0.01 m by 2015 and net subsidence by 2019. Over the 8-year period (2011 - 2019) the subsidence trend has resulted in mean surface elevation losses of 41 mm yr^{-1} at Ferrymead, 39 mm yr^{-1} at South Brighton, and 32 mm yr^{-1} at Southshore (Fig. 2.3). These ground level changes were significantly different between sampling periods and locations and there was a significant interaction between the two (ANOVA, $p < 0.001$). Post-hoc comparisons showed that ground level changes at South Brighton differed significantly from both other areas ($p < 0.001$), but there was no significant difference between Ferrymead and Southshore overall ($p = 0.61$).

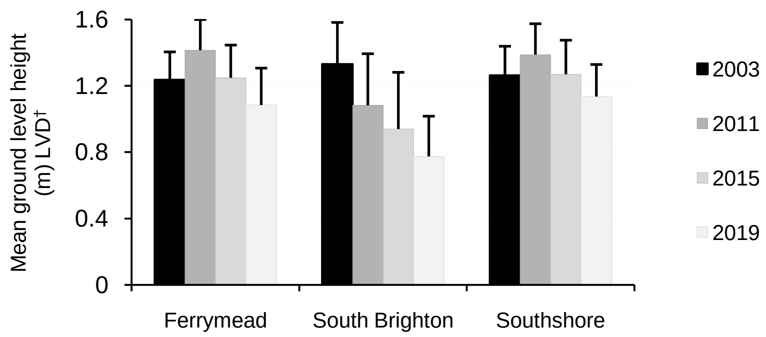

Fig. 2.3 Changes in ground levels within the pre-earthquake intertidal vegetation footprint for three localities within the Avon Heathcote Estuary / Ihutai over the period 2003 – 2019. The Canterbury Earthquake Sequence (CES) included two major earthquakes in September 2010 and February 2011 that preceded the 2011 ground level measurement date.

These changes within VCEs are consistent with the general pattern of ground displacements in the eastern Christchurch area revealed by LiDAR surveys to 2015 (Fig. 2.4). The key trends are a tilting effect of the February 2011 earthquake with subsidence of the lower Avon River catchment and South Brighton area, uplift in the Heathcote catchment and Ferrymead area, and a mixture of effects at Southshore (Fig. 2.4a). As the CES progressed, further subsidence was evident throughout the wider area (Fig. 2.4b). These measurements were, however, partly influenced by horizontal ground movement in an easterly direction, as evidenced by the pattern of movements in valley systems towards the south of the study area between the 2011 and 2015 LiDAR acquisition date (Fig. 2.4b). These horizontal movements may have led to overestimates of subsidence on the eastern shoreline and the reverse effect on the west, especially in areas with steeper shore profiles. Despite this, results from the 2019 GPS survey indicate that the coastal wetlands have continued to lose surface elevation over the past four years (Fig. 2.3). Further investigations are needed to determine the cause of these trends given that they do not appear to be related to further tectonic movement.

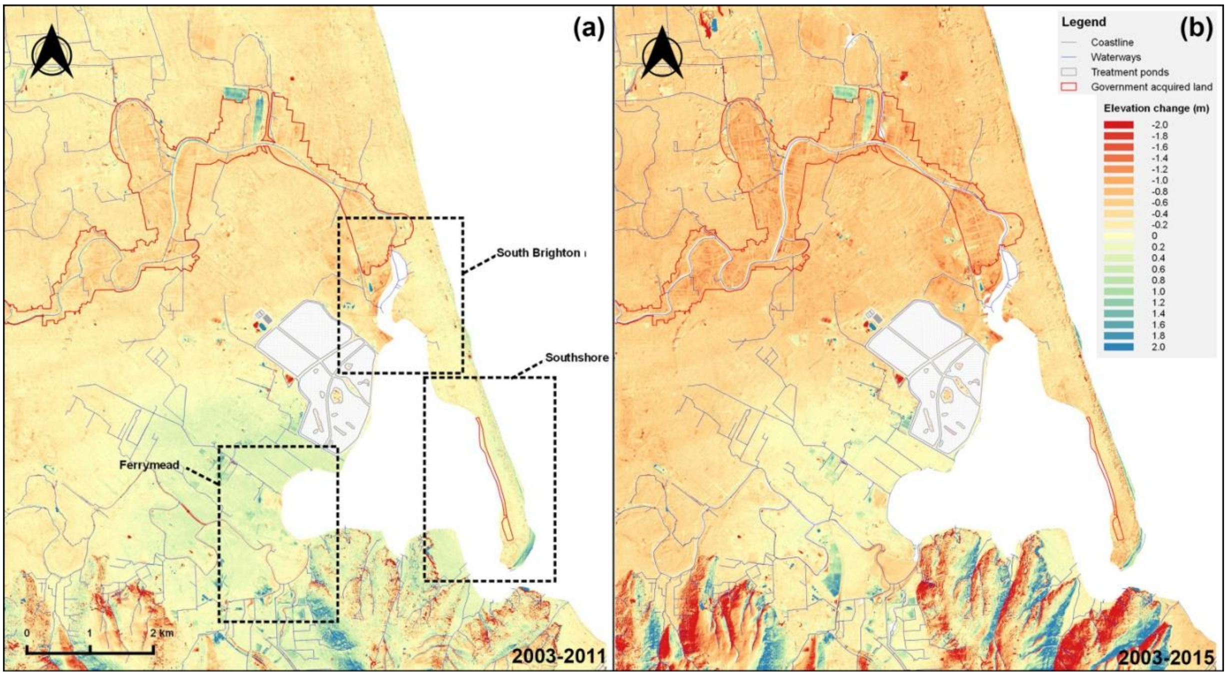

Fig. 2.4 Ground level differences relative to 2003 ground elevations measured by airborne light detecting and ranging radar (LiDAR) in the vicinity of the Avon Heathcote Estuary Ihutai in eastern Christchurch. (a) 2003-2011 differences as measured after the February 22nd 2011 earthquake that caused widespread damage to infrastructure and land. (b) 2003 – 2015 differences showing cumulative effects of all major earthquakes in the CES. Note the effects of horizontal movement in an easterly direction evident on hillslopes to the south.

2.3.2 Vegetation change

Across all transects, the percentage change in shore-perpendicular vegetation extent increased by an average of 151% for the period 2008 – 2015, and there was little further change from 2015 to 2019 (Table 2.2). Within each area, there were substantial differences from this overall trend. Ferrymead study sites experienced a 17% reduction in the 2015 – 2019 period, which differed significantly ($p < 0.05$) from the relatively small initial response. In Southshore, similar reductions (24%) were experienced in the 2015 - 2019 period that had the effect of reducing initial gains. However, this trend was not statistically significant due to highly variable effects at the different wetlands within this area, particularly in the 2008 – 2015 comparison. Only small changes were detected for both post-quake periods at South Brighton (Table 2.2).

Changes in the total extent of wetlands showed a much greater reduction than suggested by the percentage changes on individual transects, reflecting the different sizes of the wetlands involved. Total pre-quake vegetation extent (2633 m) was relatively unchanged by 2015 (2558 m), but then reduced to 2233 m by 2019. This 15% overall loss compares to the mean percentage change on individual transects of only 1.1% (Table 2.2). At Ferrymead, the vegetation reduced from 1174 m to 917 m, with most of this 22% loss occurring between 2015 and 2019. South Brighton also experienced a reduction, from 1069 m (pre-quake) to 873 m by 2015, and 815 m by 2019, a 24% loss overall. At Southshore the pre-quake extent (390 m) initially increased to 532 m in 2015, and then decreased to 501 m by 2019, resulting in a 111 m (28%) gain overall.

Table 2.2 Changes in the shore-perpendicular extent of coastal wetland vegetation at three locations within the Avon Heathcote Estuary Ihutai in relation to the Canterbury Earthquake Sequence beginning 2010. Measurements taken in 2008 are representative of pre-earthquake conditions.

Percentage change† on individual transect lines (mean ± SE)					
	2008 - 2015	**2015 - 2019**	**df**	**Kruskal-Wallis χ^2**	**p-value‡**
Ferrymead	103.7 ± 6.1	83.0 ± 5.0	**2**	**5.8514**	**0.0156**
South Brighton	100.1 ± 11.7	99.1 ± 16.9	2	0.0915	0.7623
Southshore	155.3 ± 45.8	118.4 ± 18.8	2	0.0914	0.7624
TOTAL	151.5 ± 35.1	98.9 ± 7.5	2	3.0698	0.0798
Total change (m)					
	2008 - 2015	**2015 - 2019**	**overall**	**overall (%)**	
Ferrymead	-21	-236	-257	-21.9	
South Brighton	-196	-58	-254	-23.8	
Southshore	142	-31	111	+28.5	
TOTAL	-75	-325	-400	-15.2	

† expressed as percentage of 2008 figures

‡ results shown in bold are significant at $\alpha = 0.05$

2.3.3 Zonation patterns and structural classes

Zonation model

The 2008 vegetation data set contains 347 mapped polygons representing the 26 vegetation types with a total area of 55.8 ha and mean pre-quake ground elevation of 1.26 m LVD (±0.26 SD), based on the 2003 DEM (Table 2.3). Previous studies of the Avon Heathcote Estuary Ihutai have reported a mid-marsh vegetation band characterised by herbfield species such as glasswort (*Salicornia quinqueflora*), buck's horn plantain (*Plantago coronopus*), and remuremu (*Selliera radicans*) (Jones & Marsden 2007). The intertidal zonation model uses the presence of two indicator species to define the zone boundaries. The mid zone lower limit is defined by the presence of glasswort (*Salicornia quinqueflora*) herbfield, and the upper limit by the transition to saltmarsh ribbonwood (*Plagianthus divaricatus*) shrubland (Table 2.3).

Table 2.3 Intertidal zonation model for coastal wetland vegetation in the Avon Heathcote Estuary Ihutai.

Vegetation type †	Ground elevation (LVD†) Mean	SD	Zonation model
Native musk (*Thyridia repens*) herbfield	0.87	0.28	Low marsh
Mixed saltmarsh herbfield, glasswort (*Sarcocornia quinqueflora*) absent	1.00	0.18	
Lake clubrush (Schoenoplectus tabernaemontani) reedland	1.02	0.21	
Caldwell'clubrush *(Bolboschoenus caldwellii*) with native saltmarsh species	1.08	0.36	
Grass-herbfield	1.11	0.26	
Sea rush (*Juncus kraussii*) with exotic grasses	1.12	0.06	
Sea rush (*J. kraussii*) essentially alone	1.16	0.44	
Three square (*Schoenoplectus pungens*) reedland	1.17	0.42	
Sea rush (*J. kraussii*) with saltmarsh herbfield, glasswort (*S. quinqueflora*) present	1.22	0.16	Mid marsh
Glasswort (*S. quinqueflora*) and sea lavender (Limonium companyonis) herbfield	1.22	0.01	
Oioi *(Apodasmia similis*) with saltmarsh herbfield	1.24	0.20	
Creeping bent (*Agrostis stolonifera*) grassland	1.26	0.26	
Oioi (*A. similis*) with marsh ribbonwood (*Plagianthus divaricatus*) and sea rush *(J. kraussii*)	1.29	0.31	
Three square (*S. pungens*) with native saltmarsh shrubs and rushes	1.29	0.12	
Caldwell's clubrush (*B. caldwellii*) reedland	1.30	0.48	
Mixed saltmarsh herbfield, glasswort (*S. quinqueflora*) present	1.30	0.58	
Oioi (*A. similis*) with introduced grasses and jointed rush (*Juncus articulatus*)	1.30	0.29	
Sea rush (*J. krausii*), marsh ribbonwood *(P. divaricatus*), oioi (*A. similis*), three square (*S. pungens*)	1.32	0.36	
Oioi (*A. similis*) restiad rushland	1.35	0.49	
Tall fescue (*Schedonorus arundinaceus*) dominant grassland with exotic associates	1.35	0.21	
Marsh ribbonwood *(P. divaricatus*)shrubland with oioi (*A. similis)*	1.40	0.14	High marsh
Marsh ribbonwood *(P. divaricatus*) with sea rush *(J. kraussi*)	1.40	0.34	
Sea primrose (*Samolus repens*) turf	1.41	1.03	
Marsh ribbonwood *(P. divaricatus*) with exotic grass	1.52	0.34	
Tall fescue (*S. arundinaceus*) dominant grassland with native associates	1.52	0.41	
Marsh ribbonwood *(P. divaricatus*) shrubland	1.64	0.37	

† vegetation type descriptions follow naming convention of Grove et al. (2012).

Earthquake effects on zonation

The greatest absolute changes were found in the low zone, which was reduced by 318 m (28%). The high zone experienced a 132 m loss (32%), and the mid zone a 50 m (5%) gain across all sites (Fig. 2.5a). This overall pattern varied considerably between locations. At Ferrymead, losses occurred in the low and mid zones (280 m and 56 m, respectively), with the high zone gaining 79 m by 2019 (Fig. 2.5a). At South Brighton all zones had reductions, with the greatest losses occurring in the low and high zones (125 and 108 m, respectively, for 2019), with the mid zone less affected (21 m). The overall loss was 245 m (24%) in 2019. Southshore showed gains in the low and mid zones (87 m and 127 m, respectively) and losses of 103 m in the high zone. By 2019 the overall gain was 111 m (29%).

Structural classes

Impacts on the five structural vegetation classes were generally consistent with the zonation pattern changes in consideration of the characteristic habitat types. Across all sites, the greatest losses occurred in reedland, herbfield and shrubland structural classes (Fig. 2.5b). At Ferrymead the greatest losses included reedland (272 m, 35%), herbfield (76 m, 26%) and grassland (17 m, 29%), accompanied by a gain in shrubland (36 m, 198%) consistent with gains in the upper zone. At South Brighton the largest losses involved herbfield (149 m, 64%), rushland (135 m, 22%) and shrubland (120 m, 55%). At Southshore, where gains outweighed losses, there were increases in rushland (96 m, 160%) and herbfield (65 m, 180%), accompanied by losses of shrubland in the high zone (63 m, 54%). Sparsely vegetated areas were found to have increased at all locations by 2019. These include both areas of die-off and new colonisation in which vegetation changes are still occurring and a stable state has yet to be reached.

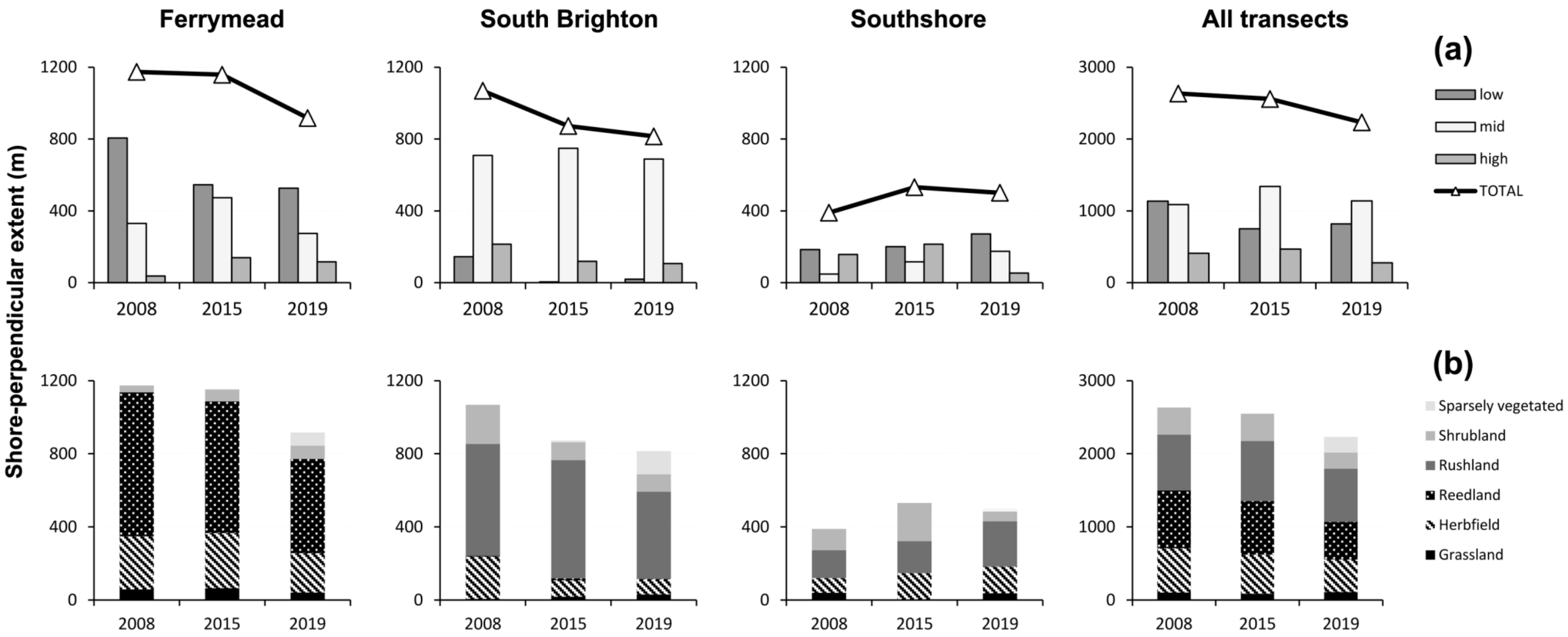

Fig. 2.5 Shore-perpendicular extent of coastal wetland vegetation at three locations in the Avon Heathcote Estuary Ihutai in relation to the Canterbury Earthquake Sequence showing effects on (a) intertidal zonation patterns, and (b) structural vegetation classes. Measurements taken in 2008 are representative of pre-earthquake conditions. Composition of the low, mid and high zonation vegetation communities is detailed in Table 2.3.

2.3.4 Responses of individual wetlands

Responses of the three largest individual wetlands (Bexley, Settlers and Charlesworth) exerted disproportionate effects on the impacts as a whole, and these have been exacerbated since 2015 (Fig. 2.6). All involved vegetation loss, although the temporal sequence of events differed markedly. At Bexley Wetland, where the largest area of vegetation was recorded in 2008, severe reductions occurred by 2015 and there were further reductions by 2019 (Fig. 2.7a). In contrast, Charlesworth and Settlers wetlands in Ferrymead experienced little change by 2015 and then lost vegetation area in the subsequent period to 2019.

Where they occurred, gains were typically recorded only in the smaller wetlands. An exception was the Upper Bridge wetland in South Brighton where subsidence effects of the earthquake initiated landward migration of intertidal vegetation into an area of former pine forest located seaward of a large embankment that provides the primary flood defence (Fig.2.7b). In contrast to Bexley, saltmarsh species were able to colonise the available space after mortality of the pines. Nonetheless, the inland limit of this migration sequence is constrained by the embankment system, and this is reflected by the lack of further change between 2015 and 2019 (Fig 2.6).

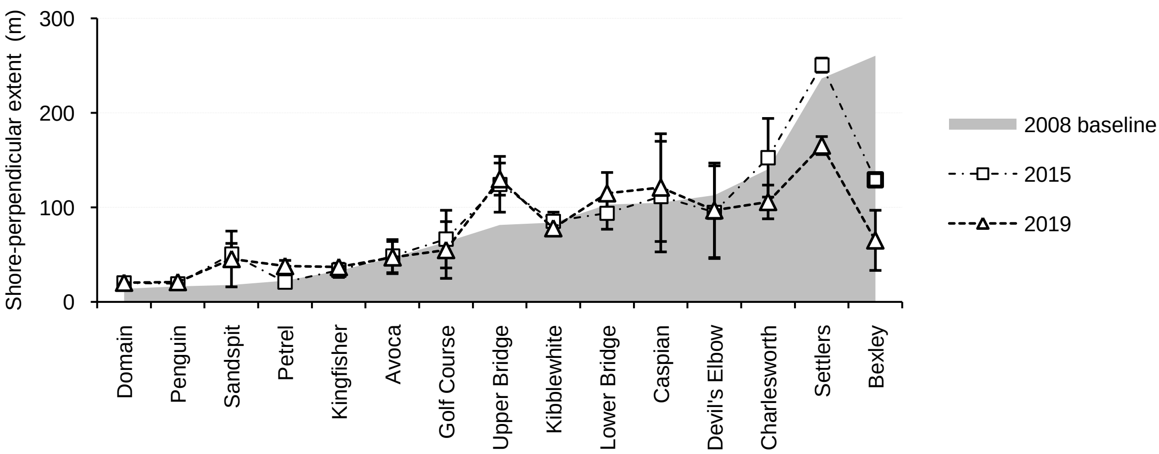

Fig. 2.6 Changes in the shore-perpendicular extent of intertidal vegetation at 15 coastal wetlands in the Avon Heathcote Estuary Ihutai over the period 2008 – 2019. Error bars are standard error of the mean. Locations of each wetland are shown in Fig 2.2.

Fig. 2.7 Site photographs. (a) Bexley Wetland in the South Brighton area where large areas of wetland were eliminated by drowning. Landward migration of riparian wetlands was constrained by land reclamation and seawalls associated with a previous housing development (arrowed). (b) an extensive new area of saltmarsh herbfield now occupies a former area of pine forest following landward migration in response to relative sea-level rise. In this location flood defences (arrowed) were set back from the pre-earthquake high tide shoreline. Saltmarsh vegetation now extends to the foot of the embankment. (c) temporary stop-banks were erected in the Southshore area that initially experienced uplift before beginning to subside. Government acquisition of waterfront properties in this area has now created space for estuarine movement and the managed retreat of these defences could be readily achieved. (d) large expanses of lower intertidal zone rushland perished in areas of subsidence. Below-ground remnants are now eroding releasing accumulated carbon stores. (e) areas of former shrubland have been slow to recover in many areas indicating complex turnover dynamics as conditions change. (f) yellow flowers of *Cotula coronopifolia* indicate large areas of saltmarsh herbfield that have only recently invaded pastoral land on the southwestern shoreline in response to ground surface subsidence and relative sea-level rise. The lack of major barriers to coastal wetland movement in this farmed landscape suggests that it may be a suitable area for accommodating natural environment change. Credits: Shane Orchard (a-e), Andrew Crossland (f).

2.3.5 Vegetation change in relation to uplift and subsidence

Vegetation responses were variable in relation to ground level changes over 2008 – 2015, with no clear patterns in magnitude or direction (Fig.2.8a). For example, widespread subsidence at South Brighton resulted in both losses and gains at individual wetlands, indicating the importance of site-specific effects. By 2015, net uplift was only present at three sites and these also showed variable vegetation responses. At many wetlands, however, the direction of responses measured in 2015 continued to 2019 (Fig. 2.8a).

Responses of the structural vegetation classes were also highly variable with the exception of herbfield which was consistently reduced by RSL rise in both time periods (Fig. 2.8b). The variable effects on other classes as measured in 2015 may reflect different responses of the sites to RSL changes but also the potential for lag effects in response to the ground displacements. By 2019 the structural class responses had developed a more consistent relationship with ground level change which highlighted an association between subsidence and negative impacts on herbfield, reedland and shrubland (Fig. 2.8b).

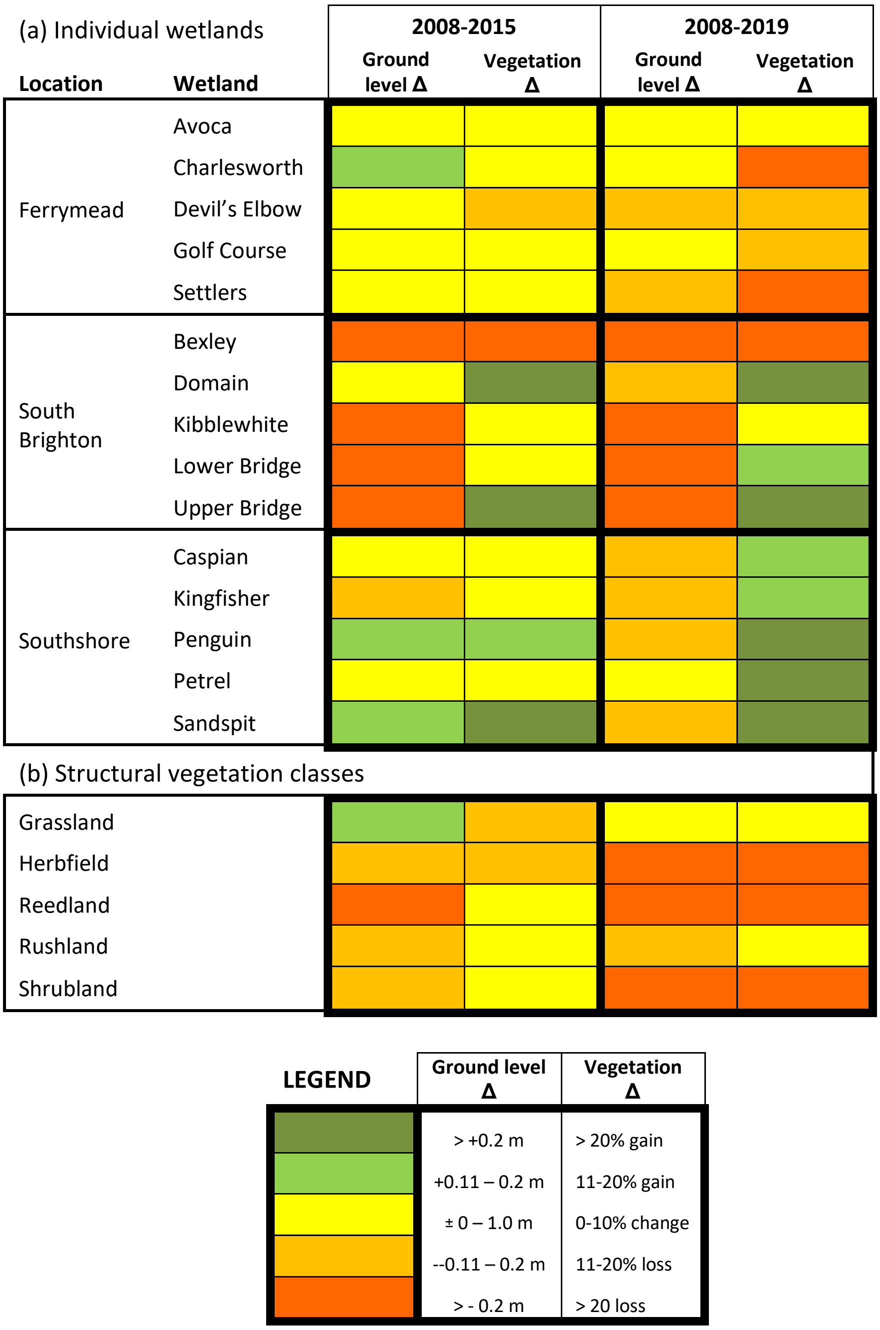

Fig. 2.8 Summary of vegetation responses in relation to ground level change associated with the Canterbury Earthquake Sequence beginning 2010 at (a) 15 individual wetlands on the estuary shoreline, (b) within five key structural classes.

2.4 Discussion

Globally, very few studies have concurrently measured RSL changes and the persistence of VCEs. This is also the first New Zealand study of the effects of RSL changes on contemporary VCEs. In this case we were able to observe variable RSL effects acting on the same wetlands over different time periods as well as variation between sites. Following the February 2011 (6.4 M_w) earthquake, the general pattern of tectonic displacement involved subsidence towards the north and uplift towards the southwest. By 2015, widespread subsidence had returned ground level elevations to their pre-quake levels in uplifted areas (e.g., Ferrymead and Southshore). Subsidence continued across the whole system to 2019 resulting in an eight-year period of RSL rise. Over the 11-year study, there was a 15% loss of wetland extent that mostly reflects rapid changes between 2015 and 2019. In the following sections we discuss the key contributing factors and management implications. We begin with a discussion of uplift before turning attention to subsidence and RSL rise.

2.4.1 Uplift and resilience gains

Uplifted areas showed changes in the relative proportions of wetland types, even where there was little change in the overall extent of wetlands (e.g., Ferrymead). Turnover was evident in both structural vegetation classes and the intertidal zonation pattern resulting in new configurations of the characteristic wetland species. A lag effect between the timing of ground level changes and vegetation responses was evident for shrubland and consistent with slow establishment times in comparison to other classes. The 2015 measurements showed an expansion of shrubland in the initially uplifted areas likely facilitated by drier conditions and also consistent with increases in the extent of the upper intertidal plant community (i.e., the ‘high’ zone). At this time, however, ground levels had essentially returned to pre-earthquake conditions. Assuming a continuation of lag effects, the future contraction of shrubland was expected and indeed was detected in the 2019 surveys.

Key implications for management include the need to avoid the encroachment of anthropogenic land-uses into uplifted areas that initially became drier and further from the tide. This space is now needed to accommodate a reversal of the initial (February 2011) ground displacements. Higher intertidal areas have historically been easy targets for land reclamation, yet this comes at the expense of high tide beaches and biodiversity impacts such as shorebird habitat loss (Green et al. 2015; Woodley 2012). In this case, further encroachment has the effect of exacerbating coastal squeeze pressures by introducing new barriers to wetland migration that did not exist before the earthquake (Fig. 2.7c). It also suggests that coastal uplift should be embraced as a resilience gain for future sea-level rise.

Seaward shoreline movements present rare opportunities for a natural resetting of the system, offsetting the effects of historical land development trends.

2.4.2 Subsidence and relative sea-level rise

In New Zealand and elsewhere, conservation policy objectives are primarily focussed on the protection of current VCEs through the identification of important locations and establishment of protected areas. Achieving protection has the potential to improve outcomes for biodiversity and climate change mitigation yet is challenging to integrate with other important land-use under conditions of RSL rise (Doughty et al. 2019; Lovelock et al. 2017; Pendleton et al. 2012). Our empirical observations confirm beyond doubt that rapid RSL rise can pose major risks to the persistence of tidal wetlands. Furthermore, the observation of a sustained and relatively constant rate of wetland loss over an eight year period of subsidence (2011 – 2019) in all three areas contributes valuable information on the question of whether compensatory accretion processes can keep pace.

The annualised RSL rise rates show that the lowest rate observed (32 mm year^{-1}) was beyond the threshold of resilience. Additionally, lower RSL rates were associated with less vegetation loss than higher subsidence rates, consistent with the notion of a threshold at which compensatory accretion could be achieved (Morris et al. 2002). These are very high rates of RSL rise in comparison to background sea-level changes (~ 2 mm year^{-1}) and future climate change scenarios (Church et al. 2013), and therefore, are not directly illustrative of the progression of changes that might be expected under gradual climate change. Instead, they are informative as scenarios that empirically illustrate the outcomes of RSL rise that exceeds the threshold for persistence of characteristic VCEs. In the following sections we identify mechanisms that have contributed to wetland loss in this example of RSL rise, and conversely, opportunities for human agency to ameliorate and build resilience to similar future effects.

As was confirmed in this study, elimination effects are more likely to be found in the lower intertidal zone, consistent with established theory (Kirwan & Guntenspergen 2010), even though vegetation changes occurred across the full intertidal range. Vegetation losses adjacent to active tidal channels are likely to increase the risk of below-ground carbon deposits being eroded and exported from the wetland (Theuerkauf et al. 2015). However, we observed that the root mat of rushland species such as oioi (*Apodasmia similis*) could persist for several years after mortality of the vegetation and this may present an opportunity for restoration initiatives to save carbon deposits (Fig. 2.7d). We also observed elimination effects higher in the intertidal range associated with the die-off of shrubland (Fig. 2.7e). The newly exposed areas have been slow to recover and these could also present targets for beneficial restoration. Previous studies have shown that small-scale disturbance dynamics may significantly influence vegetation trajectories (Macreadie et al. 2013; Martinetto et al. 2016), and marsh-scale vegetation differences influence the rate of accretion overall (Chmura & Hung 2004; Rodríguez et al. 2017; Roner et al. 2016).

Practical strategies for the management of rising sea levels may include the maintenance of conditions that promote compensatory accretion because this process provides the most promising potential mechanism by which current wetland footprints could be maintained (Macreadie et al. 2017; Voss et al. 2013). As was noted by Kirwan et al. (2016), biogeophysical feedbacks have the potential to drive higher accretion rates with increasing inundation duration. Threshold rates for tidal marsh survival under global sea-level rise were found to be highly influenced by sediment availability (Kirwan et al. 2016), which may be linked with anthropogenic influences such as sediment impoundment in regulated rivers (Weston 2014). In this case previous studies have shown that sediment supply was sufficient to infill and eliminate tidal marsh communities in the Ferrymead area over a 14 year prior to the earthquakes (Jupp et al. 2007). However, sediment accumulation rates for the estuary as a whole were found to be surprisingly low (< 1mm yr^{-1}) based on measurements made between 1970 and 2000 (Burge 2007), and were much lower than typical of other NZ estuaries (Morrison et al. 2009). Additionally, organic matter loading has been reported to be less than the rate of *in situ* mineralisation since the cessation of treated wastewater discharges to the estuary in 2010 (Zeldis et al. 2019). These factors suggest that the availability of suspended particulate matter was likely to have contributed to the observed lack of compensatory accretion in this example of RSL rise. However, the progressive loss of surface elevation long after the period of seismic activity introduces another as yet unexplained factor that requires further investigation. Potential explanations include erosion from areas of marsh break-up in the low intertidal zone or from areas of degraded vegetation, both indicative of a lack of VCE recovery. Although measures of vegetation condition are not directly available from the methodology used in this study these explanations are consistent with the observed increase in sparsely-vegetated areas (Fig. 2.5b).

Although the rates of RSL rise observed are high in comparison to expectations for climate change (Kirwan et al. 2016), the anthropogenic contributing factors are transferable to the general case of managing RSL rise that may exceed plant community thresholds. Our results point to the need for prioritisation exercises to identify at-risk locations, with specific attention to the size of the areas involved. This spatially-explicit approach offers practical benefits for the design of intervention strategies targeting risk factors other than hydrological changes that contribute to marsh-scale vulnerability and the landscape-scale risk profile. The urgent need for such vulnerability assessments has been similarly highlighted in other studies (Doughty et al. 2019; Osland et al. 2017).

Useful management techniques could include the restoration of vegetation communities on the cusp of die-back and loss, supplementing sediment supply shortages important for accretion potential, and restoring allochthonous organic material inputs from higher in the catchment where these have been reduced by deforestation or other land-use change (Duarte et al. 2013; Macreadie et al. 2017; Morris et al. 2016). These principles offer promise for a new ecological engineering paradigm that differs from, and yet is complementary to the assisted migration approach based on the facilitation of range shifts (Hällfors et al. 2017). However, the latter is also an important strategy to ensure the continuity of accommodation space and relies on the identification of suitable areas for VCE migration under a changing climate (Spencer et al. 2016).

2.4.3 Connectivity, land-use planning, and infrastructure design

Despite the self-maintenance potential of vertical accretion, the management of connectivity between potential areas of tidal inundation is an important principle for the sustainability of coastal wetlands. In the upper intertidal zone this may rely on small waterway connections or overland flowpaths only activated at high tidal levels and which are easy to overlook. Simple engineering such as attention to hydrological barriers or planned breaches of existing flood defences provide useful strategies for assisting the movement of wetlands as conditions change (Duarte et al. 2013). To enable this, the suitability of existing land uses for retirement and naturalisation becomes a key consideration, and this is an excellent example of where an opportunistic conservation paradigm could be beneficial for climate change (Knight & Cowling 2007). In the present study an example became evident only recently on the southwestern shoreline. This area had previously uplifted before subsiding more recently, activating the landward migration of saltmarsh herbfield species via an existing network of drainage channels. In response, large areas of pasture have begun a transformation to coastal wetland vegetation (Fig. 2.7f). The lesson here is two-fold. First, this illustrates a natural environment response in the absence of connectivity barriers. Second, it highlights the need to identify and protect such areas to build resilience to sea-level rise.

2.4.4 Risk factors that can inform climate change adaptation

Although the RSL rise rates observed in this study represent extreme scenarios the empirical observation of VCE responses have illuminated risk factors that are transferable to other locations facing RSL rise. We conclude by summarising four risk factors amenable to management that can be identified from the observed patterns of VCE loss.

1. Anthropogenic encroachment results in resilience loss

When predicted sea-level rise is considered, further encroachment of anthropogenic barriers results in a loss of resilience for tidal lagoons. Although this principle may appear obvious, in reality it is seldom embraced. Reclamation and armouring continue to occur and managed retreat is rare. The severity of effects depends on the size of the areas that are lost from the natural environment and the specific elevation ranges involved that are influenced by both of the location and design of built-environment modifications. As with our example, the use of temporary barriers for flood defence while longer term solutions are being designed may offer a pragmatic response for dealing with unexpected extreme events. In the general case, however, the reversibility of any such new infrastructure is an important consideration for socio-ecological resilience in the longer term.

2. Connectivity losses exacerbate encroachment effects, and conversely, are a practical focus for improving resilience

The context of rising sea levels demands attention to connectivity effects. As observed here, higher water levels cause impacts on the lower elevational limit of coastal wetlands and drive compensatory landward movement across the shore profile. Anthropogenic modifications may have the effect of truncating the landward migration of VCEs as conditions change. Although opportunities for resilience gains may be identified and modelled as suitable ground elevations behind the line of existing defences, they are highly dependent on connectivity effects. This demonstrates a relationship between connectivity-related impacts and the design of infrastructure. For example, storm defences could be designed to allow for connectivity to support natural environment process and deployed only as needed during extreme events.

3. Risk exposure is disproportionately influenced by the largest wetland remnants
Vulnerability acts unevenly across the spatiotemporal domain, and yet current wetland values are associated with a specific footprint. In this study, vulnerability of the largest wetlands proved to be an important contributor to the overall impacts and was a function of their vegetation composition and position in the socio-ecological landscape. These results suggest a need for spatially-explicit vulnerability assessments at the scale of individual wetlands, with a focus on the larger remnants being important to risk reduction. Similarly, site-specific vulnerability assessments are required to evaluate effects on key wildlife species with regards to critical habitats. They may also be useful in considering climate effects on human activities such as wild harvest and recreational use.

4. There is a need to protect new areas to accommodate the future movement of ecosystems
Despite the promise of wetland self-maintenance, this study showed the actual movement of wetlands exposed to prolonged RSL rise. Similar effects may occur in other VCEs given current climate change projections and are particularly likely where sediment availability is limited. In combination with connectivity management, the availability of space poses a critical risk. Land-use change initiatives must be initiated without delay to improve the potential for workable solutions to be identified, and as highlighted here, they include taking advantage of opportunities to future-proof available land. Arguably, the most effective formula for long term sustainability integrates new ecological engineering paradigms with resilience-based land-use planning and infrastructure design. Failure to embrace these concepts poses a major risk to the persistence of coastal wetlands and the multiple benefits they provide.

2.5 Acknowledgements

We thank Land Information New Zealand, Environment Canterbury and Christchurch City Council for access to datasets, and Philip Grove, Mark Parker, and Andrew Crossland for sharing field observations and helpful discussions. Thanks to Thomas Falconer, Zoe Smeele and Irene Setiawan for field assistance. Financial support for this study was provided by the Ngāi Tahu Research Centre at the University of Canterbury.

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Chapter 3

Nature-based solutions for climate change on a peri-urban sandspit

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Abstract

Barrier sandspits are biodiverse natural features that regulate the development of lagoon systems and are popular areas for human settlement. Despite many studies on barrier island dynamics few have investigated the impacts of sea-level rise (SLR) on sandspits. In peri-urban settings, we hypothesised that shoreline environment change would be strongly dependant on contemporary land-use decisions, with modern engineering capabilities marking a significant step-change from evolutionary dynamics of the past. We evaluated these factors in a case study from Christchurch, New Zealand, that presented a unique example of SLR caused by tectonic subsidence and included new opportunities for managed retreat created by government acquisition of affected properties. We identified plausible futures for the post-earthquake landscape using a novel scenario modelling approach considering both shorelines of the sandspit simultaneously for 0.25 m increments of SLR and with varying positions of coastal defences. The results identify challenges for dune conservation following a reversal of the current shoreline progradation trend. A third of the dune system is eliminated under a 1 m SLR in 100 year scenario, leading to increased exposure to natural hazards such as extreme storms and tsunami. Increasing demand for seawalls is a likely consequence unless natural alternatives can be progressed. In contrast, the managed retreat initiative on the backshore presents an opportunity to restart saltmarsh accretion processes seaward of coastal defences with the potential to reverse decades of degradation. Considering both shorelines simultaneously highlights the existence of pinch-points from opposing forces that result in small land volumes above the tidal range. Societal adaptation is delicately poised between the paradigms of resisting or accommodating nature and challenged by the long perimeter and confined nature of the sandspit feature. We suggest

that the further use of innovative policy measures in disaster recovery contexts may offer a beneficial, though reactive, framework for progressing adaptation to climate change.

Keywords

Natural hazards, natural features, socio-ecological systems, shoreline management, climate change adaptation, sea-level rise.

3.1 Introduction

Aquatic margins are dynamic boundary environments that have become a global conservation priority in the face of widespread environmental change (Brown & McLachlan 2002; Strayer & Findlay 2010). Pressures have become critical in lowland and coastal regions due to a rising population and the intensification of human land-use (McGranahan et al. 2007; Neumann et al. 2015). Sandy beaches are an important subset of these considerations due to their worldwide distribution (Short 1999), and are typically of high value for local communities wherever they are found (Klein et al. 2004). Barrier sandspits represent a special case due to their narrow low-lying configuration and characteristic twin shorelines that are often approximately parallel (Otvos 2012). They play important roles in protecting mainland shorelines and structuring tidal inlets and lagoons (Lorenzo-Trueba & Ashton 2014). They also support many anthropogenic values, creating a difficult and unique challenge for environmental planning under conditions of sea-level rise (Fallon et al. 2017; Moore et al. 2010).

Beach management is nuanced by highly diverse stakeholder perspectives (Martinez et al. 2017). These are underpinned by the many benefits of beaches as natural coastal defences, key sites for recreation and tourism, and local amenities linked to property values (Parsons & Powell 2001; Schleupner 2008). Beaches are also ecosystems that may require protection from anthropogenic use (Brown & McLachlan 2002; Schlacher et al. 2008). Management difficulties often arise from trade-offs between historical and anticipated human uses and the underlying natural system in which they occur (Defeo et al. 2009; McLachlan et al. 2013). Despite these inter-dependencies, the prevailing management paradigm has become focused on engineering approaches that seek to control beach function and form (Schlacher et al. 2007; Schlacher et al. 2008). Examples include the use of armouring to protect nearby built environments (Cooper & Pilkey 2012), beach nourishment programmes to maintain sediment budgets (Nordstrom 2005), and a wide variety of modifications for attenuating sediment deposition trends (Nordstrom 2014). Although the latter may include the purposeful use of vegetation (Martinez et al. 2016), the conservation of natural features and habitats is often a minor consideration, which suggests a general lack of appreciation for beaches in their natural form (Defeo et al. 2009; Dugan et al. 2010; Peterson & Bishop 2005).

The scientific literature is symptomatic of these trends with the research effort being skewed towards engineering and geomorphology (Jackson & Nordstrom 2019). A review by Nel et al. (2014) found that only 22% of sandy beach publications in the period 1950 - 2013 had a focus on anthropogenic impacts, conservation, or management. Furthermore, sandy beach ecosystems are poorly represented in the conservation ecology literature (Schoeman et al. 2014), and often neglected in protected area networks (Schlacher et al. 2006). Contributing factors may include their narrow configuration combined with jurisdictional overlaps and associated management complexities. For example, many marine protected areas do not incorporate a landward dimension despite the prevalence of threats that have distinctly terrestrial origins (Martinez et al. 2014). These aspects suggest a need for whole system approaches that integrate across management zones and also respond to environmental change (Harris et al. 2015; Schiel & Howard-Williams 2016).

Nature-based solutions involve working with natural ecosystems to overcome societal challenges (Cohen-Shacham et al. 2019; Kabisch et al. 2016). Climate change presents new issues associated with sea-level rise, storm events and interactions with other variables (Harley et al. 2006). On erodable coasts the potential for shoreline retreat presents a considerable threat to many current values (Martinez et al. 2017). Conservation challenges include accommodating the landward migration of natural environments and their interactions with people (Feagin et al. 2010; Lentz et al. 2016; Nicholls & Cazenave 2010). Because effective measures for habitat and ecosystem conservation are already lacking (Hoekstra et al. 2005), there is an urgent need to address the plight of natural systems (Betts et al. 2009; Jennings & Harris 2017). The concept of ecosystem-based adaptation is a solution-oriented paradigm that provides a framing for proactive measures (Roberts et al. 2012). It has been defined as “adaptation that integrates ecosystem services and biodiversity into a strategy to limit the adverse impacts of climate change” (UNEP 2010). In contrast to hard engineering it aims to maintain the properties and services of natural ecosystems and to incorporate natural defences in the design of hazard management projects (Colls et al. 2009;

Dudley et al. 2010; Spalding et al. 2014). Experience to date, however, suggests that implementation remains difficult (Wamsler 2015). Typical challenges include attracting buy-in from current stakeholders and identifying solutions that avoid unnecessary trade-offs (Adams et al. 2016; Di Marco et al. 2016).

In this study, we explore these issues in relation to shoreline conservation on a peri-urban sandspit with a focus on the prospects for ecosystem-based adaptation. Additional impetus for the investigation arose from a recent natural disaster involving major earthquakes and subsidence of the backshore, creating an immediate need for recovery (Chapters 1 and 2). This unique situation led to the government acquisition of residential land in two different areas on the estuarine backshore in the city of Christchurch, New Zealand (Fig. 3.1). These properties have since been demolished creating new land-use opportunities together with a raised awareness of natural hazards and climate change implications. This study was timed to inform the difficult decisions faced by affected communities. Our specific objectives were to a) investigate potential shoreline changes on both coasts simultaneously with incrementally rising sea levels from the current point in time, (b) identify the potential implications of status quo land-uses for conservation of the margins, and (c) evaluate the opportunities for ecosystem-based adaptation with an emphasis on integration with natural hazard management and the current disaster recovery context.

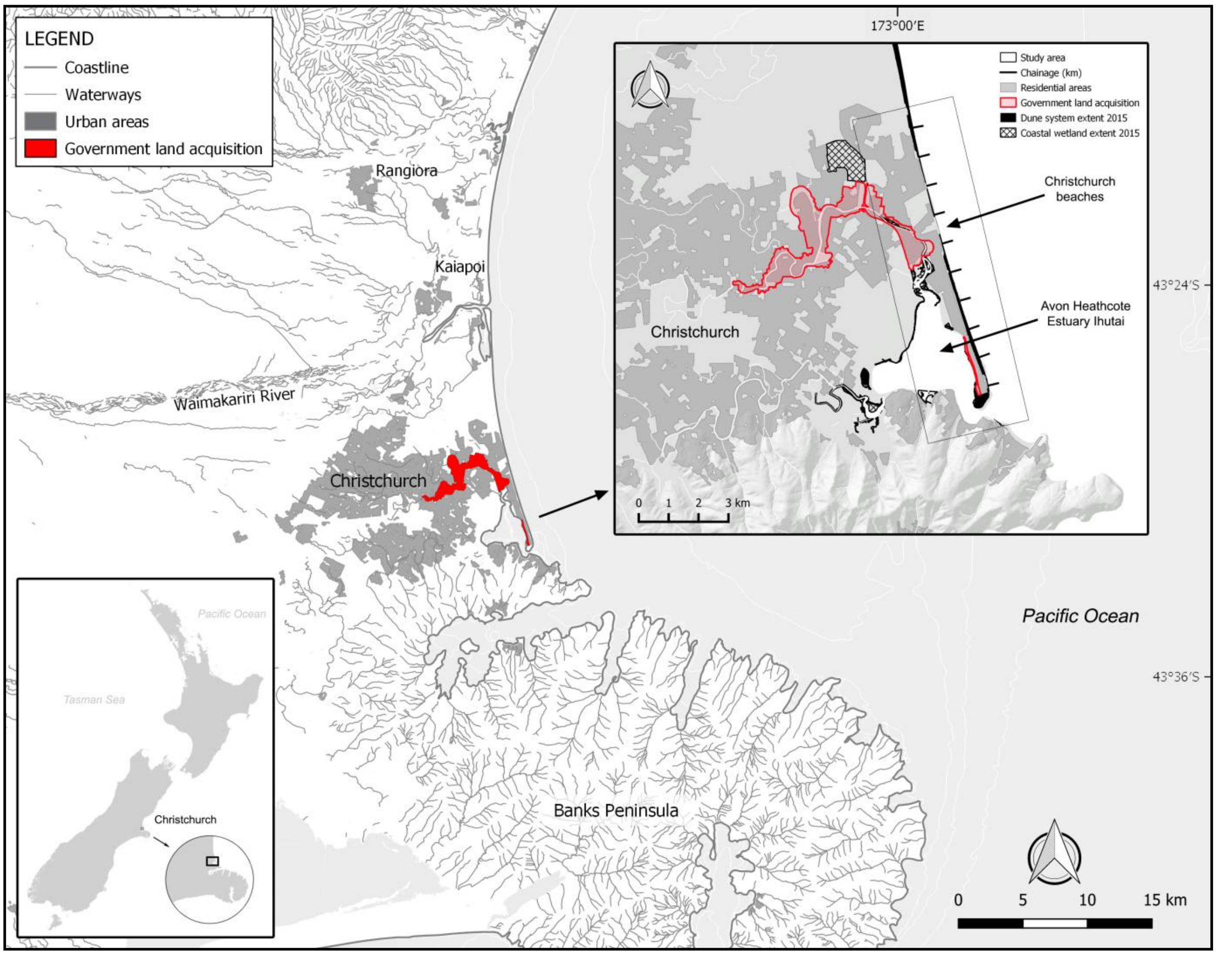

Fig. 3.1 Location of the Avon Heathcote Estuary Ihutai, Christchurch beaches and barrier sandspit on the east coast of New Zealand's South Island.

3.2 Methods

3.2.1 Study area

The study area is the barrier sandspit of the Avon Heathcote Estuary Ihutai which enters the Pacific Ocean at the southern end of Pegasus Bay on the South Island's east coast (Fig. 3.1). The sediment composition of the beaches exhibits a gradient from finer sands in the south to mixed sand-gravel beaches further north in Pegasus Bay (Kirk 1979). Sediment sources are primarily the Waimakariri and Ashley Rivers (Blake 1968; Kirk 1979), and a small supply of fine sediments from the Avon-Heathcote Estuary catchment (Begg et al. 2015). The wave climate is characterised by a mixture of southerly swell refracted around Banks Peninsula, less frequent easterly and north-easterly long period swells and frequent locally-generated wind waves (Hart & Knight 2009). The predominant longshore current is north-south along the Christchurch beaches under the influence of a large eddy in the offshore Southland current in the lee of Banks Peninsula together with locally generated swell from north-east wind patterns (Kirk 1979; Reynolds-Fleming & Fleming 2005). The semidiurnal tidal regime is typical of a micro-tidal lagoon-type estuary with spring tidal range of 2.2 m at the entrance based on gauge records from nearby Port of Lyttelton (LINZ, 2019). In recent decades the open beaches have experienced a progradation trend despite a small rise in eustatic sea level (Tonkin & Taylor 2017). The dune system is dominated by introduced marram grass (*Ammophila arenaria*), iceplant (*Carpobrotus edulis*) and lupin (*Lupinus arboreus*) with small areas restoration planting to re-establish indigenous sand dune species such as spinifex (*Spinifex sericeus*) and pīngao (*Ficinia spiralis*) (Bergin 2008; Orchard 2014). The current ecosystem is organised around a central marram-dominated foredune with variable front-slope gradients, occasional secondary dunes, and crest elevations of up to 7 m above the reach of the tide (Fig. 3.2). The width of current system is constrained landward by residential and commercial areas along the full of the sandspit with the exception of a public reserve at the distal tip.

Christchurch is New Zealand's second largest city with a population exceeding 400, 000. It was struck by a series of large earthquakes (up to M_w 7.1) on previously unknown fault lines beginning in late 2010 and tapering off in 2012 (Beavan et al. 2012). Four earthquakes exceeded M_W 6.0 over the Canterbury Earthquake Sequence (CES) with profound effects on the city and natural environment (Quigley et al. 2016). The most serious event occurred on 22 February 2011 causing 185 fatalities and capital costs of approximately NZ$40 billion (Kaiser et al. 2012; Potter et al. 2015). The process of disaster recovery has taken many years due to the extensive infrastructure damage and complex decisions on future land use patterns in relation to natural hazards (Hughes et al. 2015; Orchard 2017a). Major environmental changes in the study area and nearby Avon Heathcote Estuary Ihutai included lateral spread and liquefaction (Robinson et al. 2012; Zeldis et al. 2011), tidal prism and salinity changes (Measures & Bind 2013; Orchard & Measures 2016), and new patterns of inundation (Chapter 1). Of particular relevance to this study, recovery planning has included the establishment of new legislation and the government acquisition of thousands of residential properties on the estuary shoreline and lower river floodplains (Regenerate Christchurch 2019). An increasing awareness of coastal hazards has been supported by recent studies on climate change aspects (Orchard & Challies in press; Tonkin & Taylor 2017).

3.2.2 Climate change policy context

Current climate change policy for New Zealand requires the avoidance of redevelopment or land-use change that would increase the risk of adverse effects in areas potentially affected by coastal hazards (Department of Conservation 2010). Hazard risk assessments are required for a minimum 100 year planning horizon along with responses that integrate across administrative boundaries and are inclusive of the built environment, public use and enjoyment values, and the fate of natural ecosystems and characteristic species (Orchard 2011). Recent national guidance has suggested a risk tolerance approach structured by land-use type as a basis for selecting sea levels for consideration in risk assessments (Bell et al. 2017). This includes an accelerated warming scenario based on the K14 projection of Kopp et al. (2014) for the assessment of greenfield developments or to 'stress test' the potential impacts of high-end sea-level rise rates on other land uses. Other scenarios recommended for testing are based on the representative concentration pathway (RCP) median projections following Church et al. (2013). Together, the recommended scenarios range from 0.32 – 0.61 m for 2070, and 0.55 – 1.36 m for 2120 relative to an average mean sea level baseline for the period 1986–2005 (Bell et al. 2017).

3.2.3 Scenario modelling

We used a GIS-based scenario modelling methodology to investigate potential impacts whilst simultaneously considering both open coast and estuarine shorelines. The scenario approach focuses on the conceptualisation of plausible futures which may lie outside the realm of past experience and yet are distinctly possible (Cuddington et al. 2013; March et al. 2012). Uncertainty is addressed through the exploration of alternative futures. This differs markedly from probabilistic approaches that attempt to directly incorporate uncertainties associated with a future event (Peterson et al. 2003). Each scenario was constructed using numerical simulations on 900 individual transects developed using the *ambur* package for shoreline change analysis in R (Jackson et al. 2012). Transects were cast perpendicular to a smoothed open coast shoreline at 10 m intervals and extrapolated to the estuarine shoreline (Fig. 3.2). Changes were assessed relative to a 2015-2016 baseline, and spatiotemporal differences analysed for contiguous one kilometre chainage units each containing 100 transects. Baseline measurements were compiled using a combination of historical trend extrapolation, tidal height modelling and field survey. We followed two sets of environmental changes: dune ecosystem movement on the open ocean shoreline and tidal saltmarsh inundation changes on the estuarine shoreline. Changes were modelled simultaneously on full-width cross sections to consider their combined effects on the sandspit feature.

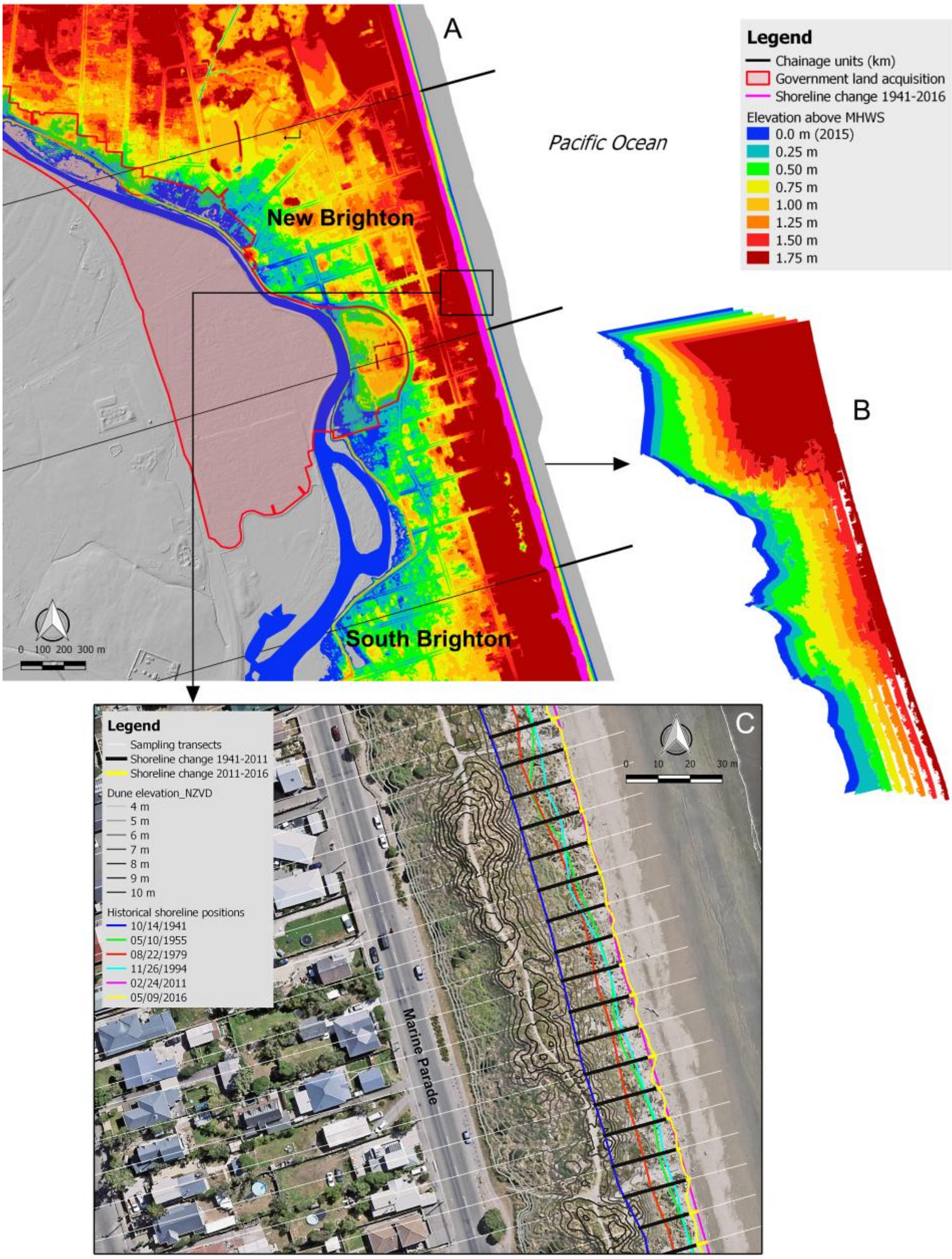

Fig. 3.2 Linked scenario models. (a) the barrier sandspit at New Brighton showing ground elevations above Mean High Water Springs (MHWS). (b) pour-point scenario models representing inundation of hydrologically connected areas in 0.25 m sea-level rise increments relative to MHWS. (c) shore-perpendicular transect array used to simulate open coast responses showing historical shoreline positions and a typical section of dunes near New Brighton village.

Open coast shoreline change

Open coast scenarios were simulated using a combination of historical shoreline change extrapolation and a geometric beach response model. For the latter we used an adaptation of Bruun's rule based on the conservation of beach volume (Bruun 1962), providing a first order approximation of the relationship between shoreline movement Δy, to sea-level rise S, where W* is the width of the active profile perpendicular to the shoreline, h is the depth of the active profile base, and B is the berm crest elevation (Equation 1).

$$\Delta y = -\Delta S \times W^* / h + B \qquad \text{(Equation 1)}$$

Alternatively, this relationship can be expressed as Equation (2) where $\tan\theta$ is the average slope across the active beach profile of width W* (Komar 1998; Scientific Committee on Ocean Research 1991).

$$\Delta y = -\Delta S / \tan\theta \qquad \text{(Equation 2)}$$

Beach profiles undergoing Bruun–type behaviour are expected to equilibrate in response to sea-level rise following a process of sediment erosion from the upper beach and deposition into the lower portion of the profile such that the profile shape is maintained over time. Many authors have asserted that Bruun–type behaviour is the exception rather than the rule due to the underlying assumptions being seldom met (Cooper & Pilkey 2004; Dean & Houston 2016). For example, the rule assumes there are no other sediment exchanges with respect to the profile area and adjacent coastal compartments (Bruun 1983). To empirically account for the likelihood of sediment flux effects, we incorporated recent historical shoreline movement rates as used in similar studies (Anderson et al. 2015), and recent coastal hazards assessments in the study area (Tonkin & Taylor 2017). This provides a computationally efficient approach for addressing Bruun rule limitations, although a key assumption is the continuation of historical trends.

Historic shoreline data were available from aerial photography compiled by the Canterbury Regional Council for five dates over 1941-2011 with shoreline positions digitised as polylines. An additional recent shoreline was digitised using recent imagery (LINZ 2016), to create a set of six shorelines spanning a 75 year period. Each shoreline represents the seaward limit of vegetation at the foredune toe which is a characteristic position visible in all of the historical imagery. Shoreline positions and rates of change were calculated using the linear regression functions in *ambur* (Jackson et al. 2012).

Additionally, we applied the conceptual model of Davidson-Arnott (2005) to better account for relationships between the beach and foredune which are prominent in this case. This proposes a similar relationship to the Bruun rule for shoreline response to sea-level rise but calculates the nearshore slope (tanθ) for only the inundated portion of the active beach profile (*h*) and therefore independent of the dune height (B). With rising sea levels, sand eroded from the dune is available for both aeolian transport and exchange with the nearshore environment providing a plausible mechanism for landward migration of the foredune that is not explicitly accommodated within Bruun rule assumptions (Davidson-Arnott 2005). We adopted the 10 m depth contour as the lower limit of the active profile as is commonly used in other studies (Stive 2004) and used the 2016 shoreline position for the upper limit. Elevation of the latter was extracted from a 1 x 1 m DEM prepared from airborne light detection and ranging (LiDAR) data acquired October 2015 with a horizontal and vertical accuracy of ±1.00 and 0.20 m, respectively. This represents the most recent of several LiDAR acquisitions made during the course of the CES (LINZ 2017). The mean elevation (3.12 ± 0.5 m SD) was used to calculate a tanθ value of 0.01 which was applied to all simulations.

Following the above approach, sea-level rise scenarios were generated in 0.25 m increments with sediment supply simulated as the continuation of historical accretion rates within periods of 25 years. An adjustment term was required to account for the rate of sea level change that occurred within the period of historical observations, which is estimated at 1.9 mm yr^{-1} based on Lyttelton tidal gauge measurements adjustments for vertical land motion (Hannah & Bell 2012). Each resulting scenario represents a unique combination of sea level height and historical rate of change extrapolation within a defined period of time and for which the sea-level rise increment can be compared for both the open and estuarine coast. In relation to sea-level rise projections using RCP scenarios (Church et al. 2013), these represent plausible high-end scenarios over shorter time frames of < 50 years and RCP 8.5 median scenarios for 100 years and beyond (Table 3.1).

Table 3.1 Specification of scenarios.

(a) Sea-level rise scenarios and adjustment factors used for the estimation of beach responses on the open coast using a combination of the Bruun rule and historical trend extrapolation. SLR = sea-level rise. SLR increments are relative to baseline conditions in 2015. ΔS is the adjusted SLR increment incorporating the historical accretion trend. RCP = representative concentration pathway calculations presented in recent New Zealand guidance (Bell et al. 2017). M = median. H^+ = 83rd percentile (Church et al. 2013).

Time frame (years)	SLR† increments (m)	Historical rate of change adjustment	ΔS	Comparable RCP scenarios‡
25	0.25	0.05	0.20	RCP8.5 H^+
50	0.50	0.10	0.40	RCP8.5 H^+
75	0.75	0.14	0.61	RCP 8.5/8.5 H^+
100	1.00	0.19	0.81	RCP8.5 M
125	1.25	0.24	1.01	RCP8.5 M

(b) Orthographic heights of interest.

Description	Elevation † (m)	Data source
Lower marsh limit	0.35	this study
Mean High Water Springs (MHWS)	0.84	Land Information New Zealand (2019)
Highest Astronomical Tide (HAT)	1.07	Land Information New Zealand (2019)
Upper marsh limit	1.36	this study

†all heights are in New Zealand Vertical Datum 2016.

Estuarine shoreline change

Estuarine shoreline scenarios were developed using static inundation models based on extraction of hydrological connected areas from the 2015 DEM and applying the same SLR increments used on the open coast (Table 3.1). Here, the RCP analogies are less important for our purpose of simulating the temporal progression of change towards plausible endpoints with a focus on exploring alternatives to the current pattern of defences. These simulations also assume instantaneous inundation of all areas below the elevation of interest which overestimates inundation at the inland limit due to the effects of ground surface friction (Passeri et al. 2015). In this case friction effects are expected to have a limited influence on the local variation in inundation at high tidal levels due to the short flow path distances between riparian land and the estuary basin (Fig. 3.1). As other effects such as the estuarine channel alignment, wind and wave set-up, barometric pressure and river discharge also influence observed water heights, each scenario is interpreted as a discrete plausible future independent of the frequency and conditions under which it may occur.

We considered four orthographic heights of interest in relation to tidal heights and coastal vegetation limits (Table 3.1b). Mean High Water Springs (MWHS) and Highest Astronomic

Tide (HAT) heights were identified from published figures for the current 18.6 year tidal cycle (2000-2018) at Port Lyttelton (LINZ 2018a). Although HAT is defined by the full tidal cycle, MHWS is subject to yearly variations of up to 15 cm across the cycle and the current position is 11 cm above the mean (LINZ 2018b).

To investigate effects on vegetation limits we developed a vegetation model representative of stable pre-earthquake conditions to circumvent problems with earthquake-induced ground level changes and lag effects in the pattern of vegetation responses (Chapter 2). We used a 5 x 5 m DEM (horizontal and vertical accuracy ±0.55 and 0.15 m, respectively) based on LiDAR data acquired in 2003 (Canterbury Geotechnical Database 2014), and the results of a 2008 vegetation survey that mapped all coastal wetlands in the estuary (Grove et al. 2012). Characteristic estuarine vegetation includes extensive areas of sea rush (*Juncus kraussii*), jointed wire rush / oioi (*Apodasmia similis*) and saltmarsh herbfield grading to saltmarsh grasslands and coastal riparian shrubland (Grove et al. 2012; Jones & Marsden 2007). Upper and lower vegetation limits were defined empirically using zonal statistics analysis of all mapped estuarine vegetation polygons (n = 347) and adoption of the 2.5th and 97.5th ground elevation percentiles. This recognises that isolated features such as channels and embankments may be present within the wetland dataset and may have been sampled in the raster analysis. We also reduced the effect of inaccuracies in the digitisation of vegetation unit boundaries by applying a 1m negative buffer to all polygons to reduce edge effects. All analyses were conducted in NZTM 2000: ESPG 2193 projection using QGIS v 2.18 (QGIS Development Team 2017). Statistical analyses were conducted in R v3.3.3 (R Core Team 2017).

3.3 Results

3.3.1 Open coast shoreline

Shoreline change rates over the 1941 – 2016 period show a net accretion trend throughout the study area with a mean of 0.29 m yr^{-1}, maximum of 0.71 m yr^{-1} and minimum of 0.003 m yr^{-1} on individual transects (Fig. 3.3).

Average values for the 1 km chainage units show considerable variation in mean accretion rates (ANOVA, F = 297.8, p < 0.001), which range from 0.17 – 0.49 m across the study area (Table 3.3). Sea-level rise scenarios show a reversal of the historical accretion trend across the study area as a whole, although to varying degrees. Two examples (0.5 m and 1.0 m SLR) are explored further to illustrate consequences of these erosion scenarios which are more severe within the middle portion of the study area (Table 3.3).

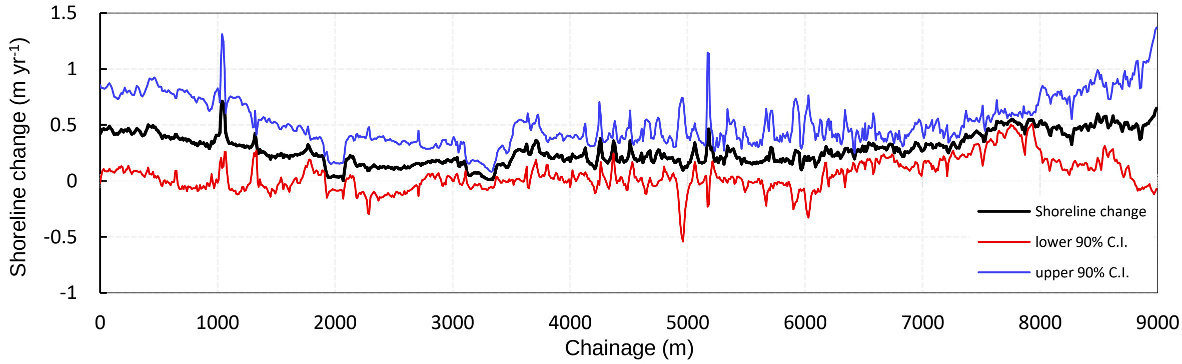

Fig. 3.3 Shoreline rates of change over a 75 year period at New Brighton Beach in Christchurch calculated from a linear regression of six historical shoreline positions (1941 – 2016) on 900 individual shore-perpendicular transects with 90% confidence intervals (C.I.) shown. Positive values indicate accretion.

Table 3.3 Summary of shoreline changes for over the historical period 1941-2016 within contiguous 1 km chainage units at New Brighton Beach in Christchurch together with two SLR scenarios that assume continuation of net sediment fluxes inherent in the historical accretion trend. Each calculation represents mean values from 100 shore-perpendicular transects. SLR = sea-level rise.

Chainage unit	Historical change 1941-2016			Example SLR scenarios	
	Rate of change (m yr^{-1})		Envelope of change (m)	Shoreline change from 2016 (mean ± SD)†	
	mean	SD		50 year_SLR0.5	100 year_SLR1.0
1	0.41	0.05	38.8	-20.1 (± 2.6)	-40.3 (± 5.2)
2	0.27	0.12	26.5	-26.8 (± 5.8)	-53.7 (± 11.5)
3	0.14	0.04	16.3	-33.7 (± 2.2)	-67.4 (± 4.3)
4	0.17	0.09	16.6	-31.8 (± 4.5	-63.6 (± 9.1)
5	0.21	0.05	21.3	-30.1 (± 2.6)	-60.2 (± 5.1)
6	0.21	0.06	21.7	-30.2 (± 3.0)	-60.3 (± 6.1)
7	0.26	0.05	23.0	-27.3 (± 2.3)	-54.6 (± 4.6)
8	0.41	0.09	32.1	-19.8 (± 4.6)	-39.5 (± 9.2)
9	0.49	0.05	43.4	-15.9 (± 2.7)	-31.9 (± 5.4)
All units	**0.29**	**0.14**	**26.6**	**-26.2 (± 4.8)**	**-52.4 (± 8.9)**

† negative values indicate shoreline retreat

Under current (2016) conditions, accommodation space for the dune ecosystem varies considerably due to the position of fixed infrastructure (primarily roads and car parks) in relation to the beach. The shore-perpendicular accommodation space is between 0.6 and 141 m with the low end being representative of a virtual lack of dunes where seawalls have been built opposite New Brighton village and at a small number of other access points. In addition, a contiguous stretch of more than 3 km (chainage ca. 1050 – 4250) is characterised by a dune accommodation space of 60 m or less (Fig. 3.4). This becomes more significant in consideration of the space required to maintain the current foredune size and crest height which is in the vicinity of 6 – 9 m above HAT along the majority of the coastline. Field measurements showed that this minimum space is in the order of 40 m from the 2016 shoreline position (dune toe) as depicted by the orange line in Fig. 3.4. Under the 0.5 m sea-level rise scenario this space reduces markedly to become less than the 40 m threshold over the entire above-mentioned gap. The 1.0 m SLR scenario is characterised by further loss of dunes in this section of the coastline. At the far south of the study area a small area of housing in relatively close proximity to the beach is associated with a further localised area of loss (Fig. 3.4). Other sections of the dune system approach the 40 m width threshold (e.g. chainage 5 – 6000) and are also cause for concern.

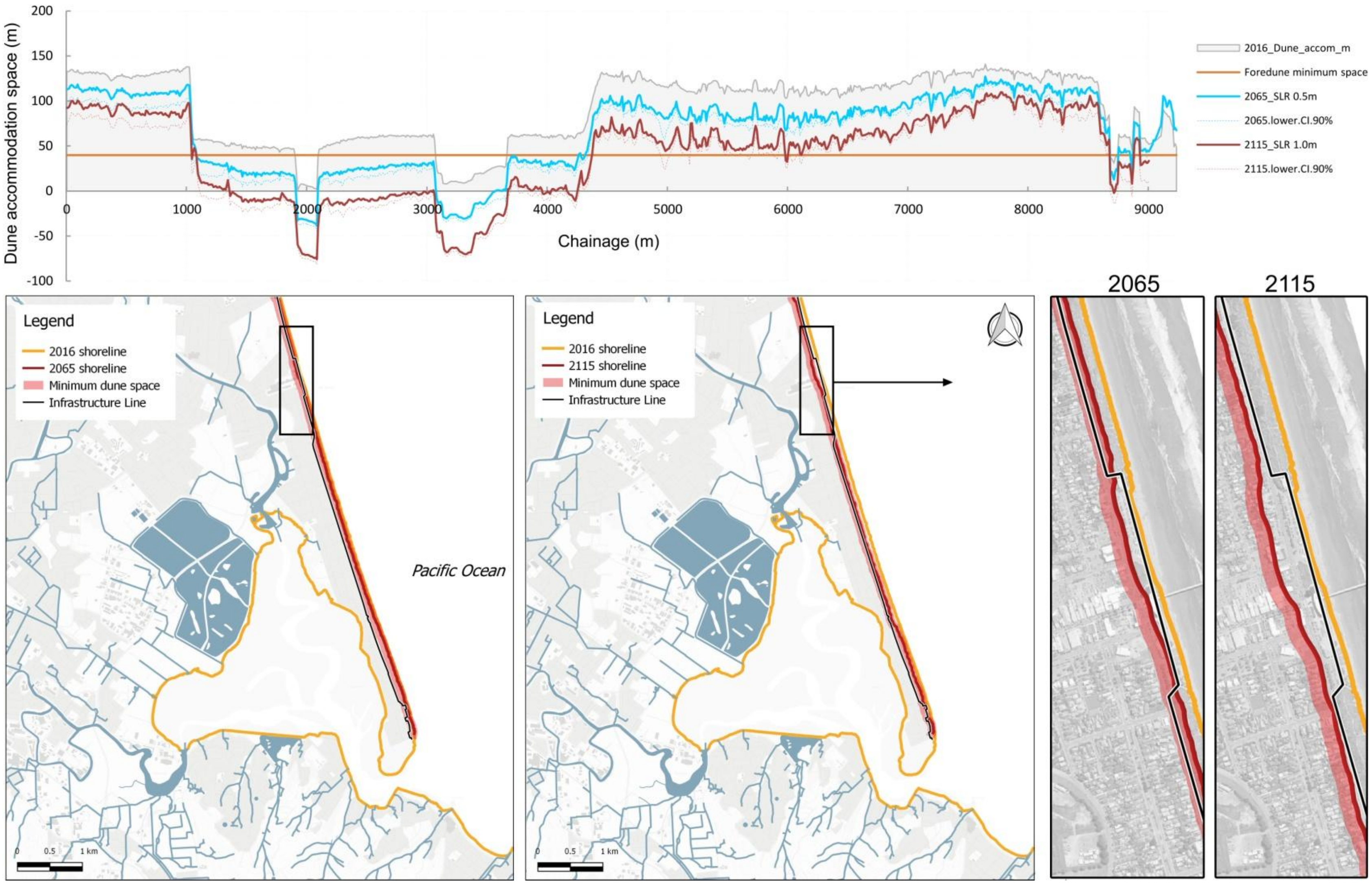

Fig. 3.4 Modelled effects of two sea-level rise scenarios on position of the dune toe in relation to current fixed infrastructure (primarily roads) for a 9 km stretch of coastline. Dune toe positions landward of the 40 m line are associated with loss of the foredune. Negative values indicate the expected position in absence of shoreline armouring.

3.3.2 Estuarine shoreline

The backshore inundation scenarios show that a large proportion of the sandspit is characterised by low elevation topography requiring defences against flooding from the estuary. These effects diminish towards the north (chainage 0 – 3000) where the base of the sandspit is delineated by the current alignment of the lower Avon River / Ōtākaro (Fig. 3.1). Further south, vulnerability increases from the point at which the overall feature narrows to < 1 km. All of the remaining 6 km of the study site is characterised by considerable exposure to flooding (Fig. 3.5a) with a high percentage of current land areas being vulnerable to SLR unless actively defended (Fig. 3.5b). Currently, 27 ha of land occupies the elevation range between HAT and MHWS, which is significant since the position of existing defences is close to MHWS.

Consequences of the CES included the expansion of riparian wetlands along the backshore from 4.2 to 11.1 ha under the influence of ground level subsidence. These changes were driven by a small number of remnant wetlands that were able to migrate to new positions located forward of the line of coastal defences (Chapter 2). However, the overall situation remains very similar to the historical pattern of wetland degradation. For example, the modelled elevation range suitable for wetland vegetation comprises 108 ha at present, indicating that the current extent is around 10% of potential (Fig. 3.5d). Alternatively, if the upper limit of wetlands was confined to HAT, the current extent is still only 17% of the available area. The discrepancies arise from the location of current land uses in relation to tidal patterns. Under SLR, the elevation range suitable for wetlands expands to maximum of 318 ha with 0.75 m SLR and then declines (e.g. 243 ha with 1.25 m SLR). In these higher SLR scenarios, water depths have begun to exceed the threshold for vegetation persistence forcing movement of the lower limit. Importantly, the physical position of this lower limit stays relatively static with SLR increments of < 0.75 m. This information is practically useful since it suggests that saltmarsh re-establishment will remain possible in the vicinity of the current coastal margin for a considerable time, even under conditions of sea-level rise (Fig. 3.5d).

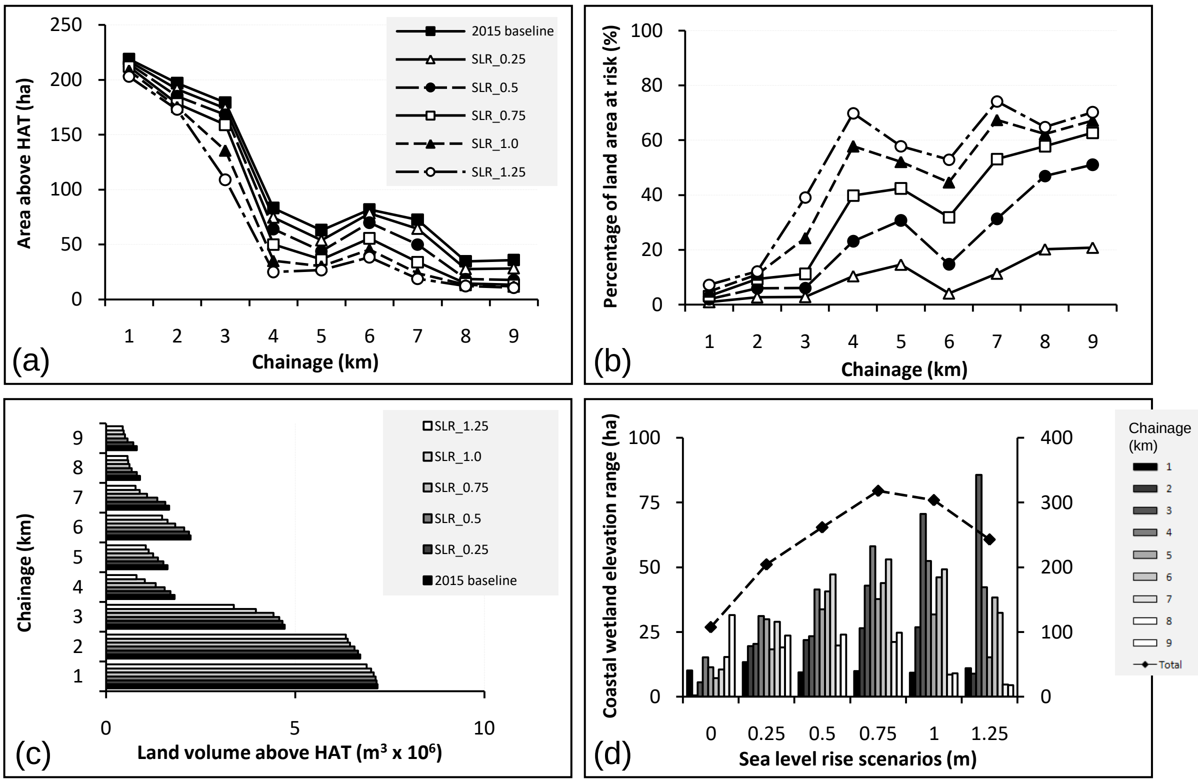

Fig. 3.5 Spatiotemporal scenarios of flood risk development on Southshore sandspit backshore with 0.25 m increments of sea-level rise (SLR). (a) area of dry land above the elevation of hydrologically-connected inundation pathways at highest astronomic tide (HAT). (b) percentage of 2015 land area at risk. (c) land volume above HAT. Calculations reflect the open coast dune system in its current (2016) position. (d) Changes in the elevation range suitable for coastal wetland vegetation in relation to sea-level rise. Each scenario assumes current topography and negligible water-slope effects that may arise from ground friction. Note different scales on Y axes in (d), and legend shared by (a) and (b).

3.3.3 Twin coast effects

The combined effects of sea-level rise acting on opposite shorelines are considered using the linked scenarios to simulate shoreline change with and without anthropogenic intervention. With a lack of intervention several pinch-points can be identified with implications for persistence of the landform. Under the 0.5 m SLR scenario a 2.5 km area of heightened flood risk develops on the backshore that overlaps with the area most vulnerable to dune system losses (chainage 3 – 5500). South of chainage 7000 similar effects are severe with approximately two-thirds of the land area being vulnerable without backshore defences (Fig. 3.5). With a 1m SLR these risks are exacerbated in some areas (e.g. between chainage 2 – 4000 and 5 – 7000) but to a lesser extent elsewhere, providing useful information for the timing of responses (Fig. 3.6).

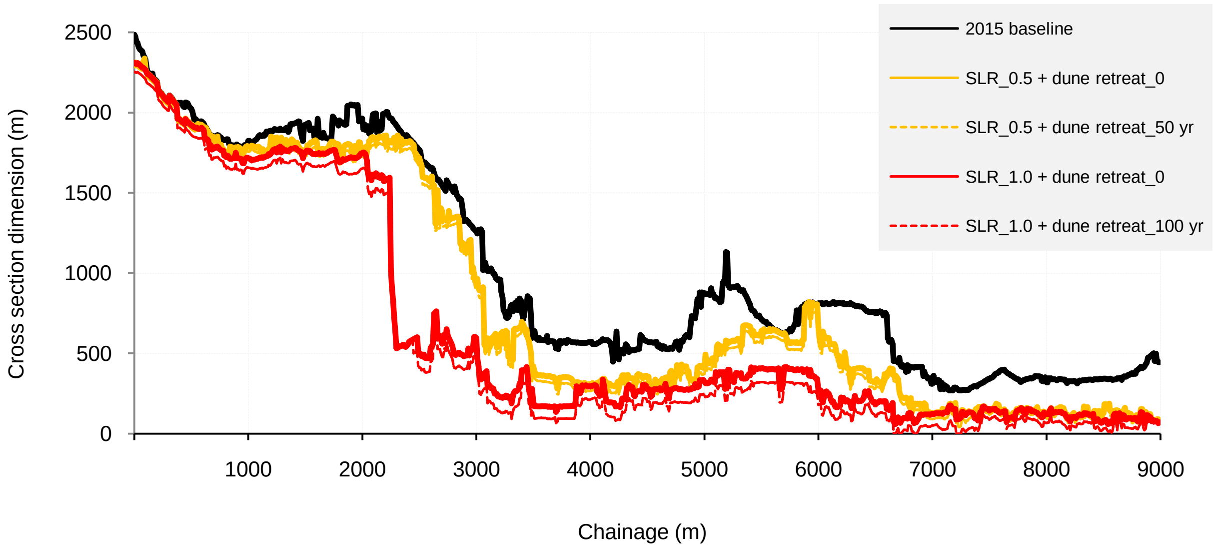

Fig. 3.6 Cross-section scenarios for the barrier-forming sandspit at the Avon Heathcote Estuary Ihutai, Christchurch under sea-level rise. Potential changes are simulated for both shorelines concurrently on 900 transects at 10 m spacing and reflect the distance between the highest astronomic tide (HAT) elevation on the estuarine backshore and position of the dune toe on the open ocean beach.

3.4 Discussion

3.4.1 Sea-level rise on a sandspit

Barrier translation studies have shown that there is no single universal model that accounts for their historical behaviour and similarly, that could reliably predict change under future sea-level rise. However, it is clear that landward translation can only result from sediment deposition in the backshore (Lorenzo-Trueba & Ashton 2014). There are essentially four mechanisms by which this may occur under conditions of sea-level rise: barrier rollover via overwash, sediment injection via inlet formation, aeolian dune transgression and backshore accretion from suspended sediments (Heward 1981). Sedimentary records have also shown that barrier translation within human timeframes (< 100 years) may not naturally occur. For example, Leatherman (1983) reported cases where both coasts of a barrier have been submerging over > 1000 years, implying a progressive narrowing of the landform. Despite this, modern-day engineering capabilities convey a strong potential for anthropogenic influence that marks a step-change for the consideration of future changes in relation to the past.

3.4.2 Socio-ecological aspects

A review of barrier translation mechanisms is illustrative of the potential effects of people on a peri-urban sandspit. Since defence against overwash from large storms and tsunami is desirable for hazard management (Kron 2013), this is an unlikely mechanism for nourishment of the backshore. Anthropogenic interventions will strive to prevent its occurrence over at least the short to medium term. Similarly, new inlet creation is unlikely to be accommodated within the current development pattern, which in this case features a near-continuous strip of beachfront residential property as is common elsewhere (Cooper & Pilkey 2012). Should this occur, the most likely breach point is at New Brighton village where there is a high density of commercial property. The inundation modelling shows evidence of a backshore scour basin in this location (Fig. 3.2) in addition to the current dune system gap. Although it is possible that breaches may have occurred here before, such events are now incompatible with the development trend.

The socio-ecological context also suggests the absence of a realistic mechanism for dune transgression, at least in the short to medium term. Previous experience with dune narrowing in the north of the study area included frequent episodes of sand loss from the backdune (Fig. 3.7a). This had a damaging effect on residential areas and posed a vehicular hazard on roads that necessitated regular physical removal after storm events (R. Chambers, pers. comm.). Although this history demonstrates the evolution of a sediment transport pathway towards the opposite shore that was curtailed by dune regrading, it also illustrates tensions between natural processes and the built environment. The only realistic transgression mechanisms involve the creation of dune migration pathways where the system is permitted to move inland towards the estuarine shore. This is very unlikely to achieve social acceptability though in theory could be facilitated at narrow pinch-points if combined with partial retreat strategies as a response to climate change.

In contrast to the above three mechanisms, there are fewer impediments to the encouragement of backshore accretion from suspended sediment sources. The socio-ecological context suggests that the goal of saltmarsh conservation could be readily supported provided that inundation hazard concerns are simultaneously addressed. Several contributing aspects suggest that the disaster recovery setting offers new opportunities for building climate change resilience. Firstly, despite the historical protection of saltmarsh ecosystems under statute, actual development patterns have had deleterious effects. This is evidenced by the fragmented pre-earthquake distribution of wetlands associated with extensive shoreline armouring and associated vegetation gaps (Fig. 3.7b). In response to subsidence some sites have retained intact saltmarsh ecosystems seaward of built defences (Fig. 3.7c), whereas elimination occurred at others contributing to gaps and erosion (Fig. 3.7d).

Fig. 3.7 Displacement of natural sandspit environments. (a) dune narrowing on the open coast beach at New Brighton. (b) armouring on the estuarine backshore with associated lack of intertidal vegetation. (c) example of recovering saltmarsh seaward of armouring in an area that experienced subsidence. (d) a nearby site where vegetation has not recovered indicating the exceedance of a local tipping point (photograph taken November 2019).

Second, and perhaps most importantly, the advent of earthquake-induced subsidence and government-led managed retreat may enable saltmarsh accretion processes to be re-started. The relaxation of oversteepened shore profiles associated with current defences provides an opportunity to move towards natural defences that encourage vegetation-mediated sediment sequestration. These can help maintain the current landform and potentially assist longer term barrier transgression (Kirwan et al. 2010; McKee et al. 2007). This differs from the strategy of reactivating floodplains behind existing defences as it creates the potential for marsh expansion seaward of their current positions and could potentially help prevent the break-up of recently drowned marsh with consequent loss of blue carbon (Chapter 2).

3.4.3 Challenges for dune conservation

Despite the opportunities for ecosystem-based adaptation on the estuarine shoreline we also identify a perilous situation for the conservation of dunes. The open coast scenarios indicate that over a third of the study area is in need of urgent attention. Risk reduction is required since the implications of dune loss include lower levels of coastal hazard protection. We highlight two aspects of risk management that are relevant to the design of adaptation pathways. First, the principle of risk accrual in both built and natural environments is exemplified by the coastal squeeze situation. Failure to address risk in its totality is likely to result in losses in the neglected dimension and has historically lead to beach and dune degradation due to trade-offs with other desirable land uses (Berry et al. 2013). Second, adaptation pathways must be attuned to, and specifically geared for, natural hazards and disaster recovery activities since deleterious effects are likely to be manifested in stochastic events (Cowell et al. 2006). These non-linear aspects suggest the need for proactive approaches in which key strategies are enabled prior to the occurrence of extreme events.

Specific challenges for dune conservation include the level of commitment to dune restoration following storm events in which foredunes become narrowed or lowered. Under natural conditions, landward transgression of the foredune is expected to occur via sand movement through low points following erosion events and a generally high capacity for self-recovery (Martinez et al. 2016). However the minimum space principle illustrated here indicates a threshold beyond which these processes become incompatible with current land-uses. Decisions made before impacts become critical are important to enabling a full consideration of potential solutions. Useful approaches may include retaining narrow sections of dune system to help maintain height and prevent sand loss from the backdune (Fig. 3.7a). Dune plant restoration can also be practiced forward of engineered defences, including after periodic erosion though with the expectation of diminishing returns on investment with rising sea levels (Orchard 2014).

Realistically, however, there are only two plausible pathways for saving dunes and their associated beaches under progressive sea-level rise, assuming the absence of any natural increase in sediment supply. These are anthropogenic nourishment and assisted dune migration. Despite the apparent synergy between dune conservation and coastal hazard protection, the long term prospects for dune persistence appear to hinge on the commitment to one of these approaches (Cooper & Pilkey 2012). In the present case, neither option has been contemplated in the discourse to date, suggesting that the current adaptation pathway is aligned with reliance on seawalls. Although, the natural sediment nourishment rate might accelerate in event of greater expulsion from nearby rivers, the reverse trend is also possible (Hicks 1993). Therefore, while the monitoring of accretion rates will remain important, it is essential to progress active strategies for climate change adaptation. Failure to do so is likely to limit the opportunities for incorporating nature-based solutions. Taking early action on sea-level rise affords the best chance of devising adaptation strategies that are effective across a range of values and are workable for the communities involved.

3.4.4 Limitations of scenario models

Although scenario models are useful tools it is important to emphasise that each scenario provides a representation of potential outcomes in consideration of a limited number of parameters that simulate a complex trajectory of changes (Yohe et al. 1999). Time varying aspects are implicitly incorporated though not directly simulated in their entirety. For example, inundation scenarios on the backshore reflect a lack of defences which is unikely. Instead, various configurations of defences are expected and the scenarios provide a touchstone to inform potential strategies. On the open coast shoreline, historical accretion rates are directly accounted for in the projected scenarios. However, since their future trajectory is also unknown, these simulations are also ‘plausible futures’ designed to assist decision-making.

3.4.5 Concluding remarks

This scenario-based assessment identifies major challenges for the protection of beach and backshore environments in peri-urban settings. The overriding conclusion is that these features are likely to become degraded over time unless society can effectively integrate conservation and natural hazard management in a changing climate. The perspective that some conservation objectives are better pursued elsewhere is valid though implies accepting the loss of natural assets such as local beaches (Schlacher et al. 2007). Since other intensive land uses are present along the coastal margin within a wide radius of the city, early and strategic planning will be essential to identify opportunities for this form of offsetting, and is recommended.

The conservation of peri-urban beaches and dunes presents a difficult situation due to the high intensity of existing land-uses and need for additional space (Berry et al. 2013). Although beach nourishment is a relatively common practice for similar situations worldwide (Nordstrom 2005) its adverse effects can also be substantial (Peterson & Bishop 2005). One of the best prospects in the present study lies on the estuarine shoreline where government land acquisition creates an opportunity to work with nature to reverse the plight of riparian wetlands. There is potential both to restore intertidal habitat and restart saltmarsh accretion. The major decisions required relate to the reconfiguration of shoreline defences to achieve a more natural solution.

In absence of a major open coast erosion event, the long-term viability of defending the backshore may prove to be a tipping point for perspectives on adaptation and resilience. Here, the temporal progression aspects of the scenario analyses can provide useful insights. They highlight the efficacy of maintaining a perimeter defence on a long narrow landform, as well as the concurrent challenge of draining ponding areas behind the line of such defences. The potential effect of groundwater connections on riparian water tables may also be considerable (Smith et al. 2016), creating a further consideration for the design of hydrological defences. These aspects also suggest that natural barrier translation processes are likely be required in the longer term to prevent eventual submergence. On both affected shorelines, naturally occurring ecosystems have the potential to modulate the effects of sea-level rise due to their influence on sediment budgets and sequestration. Although locally preferred approaches are likely to include a degree of additional fortification, consideration should be given to the role of nature-based solutions, especially over longer timeframes.

3.5 Acknowledgements

We thank Rodney Chambers, Jason Roberts, Pieter Borcherds, Justin Cope, Bruce Gabites, Philip Grove and Mark Parker. LiDAR datasets were provided by the Canterbury Regional Council and Christchurch City Council. This work was funded by the Ngāi Tahu Research Centre, University of Canterbury. Dune conservation research was supported by the Coastal Restoration Trust of New Zealand.

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PART 2

SALT WATER INTRUSION AND HABITAT SHIFTS

Part 2 of the thesis investigates the potential for salt water intrusion effects associated with bathymetric changes in the lower rivers and tidal lagoon basin. The issue of whitebait conservation was the specific focus of this work due to a previously reported relationship between the location of īnanga (*Galaxias maculatus*) spawning sites and the upstream limit of salt water intrusion in coastal waterways. This species makes up the bulk of the whitebait catch, a traditional open-access fishery of significant cultural and economic value. Key objectives for these studies were the detection and quantification of habitat shifts that may result from salinity changes, and the evaluation of new vulnerabilities and associated implications for disaster recovery activities and longer term climate change adaptation.

Chapters 4 and 5 describe the development of new methodologies and their application to test a hypothesis of habitat migration associated with salt water intrusion changes. Chapter 4 presents a comprehensive survey methodology to quantify spawning activity for spatiotemporal comparisons. Chapter 5 describes the novel use of artificial habitats as detection tools to help overcome field survey limitations in degraded environments where egg mortality can be high. Following a successful pilot study in the first year of research this technique was up-scaled as part of a collaborative project funded by the Department of Conservation and involving a wide range of participating organisations. Chapter 6 presents results from a two year study of spawning habitat dynamics using the above methodologies. Major habitat shifts were detected along with new patterns of vulnerability resulting from the new distribution of spawning habitat in relation to human land-uses. Subsequent work with stakeholders beginning 2017 has implemented adaptive management measures through spatial planning informed by the post-earthquake research. Chapter 7 investigates the effectiveness of conservation planning methods for the protection of spawning sites as is required by higher level policy. This study was conceived as a comparative evaluation of three contemporary approaches in relation to the observed habitat shifts and with regards to the likelihood of future changes driven by sea-level rise.

Whitebait find new sites to lay eggs post Canterbury earthquakes

Published: 10:48AM Monday March 16, 2015 Source: ONE News

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Whitebait may be the unsuspecting beneficiaries of the Canterbury earthquakes.

Whitebait eggs have been found in areas where none have previously been seen four years after the quakes devastated the lower reaches of the Heathcote River and its mouth at the Avon-Heathcote estuary.

Shane Orchard carrying out whitebait research work in Christchurch - Source: Supplied

Adult whitebait lay their eggs in streamside vegetation in the area where saltwater mixes with fresh river water, says University of Canterbury doctoral researcher Shane Orchard.

The PhD research investigates the vulnerability of coastal conservation areas and Mr Orchard is using earthquake effects to simulate the type of issues that might occur with climate change.

Whitebait have been in decline and management is an ongoing conservation issue.

Because of uplifting and liquefaction at the river mouth, the local pattern of saltwater intrusion is thought to have changed and large numbers of whitebait eggs have now been found in the lower Heathcote River, closer to the estuary than ever seen before.

"I am exploring the salinity environment in the lower reaches of the Christchurch rivers to get a better picture how this might affect whitebait spawning sites and ways they can be protected," Mr Orchard says.

Box 1 TVNZ ONE News coverage of early results from the whitebait research programme and outreach brochures produced in collaboration with the Department of Conservation in English and Te Reo.

Chapter 4

Census survey approach to quantifying īnanga spawning habitat for conservation and management

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Abstract

Here we describe a methodology for detecting and quantifying the spawning habitat of īnanga (*Galaxias maculatus*), a protected native fish species. Our approach is demonstrated with a survey of the Heathcote River Ōpāwaho following the Canterbury earthquakes that produced unexpected findings. The survey methodology detected spawning habitat over a 2.5 km reach, and the area occupied by spawning sites (75 m^2) was much larger than in previous records (ca. 21 m^2). Sites dominated by the invasive *Phalaris arundinacea* were found to support high egg numbers. Spawning has not previously been reported on this species and it is currently identified in the literature as a threat to spawning habitat. Considerable spatio-temporal variability was also detected in the location of spawning sites and pattern of egg production. Together these aspects illustrate the need for a comprehensive survey methodology to reliably quantify spawning habitat. The Heathcote Ōpāwaho example shows the utility of our census approach for achieving this, and supporting habitat conservation objectives.

Keywords

Conservation planning, habitat protection, waterway management, survey methodology, whitebait, *Galaxias maculatus*, New Zealand.

4.1 Introduction

Īnanga (*Galaxias maculatus*, Jenyns 1842) is a riparian spawning diadromous fish species that comprises most of New Zealand's whitebait fishery (McDowall 1984). However, īnanga are currently listed as 'at risk - declining' in the New Zealand Threat Classification System (Goodman et al. 2014). National policy under the Resource Management Act 1991 includes protection requirements for at risk species, and there are further statutory protections under the Conservation Act 1987. In practice, this creates a pronounced tension between conservation and the fishery value of the species. This has been recently highlighted in a range of policy contexts including concerns presented to the New Zealand Conservation Authority regarding the sustainability of the fishery (Goodman 2016).

The protection and enhancement of īnanga spawning habitat is an important focus for conservation as well as being a policy requirement. Globally, lowland riparian zones have been increasingly modified by urbanisation, impoundment, and floodplain development (Kennish 2002). These changes have contributed to the decline of species such as īnanga that rely on riparian habitat for critical aspects of their life cycle (Hickford et al. 2010; Mitchell 1994). Despite being a well-known species, there are few examples of field studies that have comprehensively quantified the location and extent of īnanga spawning habitat. This is essential information for managing threats through approaches such as spatial planning and the design of protected areas.

Obtaining improved spatial data is not straightforward because īnanga spawning habitat is strongly structured by salinity and water level, both of which are highly dynamic. As a consequence, field studies need to account for fluctuations in time and space. At the catchment scale, spawning sites are usually found close to the interface between saline and freshwater (Burnet 1965; Taylor 2002). At the river reach scale, the distribution is finely structured vertically with eggs being laid very close to the spring tide high-water level (Hickford & Schiel 2011a; Richardson & Taylor 2002). These factors vary considerably in accordance with tidal cycles and river discharge. There is also a defined peak in seasonal spawning activity that varies around the country (Taylor 2002) and potentially between years. All of these factors must be addressed simultaneously to quantify spawning habitat and prioritise areas for protection as part of an effective conservation strategy.

The objective of this paper is to describe a research approach that addresses these needs. First, we describe a census survey methodology for quantifying spawning habitat at catchment scale. Second, we demonstrate its application and utility for management using a survey of the Heathcote River Ōpāwaho as an example.

4.2 Methods

4.2.1 Survey approach

Previously reported trends in peak spawning months (Taylor 2002) were used to establish the survey period. On each month of the survey, the extent of potential spawning habitat was assessed using salinity data as a guide to determining its location in the catchment. Spawning was detected using direct searches of riparian vegetation and spawning habitat quantified as the Area of Occupancy (AOO) of eggs as observed. The remainder of this section describes the methodology used in applying this approach to a survey of the Heathcote Ōpāwaho over four months (Feb–May) in 2015 (Fig. 4.1).

The Heathcote Ōpāwaho is a spring-fed, lowland waterway with an average base flow of approximately 1 cumec (White et al. 2007). Riparian vegetation species in the study area included pasture grasses such as tall fescue (*Schedonorus arundinaceus*) and creeping bent (*Agrostis stolonifera*), together with herbs such as monkey musk (*Erythranthe guttata*), watercress (*Nasturtium officinale*) and mint (*Mentha* spp.). The invasive exotic reed canary grass (*Phalaris arundinacea*) was widespread throughout the study area in the īnanga spawning habitat elevation range. Infestations of *Glyceria maxima* were also present downstream. Indigenous plant species were generally scarce, but included rushes (*Juncus* spp.), occasional sedges (e.g., *Carex secta*) and harakeke (*Phormium tenax*). Oioi (*Apodasmia similis*) became more common near the downstream limit of the study area along with saltmarsh ribbonwood (*Plagianthus divaricatus*) and other saltmarsh species. Small stands of rāupo (*Typha orientalis*) and kuawa (*Schoenoplectus tabernaemontani*) were also present, generally at slightly lower elevations than īnanga spawning habitat and often emergent in the main channel.

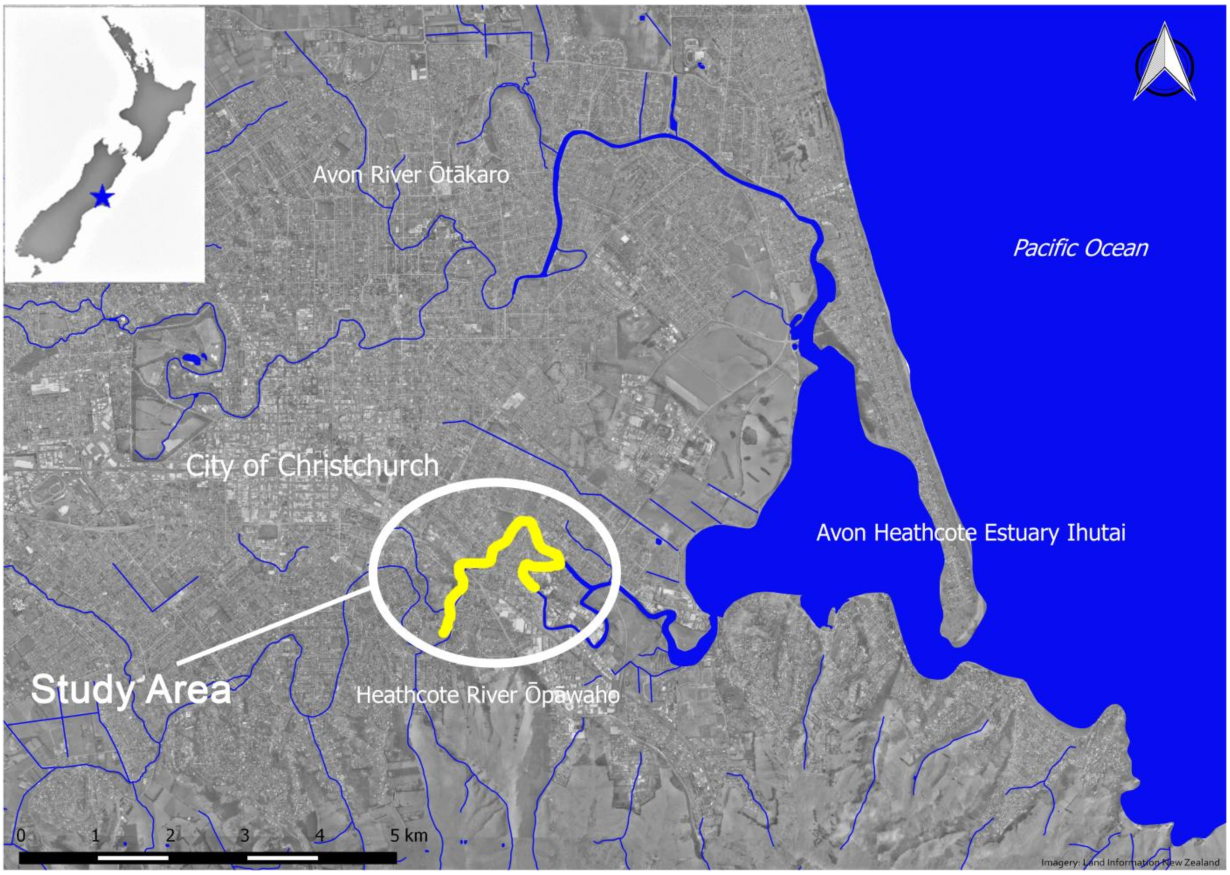

Fig. 4.1 Location of the study area in the Heathcote River Ōpāwaho catchment with survey extent shown in yellow.

4.2.2 Establishing the survey area

During the December 2014 and January 2015 new moon spring tide sequence, the upstream extent of saltwater intrusion was determined using a handheld salinity-meter (YSI Model 30, YSI Inc., USA). The progression of the flood tide was followed upstream as described in Richardson & Taylor (2002) but using kayaks. Measurements were taken 10 cm from the bottom and top of the water column at locations recorded with GPS. Three conductivity/temperature loggers (Odyssey, Dataflow Systems Ltd, NZ) were deployed over the spring tide sequence with each logger secured 10 cm above the riverbed. The loggers were deployed near the upstream limit of salt water, near the saltmarsh vegetation transition zone (downstream), and at an intermediate position. Logger data were used to verify the peak salinity intrusion days across the tidal cycle while interpreting the handheld meter data.

The upstream margin of the survey area was set 500 m upstream of the 0.2 ppt saline intrusion peak based on all measurements. This was interpreted as the upstream limit of seawater in the Heathcote Ōpāwaho because a 0.1 ppt measurement was consistently recorded for >2 km further upstream. The downstream extent was set at the transition to dominant saltmarsh vegetation which is considered to be unsuitable for spawning (Mitchell & Eldon 1991). In the Heathcote/Ōpāwaho this transition is subtle, but this is not the case in all rivers. These steps resulted in the selection of a ca. 4 km reach for surveying (Fig. 4.1).

4.2.3 Detection of spawning sites

To achieve coverage of the entire survey reach within time restrictions, areas of potential habitat were first identified in a subjective survey similar to the expert judgement approach of Hicks et al. (2010), but using set criteria to define areas of high, moderate and poor quality habitat (Table 4.1). All areas of moderate and high quality habitat were surveyed systematically to detect egg occurrence. For each 5 m length of riverbank, three egg searches were completed at random locations at least 1 m apart. This resulted in a large number of independent searches. Each search involved inspecting the stems/tillers and root mat of riparian vegetation along a transect that ran perpendicular to the high water mark. A 0.5 m wide and 2 m long swathe was inspected (1 m either side of the high water mark). Where īnanga eggs were found the search area was extended by at least 50 m either side of the last occurrence to confirm the full extent of spawning.

When quantifying spawning habitat, mortality of eggs between spawning and the survey is a potential confounding factor (Hickford & Schiel 2011b). To minimise this, a standardised schedule was used to improve comparability between months (Table 4.2). Surveys commenced six days after the peak tide sequence each month. Four to five days of surveying were required each month using a team of three people..

Table 4.1 Īnanga spawning habitat quality classes and criteria.

Class	Quality of habitat for supporting spawning	Expected egg mortality rate	Criteria
1	Poor	High	Vegetation cover <100% or Stem density <0.2cm^{-2}
2	Moderate	Moderate	Vegetation cover 100% Stem density >0.2cm^{-2} Aerial root mat depth <0.5cm
3	High	Low	Vegetation cover 100% Stem density >0.2cm^{-2} Aerial root mat depth >0.5cm

Table 4.2 Tidal cycle data and survey periods for the 2015 Heathcote River Ōpāwaho survey.

Month of spawning	Peak tidal cycle start	Peak tidal cycle end	Lunar phase	Peak tidal height† (m)	Survey period
February	22/2/2015	25/2/2015	new moon	2.6	March 3-6
March	20/3/2015	23/3/2015	new moon	2.6	March 29 - April 3
April	18/4/2015	20/4/2015	new moon	2.6	April 26-30
May	17/5/2015	19/5/2015	new moon	2.6	May 26-30

† predicted tide levels at Port of Lyttelton (Lat. 43° 36' S Long. 172° 43' E) in metres above Chart Datum (Source: LINZ).

4.2.4 Spatial data

Spawning ‘sites’ were defined as continuous or semi-continuous patches of eggs with dimensions defined by the pattern of occupancy. For all sites, the upstream and downstream extent of the site were established, coordinates recorded and length measured. The width of the egg band was measured at the centreline of the search transects falling within the site (minimum of three, adding additional transects where needed). Zero counts were included where they occurred (i.e., at discontinuous sites). Mean width of the egg band was calculated for each site. Area of occupancy (AOO) was calculated as length x mean width. For temporal analysis, spawning was considered to occur at the same ‘site’ where the AOO overlapped between months.

4.2.5 Spawning site productivity

Productivity of each site was assessed by direct eggs counts using a sub-sampling method adapted from Hickford & Schiel (2011a). For each transect, a 10 x 10 cm quadrat was placed in the centre of the egg band and all eggs within the quadrat counted. For atypical locations where the egg patch was not a narrow band (e.g., in low bank gradient areas), a 1 m^2 grid was overlaid on the site and a 10 x 10 cm quadrat sampled at the centre of the egg band within each grid. This improved sampling spread along the vertical bank profile. For quadrats with very high egg densities (> 200 $quadrat^{-1}$), egg numbers were estimated by further sub-sampling using five randomly located 2 x 2 cm quadrats and the average egg density of these sub-units used to estimate egg density for the larger 10 x 10 cm quadrat. Mean egg density was calculated from all 10 x 10 cm quadrats sampled within the site. Productivity was calculated as mean egg density x AOO.

4.3 Results

4.3.1 Habitat quality assessment

Within the study area, over 1.5 km of the true right bank was assessed as having high or moderate quality and was selected for egg surveys (Fig. 4.2). These areas were not contiguous being spread out over a ca. 4 km reach extending from above the known spawning site near Opawa Road to near the downstream limit of the survey area. Habitat quality was assessed as poor for a large proportion of the true left bank as a result of steep bank profiles (often vertical) and erosion in the vicinity of the spring tide high water mark with typically sparse vegetation. Only small sections supporting more established riparian vegetation were selected for egg surveys with the exception of a contiguous reach of ca. 500 m near the downstream limit of the study area (Fig. 4.2). The reach between Radley Street and the tidal barrage offered mostly poor habitat quality on both banks with tall trees limiting the amount and density of understorey vegetation in the riparian zone.

4.3.2 Egg surveys

Thirty-two spawning sites were identified over the four month survey period distributed over a reach of approximately 2.5 km with the upstream extent being Opawa Road (Fig. 4.2). All spawning was found in 'high' quality habitat validating the habitat quality classification (Table 4.1). The number of spawning sites detected varied considerably between months with nine found in February, 27 in March, 27 in April, and six in May (Fig. 4.2). Repeated use of the same area of riparian vegetation was observed at some sites, but other sites were used only once. Of the 27 sites found in March, 24 were re-used in April, but the AOO often differed. Six sites (19%) were used only once, 23 sites (69%) twice, and four sites (12%) three times, with no site used on all four months of the survey.

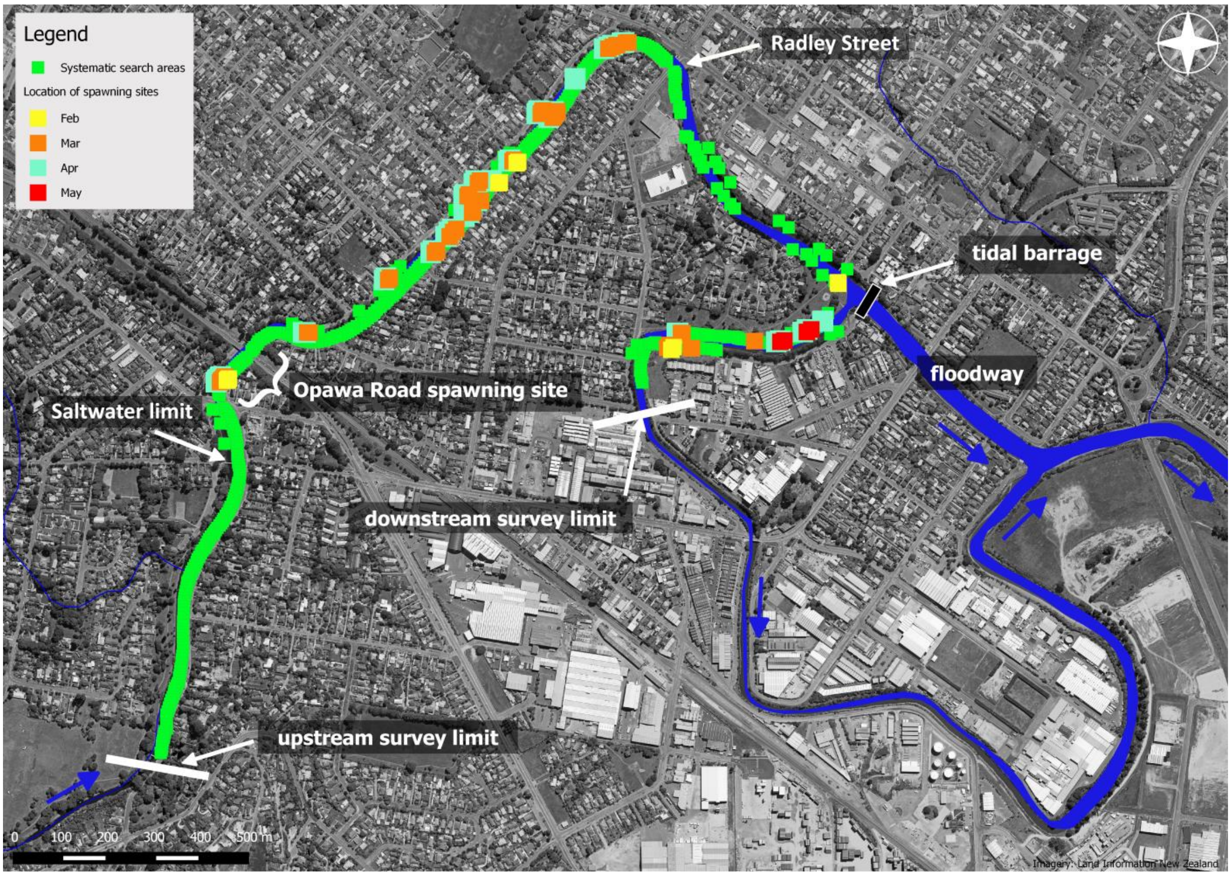

Fig. 4.2 Areas subject to systematic egg searches and locations of spawning sites detected in the Heathcote River Ōpāwaho catchment in each month (February – May) of the survey. The upstream limit of salt water (as measured) is also shown.

Area of occupancy showed a defined peak in March (57.7 m^2 ± 2.7; $\bar{x}$ ± SE) (Fig. 4.3a). Very little spawning was recorded in May (1.4 m^2 ± 0.2). The productivity peak was also in March with 3.3 x 10^6 eggs recorded (Fig. 4.3b), three times that of the next highest month (April). Total egg production over all months was 4.93 x 10^6 ± 0.44 x 10^6 eggs. Areas of occupancy varied considerably across the study area with large areas found a long distance apart (Fig. 4.4a). The largest area was 13.3 m^2 recorded in February and in general not all sites were used in all months. Using the maximum AOO recorded at each of the 32 sites over all months, the cumulative AOO of spawning habitat utilised at least once was 74.6 m^2. The spatial distribution of egg production (Fig. 4.4b) often reflected the AOO of spawning sites, but also showed evidence of highly variable egg densities. The highest mean density was 13.5 eggs cm^{-2} near Opawa Road, but >10 eggs cm^{-2} were also recorded at two other sites. The average mean egg density across all spawning sites and months was 2.9 eggs cm^{-2} ± 0.71.

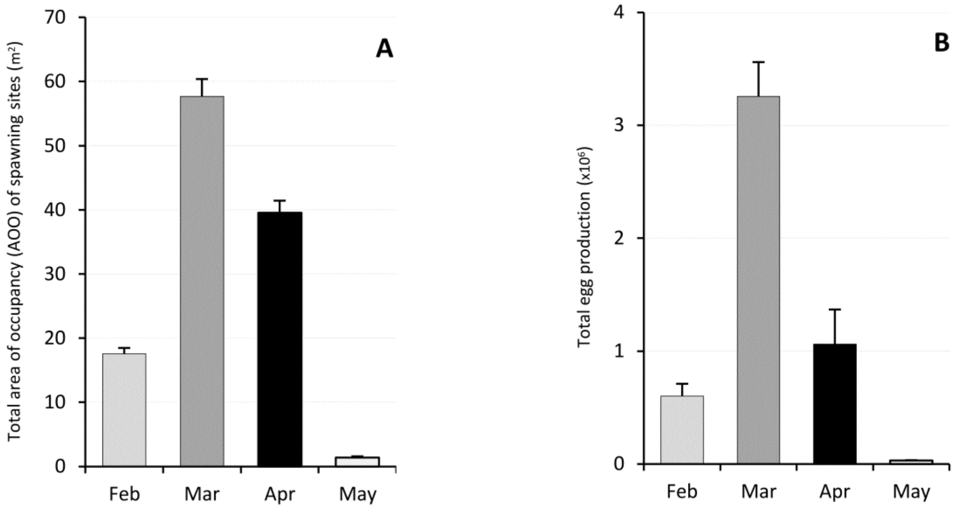

Fig. 4.3 Comparison of monthly totals for īnanga spawning site metrics in the Heathcote River Ōpāwaho catchment in 2015. (a) Total area of occupancy. (b) Total egg production. Each month represents the new moon spawning event. Error bars reflect standard errors of the mean for sub-sampled metrics at each site.

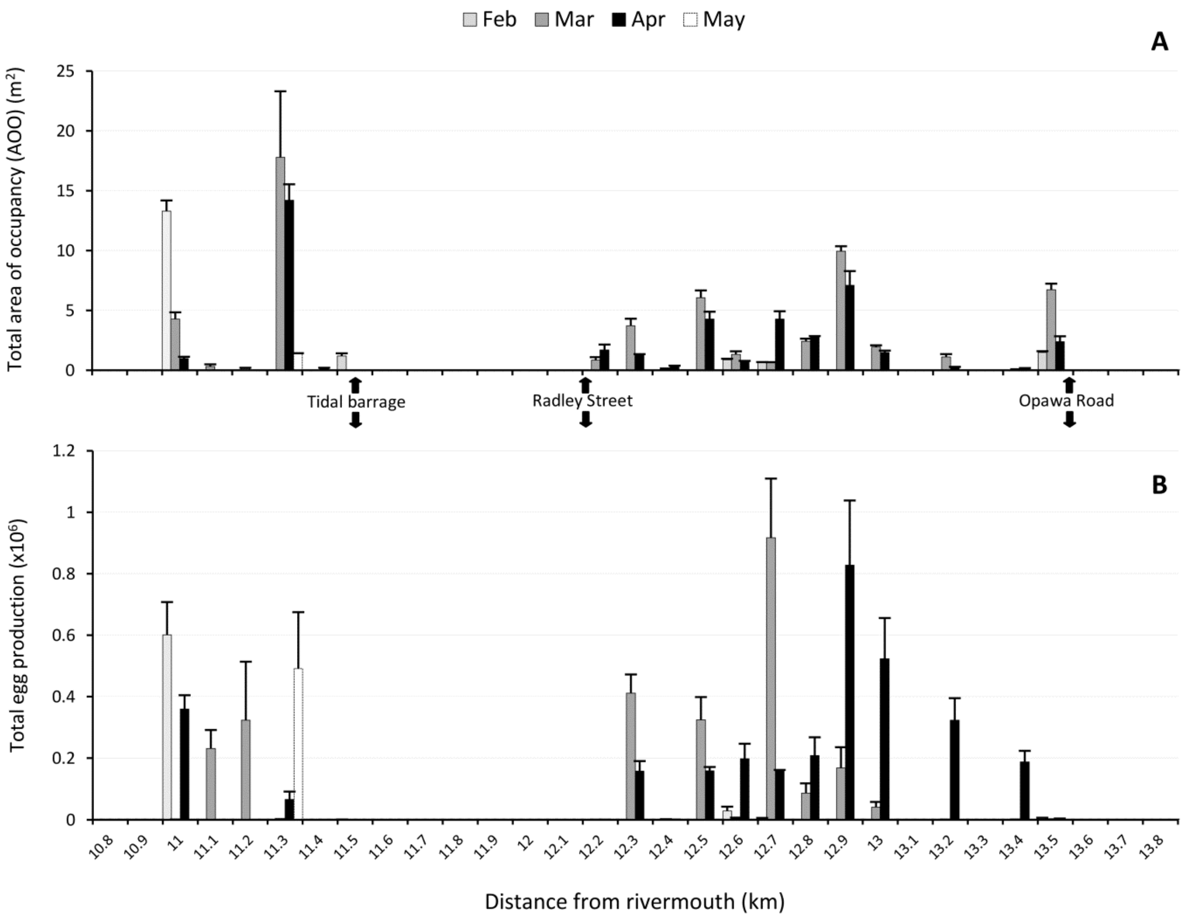

Fig. 4.4 Īnanga spawning site metrics binned into contiguous 100 m reaches across the study area for each of the four months surveyed. (a) Total area of occupancy. (b) Total egg production. Assignment to bins was based on the centrepoint of each spawning site. Error bars reflect standard errors of the mean for sub-sampled metrics at each site. The full extent of the survey reach has been truncated for clarity.

4.4 Discussion

Application of the census approach has improved knowledge of īnanga spawning habitat in several ways. In particular, the Heathcote Ōpāwaho example identified many previously unknown spawning sites in a well-studied catchment. This is notable because the literature has often reported the use of the same spawning sites year after year and this may simplify their protection and management (Richardson & Taylor 2002). However, there were also many more spawning sites than ever previously reported and they collectively occupied a much greater area. For example, the AOO has historically been in the order of 21 m^2 (Taylor 2002). In comparison, the area of spawning habitat used at least once in 2015 was 74.6 m^2. Potential reasons for the discrepancy between our results and previous estimates for the catchment include: i) exceptionally high numbers of spawning īnanga in 2015 when we undertook our survey, and/or ii) that previous survey efforts were not extensive or intensive enough to capture the full pattern. In addition, if the pre-earthquake spawning habitat had included degraded areas, the discovery of spawning sites in those areas would have been difficult due to greater egg mortality (Hickford & Schiel 2011b), especially in sporadic surveys. Bank slumping and other effects of the earthquakes may have increased the availability of high quality habitat and in turn made detection more likely (Orchard & Hickford 2016).

The literature on īnanga spawning site ecology suggests that most spawning is found close to the upstream limit of the saltwater wedge associated with high tides (Richardson & Taylor 2002). In an analysis of the National Īnanga Spawning Database, Taylor (2002) calculated a median distance of 107 m between the location of spawning sites and the saltwater wedge position from all available data where both had been recorded (n = 84), although a few studies showing anomalies to this trend were identified. In Australia, Hicks et al. (2010) reported spawning site distributions spanning several kilometres in some rivers although the saltwater limit position was not reported. In the Heathcote Ōpāwaho we consistently detected spawning habitat over 3 km downstream from the saltwater limit.

The census approach initially assumes that an extensive reach of the river may be capable of supporting īnanga spawning. Investigations are required to determine the actual reach to be surveyed. These focus on the saltmarsh vegetation transition (downstream) and the saltwater limit (upstream) on high tides. As was the case in our example, some judgement may be required in setting the downstream survey limit if there is a wide transition zone before salt marsh vegetation becomes dominant. However, due to the effect of river engineering works over many years, the Heathcote Ōpāwaho is a relatively complex example and the transition zone is typically more clear-cut in other rivers. In general, the downstream survey limit can be set to include all potential spawning habitat based on vegetation condition until at least such time as the peak spawning month has been identified. The pattern of spawning observed on peak months may assist decisions on whether to exclude some reaches in future surveys to reduce the survey effort.

Although a similar rationale may be applied to setting the upstream limit of the survey area, a pragmatic decision on the cut-off point will be required if the vegetation appears suitable for a long distance upstream. We chose a point 500 m upstream of the saltwater limit consistent with Richardson & Taylor (2002). Determining the saltwater limit is a crucial step and it is important that the field methodology is appropriate. We used data loggers to confirm the date(s) of maximum saltwater intrusion during the spring tide cycle on which spawning occurred. This provided a degree of validation for the salinity characterisation method using hand-held meters, which was only applied to one or two tides. Should a data logger be unavailable, we believe the salinity characterisation could also be reliably conducted by careful attention to the predicted peak tides and the possible influence of river flows when planning the field salinity measurements. The measurements should be taken on the largest tides and with low river flows to increase the chance that they will capture the maximum saltwater intrusion over the period when spawning may have occurred.

Once the survey limits were established, we used a subjective habitat assessment to identify areas for intensive searches similar to Hicks et al. (2010). However, a key aspect of our approach is the use of set criteria to guide the identification of potentially important areas. The Heathcote Ōpāwaho study shows that the criteria and thresholds chosen were useful in refining the search area with no evidence that spawning sites were missed through their application. This aspect helps improve the repeatability of the survey methodology in practice. Attention to temporal replication is a further defining feature of our census survey approach. Between-month comparisons in the Heathcote Ōpāwaho show that the distribution and number of spawning sites varied considerably despite the repeat use of some sites. AOO comparisons show major differences in the area occupied each month further highlighting the importance of temporal replication when quantifying spawning habitat. March spawning exceeded February by a factor of three in the number of sites and their combined AOO. As exemplified by the

March – April comparison, the AOO can vary markedly even with the same number of spawning sites being used. These findings suggest that the number of monthly surveys and seasonal timing aspects are critical considerations.

Where temporal trends are of interest there is a practical necessity to discover all of the spawning sites. For egg production this may be especially crucial due to the influence of varying egg densities. In this study the average egg densities per site varied by up to three orders of magnitude (i.e., from 10^2 to in excess of 10^5 eggs m^{-2}). There were many examples of relatively large sites with low egg densities whereas several small sites had very high egg densities and contributed greatly to total production in the catchment. Very few studies have quantified egg numbers, especially on a catchment basis, and we are not aware of any studies that have quantified spatiotemporal variation in patterns of egg production. However, the failure to detect high density sites is potentially a key issue. Our census survey methodology provides a comprehensive and replicable means to address spatiotemporal variation.

Lastly, the Heathcote Ōpāwaho study identified a high proportion of the total spawning at sites where reed canary grass was the dominant vegetation. Spawning has not previously been recorded on this invasive species and other studies have identified it as a threat and recommended removal (Taylor 2002; Taylor & Chapman 2007). The control of reed canary grass is certainly more complex in light of this finding. However, it may be possible to undertake a staged approach to reed canary grass removal guided by regular monitoring of egg production and habitat occupancy, and potentially associated with the restoration of native species suitable for īnanga spawning.

Taken together these findings suggest that the census survey methodology can make a useful contribution to habitat management. It may assist integration with other land uses by more precisely informing tools such as spatial planning. Monitoring applications include the evaluation of habitat protection initiatives and trends in egg production.

4.5 Conclusions

Our aim was to develop a survey methodology that can determine the full extent of īnanga spawning habitat within a river system. This information is lacking in most of New Zealand’s river systems and will improve the implementation of current conservation policy. Although the survey effort is considerable, the Heathcote Ōpāwaho example shows that it can be readily applied to a large study area. Limitations include the availability of sufficient personnel. There is a practical necessity for completion before the hatch date and egg mortality rates introduce an additional confounding factor, especially at degraded sites. Completing the survey within a standardised time frame relative to the date of spawning is important to improve the comparability of repeated surveys at the same study site. Practical strategies for larger sites include increasing the number of researchers and the potential use of volunteers. Results gained in this first application of the methodology confirm its utility and underlying rationale. The findings suggest that spatiotemporal variability in spawning combined with the timing and extent of surveys have the potential to exert large effects on the results, directly influencing the inference that can be draw from them.

4.6 Acknowledgements

Support for this study was provided by the Ngāi Tahu Research Centre, Engineering New Zealand / Water NZ Rivers Group and a New Zealand Ministry of Business, Innovation and Employment grant in conjunction with NIWA (C01X1002). We thank Jesse Burns, Jason Telford, Nicole Wehner, Eimear Egan and Judith Rikmanspoel for assisting with field surveys and the staff of the Waterways Centre for Freshwater Management and Marine Ecology Research Group for technical support.

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Chapter 5

Use of artificial habitats to detect spawning sites for the conservation of *Galaxias maculatus*, a riparian-spawning fish

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Abstract

Galaxias maculatus is a diadromous riparian-spawning fish that supports an important fishery. Eggs develop terrestrially as with several other teleost fishes. Spawning habitat occurs in specific locations near rivermouths and its protection is a conservation priority. However, quantifying the areas involved is hampered by high egg mortality rates on degraded waterway margins. We hypothesised that temporary artificial habitat would detect spawning in these situations, producing a useful indicator for riparian management. We installed arrays of straw bales as artificial habitat in two independent experiments over consecutive years and assessed their impact using pairwise Before-After-Control-Impact (BACI) experimental designs. We tested degraded gaps within the distribution of known spawning sites and also areas further upstream and downstream. Nine spawning occurrences were recorded on artificial habitats in 2015, 22 in 2016, and two on paired controls. Both experiments produced a significant effect for artificial habitats deployed in degraded gaps within the known spawning site distribution (p = 0.0001), providing evidence that these locations should be regarded as actual or potential spawning sites. In 2016 the technique also produced a significant effect downstream of known sites in one of the study catchments (p = 0.0375). We believe the use of artificial habitats as a detection tool could be useful in a variety of management contexts. These include identifying areas for protection, as confirmation of site suitability prior to making restoration investments, and in investigations to support the migration of habitats to new locations under climate change, since these may currently be degraded.

Keywords
Habitat protection, detection issues, degraded environments, fisheries management, whitebait, New Zealand.

5.1 Introduction

Galaxias maculatus (Jenyns 1842) is a diadromous fish species that is widely distributed in the Southern Hemisphere (Berra et al. 1996). The harvesting of juveniles during their upstream migration supports lucrative fisheries in several countries (Barbee et al. 2011). However, the species is in decline in New Zealand (Goodman et al. 2014) and South America (Encina-Montoya et al. 2011) prompting concern for the fishery and a range of conservation measures. A major contributing factor is the degradation of spawning habitat associated with land use change in lowland catchments (Hickford et al. 2010). Due to a specialised reproductive strategy the eggs develop in a terrestrial environment (McDowall & Charteris 2006). This is associated with delayed hatching to coincide with favourable conditions for larval survival (Martin 1999). Conversely, this increases vulnerability to anthropogenic threats (Hickford & Schiel 2011a). Other examples of terrestrial egg development in teleost fishes include Mummichog (*Fundulus heteroclitus*), Diamond killifish (*Aidinia xenica*), California grunion (*Leuresthes tenuis*), Gulf grunion (*L. sardine*), and Giant kōkupu (*Galaxias argenteus*) (Franklin et al. 2015; Martin 1999). Spawning occurs in riparian vegetation inundated during spring high tides and close to the upstream limit of salt water intrusion (Benzie 1968a). Spatiotemporal variance may result from interactions between salinity, water level, topography and the timing of fish movements and spawning events, making detection of the sites used more difficult (Orchard & Hickford 2018a). This is a significant issue for management and is usually attempted by direct observations of adult fish during spawning events, or searches of riparian vegetation for eggs (Taylor 2002). However, both of these approaches have conceptual and practical weaknesses.

Using observations of adult fish as an indication of spawning sites is problematic unless spawning is actually observed. Adult *G. maculatus* spend several days shoaling in pre-spawning aggregations and devote considerable energy to searching riparian vegetation before selecting a spawning site (Benzie 1968a). There may be a large area in which an aggregation is observed prior to spawning that is relatively imprecise compared to the sites actually used. In comparison, direct searches for eggs provide indisputable evidence that spawning has occurred. However, egg mortality between the date of spawning and the field survey reduces the effectiveness of this approach. Recent research has found that spawning may occur irrespective of whether the habitat is favourable for egg survival (Hickford & Schiel 2011a) and egg mortality can be extremely high (Hickford & Schiel 2011b). This suggests that egg mortality is a major management issue rather than the absence of spawning *per se*, and the same issue makes the detection of spawning sites more difficult. Once dead, the tiny eggs (approximately 1.2 mm Ø) dehydrate and rapidly disappear (Harzmeyer 2006). In degraded environments, surveys reliant on egg discovery may fail to detect spawning sites or underestimate the areas involved.

Further research has shown that artificial habitats such as installations of straw bales can provide favourable spawning sites and support high egg survival rates (Hickford & Schiel 2013). We predicted that temporary installations of artificial habitats could also be used as a detection tool in degraded areas. In particular, we expected that experimental arrays might produce a useful indicator for management to help identify unknown spawning locations or establish the full extent of potential spawning habitat on degraded riparian margins. To test this, we hypothesised that artificial habitats would detect spawning at locations where eggs had not been detected in previous field surveys due to either the influence of egg mortality on survey findings or avoidance of those sites by adult fish, since it is difficult to distinguish directly between the two.

In addition, a test of this hypothesis needed to account for the inability to provide a true control – a conundrum that is typical of before-after experiments (Stewart-Oaten et al. 1986). To address this we assessed the effect of installing straw bales on riverbanks using replicate treatment-control pairs in a modified Before-After-Control-Impact (BACI) experimental design (Underwood 1992). In this terminology, the experimental approach tests whether an intervention (e.g., the introduction of artificial habitats) has a statistically significant impact on a response variable of interest, such as the occurrence of eggs (Stewart-Oaten & Murtaugh 2003). We also considered the application of artificial habitats to two different management questions: whether spawning could be detected at previously unrecorded but currently degraded locations within the distribution of known spawning sites, and whether spawning could also be detected outside of the distribution of known sites where these areas also happened to be degraded. We use the term ‘spawning sites’ to refer to the geospatial position of eggs in the environment. The term ‘spawning habitat’ refers to the locations and physical conditions that support spawning. In this paper our objectives are to i) demonstrate the use of artificial habitats to overcome egg detection issues at degraded locations, and ii) discuss applications of this approach to support conservation planning in the wider management context.

5.2 Materials and Methods

5.2.1 Study areas and context

The study areas are located in the Avon Heathcote Estuary Ihutai catchment in the city of Christchurch on the east coast of New Zealand's South Island (Fig. 5.1). This is a barrier-enclosed tidal lagoon system associated with a dynamic sand spit at the southern end of Pegasus Bay (Kirk 1979). The Avon (Ōtākaro) and Heathcote (Ōpāwaho) rivers are spring-fed lowland waterways with average base flows of approximately 2 and 1 m^3s^{-1} respectively (White et al. 2007). Anzac Creek and an area of interconnected swamps and small lakes are tributaries of the Avon Ōtākaro. The total study area included a reach of 3.5 km in the Heathcote Ōpāwaho mainstem, 3.5 km in the Avon Ōtākaro mainstem, and an additional 0.7 km in the Anzac sub-catchment (Fig. 5.1).

Although many aspects of the main waterways are similar (White et al. 2007), the Heathcote Ōpāwaho catchment is modified by a tidal barrage that limits the upstream progression of the tide (Christchurch City Council 2016). In comparison to the Avon Ōtākaro, this reduces salinity in the reach used for spawning. Recent studies have found spawning sites in salinities of up to 18 ppt in the Avon Ōtākaro versus only 10 ppt in the Heathcote Ōpāwaho (Orchard & Measures 2017). These studies also showed that the downstream limit of spawning was governed by habitat quality characteristics rather than being a direct effect of salinity tolerance, and that upstream limit of spawning coincided with the upstream extent of salt water intrusion in the majority of spawning events (Orchard & Measures 2017). In other respects the waterway morphologies and riparian vegetation characteristics are similar and these are of particular interest to the technique described here. A considerable proportion of the waterway margins are degraded as a consequence of engineered armouring or vegetation clearance activities of several types (Orchard et al. 2018a). Each catchment provides an independent test of the technique which is generally applicable to any tidal river where there is a) a known or suspected fish population, and b) degraded spawning habitat.

The Ihutai catchment is among New Zealand's most intensively studied area for *G. maculatus* spawning habitat (Taylor 2002). However, a sequence of large earthquakes in 2010-2011 caused severe impacts in the east of the city that were pronounced in the vicinity of the estuary (Beavan et al. 2012). Ecological changes were expected at a variety of scales in association with altered hydrodynamics, ground and water levels, and substrates (Measures et al. 2011). This created an imperative for updated information on the distribution of *G. maculatus* spawning sites to inform earthquake recovery planning and longer term waterway management (Orchard & Hickford 2016, 2018b). The potential for a large scale shift to have occurred was initially investigated in 2015 using intensive surveys of riparian vegetation (Orchard & Hickford 2018a). This study expands on that work to address the aforementioned detection issues posed by degraded riparian vegetation.

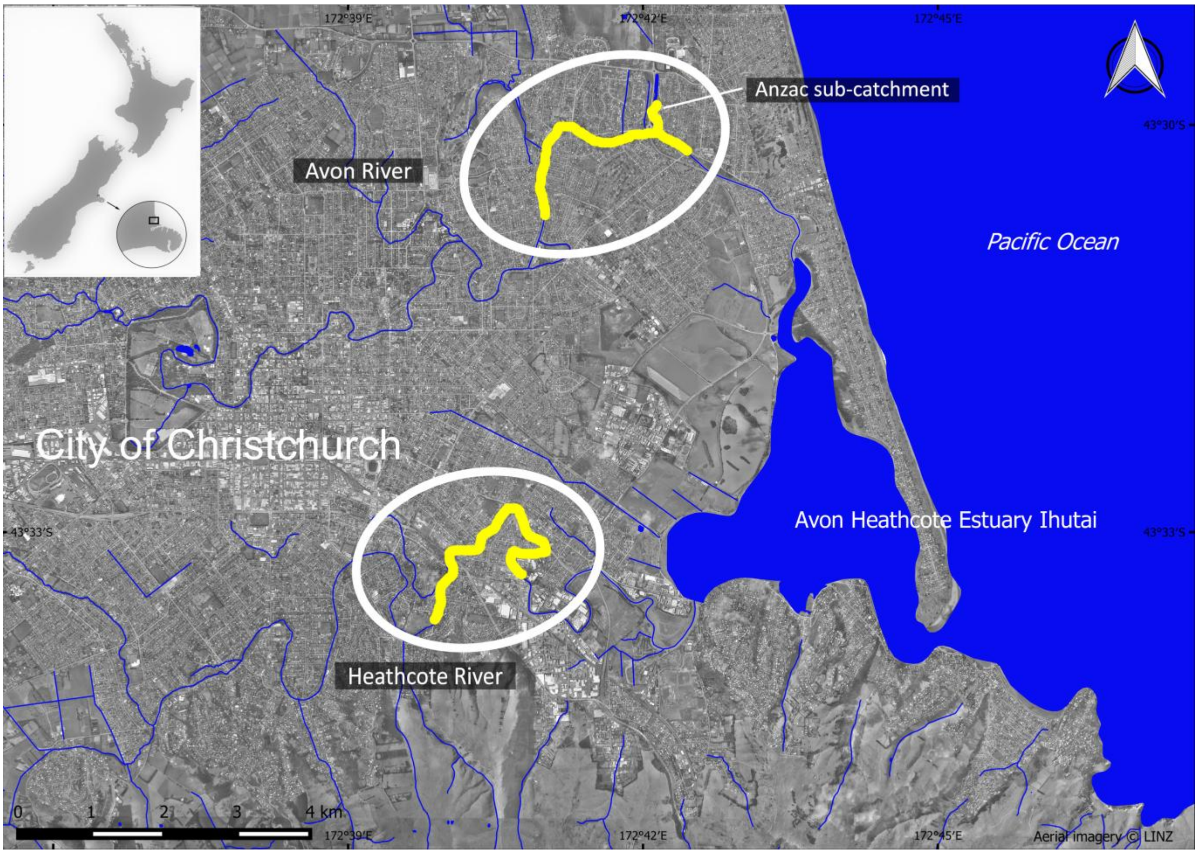

Fig. 5.1 Location of the city of Christchurch and Avon Heathcote Estuary Ihutai on the east coast of the South Island, New Zealand, showing the study areas in the Avon Ōtākaro and Heathcote Ōpāwaho catchments. Yellow lines indicate reaches that were searched for *G. maculatus* spawning sites after the 2010 – 2011 Canterbury earthquakes. The results of these surveys informed the artificial habitat experiment design.

5.2.2 Artificial habitats

Following Hickford & Schiel (2013) each artificial habitat installation was a set of straw bales oriented with the longest side perpendicular to the river bank and a 5 cm gap between the faces of adjacent bales. Spawning occurs in the microclimate provided by this gap. Each bale set was secured with a pair of steel stakes between which a loop of wire was fixed and driven down to pin the installation to the riverbank (Fig. 5.2). The bales were placed to straddle the high waterline on the spring tides of each month (Fig. 5.3).

Water levels were monitored over the spring tide sequences and the bales were repositioned accordingly. In the 2015 experiment, each installation used three bales. In 2016, sets of two bales were used to improve spatial coverage with the resources available. In both years, three replicates were installed at each site and placed at 10-20 m spacing along the bank depending on logistical constraints.

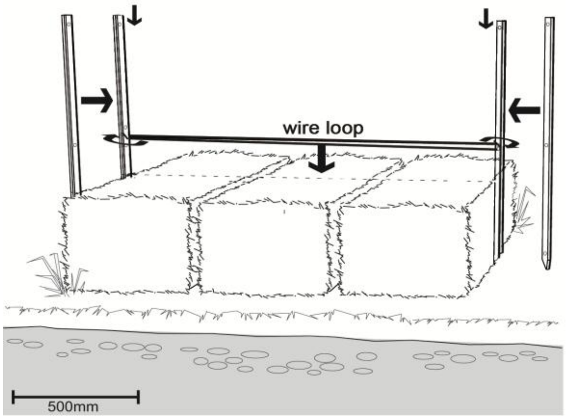

Fig. 5.2 Design of the artificial spawning habitats using straw bales. Bales are secured by a loop of wire attached to steel stakes which are driven down to pin bales to the ground. The long axis of the bales is oriented perpendicular to the riverbank. Adapted from Hickford & Schiel (2013).

5.2.3 Experimental design

We conducted two independent experiments in successive years. A pilot study was completed in 2015 using 24 installations followed by a larger-scale experiment (n = 90) in 2016 known as the Whaka Inaka – Causing Whitebait project. The design of both experiments was informed by prior field surveys that identified a) habitat quality for *G. maculatus* spawning across the entire study area, and b) the location of natural spawning sites based on the detection of eggs in riparian vegetation.

Fig. 5.3 An artificial habitat installation on a degraded section of riverbank in the Avon catchment in 2015.

Habitat quality was assessed by field survey following habitat quality classification schema (Supplementary Material, Table S5.1) with a minimum classification unit of 5 m riverbank length. Intensive egg surveys were completed for the entire study area over the four peak months of spawning activity in 2015 (Feb-May) and three months in 2016 (Feb-Apr), following Orchard & Hickford (2018a). Each survey commenced six days after the highest tides in the spring tide sequence each month (Supplementary Material, Table S5.2) and took 10–14 days to complete depending on the number of spawning sites found and weather conditions. In each experiment, three situations of interest for management were identified: degraded locations within the known distribution of naturally occurring spawning sites (i.e., degraded gaps), and both upstream and downstream of the known distributional limits of spawning. In each case 'degraded' was defined as areas of Class 1 or 2 ('poor' or 'moderate') habitat quality. The distribution of naturally occurring spawning sites was obtained from the results of egg surveys (Orchard et al. 2018a). Each spawning site was a continuous or semi-continuous patch of eggs for which the upstream and downstream limits were recorded as point coordinates. In all cases, natural spawning sites did not occur in areas that had been assessed as degraded.

Spatial data for spawning sites and habitat quality were visualised in QGIS v2.18 (QGIS Development Team 2017). For each waterway where spawning was found (Avon, Heathcote, and Anzac) three experimental reaches were defined according to the portion of the study area i) within, ii) upstream, and iii) downstream, of the known spawning site distribution (Figure 5.4). Throughout this paper these are referred to as the 'known distribution', 'upstream' and 'downstream' experimental reaches. Reach lengths for each were calculated on the centrelines of waterway channels digitised from 0.075 m resolution post-quake aerial photographs (LINZ 2016). Overlay analysis was used to delineate degraded locations with a minimum bank length of 10 m. These locations were the focus of the experimental design.

2015 Pilot study design

Egg survey results from February and March 2015 were used to delineate the experimental reaches for design of the pilot study, and artificial habitats were installed in April. A 'test site' was selected at random within each of the three experimental reaches in the Avon Ōtākaro and Heathcote Ōpāwaho mainstems, and for the 'known distribution' and 'upstream' reaches in the Anzac sub-catchment resulting in a total of eight test sites (Supplementary Material, Table S5.3). Following a replicated paired design (Underwood 1992), three replicate treatment-control pairs were established at the centre point of the degraded locations closest to each test site resulting in a total of 24 artificial habitat installations and their paired controls (Figure 5.4a & b). Each replicate comprised of a straw bale set and 1 m of vegetation either side (addressing a halo effect that is sometimes observed) giving a total bank length of 4 m which was considered to be the 'treatment', a paired 'control' area of the same dimensions, and a 2 m separation between treatment-control pairs. The footprint of each treatment-control pair was monitored in all months (Feb-May). Monthly monitoring took place 7-14 days after the peak spring tide and involved inspection of the stems and root mats of all vegetation in the test areas. For the artificial habitats this included all surfaces of the straw bale sets and the 1 m halo of adjacent vegetation, as described above. Eggs were recorded on a presence-absence basis.

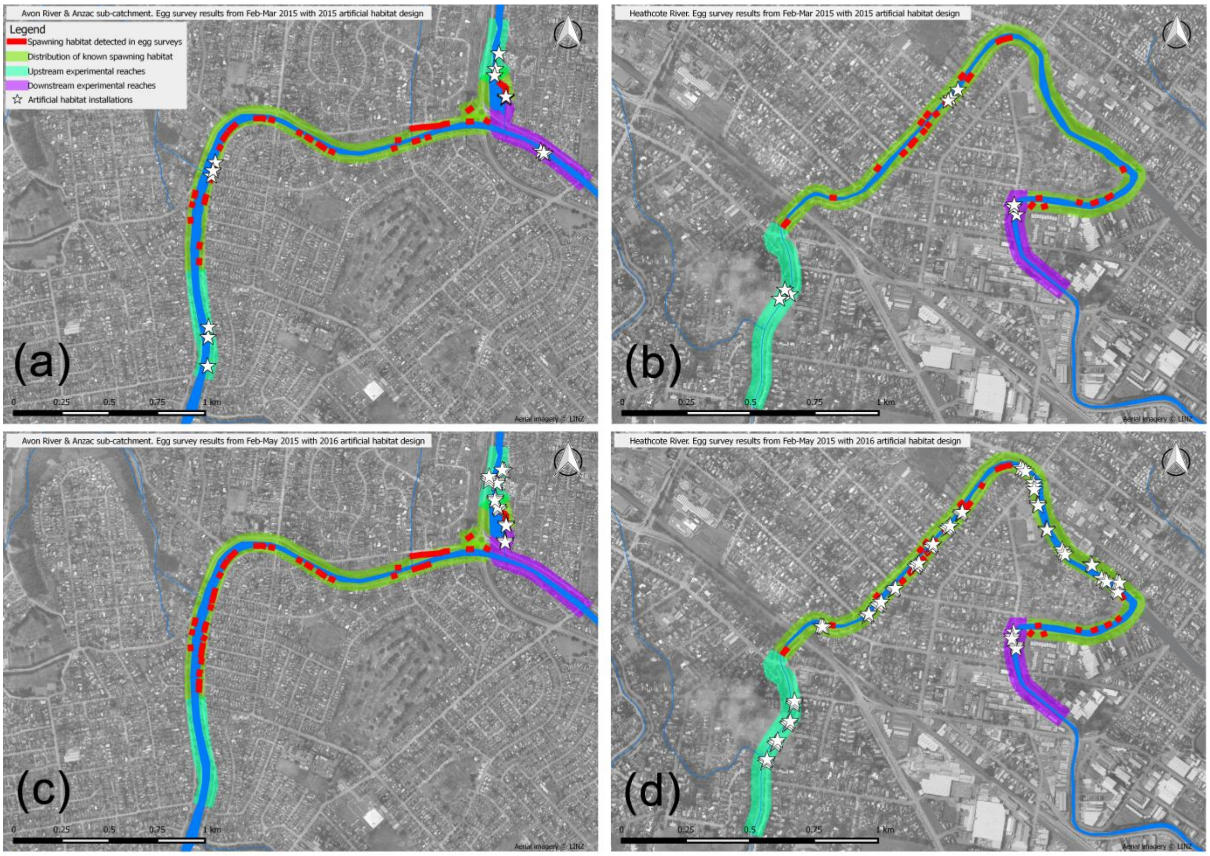

Fig. 5.4 Location of the experimental reaches as defined by the results of egg surveys completed prior to the installation of artificial habitats. Red lines show the spatial extent of all naturally occurring spawning sites detected in 2015 (n=85). Coloured areas show the three experimental reaches defined in relation to the location of the confirmed spawning sites. White stars are the locations of individual treatment-control replicates: (a) Avon mainstem and Anzac sub-catchment, 2015, (b) Heathcote mainstem, 2015, (c) Avon mainstem and Anzac sub-catchment, 2016, (d) Heathcote mainstem, 2016. Downstream is to the right in all maps.

2016 Whaka Inaka – Causing Whitebait project

In 2016, the Heathcote Ōpāwaho mainstem and Anzac sub-catchment of the Avon Ōtākaro were chosen for a larger scale experiment (Figure 5.4). Results from four months of 2015 egg surveys were used to inform the experimental design. However, this resulted in the same experimental reaches (as used in 2015) being defined in these study areas since the spawning sites found in April and May did not extend the known distribution beyond the sites recorded in February and March (Supplementary Material, Table S5.3). Compared to the pilot study, a major difference was the installation of artificial habitats in January to facilitate monitoring over the three peak months of spawning activity (February-April) as identified in the 2015 egg surveys (Orchard & Hickford 2016). The 2016 design also markedly increased sampling effort in each of the two study areas selected with a total of 21 artificial habitats installed in the Anzac sub-catchment and 69 in the Heathcote Ōpāwaho. A randomised block design (*sensu* Hurlbert 1984) was used to select the test sites with the blocks being 100 m contiguous reaches within the experimental reaches to be sampled. Due to limitations on materials and monthly monitoring resources only 23 of 35 blocks were sampled in the Heathcote Ōpāwaho together with seven in the Anzac study area for a total of 30 test sites (Supplementary Material, Table S5.3). Other details are as per the 2015 experiment except the dimensions of each artificial habitat installation (and its paired control) were smaller due to the use of two straw bales per installation instead of three.

5.2.4 Data analysis

Our analysis followed a pairwise BACI analytical framework (Stewart-Oaten et al. 1986; Underwood 1992). Before and after sampling was provided by the results of egg occurrence monitoring at each of the test sites in relation to the study period and timing of the treatment being tested (being the introduction of artificial habitats). Statistical analysis was carried out following a quasi-likelihood generalised linear model method (McDonald et al. 2000) in R v3.3.1 (R Core Team 2017) using the packages *lme4* (Bates et al. 2015) and *lsmeans* (Lenth 2016). The statistical model in R notation is:

$$Y = Period + Treatment + Period*Treatment + (1|Month) + (1|Catchment) + (1|Month*Catchment) + Period*Treatment(R)$$

where *Period* is the classification of time as Before or After, *Treatment* is the classification of sites as either Impact or Control, and their interaction *Period*Treatment* represents the BACI contrast of interest. Random effects for month, catchment and their interaction are modelled as (*1|Month*), (*1|Catchment*) and (*1|Month*Catchment*), and *Period*Treatment(R)* represents the residual variance.

Analysis was completed on the raw data with the response variable being the count of egg occurrences for each installation together with its control within each Period. All sites were measured in each month of the experiment. The fixed effects are Treatment, Period and their interaction (which is the BACI contrast of interest). The random effects were Catchment and Month. For models specific to one catchment the Catchment term was replaced with Site. Visual inspection of residual plots did not reveal obvious deviations from homoscedasticity or normality. P-values were obtained by likelihood ratio tests in *lsmeans* using the full model containing the effect of interest (Period*Treatment) against the model with this effect dropped following Bolker et al. (2009).

5.3 Results

5.3.1 Spawning occurrences on artificial habitats

Eggs were recorded on artificial habitats in both years (Table 5.1). Nine occurrences were recorded from a potential maximum of 48 (installation-months) in 2015, and 22 from a potential maximum of 270 in 2016. The highest number of occurrences per month was recorded in March 2016 consistent with the previously reported timing of peak spawning activity (Orchard & Hickford 2016). Within the known spawning site distribution the highest monthly 'strike rate' (egg occurrences per installation) was 0.56 in 2015, and 0.24 in 2016 with means across all installations per year of 0.44 and 0.11 respectively (Table 5.1). In the upstream experimental reaches only one occurrence was recorded (Anzac sub-catchment, April 2015). None were recorded in any of the upstream experimental reaches in 2016 despite greater sampling effort. In the downstream experimental reaches no occurrences were recorded in the 2015, but there were five in 2016 (Table 5.1). These comprised of three in the Heathcote Ōpāwaho and two in the Anzac study areas. This represented a mean strike rate of 0.19 and was higher than that (0.11) recorded on artificial habitats within the known distribution that year (Table 5.1).

Table 5.1 Summary of spawning occurrences on artificial habitats.

Experimental reach		**Spawning occurrences / month**			**Total**	**Mean occurrence / month**
2015	***n***	**Month 1 (April)**	**Month 2 (May)**			
Known distribution	9	5	3		8	0.44
Upstream	9	1	0		1	0.06
Downstream	6	0	0		0	0
2016	***n***	**Month 1 (February)**	**Month 2 (March)**	**Month 3 (April)**		
Known distribution	54	0	5	13	18	0.11
Upstream	27	0	0	0	0	0
Downstream	9	0	2	3	5	0.19

5.3.2 BACI analysis

The control site results add important information for the assessment of artificial habitats as an indicator of previously undetected spawning. In both experiments there were no eggs recorded on any of the controls for test sites within the known spawning distribution (Fig. 5.5). However, there were a total of eight spawning occurrences on artificial habitats in 2015 (Fig. 5.5a & b), and 18 in 2016 (Fig. 5.5c & d) in these locations. A single upstream occurrence was recorded in the Anzac sub-catchment in 2015 that was also associated with a negative result on the control (Fig. 5.5a). In the downstream experimental reaches there were three spawning occurrences recorded in the Heathcote Ōpāwaho in 2016 and again none on the controls (Fig. 5.5d). Over the same period in the Anzac sub-catchment two occurrences were recorded on artificial habitats but in both cases there were concurrent occurrences recorded on the paired controls (Fig. 5.5c).

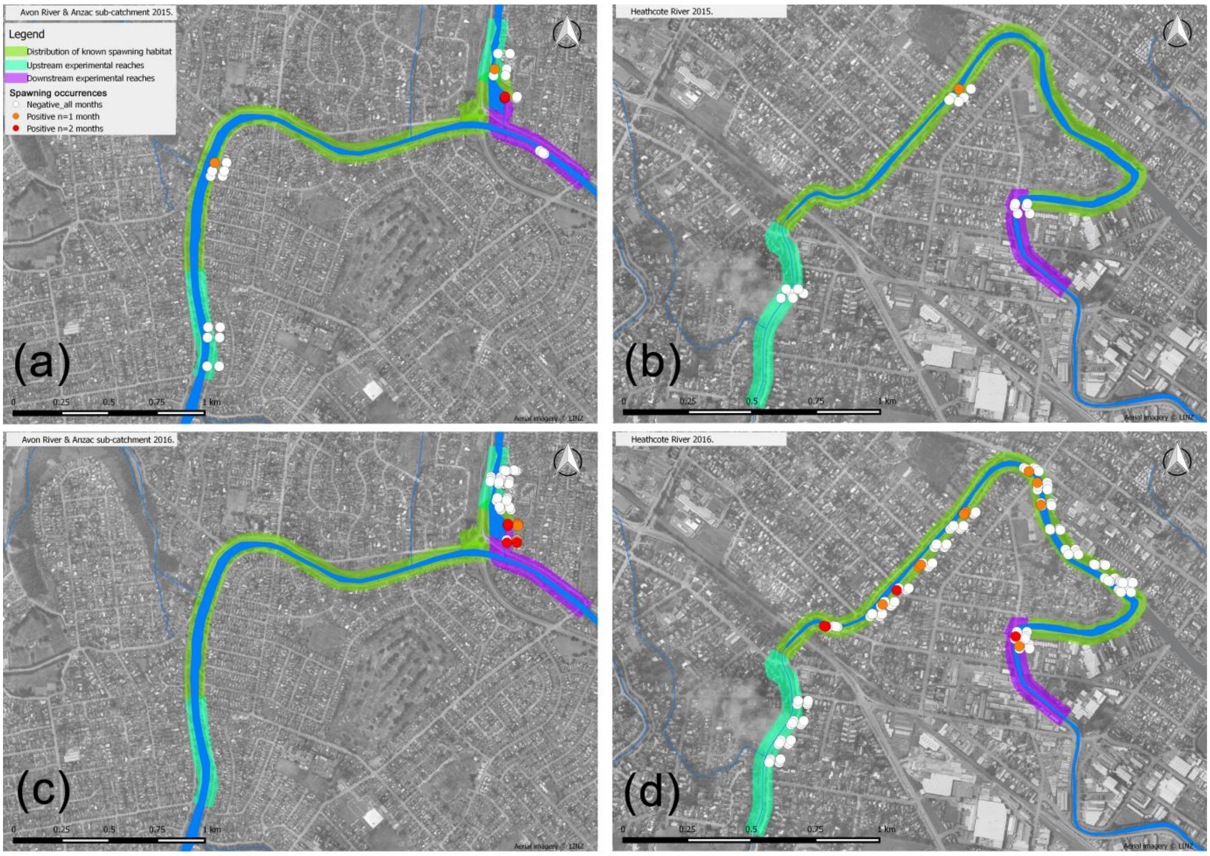

Fig. 5.5 Number of egg occurrences recorded for each treatment-control pair in the two independent artificial habitat experiments. (a) Avon mainstem and Anzac sub-catchment, 2015, (b) Heathcote mainstem, 2015, (c) Avon mainstem and Anzac sub-catchment, 2016, (d) Heathcote mainstem, 2016. Downstream is to the right in all maps. At each location the left hand dot shows treatment results, the right hand dot is the control.

The BACI analysis confirms the significance of these trends (Table 5.2). The artificial habitats were found to have a significant effect (p = 0.0001) on the detection of egg occurrences in both years when used to test degraded locations from within the known spawning distribution (Table 5.2a & b). In 2015, the artificial habitats did not produce a significant effect (p < 0.05) when used to test degraded locations outside of the known distribution, despite that eggs were recorded on one installation (Table 5.2a). In 2016, the artificial habitats produced a significant effect in the Heathcote Ōpāwaho downstream reach (p = 0.0375, Table 5.2b). In the Anzac Creek downstream reach, the artificial habitats did not produce a significant effect despite egg occurrences being recorded (Table 5.2b). This reflects the control results and was associated with a change in vegetation condition resulting from the recovery of riparian grasses in a previously disturbed area during the summer months.

Table 5.2 Selected BACI analysis results. Means are those predicted by the model for the impact + after (artificial habitat) versus control + after (control site) comparisons. The BACI estimate (also known as the 'BACI contrast') is the modelled difference in the means attributable to the treatment (being introduction of the artificial habitats). The χ^2 statistic is calculated on the likelihood ratio test for fitted vs. null models.

(a) 2015 Pilot study

Catchment	Experimental Reach	Model †	lsmean^{ImpactAfter}	SE	lsmean^{ControlAfter}	SE	χ^2	*p* (> F) #	BACI estimate ‡	lower CI	upper CI
All	Known distribution	Period + Treatment + Period*Treatment + (1 | Month) + (1 | Catchment) + (1 | Month*Catchment)	0.444	0.088	0.000	0.088	16.096	**0.0001**	**0.444**	**0.234**	**0.655**
All	Upstream + Downstream	Period + Treatment + Period*Treatment + (1 | Month) + (1 | Catchment) + (1 | Month*Catchment)	0.038	0.020	0.004	0.020	1.036	0.3087	0.033	-0.031	0.098
All	Upstream	Period + Treatment + Period*Treatment + (1 | Month) + (1 | Catchment) + (1 | Month*Catchment))	0.056	0.028	0.000	0.028	1.051	0.3053	0.056	-0.056	0.167
Anzac	Upstream (Data=one month 'After')	Period + Treatment + Period*Treatment + (1 | Month) + (1 | Site) + (1 | Month*Site)	0.333	0.126	0.000	0.126	2.775	0.0958	0.333	-0.138	0.805
Anzac	Upstream (Data=two months 'After')	Period + Treatment + Period*Treatment + (1 | Month) + (1 | Site) + (1 | Month*Site)	0.167	0.083	0.000	0.083	1.171	0.2792	0.167	-0.183	0.517

† lmer() model fit in R notation

bold = significant at p < 0.05

‡ estimated BACI contrast from model

(b) 2016 Whaka Inaka – Causing Whitebait project

Catchment	Experimental Reach	Model †	lsmean^{ImpactAfter}	SE	lsmean^{ControlAfter}	SE	χ^2	*p* (> F) #	BACI estimate ‡	lower CI	upper CI
All	Known distribution	Period + Treatment + Period*Treatment + (1 | Month) + (1 | Catchment) + (1 | Month*Catchment)	0.178	0.087	0.079	0.087	**14.879**	**0.0001**	**0.099**	**0.049**	**0.149**
All	Upstream + Downstream	Period + Treatment + Period*Treatment + (1 | Month) + (1 | Catchment) + (1 | Month*Catchment)	0.046	0.015	0.019	0.015	1.421	0.2333	0.028	-0.018	0.074
All	Downstream	Period + Treatment + Period*Treatment + (1 | Month) + (1 | Catchment) + (1 | Month*Catchment)	0.185	0.061	0.074	0.061	1.593	0.2069	0.111	-0.065	0.287
Heathcote	Downstream	Period + Treatment + Period*Treatment + (1 | Month) + (1 | Site) + (1 | Month*Site)	0.074	0.019	0.000	0.019	**4.330**	**0.0375**	**0.074**	**0.004**	**0.144**
Anzac	Downstream	Period + Treatment + Period*Treatment + (1 | Month)	0.222	0.128	0.333	0.128	0.299	0.5848	-0.111	-0.541	0.319

† *lmer()* model fit in R notation

bold = significant at *p* < 0.05

‡ estimated BACI contrast from model

5.4 Discussion

5.4.1 Artificial habitats as a detection tool

Results from both experiments showed that artificial habitats could detect *G. maculatus* spawning sites on degraded riparian margins at locations where spawning had previously not been recorded. In relation to our hypothesis, some of these may have been spawning sites that were previously undetected for reasons such as high egg mortality, while others may have been 'potential' spawning sites that had previously been avoided by adult fish but became favourable due to the provision of better conditions. Regardless of which of these situations the results reflect, they provide useful information for management in the form of irrefutable evidence that spawning may occur at those locations. Because the physical intervention provided by artificial habitats mimics a structural improvement in the riverbank vegetation (Hickford & Schiel 2013), sites at which spawning was recorded could be readily improved through ecological restoration techniques.

Our results demonstrate that artificial habitats can be usefully applied as a detection tool to help identify areas for protection or restoration. In many cases, cessation of vegetation clearance and other disturbances is sufficient to address the problem of high egg mortality (Orchard et al. 2018a), suggesting that protection is more fundamentally important than active restoration. As such, the technique could help overcome the apparent gap between conservation policy objectives and implementation in practice that has been exacerbated by a lack of spawning site information. In turn, this will contribute to solving the wider conservation issues concerning *G. maculatus* decline and maintenance of a sustainable fishery. These findings extend the literature on the use of artificial habitat methodologies to identify habitat and occupancy patterns characterised by detection challenges. Examples we are aware of include the monitoring of flying (Southwood 1978) and flightless (Bowie et al. 2014) insects, and cryptic lizard species in forests (Nordberg & Schwarzkopf 2015) and shrublands (Lettink et al. 2011).

5.4.2 Do artificial habitats actually improve detection?

In our example the management question concerned the detection of degraded portions of the habitat distribution where spawning was more vulnerable to human pressures yet could be readily restored. This required some form of experimental control to monitor the trajectory of locations identified as 'degraded' with and without the introduction of the artificial habitats.

Although it is not possible to directly measure whether responses at the treatment site would have remained similar if the treatment had not been applied (Bulleri et al. 2008), the control data functions as a body of evidence that assists this estimation and practical interpretation (McDonald et al., 2000). However, if the objective was simply to test for the presence of spawning using artificial habitats as an alternative to direct searches of riparian vegetation, then a simple before-after monitoring design could have been used (Stewart-Oaten & Bence 2001).

Use of a BACI design allowed us to specifically test whether the artificial habitats had a significant effect on detection (Stewart-Oaten et al. 1986). This was provided by a pairwise sampling strategy with the arrays of artificial habitats treated as the intervention (the hypothetical impact) and analysed as a fixed effect. Although a paired experimental design strengthens the weight of evidence (Underwood 1994), the inference gained remains subject to the assumption that the trajectories of responses will be exactly parallel between treatment and control units (Murtaugh 2002). The choice of control is also critical to address the counterfactual of interest (Stewart-Oaten 2008) and may include asymmetric designs, for example with a greater number of control units than treatments (Underwood 1994). Although in our case the design was symmetrical, the experimental controls produced useful information for management. For example, in the Anzac downstream reach in 2016, the control results showed that riparian conditions did not remain degraded. Unanticipated recovery of vegetation was observed to occur over the month of March leading to a situation where a previously degraded location had recovered and spawning was detected.

5.4.3 Limitations in relation to experimental design

In the selected BACI models presented (Table 5.2) the analysis showed three independent situations where the artificial habitat technique significantly increased the detection of eggs. These were in degraded gaps within the known distribution in both years for the pooled data across all study areas (Table 5.2a & b), and downstream of the known distribution in the Heathcote Ōpāwaho in 2016 (Table 5.2b). It is also important to note that the strike rate of positive occurrences was highly variable in space and time across the study area. This was likely influenced by the location of test sites in relation to the distribution of the adult fish populations on spawning tides. However, there were few instances where more than one of the three replicates per site received concurrent positive occurrences. Together, these results suggest that it would be unwise to rely on a small number of test sites as a detection measure and that replication was important.

Other limitations of our technique, with a bearing on detectability, included the positioning of artificial habitats in relation to the waterline during spawning tides and potentially other nuances affecting their discoverability by adult fish, as well as observer error in the form of false negatives during monitoring. A further complicating factor could arise from flood events experienced within the monitoring period since larval development can occur in as little two weeks (Benzie, 1968b) and hatching will occur if the eggs are inundated after that time (Benzie, 1968a). The same situation could arise in regions where subsequent spring tides are of similar amplitude. As a result, hatching (and in some cases spawning) can occur on a two weekly cycle (Taylor, 2002) and a monitoring regime to match that period would be required. Fortunately, the pattern of spring-neap tidal variability in our study area is characterised by one tidal sequence of greater amplitude per month (Walters et al., 2001). This is associated with a monthly spawning event and allowed ample time for monitoring. In the case of *G. maculatus*, the relationship between spawning and tidal cycles also simplifies decisions on sampling frequency. For applying the technique to other species and life stages, appropriate means of addressing temporal variance are likely to take different forms.

5.4.4 Application to management questions

Accounting for spatiotemporal variance in the detectability of response variables is a particular concern for monitoring, and BACI designs are no exception (Russell et al. 2015). The novel use of artificial habitats as a detection tool offers a benefit in our case due to the installations supporting very high egg survival rates. This goes some way towards overcoming the problem that may arise if effects such as gradients act unevenly on experimental units over time (Hulbert, 1984) and the data collection process cannot be simultaneous. This also offers a practical advantage in providing flexibility for the researcher as to the exact timing of the field measurements.

Temporal aspects of the specific management question may also be of interest. An example is found in our pilot study where a single positive occurrence was recorded in the Anzac upstream reach at the Anzac site. Treated in isolation, this observation provided irrefutable evidence that spawning could occur in a degraded area beyond the known extent of the habitat distribution. Although BACI analysis may be applied to sub-units of a larger study design (Underwood & Chapman 2003), inferences drawn from a single month and small number of sites may be of limited value. For example, the result might reflect a stochastic effect such as a salinity spike that happened to occur in that month, but which is not otherwise typical. Consequently, we applied the analysis to results from all months of the pilot study since this was a better match for the management question of interest. No positive occurrences were recorded in the following month and the overall BACI effect was small and not significant at $p < 0.05$. The same area was sampled using the technique in 2016 and again no positive occurrences were recorded.

The larger scale 2016 experiment was specifically geared towards key questions for management. In this case a randomised block design was used to guide the allocation of sampling effort across a relatively extensive study area. This required a large experimental design to cover the degraded areas of interest, but results were immediately useful for management. For example, in addition to several degraded gaps that were identified as actual or potential spawning sites there was a contiguous reach in the lower Heathcote Ōpāwaho where the artificial habitats failed to detect egg occurrences (Figure 5.5d). This is a neglected part of the river where competing demands on riparian management are considerably less than in other locations creating a favourable setting for restoration works. However, the provision of isolated patches of favourable habitat did not attract spawning there despite lying within the known spawning site distribution. This suggests it would be unwise to attempt habitat restoration in this area until the effect can be explained. Adult fish may avoid this area which is mostly shaded by tall exotic trees. If enhancement for *G. maculatus* is desired, the best strategy may involve extending the areas of currently favourable habitat at either end of this reach until the potential behavioural or discoverability issues are better known.

We believe the use of artificial habitats as a detection tool could be useful in a variety of management contexts. These include identifying areas for protection to address conservation objectives where detectability is problematic. As demonstrated in this case, spawning occurrences in artificial habitats may influence the results of investigations for the detection of spawning sites. Despite the absence of previous spawning observations, these degraded locations may then be regarded as spawning habitat and this attracts statutory protection under national legislation. Identifying degraded habitat that could be readily improved is a related and practical application. In our case the areas concerned were degraded by anthropogenic activities and would likely recover naturally if these disturbances were controlled. In other contexts where active restoration is required, artificial habitat deployment could be a useful means to confirm site suitability prior to making restoration investments. Investigations using this technique are also well suited to supporting the migration of critical habitats to new locations where factors such as climate change are driving range contraction (Hovick et al. 2016). This is a topic of increasing need (Burrows et al. 2014) that requires compensatory habitat expansion into areas that may currently be degraded.

5.5 Acknowledgements

Particular thanks to EOS Ecology, Te Rūnanga o Ngāi Tahu and Conservation Volunteers NZ and to S. McMurtrie, Te Marino Lenihan, K. Brennan and T. Moore. Thanks to J. Burns, R. Taylor, E. Egan, S. Gerrity, J. Rikmanspoel, D. Keenan, J. Telford, N. Wehner, L. Vigoya, C. Macintyre, L. Clark, G. Barbour and G. Burrell for fieldwork. This research was support by funding from Ngāi Tahu Research Centre and Brian Mason Scientific & Technical Trust, and New Zealand Ministry of Business, Innovation and Employment grant C01X1002. Whaka Inaka was funded by the Department of Conservation Community Fund CCPF2-008.

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Supplementary Material

Table S5.1 Habitat quality classes.

Class	Quality of habitat for supporting spawning	Expected egg mortality rate	Criteria
1	Poor	High	Vegetation cover <100% or Stem density <$0.2cm^{-2}$
2	Moderate	Moderate	Vegetation cover 100% Stem density >$0.2cm^{-2}$ Aerial root mat depth <0.5cm
3	Good	Low	Vegetation cover 100% Stem density >$0.2cm^{-2}$ Aerial root mat depth >0.5cm

Classification schema

A. Vegetation cover <100% Class 1
   Vegetation cover >100% Class 2 or 3

B. Stem density <$0.2cm^{-2}$ Class 1
   Stem density >$0.2cm^{-2}$ Class 2 or 3

C. Aerial root mat depth <0.5cm Class 2
   Aerial root mat depth >0.5cm Class 3

Table S5.2 Tidal cycle data and survey periods for egg searches (Orchard et al. 2018a).

Year	Month of spawning	Peak tidal cycle start	Peak tidal cycle end	Lunar phase	Peak tidal height† (m)	Survey periods and study areas		
						Heathcote	Avon	Anzac
2015	February	22/2	25/2	new moon	2.6	Mar 3-6	Mar 7 -12	Mar 13-15
2015	March	20/3	23/3	new moon	2.6	Mar 29 - Apr 3	Apr 4-8	Apr 9-11
2015	April	18/4	20/4	new moon	2.6	Apr 26-30	May 1-5	May 6-8
2015	May	17/5	19/5	new moon	2.6	May 26-30	Jun 1-4	Jun 5-6
2016	February	10/2	14/2	new moon	2.5	Feb18-22	Feb 23-27	Feb 28-29
2016	March	10/3	13/2	new moon	2.6	Mar18-22	Mar23-27	Mar28-29
2016	April	7/4	11/4	new moon	2.6	Apr14-18	Apr19-23	Apr24-26

† based on predicted tide levels at Port of Lyttelton (Lat. 43° 36' S Long. 172° 43' E) in metres above Chart Datum (Source: LINZ).

Table S5.3 Experimental designs for artificial habitat installations.

(a) 2015 experiment (Pilot study)

Catchment / sub-catchment	Management context	Reach length sampled (m)	Number of test sites†	Artificial habitat installations‡ (3 per test site)
Heathcote	Within known spawning site distribution	2500	1	3
Heathcote	Upstream of known spawning site distribution	600	1	3
Heathcote	Downstream of known spawning site distribution	400	1	3
Avon	Within known spawning site distribution	2300	1	3
Avon	Upstream of known spawning site distribution	700	1	3
Avon	Downstream of known spawning site distribution	500	1	3
Anzac	Within known spawning site distribution	100	1	3
Anzac	Upstream of known spawning site distribution	500	1	3
TOTALS		**7600**	**8**	**24**

† BACI analysis notation = 'site'

‡ BACI analysis notation = 'replicates'

(b) 2016 experiment (Whaka Inaka – Causing Whitebait project)

Catchment / sub-catchment	Management context	Reach length sampled (m)	Number of test sites†	Artificial habitat installations‡ (3 per test site)
Heathcote	Within known spawning site distribution	2500	17	51
Heathcote	Upstream of known spawning site distribution	600	4	12
Heathcote	Downstream of known spawning site distribution	400	2	6
Anzac	Within known spawning site distribution	100	1	3
Anzac	Upstream of known spawning site distribution	500	5	15
Anzac	Downstream of known spawning site distribution	100	1	3
TOTALS		**4200**	**30**	**90**

† BACI analysis notation = 'site'

‡ BACI analysis notation = 'replicates'

Chapter 6

Earthquake-induced habitat migration in a riparian spawning fish has implications for conservation management

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Abstract

1. *Galaxias maculatus* is a riparian spawning fish that supports an important recreational fishery in New Zealand with spawning habitat requirements strongly structured by salinity gradients at rivermouths. This study reports changes to the spawning habitat following a series of large earthquakes that resulted in widespread deformation of ground surfaces in the vicinity of waterways.
2. Assessments of habitat recovery focused on two river systems, the Avon and Heathcote, with pre-disturbance data available over a 20 year period. Recovery dynamics were assessed by field survey and mapping of spawning habitat prior to and on seven occasions after the disturbance event. Riparian land-use and management patterns were mapped and analysed using overlay methods in a GIS.
3. Habitat migration of up to 2 km occurred in comparison to all previous records and several anthropogenic land uses have become threats due to changed patterns of co-occurrence. Incompatible activities now affect more than half of the spawning habitat in both rivers, particularly in areas managed for flood control purposes and recreational use.
4. The results are an example of landscape scale responses to salinity and water level changes driven by tectonic dynamics. These dynamics are not the source of the stress *per se*, rather, they have increased exposure to pre-existing stressors.
5. The case illustrates important principles for managing subtle, yet widespread, change. Adaptive conservation methods and investments in information are priorities for avoiding management failure following environmental change.

Keywords
Intertidal, estuary, conservation evaluation, fish, urban development, engineering.

6.1 Introduction

6.1.1 Earthquake recovery context

The Canterbury region of New Zealand was affected by a sequence of major earthquakes in 2010 and 2011. The most devastating of these was a M_w 6.3 earthquake centred beneath the city of Christchurch that caused widespread damage and loss of life (Quigley et al. 2016). After six years of recovery activities the process has entered a more strategic phase. The focus is now on longer term adaptation to environmental and societal change. Important land-use decisions remain for many geographical areas and with regards to many aspects of the natural and built environment. Examples relevant to waterway management include responses to water quality, erosion, flood risk and coastal inundation issues, and the potential re-zoning of large tracts of riparian and floodplain land. Existing statutory arrangements apply to many of the recovery activities and identify institutional responsibilities. Due to the scale and impact of the event bespoke legislation was also created. The organizations involved now include new planning entities with specific tasks (Regenerate Christchurch 2017) and a wide range of interests across central, regional, and local government, non-governmental organizations, and local community groups.

Initially, urgent decisions were made to address risks to property and life, and to reinstate essential infrastructure. Remaining decisions have the benefit of more time. There is a unique opportunity to secure benefits through earthquake recovery planning in relation to historical degradation of natural environments and improved resilience to future events. Natural values in the affected areas have thus far received less attention, but include traditional cultural uses such as the wild harvest of food and fibre (Jolly & Ngā Papatipu Rūnanga Working Group 2013; Lang et al. 2012), risk reduction functions (Orchard 2014), nature-based recreation, and habitat for many indigenous and migratory species with protected status. However, knowledge gaps are a barrier to securing benefits through the planning process. Information requirements include quantifying impacts of the earthquakes and identifying opportunities for future gains.

6.1.2 Riparian spawning habitat of īnanga

In the present study, the particular focus is *Galaxias maculatus,* or 'īnanga', a riparian spawning fish. *G. maculatus* is an amphidromous species currently listed as 'at risk - declining' in the New Zealand Threat Classification System (Dunn et al. 2018). Reversing the decline of īnanga is addressed in many statutory documents as well as non-statutory plans and it is a priority issue for Māori. Juvenile fish are harvested in an iconic recreational and culturally important fishery (McDowall 1984). The harvest of īnanga and other 'whitebait' species creates an ongoing tension between conservation and sustainable use. However, use and non-use interests share the objective of enhancing īnanga populations. The protection of spawning habitat is an urgent and practical goal due to a history of degradation associated with land-use changes near lowland waterways (McDowall 1992; McDowall & Charteris 2006).

Īnanga has a specialized reproductive strategy that is synchronized with the spring tide cycle and strongly influences the distribution of spawning sites (Burnet 1965). Spawning sites occur close to the maximum upstream extent of saltwater intrusion and occupy only a narrow elevation range (Taylor 2002). Eggs are laid in riparian vegetation just below the spring tide high-water mark and hatch in response to inundation after a 2-4 week development period (Benzie 1968a). The composition and condition of riparian margins at these specific sites is critical to spawning success (Hickford & Schiel 2011a).

This specificity suggested that earthquake-induced land deformation could affect habitat in several ways. First, disturbance could reduce the availability or condition of existing spawning sites, and enduring changes might result from vegetation recovery effects. Second, large scale impacts were possible due to physico-chemical effects. This was the particular focus of the study in light of suspected earthquake-driven hydrodynamic changes and the reported structuring of habitat by salinity (Richardson & Taylor 2002; Taylor 2002). Because there was no prior salinity baseline available, the focus was on direct detection of changes in the distribution of spawning sites. By reconstructing a spawning site distribution baseline using data from previous studies, this comparison was possible for the consideration of earthquake effects. The objectives of the study were therefore to quantify the pre- and post-quake spawning site distribution against riparian land uses and evaluate distributional effects to identify management implications.

6.2 Methods

6.2.1 Study area

The two study catchments are the Avon River (Ōtākaro) and Heathcote River (Ōpāwaho) (Fig. 6.1). These are spring-fed, lowland waterways with small average base flows (approx. 2 and 1 cumecs respectively) originating within the city of Christchurch, New Zealand (White et al. 2007). The catchments are heavily urbanized, particularly in their upstream reaches. The two waterways are extensively channelized through the use of bank stabilization engineering and flow regulation structures including flood-gates. The lower catchments support riparian saltmarsh areas that contribute to the Avon-Heathcote Estuary Ihutai (Fig. 6.1). These are remnants of a larger and relatively mobile ecosystem of coastal hydrological features (Kirk 1979). (Note: the bilingual naming convention used elsewhere in this thesis when referring to these culturally important rivers is not used hereafter in this chapter for the sake of brevity).

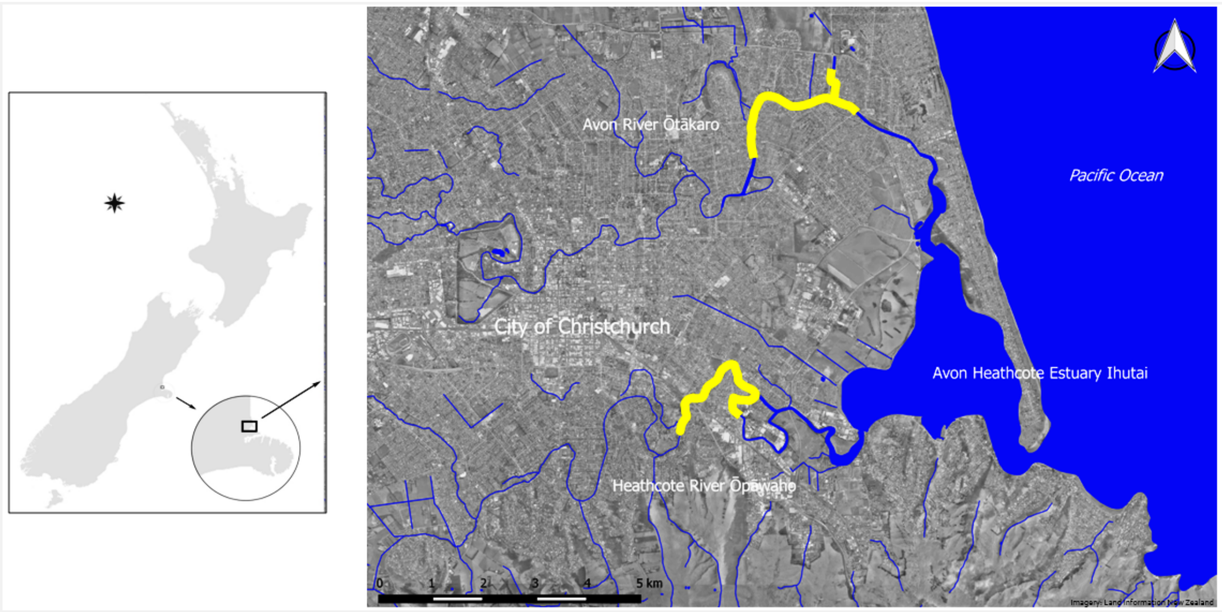

Fig. 6.1 Location of the Avon River Ōtākaro and Heathcote River Ōpāwaho in Christchurch, New Zealand, showing reaches surveyed for īnanga spawning in the post-quake studies (in yellow).

Vertical seismic shifts and lateral spread were pronounced in the vicinity of Christchurch waterways particularly towards the estuary (Hughes et al. 2015). Changes in ground levels in and around the estuary were in the order of ± 0.5 m with a trend towards uplift in the south and subsidence in the north (Beaven & Litchfield, 2012). Hydrodynamic modelling of the estuary showed extensive bathymetric change and an estimated 15% reduction in the estuarine tidal prism (Measures et al. 2011).

6.2.2 Pre-earthquake baseline

A literature review was completed to identify pre-quake spawning records augmented with information from current researchers (M. Taylor, S. McMurtrie, C. Meurk, pers. comm.). This resulted in a database of 14 technical reports and additional personal communications together with records from the National Īnanga Spawning Database (NISD). Historical spawning site data were restricted to information associated with observations of eggs in riparian vegetation. All information was digitized in GIS by identifying coordinates for the upstream and downstream extent of spatially discrete spawning sites using the original data sources. Sites were defined as semi-continuous stretches of eggs identified through riparian vegetation surveys on waterway margins. These locations were identified using the co-ordinates, maps, photographs and text descriptions provided in technical reports and direct communication with researchers.

6.2.3 Post-earthquake studies

A census-style survey methodology was used with the objective of detecting all spawning occurrences at the catchment scale following the methods of Orchard & Hickford (2018a). The search areas were approximately 4 km reaches in each river (Fig. 6.1). The survey area extended from the downstream transition to saltmarsh vegetation, which is unsuitable for spawning (Mitchell & Eldon 1991), to 500 m upstream of the inland limit of saltwater. In the Avon this included the confluence with a prominent tributary to the north. The saltwater limit was established using conductivity/temperature loggers (Odyssey, Dataflow Systems Ltd, NZ) deployed during spring tide sequences and additional spot measurements using a handheld conductivity/salinity/temperature meter (YSI Model 30, YSI Inc., USA). The survey period included the peak spawning months (Taylor 2002) over two years. Surveys commenced five days after the peak tide in the spring tide sequence and followed a set schedule to minimize temporal confounding effects between months (Supplementary Material, Table S6.1). Reaches surveyed later in the schedule were more sensitive to egg mortality effects due to the time elapsed since spawning. Results are more likely to underestimate the extent of spawning occurrences in these areas, but are comparable between months.

The search area was surveyed systematically in the first two months of the study by conducting three searches for eggs within contiguous 5 m blocks along each riverbank. Each search involved opening up the vegetation down to ground level at random locations within the block following a transect line perpendicular to and spanning the high water mark. On subsequent months, the survey effort was reduced to areas of potential habitat following a habitat classification system (Orchard & Hickford 2018a). Whenever eggs were found, the survey was extended 50 m either side of the last occurrence to confirm the full extent of the spawning site. Spawning sites were defined as the area occupied by continuous or semi-continuous patches of eggs. Upstream and downstream extents were established and the width of the egg band measured on the centreline of the search transects within the extent of the site (minimum three). Zero counts were recorded where these occurred such as when the egg patch was not continuous. Area of occupancy (AOO) was calculated as length x mean width. The total number of eggs present was calculated by sub-sampling patches. At each width measurement location, eggs were counted in a 10 x 10 cm quadrat placed in the centre of the egg band. Productivity was calculated as mean egg density x AOO.

Riparian land uses and management activities were mapped in the field using 0.075 m resolution post-quake aerial photographs (Land Information New Zealand 2016a). Anthropogenic stressors were identified based on reported incompatibility with īnanga spawning sites (Hickford & Schiel 2011a, 2011b; Mitchell 1994). Areas affected were delineated using aerial photographs in the field and digitized for overlay analysis in QGIS v2.8.18 (QGIS Development Team 2016). Four classes of land use activities were classified as threats to spawning habitat. These were bank stabilization using engineered structures, invasive species control, mowing of recreation reserves, and vegetation removal for flood management. Threats from riverbank engineering were defined on the basis of surfaces devoid of any vegetation capable of supporting spawning (Mitchell 1994). Examples include retaining walls, bridge abutments, riprap, and other bank stabilization works. Invasive species control was classed as a threat where it involved spraying or extensive mechanical clearance (e.g. using scrub cutters, line trimmers & similar). This recognizes that vegetation suitable for spawning may take several months to recover following clearance activities (Hickford & Schiel 2014). Mowing was classed as a threat where it resulted in short grass conditions at the top of the riverbank in the location of spawning habitat.

6.3 Results

6.3.1 Pre-earthquake spawning distribution

Eighteen pre-quake spawning studies spanning a 25 year period were identified, most of which involved surveys in both catchments. Thirteen of these had quantified spawning in the Avon and nine in the Heathcote (Table 6.1). In some years field surveys were conducted that did not find any spawning and these records are not shown in Table 6.1. In the Avon, most of the spawning occurrences have been in the Avondale Road area (Fig. 6.2a) and often found a short distance upstream from the road bridge on the true right (Table 6.1). The maximum extent of pre-quake spawning sites recorded in any one year was 2000 m in 2007. This also represents the maximum extent of the spawning reach based on all known records.

In the Heathcote, most of the records have been in the vicinity of Opawa Road (Fig. 6.2b). Although the downstream limit of all records is ca. 1 km further downstream this relates to only two observations of spawning below Opawa Road in the 25 year period (Table 6.1). However, the first spawning recorded in the catchment was much further upstream (> 3 km). At the time the river was under the influence of a floodway, constructed in 1986, that effectively shortened the length of the river. In 1994 a tidal barrage was installed to reduced saline intrusion and this resulted in a shift of ca. 2km downstream in the upstream limit of spawning (Taylor 2004). Although these variations in the location of pre-quake sites complicate historical analyses the location of spawning has been remarkably consistent since 1994 (Table 6.1) centred on the Opawa Road site. The maximum extent of pre-quake spawning recorded in any one year was 1050 m in 2004 (Table 6.1) associated with the discovery of small sites in a reserve ca.1 km upstream of Opawa Road.

6.3.2 Post-quake studies

Spawning distribution

A total of 85 spawning sites were identified in the 2015 post-quake survey. These were distributed along 2.4 km of riverbank in the Avon and 2.5 km in the Heathcote. In both rivers there were marked differences in the spawning distribution in comparison to previous records

(Fig. 6.2). In the Avon, the spawning reach had expanded approximately 250 m upstream and 180 m downstream of the previous extent. In the Heathcote, the changes were more dramatic with spawning recorded 1.5 km downstream of all previous records (Fig. 6.2a). The 2016 survey identified 101 spawning sites, some of which represented repeat use of 2015 sites. In the Avon, the upstream and downstream limits were very close to those recorded in 2015. In the Heathcote, the upstream limit was also similar to 2015, but the spawning reach extended a further 400 m downstream. The furthest downstream sites were found in the final month of the survey at a distance of nearly 2 km from all pre-quake spawning records and ca.3 km from the previous centre of spawning at Opawa Road (Fig. 6.2b).

Table 6.1 Extent of īnanga spawning habitat utilised in the Avon and Heathcote rivers in the city of Christchurch over the period 1989 – 2014 from all known records.

Year	Description	Extent of spawning† (m)	References
Avon			
1989	TRB 40m reach downstream and upstream of Avondale Road bridge (ARb)	80	Meurk (1989); Taylor et al. (1992)
1993	TRB 15m reach above ARb	15	Taylor (1996)
1996	TRB 90m reach above and 25m reach below ARb	115	Taylor (1996)
1997	TRB 90m reach above ARb	90	Taylor (1997)
1998	TRB 70m reach above and 20m reach below ARb	90	Taylor (1998)
1999	TRB 250m reach above ARb	250	Taylor (1999)
2000	TRB at ARb	90	Taylor (2000)
2004	TRB from Alloway Street to Orrick Crescent; TLB at Amelia Rogers Reserve, above and below ARb, and at Corsers Stream	1500	Taylor (2004)
2006	TLB Amelia Rogers Reserve TLB Lake Kate Sheppard	1070	University of Canterbury unpubl. data
2007	TRB from ARb to Sharlick Street and in Lake Kate Sheppard	2000	Taylor & Chapman (2007)
2008	TRB above ARb	250	Hickford & Schiel (2014)
2010	TRB above ARb	unknown	Taylor & Main unpubl. data
2011	TRB above ARb	90	Taylor & Blair (2011)
Heathcote			
1989	TLB 70m reach downstream and 20m reach upstream of Wilsons Road bridge, TRB 20m reach downstream of Wilsons Road bridge	90	Eldon et al. (1989)
1991	TRB 100m reach within King George V Reserve	100	Taylor et al. (1992)
1994	TRB 30m reach below Opawa Road bridge (ORb)	30	Taylor (1994)
1995	TRB 50m reach below ORb	50	Taylor (1995)
1998	TRB 50m reach below ORb	50	Taylor (1998)
1999	TRB from ORb to downstream of rail bridge	70	Taylor (1999)
2002	TRB small patch in King George V Reserve	10	University of Canterbury unpubl. data
2004	TRB in King George V Reserve, TLB and TRB below ORb	1050	Taylor (2004)
2010	TLB 12m reach adjacent to Woolston Park	12	Taylor & Blair (2011)

† Calculated as the distance between upstream and downstream limits of spawning as measured on the centreline of the mainstem for each river. Where spawning also occurred in tributaries the location of the confluence was used for this calculation. TRB = true right bank. TLB = true left bank.i

Fig. 6.2 Post-quake īnanga spawning distribution overlaid on the maximum pre-quake extent of spawning from all known records. Well-known pre-quake spawning sites are shown. (a) Avon River. (b) Heathcote River.

Distribution of threats and protected areas

There are three areas managed specifically to protect spawning habitat at well-known sites (Fig. 6.3). The protection mechanisms include recognition in local authority plans and implementation of compatible riparian management on the ground. There is also a considerable reach in the lower Heathcote that is not subject to vegetation clearance for flood or reserves management purposes. Part of this reach is characterized by tall woody riverbank vegetation and the remainder is downstream of the tidal barrage where there is less need for channel works associated with flood management.

Collectively, the four classes of threats affect a large proportion of the study area (Fig. 6.3). In both rivers, threats from riverbank engineering occupied only a small proportion of the post-quake spawning extent. Extensive channelization using gravel embankments also occurs in the Avon. Although the area available for spawning may be reduced by these structures they were not classified as threats based on observations of spawning if suitable vegetation co-occurred. Invasive plant species that have historically been the subject of spraying or mechanical clearance are widespread throughout the study area. In the Avon the major concern is yellow flag iris (*Iris pseudacorus*). It is distributed throughout the spawning reach with the exception of sections engineered with gabion baskets and in Lake Kate Sheppard. This species is largely absent from the Heathcote and instead reed canary grass (*Phalaris arundinacea*) is the major concern and is the dominant canopy species in many areas. In addition, *Glyceria maxima* and *Rubus fruticosus* are present there. There were no major spray eradication campaigns during the study period despite the severe level of infestation. Decisions on control will be required in the near future under regional pest management plans. Riparian mowing occurs in discrete areas in both river systems associated with a network of parks and reserves (Fig. 6.3). Vegetation control for flood management was conducted on a semi-regular basis in both rivers using scrub cutters or line trimmers. This work is regularly scheduled through the Avon study area with the exception of the two areas protected for spawning habitat (Fig. 6.3a) and in the upper section of the Heathcote study area (Fig. 6.3b).

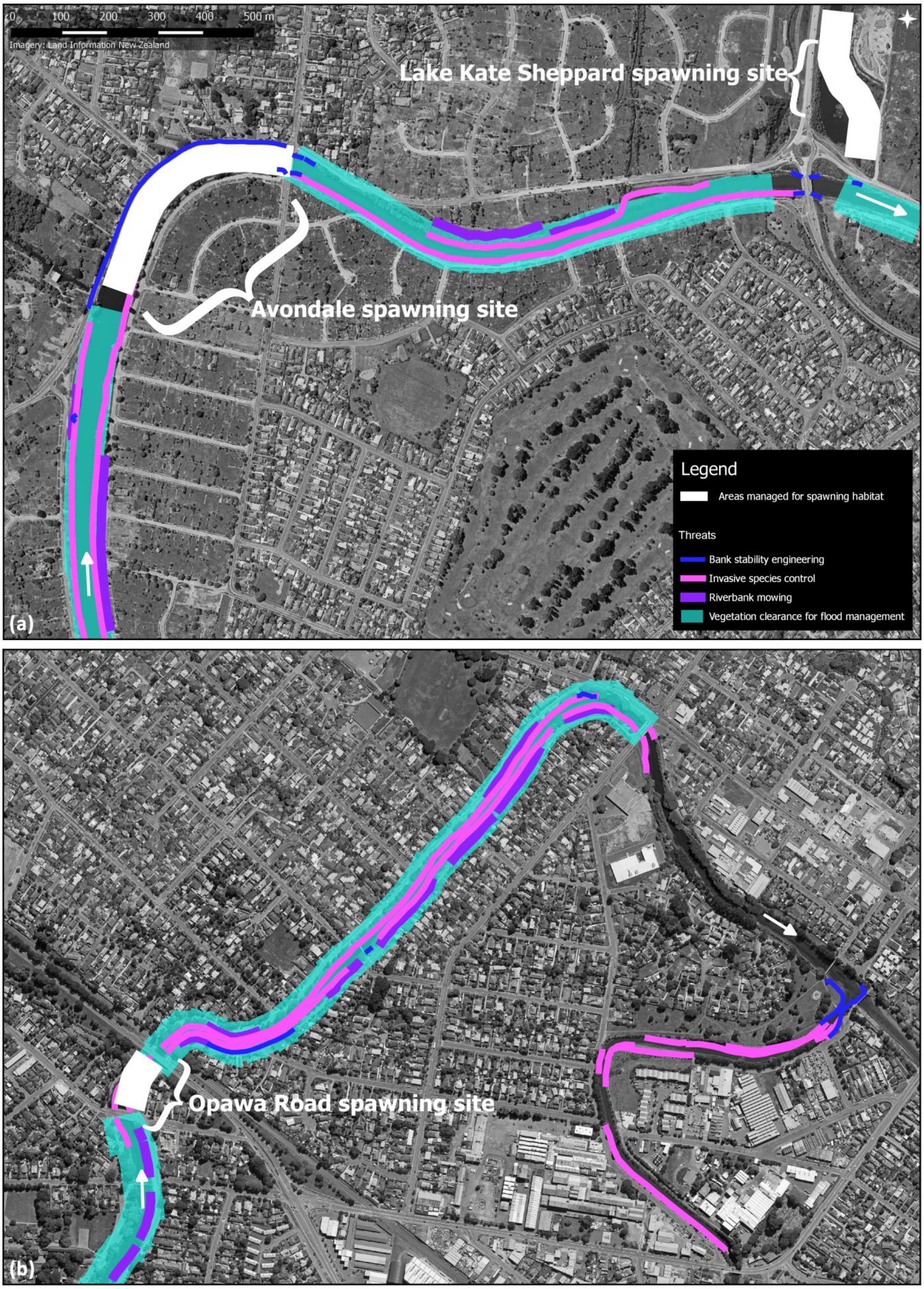

Fig. 6.3 Distribution of post-quake anthropogenic threats associated with riparian land uses and management activities in the study area. (a) Avon River. (b) Heathcote River.

Area of occupancy and egg production

In 2015, the total area of occupancy (AOO) of spawning habitat was 152.5 m^2 in the Avon and 75.4 m^2 in the Heathcote as calculated using maximum figures recorded at each site across all four surveys. Total egg production in 2015 was 11.8 million eggs (Avon 6.9 x 10^6, Heathcote 4.9 x 10^6). In 2016, egg production was higher (Avon 13.9 x 10^6, Heathcote 5.0 x 10^6) despite the survey period being reduced to only three months. The AOO was also higher in both rivers (Avon 472.9 m^2, Heathcote 99.1 m^2) although average egg densities were lower. The marked increase in AOO in the Avon was associated with several new large spawning sites that were not utilized in 2015 in addition to re-use of other sites. In both years, AOO and productivity were not evenly distributed across the study area and high production was not always correlated with AOO due to differences in egg densities (Fig. 6.4). Egg densities of >10 eggs cm^{-2} were recorded at several sites with the highest being 13.5 cm^{-2}.

6.3.3 Effectiveness of protected areas

In the Avon, the proportion of the AOO occurring in protected areas was 70% in 2015. In 2016 this figure had decreased to only 28% reflecting many new sites discovered in other locations. In the Heathcote, the proportion of AOO protected was very low (11% and 6% for the two years respectively) reflecting the discovery of many spawning sites at locations never previously known for spawning. Egg production was also considerable outside of the protected areas (Fig. 6.5). In the Avon, the proportion of egg production outside the protected areas was 28% in 2015 and 38% in 2016 (Fig. 6.5a). In the Heathcote, 82% of egg production occurred outside of the protected areas in 2015 and 98% in 2016 (Fig. 6.5b). On average across the two years of post-quake studies, only 4.5% of the spawning reach was protected in the Heathcote and 27.6% in the Avon (Fig. 6.6). Although the timing of threats was variable in relation to the presence of eggs, vegetation clearance for reserves and flood management purposes was observed at many of the unprotected spawning sites after egg deposition had occurred (see Supplementary Material). Repeat egg surveys at some of these sites after the vegetation clearance indicated close to 100% egg mortality, consistent with previous studies (Hickford & Schiel 2014).

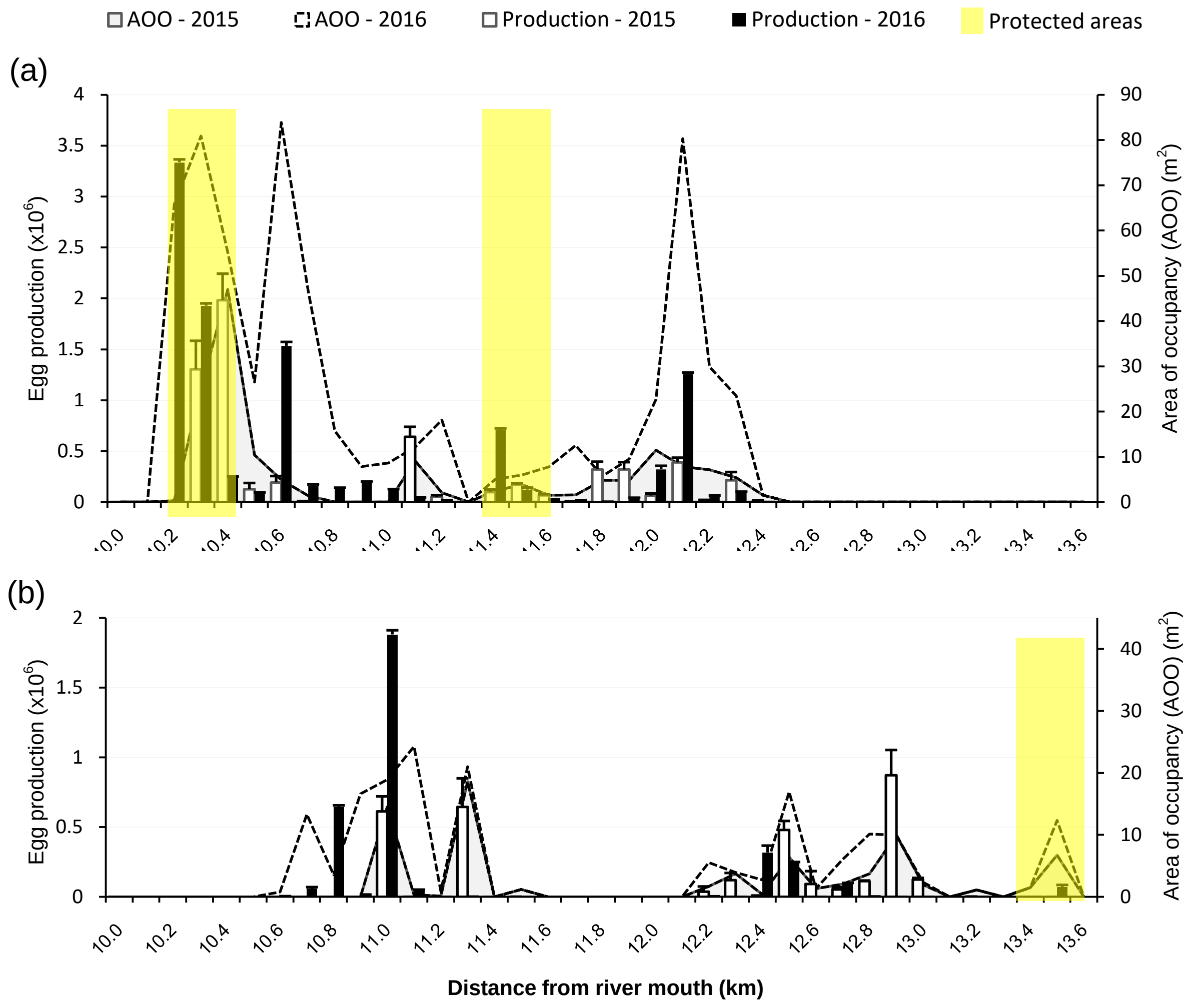

Fig. 6.4 Post-quake area of occupancy (AOO) and productivity of īnanga spawning sites presented as aggregated data for contiguous 100 m reaches. Egg production is shown as the total recorded in the surveys conducted each year (n=4, 2015; n=3, 2016). AOO is presented as the maximum area occupied by spawning sites each year. River kilometres are measured from the entrance into the Avon-Heathcote Estuary Ihutai following the main channel lines for each river. Error bars are standard error of the mean. (a) Avon River, (b) Heathcote River.

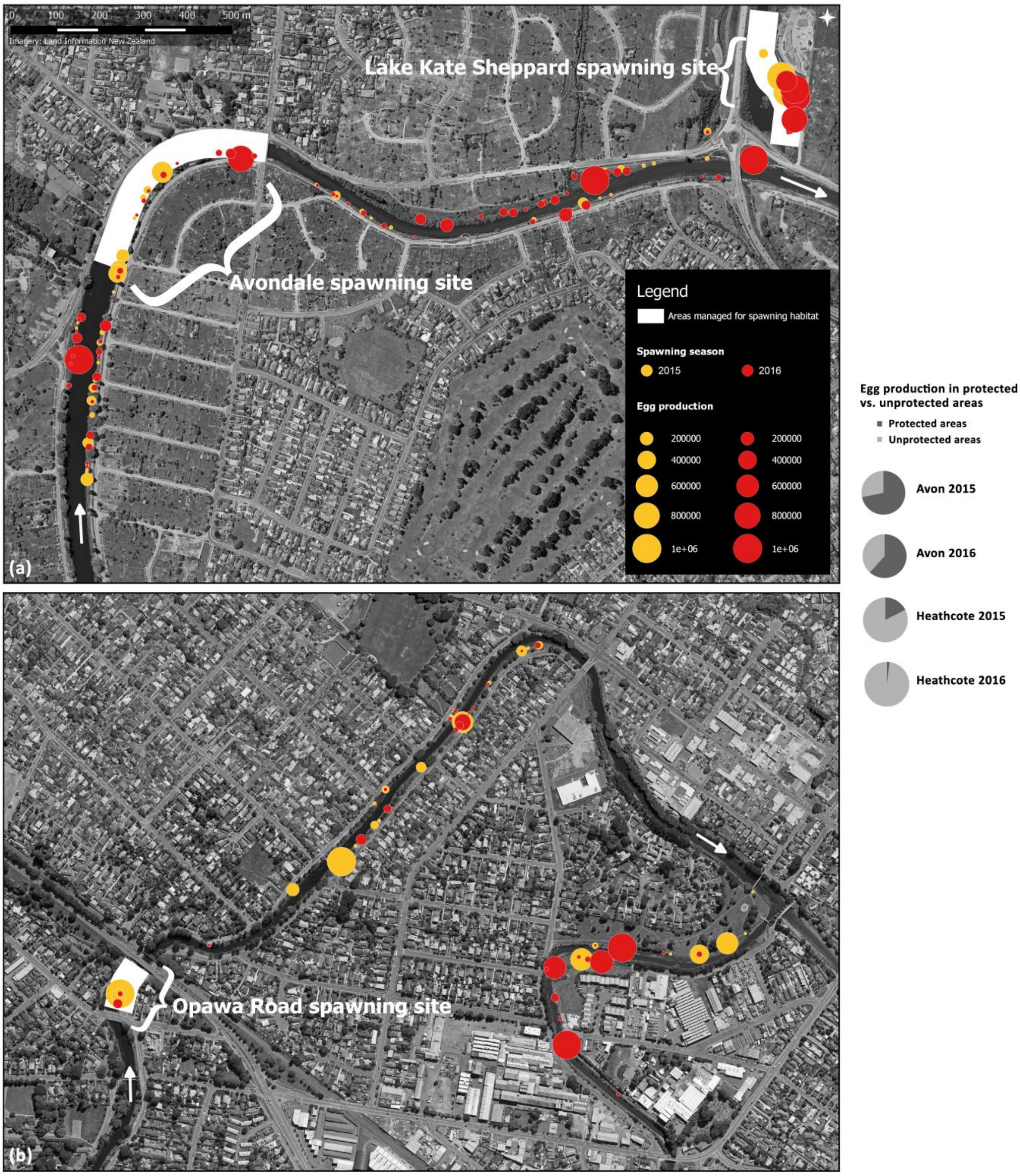

Fig. 6.5 Spatial distribution of post-quake īnanga egg production and proportion of post-quake production that occurred in protected areas. (a) Avon River. (b) Heathcote River.

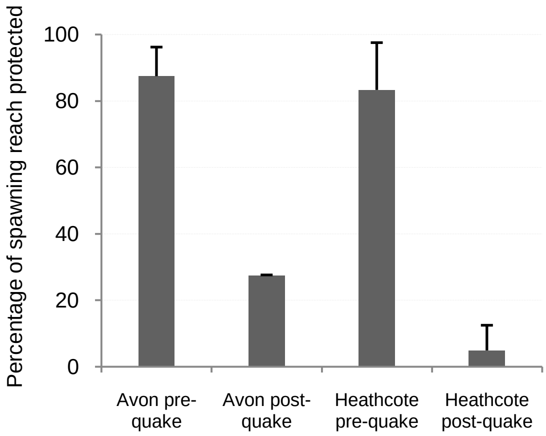

Fig. 6.6 Percentage of the known īnanga spawning reach in protected areas in the Avon and Heathcote rivers before and after the 2010–11 Canterbury earthquake sequence. Error bars are standard error of the mean for the periods.

6.4 Discussion

6.4.1 Evidence for īnanga spawning habitat migration

There are several limitations for accurately characterizing the pre-quake spawning baseline. They include the variable frequency, extent and intensity of historic surveys, and temporal effects in relation to the peak months of spawning activity, all of which may lead to under-estimation of the areas utilised. These sources of inaccuracy have a bearing on the identification of change in relation to the distribution and area of occupancy spawning habitat. Despite this, the Christchurch waterways have the most extensive record of īnanga spawning for any catchment in New Zealand in terms of the total number of surveys conducted and the length of the survey record (Taylor 2002). The relatively consistent results obtained by researchers over the historical pre-quake period are another important aspect. Additionally, we have taken the maximum values identified over all records. This produces a precautionary approach in relation to the area and extent of pre-quake spawning habitat recorded in most years. In both catchments these maximum values were atypical of the full survey record suggesting that they may over-estimate the relevant habitat parameters. However, the prevalence of degraded riparian vegetation in the study area is likely to have caused high egg mortality if spawning occurred in those areas. This effect reduces the detectability of spawning sites in field surveys (Orchard & Hickford 2018a), and was specifically addressed in the design of post-quake surveys by attention to the timing of surveys in relation to the estimated date of spawning events. For these reasons the peak detected over all pre-quake records is considered to be the best estimator of typical spawning activity over this period.

In the Avon, the majority of historical spawning has been recorded at the Avondale site (Fig. 6.2a). In this vicinity, the spatial extent of spawning steadily increased since discovery of the site in 1989 and was assisted by protection from mowing (Taylor 1999). In 2004, new sites were identified further downstream in the mainstem, and in 2006 spawning was found at Lake Kate Sheppard and then regularly thereafter. This is an area of restored riparian margins in a tributary waterway and lake system located close to the mainstem. In the Heathcote, the pre-quake distribution shifted downstream in association with construction of the tidal barrage in 1994 to reduce saltwater intrusion upstream (Taylor 1995, 1998). Subsequently, spawning has been centred on the Opawa Road site with only two sites have been recorded further downstream in all known records. Earthquake-induced migration of habitat a further 1.5 km downstream in 2015 and 1.9 km in 2016 represents a major change in spawning habitat distribution.

6.4.2 Effectiveness of protected areas

A high proportion of īnanga spawning now occurs outside of the areas designated for spawning site protection. Risk exposure is now greater due to the co-occurrence of habitat with anthropogenic threats. Earthquake-induced change is not the source of heightened vulnerability *per se*. Rather, this is an effect of natural dynamics that have increased exposure to pre-existing stressors. These activities are now threats that require management to achieve conservation objectives. Mowing of vegetation within riparian reserves co-occurs with several spawning sites in both river systems. It is a particular issue where the spring high tide water levels are sufficient to inundate riparian terraces. These provide locations where spawning habitat may be relatively expansive in comparison to areas with steeper topography. Vegetation clearance using scrub bars also occurs on the bank face throughout much of the study area for flood management purposes with the exception of locations specifically managed for īnanga spawning (Figure 6.4). Compared to reserve maintenance activities, vegetation clearance for flood management affects the upper intertidal zone of the waterway margin. At many locations this results in a direct overlap with the spawning habitat elevation band. High egg mortality from mowing and grazing has been previously reported (Hickford & Schiel, 2014). This is believed to be mostly attributable to UV irradiation or the drying out of eggs (Hickford et al. 2010; Hickford & Schiel 2011b). Recovery from vegetation clearance can take many months, with the re-establishment of sufficient cover being a critical factor (Hickford & Schiel 2014). In addition, these activities may occur after eggs have been laid in vegetation that would otherwise have been suitable for spawning. This was observed at many of the spawning sites recorded in this study and is particularly problematic for conservation. Due to the gregarious behavioural ecology of *G. maculatus* (Benzie 1968a; McDowall 1990), the majority of spawning production is typically supported by only a few sites in the catchment in each spawning event. This contributes to the vulnerability of spawning to stochastic events. Anthropogenic threats affecting these highly productive sites may have a large impact on the total egg production on a seasonal basis.

6.4.3 Learning for adaptive management

This case illustrates important principles for managing subtle yet widespread change. The results demonstrate habitat migration that was not detected by conservation management practitioners. Pre-disturbance land-use activities had continued without adaptation exposing the habitat to increased risk despite its apparent expansion. Adaptive management responses are needed to control anthropogenic stressors in areas that have now become īnanga spawning habitat. Achieving this requires further work to develop solutions that accommodate other necessary or desirable waterway management activities in the riparian zone. Although historical AOO figures are not available, the post-quake studies indicate that in both catchments the extent of spawning habitat is now greater than all previous records. This is a positive finding and suggests a potential improvement in the opportunities available for accommodating incompatible activities through tools such as spatial planning. If these are addressed and solutions identified, conservation gains could be secured in terms of increasing the area of protecting habitat and ultimately improved egg production.

This case also provides several important lessons for the wider community in relation to conservation management following major disturbance events. These include the need to fully characterise environmental change and consequences for protected species despite that information acquisition may be difficult. Challenges to overcome include the likelihood that post-disturbance landscapes will be in a state of transition until a new relatively stable state becomes re-established. In our case, this was partly addressed by commencing investigative work four years from the disturbance event but also demanded temporal replication in the post-quake studies to confirm whether the apparent effects could be related to an enduring post-disturbance change. These aspects of the study approach may be useful considerations for the design of other post-disturbance studies, and the importance of baseline measurements is also highlighted since these are essential for interpreting change. In our case, further work is also required to establish the cause-effect relationship driving the observed change. Salinity effects are thought to be the most likely driver of the large-scale catchment position changes in consideration of the literature suggesting a close relationship between spawning habitat and saltwater intrusion. As such the disturbance event offers a unique opportunity to test fundamental knowledge for conservation biology. Taking such opportunities requires a commitment to mobilize the necessary resources at the required time. However the potential gains from attention to these ‘natural laboratories’ are considerable especially given that the circumstances may represent relatively uncommon events that cannot be readily replicated or otherwise observed.

The implementation of statutory protection mechanisms for the achievement of conservation objectives adds another dimension to this case. It is important to note that protection of the post-quake habitat is a legislative requirement. However, conservation policy often suffers from implementation gaps in practice (Knight et al. 2008) which may result from a lack of attention to methods that are effective in the societal context (Knight et al. 2010). Dynamic environments and spatiotemporal variation create additional challenges for the design of conservation methods that are effective and socially acceptable. Our results illustrate that investments in information are a pivotal activity that contributes to all of these needs. In addition, contemporary information must be coupled with appropriate responses to facilitate an adaptive approach. Our case highlights that further policy-related work may be a necessary aspect of addressing the societal dimension. In particular a review of current conservation planning arrangements with a focus on role of protected areas is a practical necessity given that these are important conservation management tools.

Lastly, the effects described here are an example of landscape-scale responses to infrequent tectonic dynamics. They have likely been mediated by hydrological and salinity changes together with smaller-scale effects on ground surfaces in the riparian zone. In the Heathcote in particular, the magnitude of horizontal shift deserves further investigation and despite the current unknowns regarding causative factors the opportunity for learning is clear. Post-earthquake studies present opportunities to evaluate many aspects of socio-ecological systems for impacts and associated responses. Not only are tectonic events relatively common in evolutionary time, they may exert similar effects to climate change through influencing water levels and salinity gradients relative to existing topography (Beavan & Litchfield 2012). Earthquakes present unique and important opportunities to study vulnerable ecosystems and provide examples of real-life adaptation in action. In turn, this may assist in developing methods to achieve conservation objectives and avoid implementation failures in the face of ongoing change.

6.5 Acknowledgements

We thank Mark Taylor, Shelley McMurtrie and Colin Meurk for providing historical records. We acknowledge the many volunteers and staff of the Waterways Centre for Freshwater Research and Marine Ecology Research Group who assisted with the post-quake field studies, and local government staff for information on riparian management activities. Funding was provided by the Ngāi Tahu Research Centre, Engineering New Zealand / Water NZ Rivers Group, Brian Mason Scientific and Technical Trust, and a New Zealand Ministry of Business, Innovation and Employment grant (C01X1002) in conjunction with the National Institute of Water and Atmospheric Research.

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Supplementary Material

Table S6.1 Survey periods and tidal cycle data for post-earthquake *G. maculatus* spawning surveys.

Year	Month of spawning	Peak tidal cycle start	Peak tidal cycle end	Peak tidal height* (m)	Survey period	
					Heathcote	Avon
2015	February	Feb 22	Feb 25	2.6	Mar 3-6	Mar 7 -15
2015	March	Mar 20	Mar 23	2.6	Mar 29 - Apr 3	Apr 4-11
2015	April	Apr 18	Apr 20	2.6	Apr 26-30	May 1-8
2015	May	May 17	May 19	2.6	May 26-30	Jun 1-6
2016	February	Feb 10	Feb 14	2.5	Feb18-22	Feb 23-29
2016	March	Mar 10	Mar 13	2.6	Mar18-22	Mar23-97
2016	April	Apr 7	Apr 11	2.6	Apr14-18	Apr19-26

* predicted tide levels above Chart Datum at Port of Lyttelton (Lat. 43° 36' S Long. 172° 43' E) (Source: LINZ).

Table S6.2 Habitat quality classes.

Class	Quality of habitat for supporting spawning	Expected egg mortality rate	Criteria
1	Poor	High	Vegetation cover <100% or stem density <0.2cm^{-2}
2	Moderate	Moderate	Vegetation cover 100% Stem density >0.2cm^{-2} Aerial root mat depth <0.5cm
3	Good	Low	Vegetation cover 100% Stem density >0.2cm^{-2} Aerial root mat depth >0.5cm

Classification schema

A. Vegetation cover <100% Class 1
Vegetation cover >100% Class 2 or 3

B. Stem density <0.2cm^{-2} Class 1
Stem density >0.2cm^{-2} Class 2 or 3

C. Aerial root mat depth <0.5cm Class 2
Aerial root mat depth >0.5cm Class 3

Fig. S6.1 (a) *G. maculatus* eggs laid in riparian vegetation in the Heathcote River, February 2015. Each egg is approximately 1 mm in diameter. (b) An example of high egg densities at the Avondale site in March 2016.

Fig. S6.2 An example of good quality spawning habitat in the Avon River. In this part of the catchment ground levels had dropped by approximately 0.5 m as result of earthquake-induced subsidence and lateral spread. Prior to the earthquakes these overgrown park benches were considerably closers to the waters' edge.

Fig. S6.3 Example of a spawning site affected by mowing in a recreation reserve in the Avon catchment. The dotted line shows the area of occupancy (AOO) prior to mowing in March 2015. Egg mortality is close to 100% in these situations.

Fig. S6.4 Example of vegetation clearance for flood management. (a) Spawning site in the Heathcote River in March 2015 at which 118,000 eggs were present (located in the long grass). Mowing of a recreation reserve can also be seen in this image but did not affect the majority of the spawning site which was located lower on the bank face. (b) The same site in early May 2015 showing typical conditions following clearance of vegetation for flood management using line trimmers. This management regime is regularly applied to a large proportion of the study area.

Chapter 7

Protected area effectiveness for fish spawning habitat in relation to earthquake-induced landscape change

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Abstract

We studied the effectiveness of spatial planning methods for the conservation of *Galaxias maculatus*, a riparian spawning fish, following earthquake-induced habitat shift in the Canterbury region of New Zealand. Mapping and GIS overlay techniques were used to evaluate three protection mechanisms in operative or proposed plans in two study catchments over two years. Method 1 utilised a network of small protected areas around known spawning sites. It was the least resilient to change with only 3.9% of post-quake habitat remaining protected in the worst performing scenario. Method 2, based on mapped reaches of potential habitat, remained effective in one catchment (98%) but not in the other (52.5%). Method 3, based on a habitat model, achieved near 100% protection in both catchments but used planning areas far larger than the area of habitat actually used. This example illustrates resilience considerations for protected area design. Redundancy can help maintain effectiveness in face of dynamics and may be a pragmatic choice if planning area boundaries lack in-built adaptive capacity or require lengthy processes for amendment. However, an adaptive planning area coupled with monitoring offers high effectiveness from a smaller protected area network. Incorporating elements of both strategies provides a promising conceptual basis for adaptation to major perturbations or responding to slow change.

Keywords

Dynamic environments, landscape change, spatial planning, protected areas, conservation management, resilience, *Galaxias maculatus*, adaptation to change.

7.1 Introduction

For many species, critical life history phases create obligate habitat requirements. These may be vulnerable points in the life cycle, especially where relatively specific biophysical conditions are required (Lucas et al. 2009). Vulnerability may be associated with both periodic events and longer term change involving natural and anthropogenic processes (Turner et al. 2003). A particular concern is where human activities reduce the quality or availability of existing habitat unless counterbalanced by compensatory actions, such as the creation of suitable habitat elsewhere (Faith & Walker 2002). The concept of resilience provides a focus on thresholds in system properties that are important to their persistence (Bernhardt & Leslie 2013; Holling 1973). In linked socio-ecological systems it is related to adaptive capacity (Gallopín 2006), and actual responses to changed hazard exposure and/or sensitivity (Turner et al. 2007). Since resilience assessment is concerned with identifying the conditions required to maintain a desirable state (Gunderson et al. 2010), it may be readily applied to habitat management.

7.1.1 Protected areas for critical habitats

Protected areas (PAs) describe a desired state defined by clear objectives. They are a cornerstone of global efforts to halt biodiversity loss (United Nations 2011). The IUCN recognises six categories of PAs defined by differences in management approaches (Stolton et al. 2013). Category IV PAs aim to protect particular species or habitats (Table 7.1). They are often relatively small and are designed to protect or restore: a) flora species of international, national or local importance; b) fauna species of international, national or local importance including resident or migratory fauna; and/or c) habitats (Dudley 2008).

Effective conservation involves managing risks, yet biodiversity declines are continuing (Butchart et al. 2010). Management effectiveness evaluation is an essential activity to assess the strengths and weaknesses of the protection mechanisms in place and to consider alternatives (Stolton et al. 2007). A key area of focus is the extent to which PAs actually deliver on objectives such as the protection of important values (Hockings 2003). Under conditions of environmental change, evaluation is especially important to address whether the areas involved are functioning as an effective conservation strategy (Leverington et al. 2010). Various methodologies have been used, many of which were originally developed to the support adaptive management of PA sites and systems (Coad et al. 2015). Range shifts are a topic of particular importance since they may undermine the effectiveness of existing PA networks. In this setting, human agency is inextricably linked to the trajectory of the values identified for protection. This may require amendment of the protection mechanism itself to ensure continued performance over time.

Table 7.1 Aspects of IUCN Category IV Protected Areas (Dudley 2008).

Role in the landscape/seascape
Category IV Protected Areas frequently play a role in "plugging the gaps" in conservation strategies by protecting key species or habitats in ecosystems. They could, for instance, be used to: • Protect critically endangered populations of species that need particular management interventions to ensure their continued survival • Protect rare or threatened habitats including fragments of habitats • Secure stepping-stones (places for migratory species to feed and rest) or breeding sites • Provide flexible management strategies and options in buffer zones around, or connectivity conservation corridors between, more strictly protected areas that are more acceptable to local communities and other stakeholders
Issues for consideration
• Many category IV Protected Areas exist in crowded landscapes and seascapes, where human pressure is comparatively greater, both in terms of potential illegal use and visitor pressure • Category IV Protected Areas that rely on regular management intervention need appropriate resources from the management authority and can be relatively expensive to maintain unless management is undertaken voluntarily by local communities or other actors • Because they usually protect part of an ecosystem, successful long-term management of category IV Protected Areas necessitates careful monitoring and an even greater than-usual emphasis on overall ecosystem approaches and compatible management in other parts of the landscape or seascape

7.1.2 Protection of īnanga spawning habitat

Diadromous fishes have specific habitat requirements across several stages of their life histories, involving both freshwater and marine environments (Gross et al. 1988). In some species these may be separated by vast distances and associated with significant migrations (Metcalfe et al. 2002). There may be different conservation issues affecting each critical habitat requiring a wide range of management responses (McDowall 1999). *Galaxias maculatus* (Jenyns 1842) or 'īnanga' is a diadromous species currently listed as 'at risk – declining' under the New Zealand Threat Classification System (Dunn et al. 2018). Adult fish are found in lowland coastal waterways with the upstream distribution limited by relatively poor climbing ability (Baker & Boubee 2006; Doehring et al. 2012).

Spawning occurs in estuarine waterways with the exception of some populations that have become land-locked in lakes (Chapman et al. 2006). The locations used are highly specific as the result of specialised reproductive behaviour associated with the migration of adult fish towards rivermouths at certain times of the year (Benzie 1968a). Spawning events are strongly synchronised with the spring high tide cycle with an apparent association between spawning site distribution and the salinity regime (Burnet 1965; Orchard et al. 2018c). The majority of spawning sites have been found within 500 m of the inland limit of salt water (Richardson & Taylor 2002; Taylor 2002). In addition, spawning sites occupy only a narrow elevation range located on waterway margins just below the spring tide high-water mark (Taylor 2002). As tidal heights drop towards the neap tides these sites are no longer inundated at high-water and for most of their development period the eggs are in a terrestrial environment (Benzie 1968a, 1968b). Egg survival rates are highly dependent on the condition of the riparian vegetation in these locations until hatching in response to high water levels, usually provided by the following spring tide (Hickford et al. 2010; Hickford & Schiel 2011a; Orchard et al. 2018d).

The degradation of spawning habitat has been identified as a leading factor in the species' decline (McDowall 1992; McDowall & Charteris 2006). This has been linked to land-use intensification on coastal waterway margins (Hickford et al. 2010), as is a common trend worldwide (Kennish 2002). Protection mechanisms must often address contested-space contexts characterised by incompatible activities. Multiple-stressor situations are common with grazing, vegetation clearance, mowing, flood protection, and channelization being examples that have contributed to degradation (Hickford & Schiel 2011a; Orchard et al. 2018c).

Habitat protection is a requirement of national legislation under the Conservation Act 1987 and the Resource Management Act 1991. Implementation relies on the identification of areas for protection enforced by appropriate rules and documented in plans or management strategies prepared under the relevant Acts (Orchard 2016a). In many cases spatially explicit planning methods (e.g. maps) are used to delineate the protected areas. Although these provide a practical approach to address the conservation objective, they require reliable habitat information. In dynamic environments challenges include recognising spatiotemporal variance and accommodating it in design of the protection mechanisms used (Bengtsson et al. 2003).

7.1.3 Evaluating the effectiveness of PAs under conditions of change

In 2010 and 2011 a sequence of major earthquakes affected the Canterbury region of New Zealand. It included several large destructive events and numerous aftershocks centred beneath the city of Christchurch (Beavan et al. 2012). The magnitude of physical effects necessitated a long-term socio-ecological response associated with new ecological trajectories and a variety of land-use planning needs. Topographic and bathymetric measurements identified enduring changes in ground levels (Quigley et al. 2016), especially in the vicinity of waterways (Chapter 1). *G. maculatus* spawning was recorded at locations never previously utilised in comparison to pre-quake records (Orchard & Hickford 2016, 2018a). Vulnerability assessments identified anthropogenic threats at many of these locations and recommended review of protection methods in the operative statutory plans (Orchard et al. 2018b). This context presented a unique opportunity to evaluate conservation planning options in light of landscape-scale change whilst informing the practical needs of post-quake recovery processes and with a focus on a new statutory plan change that introduced much larger protected areas in the lower river corridors. The objectives of this paper are to (1) evaluate the efficiency and effectiveness of contemporary protection mechanisms, and (2) identify recommendations for conservation planning to address the earthquake-induced habitat shifts and likelihood of similar effects under sea-level rise.

7.2 Materials and Methods

7.2.1 Study area

The study area is the Avon Heathcote Estuary Ihutai located at 43.5^{o}S, 172.7^{o}E in the city of Christchurch (Fig. 7.1). The estuary is located between the Waimakariri River and the southern end of a large sandy bay (Pegasus Bay) where it is a prominent local feature (Kirk 1979). It is a barrier enclosed tidal lagoon type estuary (Hume et al. 2007) with high ecological and social values including cultural significance for Māori (Jolly & Ngā Papatipu Rūnanga Working Group 2013; Lang et al. 2012). The Avon (Ōtākaro) and Heathcote (Ōpāwaho) are the two major rivers of the estuarine system, both of which provide *G. maculatus* spawning habitat. (Note: the bilingual naming convention used elsewhere in this thesis when referring to these culturally important rivers is not used hereafter in this chapter for the sake of brevity). These waterways are spring-fed lowland rivers with average base flows of approx. 2 and 1 cumecs respectively (White et al. 2007). They are also among the most well-studied spawning locations in New Zealand with surveys having been conducted periodically since 1988 (Taylor et al. 1992).

7.2.2 Geospatial analyses

We analysed spawning site data from post-earthquake studies comprising of seven independent surveys conducted over two years during the peak spawning months using a census-survey methodology designed to detect all spawning in the catchment (Orchard & Hickford 2018a). The areas surveyed were approximately 4 km reaches in each river extending from the saltmarsh vegetation zone (downstream), to 500 m upstream of the inland limit of saltwater (Fig. 7.1). The dataset of 188 records provided details of 121 spawning occurrences in the Avon and 67 in the Heathcote. Each record included upstream and downstream coordinates of the spawning site, mean width of the egg band, and area of occupancy (AOO) of eggs, with each site being defined as a continuous or semi-continuous patch of eggs.

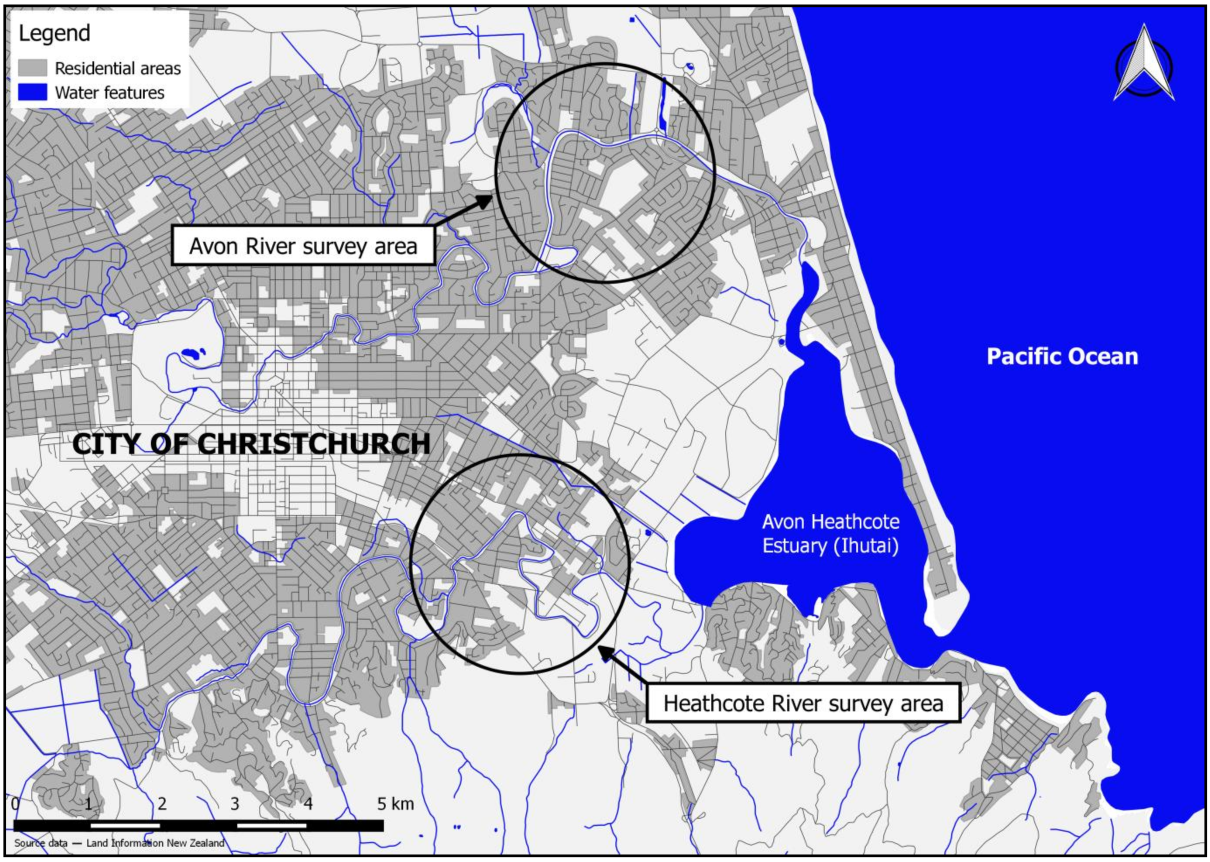

Fig. 7.1 Location of post-earthquake survey areas for *G. maculatus* spawning habitat in the Avon River Ōtākaro and Heathcote River Ōpāwaho, city of Christchurch, New Zealand.

Three spatially explicit protection mechanisms were identified in an analysis of proposed and operative resource management plans (Table 7.2). In this paper we use the term 'protected areas' to denote spatially explicit areas identified in planning methods to address conservation objectives in statutory policies and plans. The areas evaluated in this study are consistent with the IUCN definition of Category IV protected areas being 'areas to protect particular species or habitats, where management reflects this priority' (Dudley 2008). The size of these areas is often relatively small with varying management arrangements depending on protection needs (Stolton et al. 2013).

Protected area and spawning site data were visualised in QGIS v2.8.18 (QGIS Development Team 2016), and reach lengths (RL) calculated in relation to the centrelines of waterway channels digitised from 0.075 m resolution post-quake aerial photographs (Land Information New Zealand 2016b). Three comparable RL metrics were calculated to reflect (a) the RL protected under each planning method, (b) extent of occurrence (EOO) of spawning sites, and (c) the total AOO of spawning sites (Table 3).

The effectiveness of each protection mechanism was evaluated as the percentage of post-earthquake RL^{AOO} located within the PA. Efficiency was considered using two ratios: RL^{EOO} to $RL^{protected}$ and RL^{AOO} to $RL^{protected}$. These reflect the size of the area set aside for protection (in terms of reach length) versus the extent of the spawning reach, and the size of the areas actually utilised for spawning respectively. Each calculation was made on a catchment basis at a yearly temporal scale (i.e. 2015 and 2016), and also using the combined data from both years of post-earthquake surveys.

Table 7.2 Protected area mechanisms for *Galaxias maculatus* spawning habitat evaluated in this study.

Method	Protected area mechanism	Delineation of method in plans	Information source	Planning documents
1	Network of small protected areas based on known spawning sites	20 m diameter areas centred on point data coordinates of known spawning sites, identified in schedule to the plan	Point data and descriptions from NISD† and historical reports (Maw & McCallum-Clark 2015)	Environment Canterbury (2015)
2	Mapped reaches of potential spawning habitat on a catchment basis	Reaches identified in planning maps and referenced in the plan	NISD point data and historical reports coupled with field surveys of riparian vegetation (Margetts 2016)	Environment Canterbury (2014)
3	Mapped polygons of predicted spawning habitat coupled with a text description of where in the polygon the protection requirements apply	Polygons identified in planning maps and GIS layer referenced in the plan	GIS based model of predicted spawning habitat (Greer et al. 2015)	Environment Canterbury (2017)

† National Īnanga Spawning Database

Table 7.3 Metrics calculated to evaluate the effectiveness and efficiency of protected area mechanisms for *Galaxias maculatus* spawning habitat.

Metric	Definition	Calculation method
$RL^{protected}$	Reach length protected areas within a catchment	Combined length of waterway channels falling within protected areas, as calculated from channel centrelines on a catchment basis
RL^{EOO}	Reach length of the extent of occurrence (EOO) of spawning within each catchment during the timeframe under consideration	Total length of waterway channels between the upstream and downstream limits of spawning, as measured along channel centrelines on a catchment basis
RL^{AOO}	Reach length of the area of occupation (AOO) of all spawning sites within each catchment during the timeframe under consideration	Total length of all individual spawning sites, as measured along channel centrelines on a catchment basis

7.3 Results

An overlay of each protection mechanism on combined post-quake spawning site data is provided for each of the study catchments in Fig. 7.2. The three protected area mechanisms provided considerably different $RL^{protected}$ values reflecting their spatial basis (Table 7.4). However for each mechanism the $RL^{protected}$ was comparable between catchments.

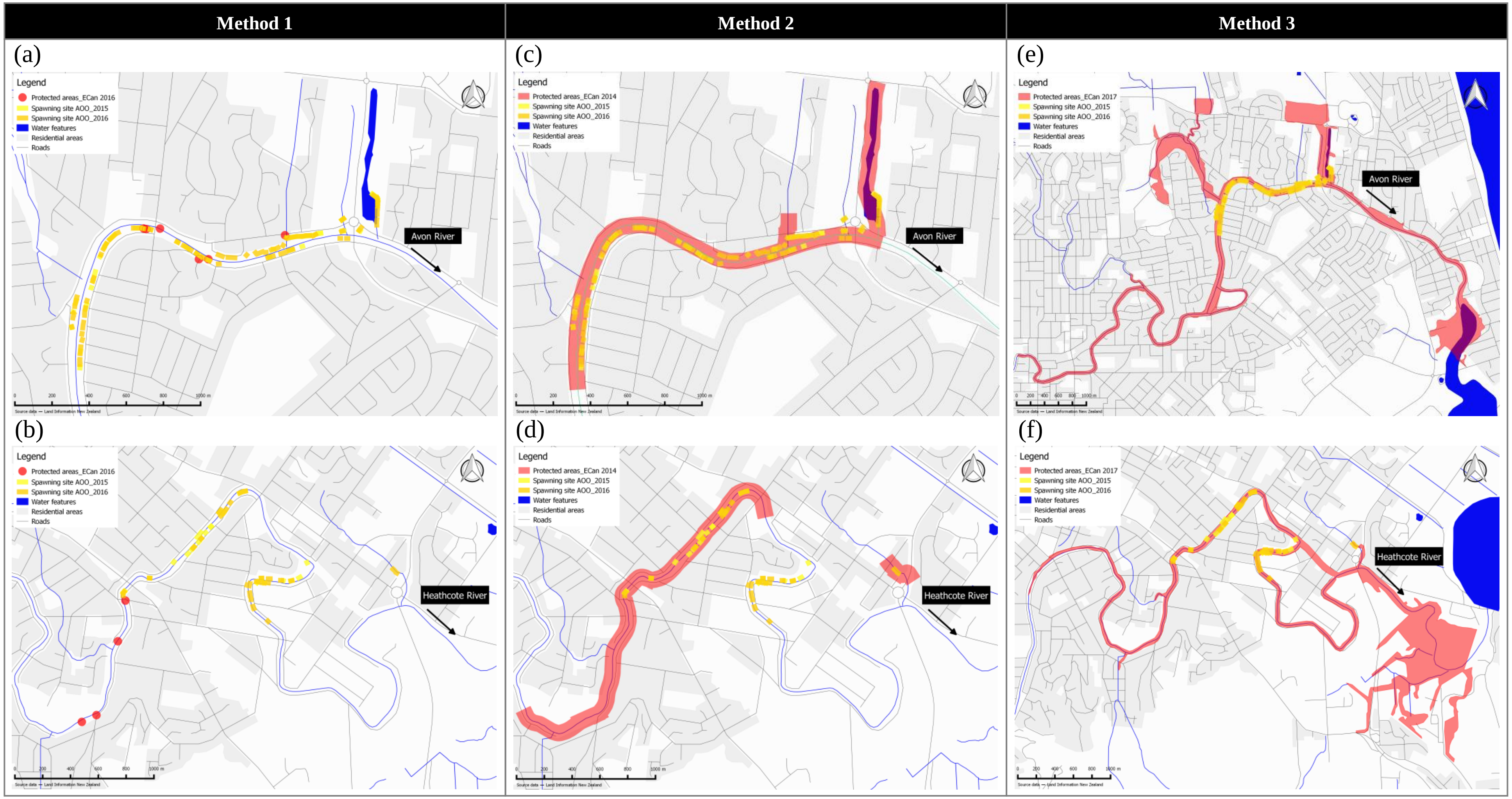

Fig. 7.2 Overlay of the spatial extent of three protection mechanisms found in conservation plans on the footprint of post-earthquake *G. maculatus* spawning sites recorded in 2015 (n = 85) and 2016 (n = 103). (a) Method 1, Avon River, (b) Method 1, Heathcote River, (c) Method 2, Avon River, (d) Method 2, Heathcote River, (e) Method 3, Avon River, (f) Method 3, Heathcote River.

Table 7.4 Reach length (RL) protected by each of the three protected area mechanisms evaluated in the two study catchments.

Method	Description of protected area mechanism	Reach length protected (m)	
		Avon River	Heathcote River
1	Network of small protected areas based on known spawning sites	120	80
2	Mapped reaches of potential spawning habitat on a catchment basis	3230	3098
3	Mapped polygons of predicted spawning habitat coupled with a text description of where in the polygon the protection requirements apply	19100	16600

Method 3 was highly effective at protecting spawning habitat, achieving 92.7% protection in the Avon and 100% in the Heathcote using the combined post-quake data (Table 7.5). The anomoly in the Avon relates to a few spawning sites that occurred outside of the mapped polygon in the vicinity of a small tributary, and this occurred in both years. In the Avon, the effectiveness of method 2 was similar with close to 100% achieved (Table 7.4). However in the Heathcote, only 69.9% of spawning habitat fell within the protected area and 45.6% in 2016. This reflected the occurrence, in both years, of spawning downstream of the protected area (Fig. 7.2d). In comparison, the effectiveness of Method 1 was low. The percentage of habitat protected ranged from 3.9–14.2% (Table 7.4). This reflected the extent to which spawning occurred at previously known sites which formed the basis for delineation of the PAs (Fig. 7.2a & 2b).

Table 7.5 Effectiveness of three protected area mechanisms for *Galaxias maculatus* spawning habitat following earthquake-induced landscape change.

Protection mechanism	Time period	Percentage of habitat protected (% RL^{AOO})	
		Avon River	Heathcote River
Method 1	2015	5.4	7.5
	2016	14.2	6.3
	2015+2016	9.3	3.9
Method 2	2015	96.9	69.9
	2016	99.0	45.6
	2015+2016	98.0	52.5
Method 3	2015	96.9	100
	2016	96.5	100
	2015+2016	97.2	100

In the efficiency evaluation, all of the protection mechanisms were relatively inefficient in terms of land use allocation when the evaluation metric was RL^{AOO} (Fig. 7.3a). For all methods, more than half of the $RL^{protected}$ was allocated to areas that were not utilised for spawning habitat over the study period, even when the areas allocated were very small and targetted at previously known spawning sites. The highest percentage overlap with RL^{AOO} was 47.5% achieved by method 1 in the Avon in 2016. However, when the evaluation metric was RL^{EOO} the percentage overlap results changed considerably. Method 1 achieved a 100% overlap in the Avon in both years but in the Heathcote only 12.5% (Fig. 7.3b). Method 2 achieved 67.6% overlap in the Avon (2016) and 48.7% in the Heathcote (2016), whilst method 3 achieved 11.5% in the Avon (2016) and 17.6% in the Heathcote (2016).

Comparing these results, Method 3 was the least efficient in terms of land use allocation for the purposes of protection in all comparisons in the Avon. However, in the Heathcote method 1 was even less efficient in terms of RL^{EOO} (Fig. 7.3b). This reflected that the protected areas identified were not well located in relation to the areas utilised for spawning (Fig. 7.2). In the Avon, the PAs under method 1 were much better located with all PAs overlapping the RL^{EOO} (Fig. 7.3b). In terms of RL^{AOO} method 1 also performed better in the Avon versus the Heathcote as a result of the PAs coinciding several of the areas actually utilised. However, even here the efficiency of the PA mechanism was rather variable with 47.5% of the $RL^{protected}$ overlapping with spawning sites in 2016 but only 17.5% in 2015 (Fig. 6.3a). This variability is associated with the repeat use of some, but not all, previously used spawning sites between years.

Overall, method 2 produced relatively consistent results in the efficiency comparisions between years. This reflects that the RL^{EOO} was similar in both catchments between years and also located in a similar position in the catchment versus the reaches mapped for protection. Within the RL^{EOO} the total RL^{AOO} was also very similar between years (Avon 386 m^2 and 410 m^2, Heathcote 133 m^2 and 158m^2 for 2015 and 2016 respectively) despite considerable variation in the location of the sites used each year.

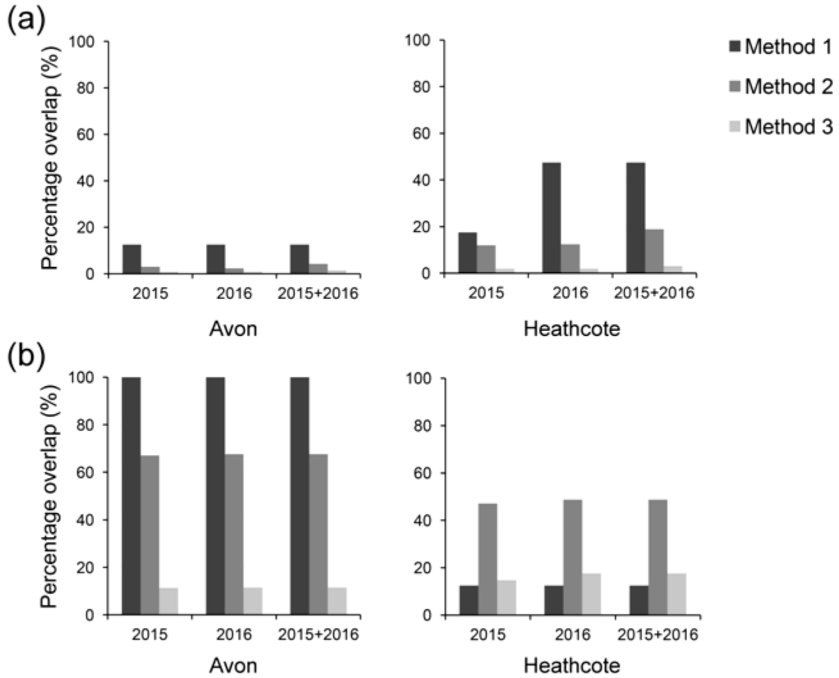

Fig. 7.3 Evaluation of the efficiency of three protected areas in terms of land use allocation using two assessment metrics. (a) percentage of reach length protected ($RL^{protected}$) overlapping the reach length of the extent of occurrence (RL^{EOO}). (b) percentage of $RL^{protected}$ overlapping the total reach length of areas occupied by spawning sites (RL^{AOO}). In all cases RL is calculated on the centreline of the waterway channel. For each calculation three time periods are considered.

7.4 Discussion

7.4.1 Addressing spatiotemporal variation

Several aspects of *G. maculatus* spawning site ecology are potential sources of spatiotemporal variation. The reported relationship with salinity results in horizontal structuring along the axis of waterway channels in relation to saltwater intrusion (Richardson & Taylor 2002; Taylor 2002). This may drive variability in the position of spawning reaches on a catchment scale when coupled with dynamism of river discharges and tidal forcing. Despite that previous studies have highlighted use of the same spawning sites for multiple years (Taylor 2002), this case was characterised by habitat shifts in both catchments in comparison to all known records. Although the potential effects of salinity changes have seldom been highlighted in the literature, this study indicates that they may important in relation to perturbations from extreme events or to incremental changes such as sea-level rise. However, a lack of historical salinity data for the reaches of interest makes the degree of variation difficult to confirm directly in our study area, and this is generally the case elsewhere.

In addition, the timing of spawning on or soon after the peak of the tide, combined with preference for shallow water depths, leads to vertical structuring of the habitat in relation to water level heights (Benzie 1968a; Mitchell & Eldon 1991). Interaction between the waterline and floodplain topography also influences the distance between spawning sites and the alignment of (i.e. perpendicular to) waterway channels. This variation may be considerable where the topography is relatively flat and is a further consideration for effective PA design.

7.4.2 Evaluating PA effectiveness for dynamic habitats

There are at least three aspects of this study that are likely to be applicable to the design and evaluation of Category IV PAs elsewhere. They include the question of PA boundary setting in relation to the habitat to be protected, the need for data to inform this and including as conditions change, and practical considerations for identifying boundaries on the ground as required by stakeholders.

Clearly, accuracy is important when setting boundaries for Category IV PAs, yet spatiotemporal variation may hamper acquisition of the necessary data in practice. For *G. maculatus* strong temporal trends are a particular consideration. Variation has been reported in relation to the peak days of activity within a tidal sequence, the tidal sequences preferred in different parts of the country, and months of most spawning activity in the year (Taylor 2002). International studies have also reported large-scale variation in traits associated with spawning (Barbee et al. 2011). In combination, these aspects suggest that spatiotemporal variability could arise at multiple scales creating practical difficulties for both empirical data collection and model-based approaches for determining habitat distribution. In this case, the study catchments are New Zealand’s best studied spawning areas yet surveys have only been periodic and seldom comprised more than one month in any given year (Taylor 2002). Consequently, the times of peak spawning activity may not have been captured in the survey record. Identification of the spawning distribution has therefore relied on the compilation of multi-year data despite the potential for confounding factors associated with both short and longer term change.

Albeit that the post-earthquake context represents a major perturbation, the impacts of spatiotemporal variance on PA effectiveness are clearly seen in planning methods 1 and 2. These methods were developed using the planning authority’s up to date information on spawning habitat in both catchments. Particularly in the Heathcote, earthquake-induced habitat shift rendered these methods relatively ineffective. Despite this, regular monitoring and amendment of the same protection mechanism could provide a strategy for maintaining effectiveness and addressing change. However, for Method 1 the data collection requirements would be onerous to achieve this in practice. This partly reflects reliance on a network of small PAs but also that the detection of spawning sites is difficult (Orchard & Hickford 2018a). The number of PAs identified appears woefully inadequate in light of the post-quake data, yet fairly represents results of the monitoring effort that was in place pre-quake. Increasing this to the level of a census-survey for peak spawning months represents a considerably scaling-up of the monitoring programme.

In comparison, Method 3 was based on considerably larger PAs and was much more resilient to earthquake change. In that case, a degree of redundancy was seen as a desirable aspect for resilience (Greer et al. 2015). However, from the perspective of PA evaluation, the three PA mechanisms share similar monitoring requirements. This arises since demonstration of PA management effectiveness requires information on the values to be protected (Stoll-Kleemann 2010). This exemplifies the need for more management-driven science to close the gap between conservation policy and practice (Knight et al. 2008), such as monitoring strategies to inform priorities for data collection and frequency (Teder et al. 2007). Potential strategies include using abiotic proxies for conservation objectives for which data acquisition is easier thus reducing the burden of repeat measurement (Lawler et al. 2015). Method 3 provides an example of this approach, using a predictive model based on elevation above sea level (Greer et al. 2015). However, the results indicate that its efficiency as a planning method is relatively low since much of the area set aside did not help achieve the stated objectives, and it could not be used as a proxy for outcomes monitoring against the relevant policy objectives. From an ecosystem-based perspective, inefficient planning methods may also hinder other potential uses of the areas involved, leading to unnecessary trade-offs (Southworth et al. 2006). The practical aspects of this relate to the rules that apply within the PA and are designed to confer protection. Where a degree of sustainable use is envisaged within PAs, the specific arrangements for management need to be well matched to intended objectives.

Efficiency may be a particular important consideration for Category IV PA evaluation where the management context is characterised by high land-use intensity in adjacent areas (Dudley 2008). In this case, Method 2 offered an alternative approach that identified the known EOO and additional areas considered be ‘potential’ habitat and included these in the areas delineated for protection (Margetts 2016). Essentially this created a buffer around the mapped EOO that served to address limitations in the information available for quantifying known habitat, as well as a providing a degree of redundancy to improve resilience. Although in the Heathcote the post-quake habitat was found to have shifted outside of these areas, they were effective in accommodating the smaller magnitude of change observed in the Avon (Fig. 7.2). Management effectiveness evaluation of methods 2 and 3 primarily requires information on EOO as could be obtained by regular census-surveys of spawning habitat (Orchard & Hickford 2018a). The combination of an evaluation-informed adaptive approach and degree of redundancy could offer an effective and efficient PA strategy for the dynamic habitat with regards to land use allocation.

Lastly, this case highlights some practical issues for the visualisation of PA boundaries. In our evaluation, spatial co-occurrence was based on coordinates describing the upstream and downstream extent of spawning sites and polygons describing PAs. In many instances the spawning site locations were very close to the PA boundaries as mapped. Unless they were clearly outside of the boundaries, such sites were assessed as being protected with the result being an optimistic view of the spatial coverage of the PA mechanism. In reality these boundaries may not be so clear. However, it is important that they *are* clear for the benefit of all stakeholders (Langhammer et al. 2007), and this depends considerably on design and communication of the planning methods. In this case the areas delineated by method 1 were interpreted by stakeholders using a location description and schedule of coordinates (Table 7.2). This is considered to offer a relatively clear mechanism for implementation of the PA requirements in practice.

Under Method 2, the areas for protection were first visualised as lines in Council planning documents (Margetts 2016) and then subsequently incorporated into ‘Sites of Ecological Significance’ (SESs) in a recent statutory plan (Christchurch City Council 2015), which is now operative. The visualisation method for plan users is a set of polygons annotated on planning maps appended to the plan (Supplementary Material, Figure S7.1). These SESs have therefore become the PAs of interest and Method 2 (as assessed in this study) can be interpreted in relation to *G. maculatus* objectives within these larger areas. However, at the scale of the mapping provided it is difficult to see exactly where the PA boundaries lie in the riparian zone, requiring considerable guesswork by plan users (Supplementary Material, Figure S7.2).

Under Method 3 the situation is improved by the provision of PA polygons as a public dataset with an online GIS viewer available, in addition to planning maps appended to the relevant plan (Environment Canterbury 2017). Nonetheless, similar boundary issues arise with regards to the exact location of the PA in relation to the spatial extent of spawning habitat.

The GIS analysis revealed a few spawning sites that were clearly outside of the PA boundary in the Avon, as reflected in effectiveness results of <100% in both years (Table 7.5), and in general many of the actual spawning locations were again very close to the PA boundary. Furthermore, the habitat may shift a considerable distance from the low flow channel on high water spawning events, and these circumstances are difficult to detect by operators (e.g. management contractors) in the field. Indeed spawning sites were found to have been destroyed by the City Council's own reserve management contractors despite being protected in the relevant statutory plan (Orchard et al. 2018b). Better guidance materials, such as interactive maps, may be required to improve PA effectiveness in practice, as was recommended in a recent management trial that aimed to avoid such damage to spawning sites (Orchard 2017b). The analysis also shows that the imprecise nature of the vertical scale in conservation area mapping is a particular issue in this case, and the inclusion of a buffer would appear warranted as an aspect of PA design in addition to the above points. The results also highlight the importance of management effectiveness evaluation as an activity that should include specific attention to efficacy of conservation measures in terms of both their design and implementation.

7.4.3 Assumptions and limitations

Several assumptions have been made in this evaluation consistent with a focus of the protection of dynamic habitats and the objective of identifying learning from the unique post-earthquake situation. Most importantly, the focus has been restricted to the spatial basis of protection mechanisms for critical habitat as found in planning documents. In all cases they were assumed to confer protection where spatial overlap occurred. In reality, this also depends considerably on the design of the rules that apply within the PA and aspects such as the provision of compliance monitoring. Also, a conservative approach has been taken in the mapping of PA boundaries and protection assumed to be effective. In the case of Method 2, the width of the riparian zone protected could not be accurately identified and all spawning sites with the protected reach were assumed to be covered. Other limitations of the study include the spatial coverage of post-quake surveys in relation to Method 3 since the full extent of those PAs was not directly surveyed. Despite this the spatial coverage of the surveys was extensive in both catchments and the methodology was designed to capture the upstream and downstream extents of the full habitat distribution (Orchard & Hickford 2018a). Different evaluation results can also be expected in light of new information. In particular the number of spawning events captured in the post-quake survey record is limited. Further spatiotemporal variation may arise from effects such as differing water heights outside of the sampled range, future vegetation change, river engineering impacts, the potential for further ground level changes, and the ongoing influence of sea-level rise.

7.5 Conclusions

This evaluation was conceived to challenge PA thinking. Firstly, our evaluation extends the discussion of PA management effectiveness towards that of resilience. Although management actions within existing PAs may help increase the resilience of natural resources, the realities of global change create a fundamental challenge that demands a range of approaches (Baron et al. 2009). In this case, the PAs involved are small and are best thought of as PA networks under the management of local and regional government entities. Yet in all respects they meet the definition of Category IV PAs and are found nationwide in recognition of their statutory role and origins.

Although a focus on critical habitats is just one dimension of protected areas management, it is an important function in terms of their role as a management tool and wider contribution to spatial planning. Importantly, attention to relatively fine scales may offer practical opportunities for integrating PA systems into the wider land and seascape (Guarnieri et al. 2016). For example, small and dynamic PAs have the potential to help fill representation gaps in PA networks as is a critical need in lowland river and floodplain systems (Tockner et al. 2008). In addition, an understanding of the role of PAs in climate change adaptation has been steadily developing but there is much work to be done. For example, new questions to assess the effects of climate change on PAs have only recently been employed in Management Effectiveness Tracking Tool (METT) evaluations despite its long history and widespread use (Stolton & Dudley 2016). Through investigations of change following an extreme event this study provides insights into similar considerations. Our findings suggest that adaptive networks of well targeted and relatively small PAs could produce an effective mechanism for responding to change thereby contributing to system resilience. Whether new or traditional PAs networks can be adapted along these lines deserves further research. We predict this will become a key topic for environmental planning and conservation management in the years ahead.

7.6 Acknowledgements

We thank Environment Canterbury and Christchurch City Council staff for information on planning methods and riparian management activities. Funding was provided by the Ngāi Tahu Research Centre and a New Zealand Ministry of Business, Innovation and Employment grant (C01X1002) in conjunction with the National Institute of Water and Atmospheric Research.

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Supplementary Material

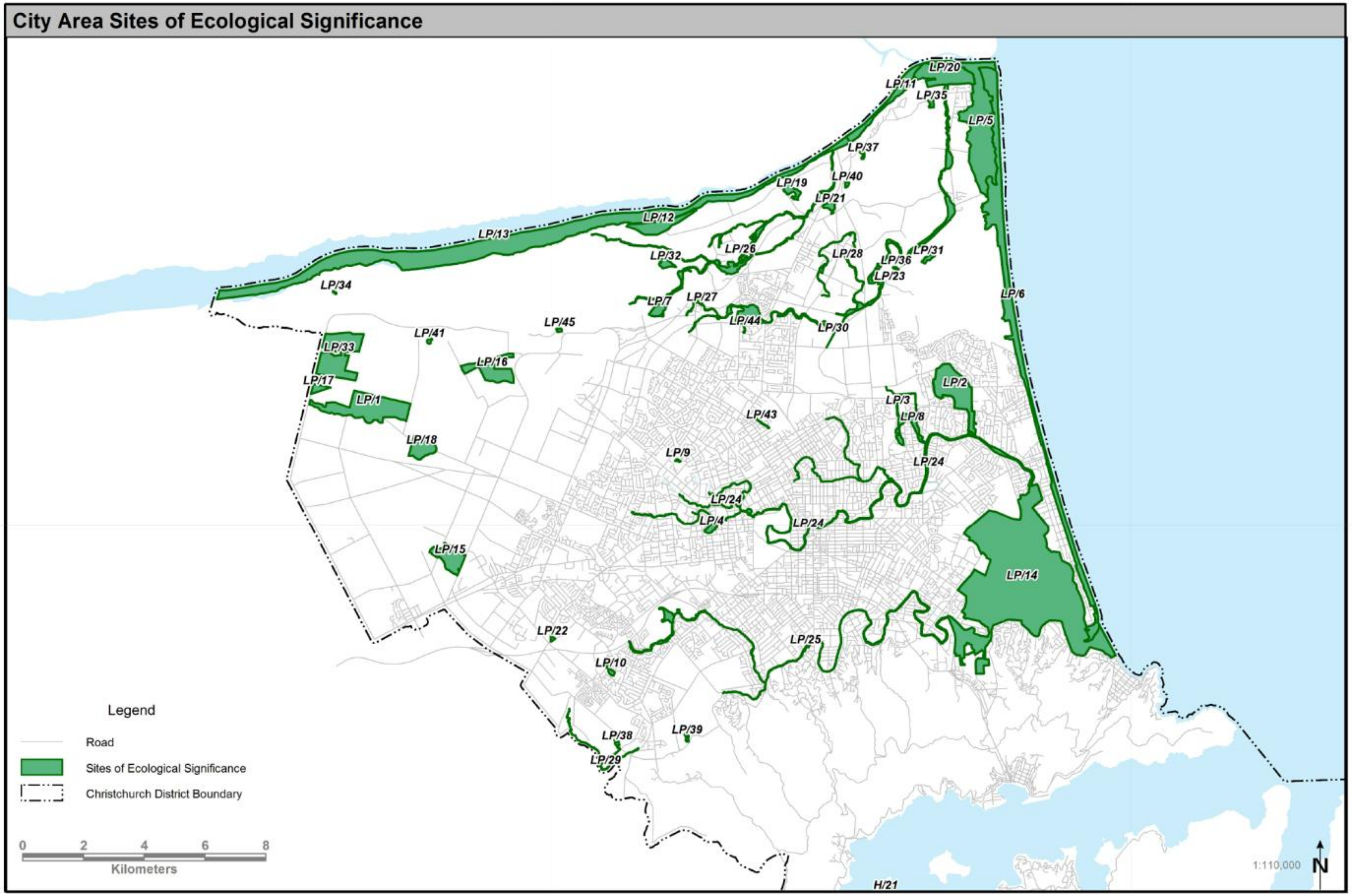

Fig. S7.1 Reference map showing Sites of Ecological Significance (SESs) in the Christchurch City area (Christchurch City Council 2015).

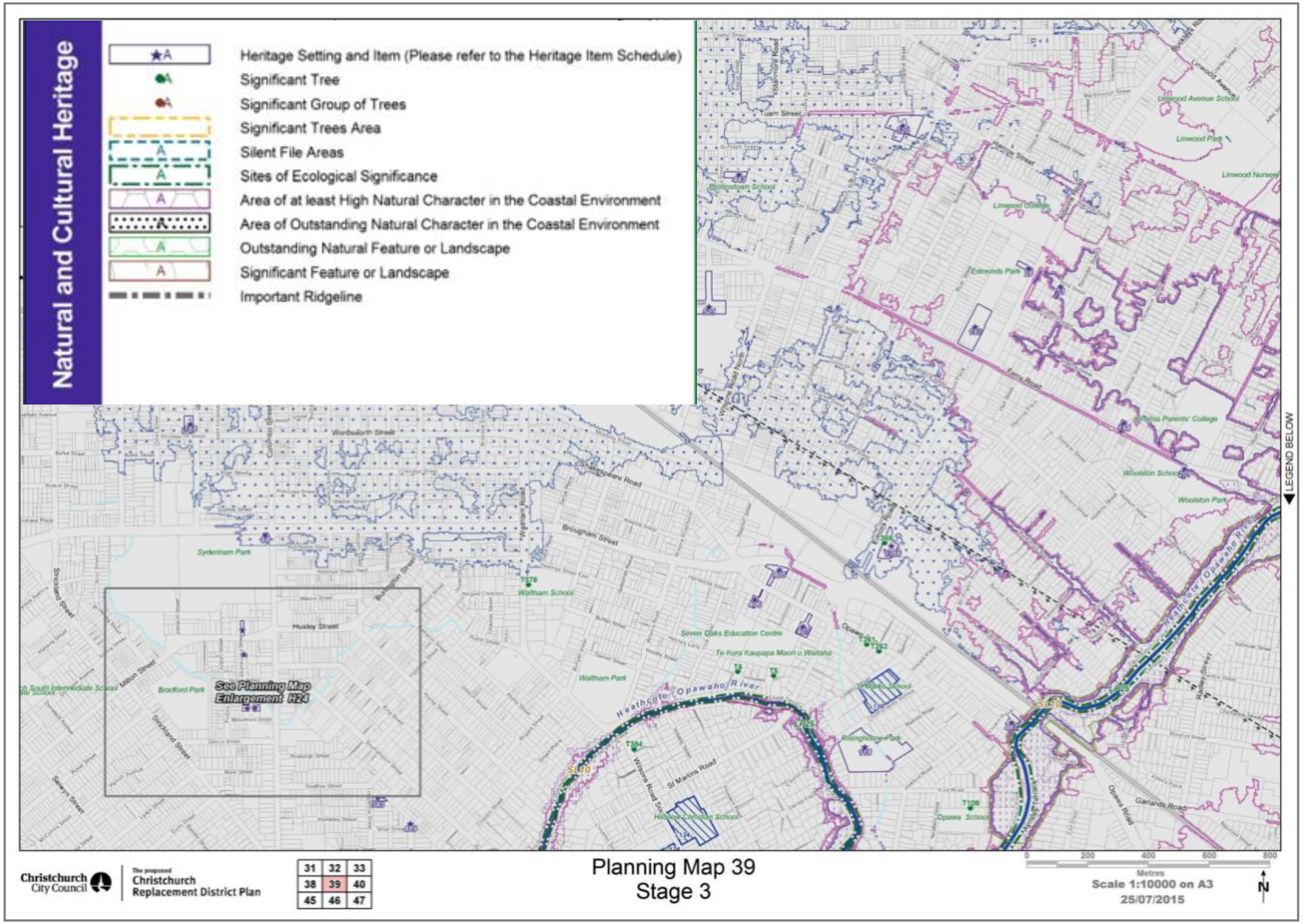

Fig. S7.2 Example of a detailed planning map showing Sites of Ecological Significance (SESs) in Christchurch. Note lack of enlargements / further detail to delineate SESs in riparian zones (e.g. for the Heathcote River Ōpāwaho, bottom right). For brevity only an excerpt of the full legend is shown (Christchurch City Council 2015).

Chapter 8

Synthesis of contributions

8.1 Introduction

This research has sought to improve the understanding of tectonic displacement associated with the Canterbury Earthquake Sequence and its implications for the management of rapid sea-level rise (Fig. 8.1). In the following sections I draw together the most significant findings in relation to the prior state of knowledge, describe their wider contribution to the global literature, and discuss their relevance for adaptation to climate change.

8.2 Land movement and climate change

The relationship between land mass displacements and eustatic sea-level is becoming better understood due to technological advances and increased attention to the sea-level rise implications of climate change (Ostanciaux et al. 2012; Pedoja et al. 2011). This is captured in the term ‘relative sea-level’ (RSL) referring to the position of the land in relation to the sea (Cahoon 2015). Deep displacement mechanisms include crustal movements associated with tectonic plates and isostatic responses to stress redistribution associated with processes such as glacial de-loading (Shugar et al. 2014; Stammer et al. 2013). Additional surface elevation changes may result from shallow sources of uplift and subsidence such as the movement of hydrocarbons and groundwater, decay of organic material and compaction of sediments (Cahoon et al. 1995; Rybczyk & Cahoon 2002; Woodroffe et al. 2016). The motion of land masses has direct relevance for the development of climate change adaptation strategies by potentially mitigating or exacerbating eustatic sea-level rise effects and driving additional interactions with erosion and sedimentation processes (Church et al. 2013; Nicholls & Cazenave 2010; Simms et al. 2013).

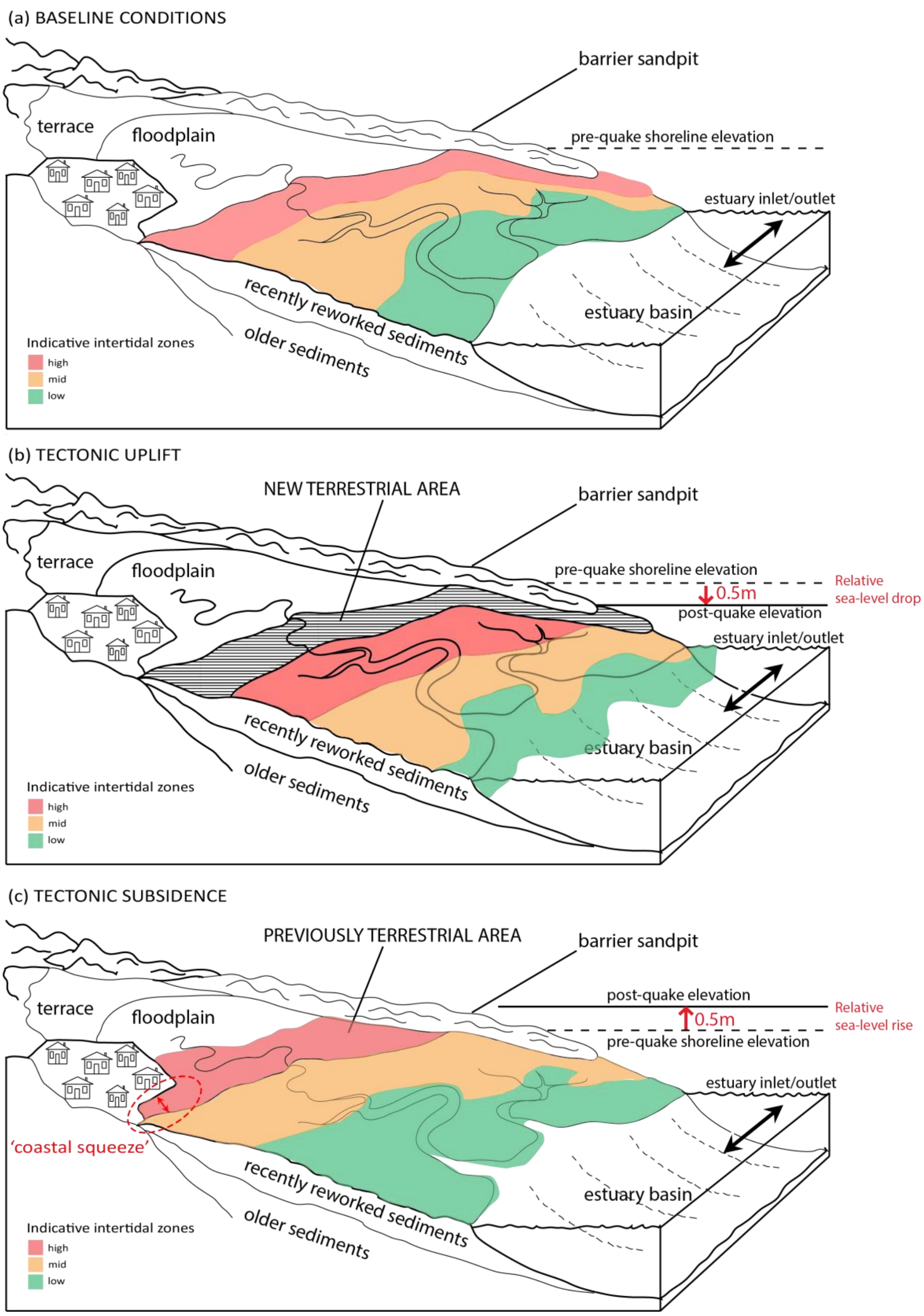

Fig 8.1 Conceptual diagram of the relationship between relative sea levels and vertical land mass movement induced by tectonic events. (a) generalised schematic of baseline (pre-earthquake) conditions in a typical barrier built tidal lagoon showing indicative tidal zones (high, mid, and low) within the intertidal range. (b) uplift of the land mass causes relative sea levels to drop on the pre-existing landscape. New terrestrial areas are formed above the reach of the tide. Intertidal zones and their characteristic habitats migrate seaward and equilibrate towards new stable states. Reworking of typography is also expected due to the influence of altered hydrology. (c) subsidence of the land mass causes relative sea levels to rise on the pre-existing landscape. Previously terrestrial areas are exposed to tidal inundation. Intertidal zones and their characteristic habitats migrate landward and equilibrate towards new stable states. 'Coastal squeeze' occur where anthropogenic (or natural) barriers act to reduce the availability of space for characteristic coastal communities to re-locate. Reworking of typography is also expected due to the influence of altered hydrology.

Globally these phenomena are relatively well studied from palaeo-environments (Heward 1981; Milne et al. 2009). The reconstruction of temporal changes from preserved sites has illustrated coastal landform evolution and ecosystem development driven by fluctuations in relative sea level (Knight et al. 2017; Norder et al. 2018). Several studies have identified relative sea-level changes associated with major earthquakes (Blanchon & Montaggioni 2003; Shchetnikov et al. 2012; Stiros & Pirazzoli 2005). These non-climatic factors have a strong bearing on vulnerability to climate change and yet may be difficult to accommodate in human-scale management cycles (Duarte 2014; Nicholls et al. 2008). For example, compared to glacial isostatic adjustments that are amenable to modelling (Peltier 2004; Spada 2017), tectonic events are unpredictable and can involve highly variable regional effects (Stammer et al. 2013; Stiros & Pirazzoli 2005).

8.3 Tectonic subsidence and rapid sea-level rise

Aside from their contribution to the longer term relative sea-level trend, rare examples of tectonic subsidence can directly demonstrate the effects of rapid sea-level rise where they occur in coastal environments (Albert et al. 2016; Reed 1990; Saunders et al. 2016). Although the rates of sea-level change are more rapid than the equivalent results of climate change, they are nonetheless illustrative of extreme scenarios. The observed responses may help to identify the mechanisms that can lead to adverse effects, and conversely, strategies to help avoid them. Recent examples provide unique opportunities to investigate sea-level rise responses in contemporary socio-ecological contexts and it is becoming increasingly important to understand these processes and their outcomes. Furthermore, the prospect of accelerated rates of sea-level rise is now a central topic within the climate change discourse (Cazenave & Llovel 2010; Nicholls et al. 2011). Important consequences for coastal communities include the reduction of timelines for adaptation processes and the potential for greater impacts in the advent of delays or poorly designed responses. Because of the severe consequences associated with run-away climate change, many authors have recommended the consideration of additional extreme scenarios as an element of preparedness and in recognition of inherent uncertainties in current predictions (Duarte 2014; Nicholls et al. 2014; Polasky et al. 2011). These aspects indicate that tectonic subsidence events can offer useful insights for longer term climate change adaptation in addition to their more immediate needs in the context of disaster recovery.

The present study investigated a significant tectonic displacement event on the east coast of New Zealand. Not only was it highly destructive with unusually high peak ground accelerations and tragic loss of life (Bradley et al. 2014; Potter et al. 2015; Quigley et al. 2016), it generated a lasting displacement of the coastal environment with accompanying RSL changes of up to 0.5 m. Globally, there have been few empirical studies of similar events due to their relative scarcity in modern times. Other contemporary examples include deep subsidence caused by a mine collapse in Australia (Rogers et al. 2019) and tectonic subsidence associated with earthquakes in South America (Reed 1990; Reed et al. 1988), California (Jacoby et al. 1995), and the Solomon Islands (Albert et al. 2016; Albert et al. 2017; Saunders et al. 2016). However, other insightful studies of rapid sea-level rise have come from dramatic examples of shallow subsidence in large river deltas (Al Mukaimi et al. 2018; Schmidt 2015). Prominent examples include the Mississippi and Mekong deltas where large scale wetland losses have been linked to subsidence trends (Day et al. 2000; Morton et al. 2010; Phan et al. 2015; Storms et al. 2008). A previous New Zealand study has also linked shallow subsidence with mangrove responses in the Firth of Thames (Swales et al. 2015).

In this study, the combination of a large magnitude tectonic displacement event and sensitive location characterised by low-lying coastal environments offerred a rare opportunity to improve the current state of knowledge on the phenomenon of RSL rise and the vulnerabilities of aquatic boundary environments. Aspects of particular interest include the very sparse distribution of such studies globally, and the relatively small number of ecosystem types and societal contexts that have been investigated. The research was positioned within the fields of water resource management, conservation ecology and the science-policy interface. A summary of the new contributions to knowledge is provided in the following sections.

8.4 Previous research

Previous research on the Canterbury Earthquake Sequence (CES) includes a wealth of studies on its unique tectonic aspects such as the activation of previously unrecognised faults (Beavan et al. 2012; Bradley & Baker 2015; Bradley et al. 2014; Kaiser et al. 2012) and surface deformation effects (Bastin et al. 2016; Cubrinovski & Robinson 2016; Quigley et al. 2013;

Reid et al. 2012; Robinson et al. 2012). Other studies include investigations of societal impacts (Hughes et al. 2015; Potter et al. 2015; Robinson et al. 2012; Taylor et al. 2012) and recovery processes (Berno 2017; Brundiers 2018; Gjerde 2017; Hayward 2013; Kenney & Phibbs 2015). In comparison, there have been relatively few studies of ecological impacts despite the potential for broad-scale changes associated with surface deformation and hydrological alterations (Allen et al. 2014; Cubrinovski & Robinson 2016; Hughes et al. 2015; Quigley et al. 2016).

Early studies on the Avon Heathcote Estuary Ihutai included mapping of topographical changes and liquefaction mounds in the estuary bed (Measures et al. 2011; Reid et al. 2012) and potential effects on benthic fauna (Zeldis et al. 2011; Zeldis et al. 2019). Measures & Bind (2013) developed a hydrodynamic model to investigate tidal inundation changes and calculated a mean tidal prism change of 14.6% based on the difference between pre-earthquake and post-earthquake bathymetries. Campbell et al. (2013) documented the degradation of a previously impounded wetland (Bexley Wetlands) following subsidence and tidal water intrusion and Hayward et al. (2015) investigated sedimentary signatures of subsidence at another nearby saltmarsh. Reid et al. (2017) monitored salt marsh vegetation changes with a focus on glasswort (*Sarcocornia quinqueflora*) to document biological responses to vertical deformation and consider their congruence with other surface elevation change data. EOS Ecology (2012) investigated the effects of untreated sewage discharges and sediment mobilisation on the fauna of the lower rivers though a lack of comparable pre-earthquake data precluded direct before-after comparisons. Orchard (2016b) documented post-earthquake vegetation patterns and salinity conditions at Anzac Drive Reserve in the Avon Ōtākaro catchment but was also unable to evaluate earthquake effects due to a lack of prior data. Similarly, Boffa Miskell (2018) compiled information on the ecology of the lower Avon Ōtākaro catchment but with a focus on summarising current terrestrial ecology values rather than determining earthquake changes. However, a cultural health assessment of the Avon Heathcote Estuary Ihutai catchment (Lang et al. 2012) compared post-earthquake conditions with baseline measurements made in 2007 and highlighted several degradation trends, and a recent study has reported the expansion of seagrass (*Zostera muelleri*) beds versus a 2005 baseline (Gibson & Marsden 2016). Related work has focussed on aspects of the post-earthquake landscape for the management of natural hazards (Bosserelle et al. 2019; Lane et al. 2017; Todd et al. 2017) and the potential effects of sea-level rise (Tonkin & Taylor 2017).

The above summary indicates that surprisingly few studies have investigated natural resource management implications of the earthquakes despite the potential for large-scale and persistent changes and their relevant for post-disaster planning. Some of the most notable research gaps are the relatively few attempts to assess the resilience of natural environments to relative sea-level effects and the lack of landscape scale studies.

8.5 New research presented in this thesis

8.5.1 Shoreline movements and their consequences

The objective of Chapter 1 was to produce an updated analysis of post-earthquake geospatial changes in the estuarine system including measures of spatiotemporal variance over a five year period since the first major earthquake. This study presents the first analysis of upper intertidal range shoreline movements over the CES and their interactions with pre-earthquake land-uses and infrastructure. The study also contributes new and relatively high resolution measures of ground displacements on the estuarine floodplain based on a dataset of 475, 000 points and quantifies spatial variability across sub-catchments. New data are presented that quantify changes across the full intertidal range at relatively fine scale using overlapping LiDAR datasets and the Delft3D hydrodynamic model developed by NIWA (Measures & Bind 2013). The DEM used in the latter was also compared with updated DEMs to identify hydrologically connected areas outside of the hydrodynamic model extent but subject to tidal inundation. These are particularly important in areas of subsidence that were previously terrestrial areas and not included in the hydrodynamic model (R. Measures, pers. comm.). These were investigated to improve the understanding of previous modelling results showing an overall shallowing of the lagoon and associated tidal prism reduction but with intertidal area gains (Measures & Bind 2013; Measures et al. 2011).

Calculations made using the additional intertidal areas showed that the total area of the estuary had indeed reduced consistent with a dominant uplift effect. A reduction of subtidal area reductions were found to account for most of the intertidal area change within the extent of the hydrodynamic model. These results strengthened the evidence for an overall tidal prism reduction though identified additional spatial variance in the contributions of major sub-basins. In particular, subsidence in the lower Avon Ōtākaro catchment associated with additional tidal inundation area may have the effect of maintaining tidal prism whereas in the Heathcote

Ōpāwaho catchment it has been reduced by uplift. The consequences of these effects are investigated further in Part 2 in relation to potential salinity changes.

These analyses also showed an empirical example of ‘coastal squeeze’ (*sensu* Doody (2004)) caused by interactions between anthropogenic shoreline armouring and RSL changes. Although this phenomenon has been relatively well studied in connection with episodic and decadal coastal erosion trends (e.g., Berry et al., 2013), few studies have shown impacts of the underlying concept under conditions of RSL rise. Such studies (e.g., Phan et al. 2015; Twilley et al. 2016), are important contributions to the global literature on coastal squeeze since the prime mechanism of impact (inundation) differs from its analogy on high energy wave-exposed coasts where erosion is the primary concern. Design principles for accommodating natural ecosystems under these conditions also differ markedly. Impacts identified in this study included a reduction in upper intertidal habitat availability that is relevant to the conservation of internationally important shorebird populations for which the estuary is renowned (Crossland 2013). Implications for land-use planning for disaster recovery and climate change adaptation include clear linkages between science and policy that arise from the conservation status of estuaries (Holdaway et al. 2012) and statutory protections applying to estuarine ecosystems and habitats (New Zealand Government 1987, 1991).

The geospatial investigations reported in Chapter 1 provide a firm basis for the remainder of the study. The combination of ground surface elevation and inundation models provides a rare example of highly resolved relative sea-level height measures at landscape scale over a considerable time period (Higgins 2016). However, because of the strong temporal variance identified over the period 2011-2015 an additional RTK-GPS survey was scheduled for 2019 and incorporated updated geodetic benchmark adjustments made the same year (Land Information New Zealand 2019). The results (reported in Chapter 2) add additional weight to the investigation of RSL effects on aquatic boundary environments and their conservation implications.

Chapter 2 presents the first comprehensive assessment of landscape scale vegetation responses associated with RSL changes in the estuary. These analyses showed that the impacts of vertical displacement varied between vegetation classes. Impacts were not always proportional to the observed RSL changes due to complexities such as lag effects in vegetation response trajectories. Edge effects and small scale dynamics resulting from differential plant tolerances were found to be important mechanisms of marsh degradation that contributed to the formation of holes and overall ecosystem effects. These aspects, also reported in other subsidence studies, indicate that marsh-scale vulnerability depends on interactions with physical drivers of change other than RSL (Macreadie et al. 2015; Mariotti 2016; Orson et al. 1985; Wilson & Allison 2008) and these limit the ability to make inferences from historical accretion rates (DeLaune & White 2012; Doughty et al. 2019; Rogers et al. 2006). The study also strengthens global evidence for spatiotemporal variability in wetland responses to RSL and concurs with the conclusions of Cahoon (2015) and Kolker et al. (2011) on the need to obtain concurrent measures of wetland status and RSL to improve the understanding of confounding factors in vulnerability assessments.

Novel aspects of this study include the identification of anthropogenic factors contributing to the patterns of wetland loss under conditions of RSL rise. This empirical evaluation of socio-ecological aspects has few precedents from southern hemisphere studies although is consistent with calls made by Doughty et al. (2019) for greater attention to local scales in sea-level rise vulnerability assessments. The four risk factors identified are directly applicable to the current post-disaster planning context in the study area and transferable to other coastal communities facing RSL rise.

Another notable finding of this study was the documentation of a long period of wetland surface elevation loss following rapid RSL rise. Although previous studies have also shown disparities between RSL and accretion responses (Anisfeld et al. 2016; Morton et al. 2010), there are a lack of studies showing the role of recovery dynamics following such disturbances (Bertness et al. 2015; Kirwan & Megonigal 2013). The need to account for these in contemporary vulnerability assessments and associated status measures has been highlighted by several authors along with the role of stochastic erosion and depositional events (Cahoon 2015; Morris et al. 2016; Rybczyk & Cahoon 2002). In this case, causes of the ongoing degradation trend are currently unresolved but likely result from interactions between sediment availability, vegetation condition and the potential contribution of shallow subsidence mechanisms, all of which are important to determining future restoration potential.

In Chapter 3, the implications of RSL rise was extended to investigate shoreline conservation prospects for the estuarine barrier sandspit using a novel scenario-based approach that considered processes operating on the two opposing shorelines simultaneously. Although there have been many studies on barrier island dynamics in relation to RSL (e.g., Leatherman 1983, Lorenzo-Trueba & Ashton 2014; Moore 2010; Otvos 2012), this study is one of few we are aware of that have addressed the conservation of peri-urban sandspits despite these being relatively common worldwide. Previous examples include a study of sandspits on the northwest coast of Spain (Lorenzo et al. 2007), storm-induced changes on a non-tidal estuarine sandspit on the southern Baltic Coast (Bugajny & Furmańczyk 2017), and a study of historic shoreline changes on an estuarine sandspit on the north Massachusetts coast (Fallon et al. 2017). Of these, only the latter included assessment of both the open ocean and estuarine backshore coastlines but the response variable was residential property values rather than impacts on the natural environment.

The present study identified significant mid- to long-term challenges for beach and dune conservation on the open ocean shoreline, despite the assumption of a continuing positive sediment balance from fluvial sources to the north. Scenario models showed that sea-level rise is likely to eliminate dunes in a third of the study area without the creation of additional space within a time frame of ca. 100 years based on current climate projections. These results extend the findings of recent a coastal hazard assessment (Tonkin & Taylor 2017) by identifying the likelihood of increasing demand for engineered armouring leading to progressive dune and beach degradation without the inclusion of ecosystem-based measures. This conclusion is consistent with other studies that have identified severe risks for sandy beach ecosystems associated with the combination of RSL rise and coastal squeeze (Berry et al. 2013; Brown & McLachlan 2002; Defeo et al. 2009; Dugan et al. 2010), though nuanced by the highly constrained socio-ecological context of barrier sandspits.

In contrast, a new nature-based solution was identified on the estuarine backshore where government land acquisition presents an opportunity to restart saltmarsh accretion processes seaward of coastal defences. This finding adds an interesting dimension to the evolving managed retreat discourse (Alexander et al. 2012; Hino et al. 2017) and will be the subject of further research to explore managed retreat in the context of sandspits. Additionally, the consideration of both coasts simultaneously highlighted the existence of opposing forces that result in small land volumes above the tidal range. These pinch-points represent areas of heightened vulnerability without adequate defences, or alternatively, potential breakpoints for managed retreat strategies. This study provides the first documentation of these aspects that we are aware of and highlights the need for attention to vulnerabilities on both coasts under climate change.

8.5.2 Salt water intrusion and habitat shifts

Part 2 of the thesis is devoted to the issue of whitebait conservation with a specific focus on īnanga (*Galaxias maculatus*), a riparian-spawning fish that provides the bulk of the New Zealand's whitebait fishery (McDowall 1984). The future sustainability of the fishery is an issue of high cultural importance (Goodman 2018) with relevance to other Pacific countries where similar traditional fisheries are also in decline (Encina-Montoya et al. 2011; Mardones et al. 2008; Vega et al. 2013). Spawning is a critical life stage and known to occur at very specific sites near coastal rivermouths that are vulnerable to degradation and land conversion pressures (Burnet 1965; Hickford & Schiel 2011a). Although spawning locations are the subject of statutory protection, the implementation of effective conservation measures requires their identification.

The overarching objective of the study addressed the hypothesis that RSL changes would drive a shift in the distribution of spawning sites due to a previously reported relationship with the limit of salt water intrusion in coastal waterways (Richardson & Taylor 2002; Taylor 2002). In this case, tectonic displacement was investigated as a natural experiment with the potential to drive habitat shifts, alter vulnerabilities, and demand new conservation strategies to address. Testing the displacement hypothesis involved the development of new methodologies for quantifying spawning habitat (Chapter 4, Orchard & Hickford 2018) that have since been used in other studies (Orchard 2018a, 2018b). The results provide the best example to date of the spatiotemporal variability of spawning habitat that includes quantifying the period of peak spawning activity and relationships between individual spawning events and the overall seasonal pattern (Orchard & Hickford 2018a; Orchard et al. 2018c). In comparison, only a handful of surveys have been completed in most rivers in New Zealand with decades elapsing between surveys and many yet to receive a baseline assessment (Orchard 2019). The present study has contributed over 300 spawning site records, whereas the entire National Īnanga Spawning Database (NISD) contains only 565 records collected over a 25 year period (Taylor 2002). Updated and improved information on spawning locations and vulnerabilities is therefore an important area of future research.

A component of the above work involved the novel use of artificial habitats to overcome the potential for mortality-related detection failures inherent in field survey techniques, as reported in Chapter 5. This approach was deployed alongside the abovementioned surveys with the objective of identifying upstream and downstream spawning limits. Following a successful pilot study in 2015, the technique was up-scaled as part of a collaborative community project funded by the Department of Conservation. In both years the results contributed useful information for the detection of spawning activity in degraded environments and identification of limits. Highly productive spawning sites were located over 2 km downstream of all pre-earthquake records in the Heathcote Ōpāwaho catchment with lesser shifts recorded in the Avon Ōtākaro catchment (Chapter 6), consistent with the tidal prism effects estimated in Chapter 1. Unexpected results included the detection of a much larger spawning site distribution than all previous records in the study area which represent New Zealand's most extensive temporal dataset (28 years). Spawning was also detected at high salinities (up to 18 ppt), contrasting markedly with previous studies that have reported a close association between spawning locations and low salinity conditions (Hicks et al. 2010; Hicks et al. 2013; Taylor 2002).

Conclusions from these studies include a new conceptual model of spawning habitat that is structured by the both the salt water intrusion limit and the occurrence of unfavourable vegetation types as dominant influences. The pronounced habitat shift in the Heathcote Ōpāwaho catchment is thought to be associated with vegetation changes in the riparian zone that were driven by a freshening of the system associated with tidal prism reduction - rather than being the direct the result of a salinity influence on spawning behaviour. In this case, the spatial distribution was influenced by the development of new habitat a considerable distance downstream of an unfavourable stretch of river illustrating the discontinuous nature of habitat migration – and characteristic of fragmented landscapes. A further unexpected result was the finding that the majority of spawning habitat in this catchment is provided by an invasive plant species previously reported as unsuitable for spawning, introducing a new conundrum for riparian management (Orchard & Hickford 2018a). Subsequent work with stakeholders has developed adaptive management measures through spatial planning informed by the post-earthquake research.

Chapter 7 concluded the research programme by evaluating the design of protection mechanisms to address the observed habitat shifts and with a focus on two new statutory planning initiatives that introduced relatively large protected areas (PAs) in the lower river corridors. This investigation contributes to knowledge on the effectiveness of IUCN Category IV PAs that aim to protect particular species or habitats (Dudley 2008). These PAs are often smaller areas that suit integration with other resource management approaches yet may become ineffective under conditions of environmental change due to range shifts of the target species (Leverington et al. 2010). This is currently an active area of global conversation research due to its high relevance for the development climate change adaptation strategies (Coad et al. 2015). However, this is first such study that investigated these aspects for *G. maculatus* in connection with RSL rise.

The previous approach in Canterbury (and common elsewhere) based on protecting small areas at known spawning sites was found to be largely ineffective at dealing with habitat shifts unless coupled with regular and extensive monitoring. It had historically relied on the notion of spawning sites occupying very similar locations between years. Although this has been reported in many rivers (Benzie 1968a; Richardson & Taylor 2002) this research has found that it may not represent the general case and is unreliable as a basis for PA strategies. On the other hand, the use of large PAs was associated with inefficiencies arising from spatial disparities between their boundaries and the values requiring protection. Additional findings related to the potential importance of fine-scale conservation area mapping, particularly in the vertical dimension in this case, and the critical role of effective methods for communicating these details to stakeholders on the ground. Adaptive PAs were identified as an alternative to reduce unnecessary trade-offs with other desirable land-uses. These may deliver high effectiveness from a network of smaller PAs by combining a degree of redundancy to address known spatiotemporal variability with the ability to respond to longer term range shift as informed by monitoring or modelling. The conceptual aspects of this study are widely transferable for the design of other Category IV PAs for protecting key species or habitats in dynamic environments where integration with other land-uses is required.

Overall the above studies have contributed valuable new information on a range of topics important to *G. maculatus* conservation including the spatial ecology of spawning habitat, patterns of vulnerability, effectiveness of management arrangements, and potential effects of RSL rise.

8.6 Opportunities for further research

Although the research presented here has increased the state of knowledge, it also highlighted the importance of quantifying spatiotemporal variability when determining changes and trends. Beneficial future research directions include increasing the extent of hydrodynamic models to facilitate more detailed analysis of inundation patterns over a greater proportion of the study area and to include simulations of RSL rise. Updated bathymetric data is also needed in view of the recent ground level changes reported here. Salinity changes could also be incorporated in hydrodynamic modelling and these have been the subject of a preliminary calibration study using the NIWA model (see Orchard & Measures 2016). Further research is also needed to monitor changes on the barrier sandspit and within the coastal wetland systems of the study area. Aspects of particular interest include land-use planning for government acquired land on the sandspit and investigating restoration strategies for both the dune system and riparian wetlands that are commensurate with other concerns of the affected communities in addressing climate change.

Within the wider lagoon basin there is an excellent opportunity to improve the understanding of saltmarsh recovery dynamics given the unusually long lag period discovered in this research. Attention to relationships between RSL changes, sediment accretion rates and vegetation composition are all important. Additionally, this research should be extended to riparian wetlands further upstream in the study area and to include future accommodation space. These areas will become increasingly important for climate change resilience with predicted sea level rise and yet key conservation planning decisions are likely to be made relatively soon. There are considerable opportunities for innovation in this space. The government acquired ‘red zoned’ lands are important subjects for research across all aspects of their socio-ecological context and potential future benefits.

There are several promising directions for further research on aspects on *G. maculatus* ecology and management. These include application of the census survey approach to other coastal waterways to facilitate a better understanding of spawning habitat. There is a wealth of opportunities to improve basic ecological knowledge important to conservation management including the relative contributions of different rivers to egg production and effectiveness of management approaches including areas closed to fishing and habitat restoration initiatives. Other opportunities include building on the salinity relationships reported in this study to test their wider applicability in different river and estuary types. These relationships are likely to be useful for the development of predictive habitat models to reduce the need for physical surveys. Opportunities for further research include investigating and testing these approaches in a range of hydrodynamic and geographical settings and exploring their utility for conservation planning and climate change vulnerability assessments.

8.7 Linkages between disaster recovery and climate change adaptation

This work used a combination of field survey, remote sensing and scenario modelling to identify the unknown impacts and implications of a rare historical event. The research programme was largely exploratory and made several unexpected findings that validate the initial premise around the significance of learning opportunities presented by the CES. Attempts were also made to evaluate the natural environment changes and their conservation implications from the perspective of a socio-ecological context characterised by recovery from a natural disaster and an increasing awareness of climate change. This focus has been maintained throughout all of the papers herein with the hope of illustrating specific science-policy linkages, and where possible, practical solutions for incorporating natural environment values in engineering projects, natural hazard management and environmental planning initiatives as fundamentally important activities for both disaster recovery and adaptation to climate change. Moreover, strong linkages between the two have been identified that relate partly to the need for responses to relative sea-level change but significantly from the upheaval, reconfiguration and re-investment context inherent in disaster recovery which presents opportunities to rethink past development patterns and explore alternative approaches that may be beneficial in the longer term.

Many of the findings of this study are widely transferable to other coastal communities and ecosystems. Tidal lagoons such as the Avon Heathcote Estuary Ihutai are found on coastlines worldwide (Hume et al. 2016; Hume et al. 2007), and the socio-ecological context is typical of many coastal communities facing the pressures of rising sea-levels and increased competition for space and resources (McGranahan et al. 2007; Neumann et al. 2015; Small & Nicholls 2003). In assessing the degree of transferability to other contexts, is important to note the existence of an enabling disaster recovery context afforded by the two managed retreat areas that resulted from government acquisition of formerly residential land. This in itself represents a significant and innovative societal response that greatly facilitates the opportunities for re-thinking the future. Although this process has yet to come to fruition it was a strong motivation for conducting the research in the hope that the information gained might promote the development of nature-based solutions.

Other disaster recovery and climate change adaptation contexts may be characterised by significant spatial constraints in aquatic boundary environments and there is considerable debate on the role of innovative policy approaches such as biodiversity offsetting (Bull et al. 2013; Curran et al. 2014; Gordon et al. 2015), managed retreat (Alexander et al. 2012; Hino et al. 2017) and ecosystem-based adaptation (Brink et al. 2016; Renaud et al. 2016). The approach taken here has illustrated the utility of high resolution monitoring techniques and scenario-based simulations as a means of identifying the spatial requirements of natural environments within dynamic and often contested landscapes.

8.8 References

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Appendix 1 Funding partners

Waterways
Centre for Freshwater Management

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