Identifying indicators of active dune condition from remotely sensed aerial imagery

Cate Ryan

PhD

2023

Identifying indicators of active dune condition from remotely sensed aerial imagery

Cate Ryan

A thesis submitted to

Auckland University of Technology

in fulfilment of the requirements for the degree of

Doctor of Philosophy (PhD)

Dec 2023
Primary Supervisor: Prof. Hannah Buckley
Second Supervisor: Assoc. Prof. Bradley Case
Third Supervisor: Dr Craig Bishop

School of Science

Abstract

Coastal active dunes, which face global threats such as sea level rise, play a pivotal role in providing ecosystem services and supporting biodiversity. Despite this, they remain inadequately studied. This thesis investigated the question: Can the ecosystem condition of active dunes be discerned from remotely sensed imagery, specifically aerial imagery, at a national scale? The research addressed the challenge of efficiently monitoring these ecosystems by investigating the potential of remotely sensed imagery for monitoring their condition. Specifically, low-cost, high-resolution red-green-blue aerial imagery at a national scale for Aotearoa New Zealand was used as a case study. First, the potential maximum extent of active dunes was estimated using geospatial datasets and aerial imagery. Second, the vegetation cover within 135 x *c*. 1 ha plots of a nationally-representative set of active dunes was characterised using Object-Based Image Analysis. This resulted in a national typology of seven, dominant vegetation types, including two native sandbinder types. Third, the spatial pattern of active dune cover classes within plots on active dunes revealed distinct patterns in dunes dominated by native sandbinders, associated with geomorphic disturbance, while patterns associated with exotic and woody vegetation indicated anthropogenic influences. These results demonstrated that native sandbinder dominance can serve as an indicator of active dune condition. Thus, this study has provided a methodology to derive useful indicators of active dune condition and offers a feasible, cost-effective approach for large-scale conservation management decisions of this ecosystem and lays the groundwork for future research in coastal active dune ecosystems at a national scale.

Contents

List of Figures

List of Tables

List of Appendices

Chapter 2

Chapter 3

Chapter 4

Attestation of Authorship

I hereby declare that this submission is my own work and that, to the best of my knowledge and belief, it contains no material previously published or written by another person (except where explicitly defined in the acknowledgements), nor material which to a substantial extent has been submitted for the award of any other degree or diploma of a university or other institution of higher learning.

Cate Ryan

11/12/2023

Co-authored works

Chapter 2 Authors: Cate Ryan, Bradley Case, Craig Bishop, Hannah Buckley Ecosystem integrity of active sand dunes: a case study to implement and test the SEEA-EA global standard, from Aotearoa New Zealand (Published in Ecological Indicators, Volume 149, May 2023)	Ryan, C. 80 % Buckley, H. 10 % Case, B. 5 % Bishop, C. 5 %
Contribution: CR, HB, BC, CB conceived the ideas and designed methodology; CR collected the data; CR and HB analysed the data; CR led the writing of the manuscript. All authors contributed critically to the drafts and gave final approval for publication.	
Chapter 3 Cate Ryan*, Hannah L. Buckley, Craig D. Bishop, G. Hinchliffe, Bradley C. Case Quantifying vegetation cover on coastal active dunes using nationwide aerial image analysis (Submitted to Remote Sensing in Ecology and Conservation, 04/12/2023)	Ryan, C. 80 % Case, B. 10 % Buckley, H. 5 % Bishop, C. 2.5 % Hinchliffe, G. 2.5 %
Contribution: CR, BC and HB conceived the ideas and designed methodology; CR collected the data; CR, HB and GH analysed the data; CR led the writing of the manuscript. All authors contributed critically to the drafts and gave final approval for publication.	

Hannah Buckley Bradley Case Craig Bishop Graham Hinchliffe

Acknowledgements

Thank you to my supervisory team, Hannah, Brad and Craig, who together brought an amazing depth of expertise and perspectives to this thesis. Hannah and Brad, thank you for believing in me and giving me the opportunity to take this PhD journey, I am very grateful. Hannah, thank you for your endless patience, enthusiasm for plants and ecology and for teaching me what it is and how to be a scientist. Brad, thank you for giving me the foundation in geospatial science for this project, and your insights. Craig, thank you for your ecological knowledge and questions. Graham, thank you for your advice and support; it made a real difference. Mike Petterson, thank you for the geology advice and readings. Nicola, thank you for the encouragement, thesis / science-chats and advice on preparing my talk for my first international conference. To the students I taught, I really enjoyed this part of my PhD, thank you for your enthusiasm and all the great questions.

To my funders, the NZ Department of Conservation (DoC), Auckland University of Technology and the NZ Coastal Restoration Trust, without your support this thesis would not have been possible. Elaine Wright, Terry Green, Meredith McKay and Kate McNutt from DoC , thank you for the interest in this topic and reviews. I would also like to thank the many people who helped me collate records about active dunes in Aotearoa especially Amy Hawcroft and Graeme LaCock from DoC, and the National Vegetation Survey Databank (NVS).

To my lab group and cohort, covid made our PhD experience truly unique! You inspired me to keep going and keep learning. Shanta, Dexter, Feby, Chloe, Fabian, Michelle, Sarah, Lynda, WS315, the Living Labs crew – without the laughs this would have been much harder!

To my friends who have supported and seen me through the toughest times, you are the very best of humans. Emma, you encouraged me to go down this route and never stopped believing in me. Ella, you reminded me to keep things real, demanding regular walks, meals and pets. Mum, I wish you could see me now.

To Steve, my husband, I dedicate this thesis to you. You've taught me to always get back up and never give in. You are my northern sky.

1. General Introduction

Coastal active dunes (active dunes) occur at the interface between land and sea in association with ice-free, sandy shorelines worldwide (Psuty et al., 2004). They can be visualised as a patchwork of constantly shifting and changing morphologies with sparse, low-growing vegetation (Castillo & Moreno-Casasola, 1996; Olsen & van der Maarel, 1997). Active dunes form above the mean high-water mark by sand blown up from the beach (Hesp, 2002) due to the geomorphic forces of the sea and wind and ecological processes (Biel et al. 2019; Durán & Moore, 2013; Hacker et al., 2012; Jay et al., 2022; Miller et al., 2010; Ruggiero et al. 2018; Zarnetske et al. 2012, 2015).

1.1 Complex, reciprocal feedbacks initiate and evolve active dunes

Active dunes are ecogeomorphic habitats whereby ecological and geomorphological processes interact, creating complex reciprocal feedbacks and characteristic ecosystem structure and function (Fig., 1; Biel et al., 2019; Charbonneau et al., 2020; Corenblit et al., 2011, 2015; Durán & Moore, 2013; Hacker et al. 2012, 2019; Jay et al., 2022; Miller et al., 2010; Ruggiero et al., 2018; Zarnetske et al., 2015) (Fig.1). Geomorphological processes result from the interaction between geomorphic drivers, which provide the conditions or materials for geomorphic change, such as sediment, and geomorphic agents, which transport sediment and shape the landscape, such as wind, waves and running water (Davidson-Arnott, 2011; Huggett., 2007). Ecological processes are activities that result from interactions and connections between organisms and their environment (Bland et al., 2016; Lovett et al., 2006; Martinez, 1996; Pettorelli et al., 2018; United Nations [UN], 2021). Ecosystem structure is the organisation and the distribution of biotic and abiotic components of ecosystems (Bowman & Hacker, 2021). Traditional measures of biotic structure from field ecology include species diversity (the number of different species and species abundance within a defined area) and the composition of species within a defined area (Bowman & Hacker, 2021). Alternative measures are living biomass, the configuration of different habitats in a landscape (spatial heterogeneity), the size and connectivity of habitat patches and the encroachment of woody vegetation into a non-woody habitat (Mücher et al., 2023; Petorelli et al., 2016; Skidmore et al., 2015; UN, 2021). Ecosystem function can be defined as the results of ecosystem processes that benefit one or more species (Lovett et al., 2006; Pettorelli et al., 2018), for example flows of energy, such as food, resistance to disease, resilience to disturbance (Bowman & Hacker, 2021; Petorelli et al., 2018) or habitat provisioning for a range of species (Pettorelli et al., 2018). Proxy measures of ecosystem structure are often used to measure functions, for example species diversity (Bowman & Hacker, 2021; Pettorelli et al., 2018), but alternative measures include indicators of ecosystem processes, for example the amount of bare soil as an indicator of erosion and sedimentation processes (Mücher et al., 2023). The relationship between structure and function can take a range of directions from positive, where the addition of species makes no difference to ecosystem functions after a certain threshold; idiosyncratic, where certain species have a more than proportionate effect on function and other non-linear directions (Bowman & Hacker, 2021). The disruption of ecosystem processes can threaten ecosystem change, often irreversible and is caused by introduced activities and components such as human settlement into new areas and invasive plants and animals (Bland et al., 2016).

1.1.1 Active dune ecosystem structure and function

1.1.1.1 Abiotic processes

Geomorphic drivers

Geomorphic drivers are external inputs and influences on ecosystems (Huggett, 2007). In active dunes, these include climate and associated sea levels, geology and geomorphology, wave climate (fetch, sea swell and storms) and tidal range (Davidson-Arnott, 2010). A supply of sand is the first input and driver of active dune formation (Biel et al., 2019; Durán & Moore, 2013; Hacker et al., 2012; Hesp, 2011; Zarnetske et al., 2012, 2015). The foundations of the sand supply of many present-day coastal dunes globally, were laid in sea level fluctuations in the late Pleistocene and Holocene. In the Pleistocene ice ages, sea level fell, the ocean floor was exposed, and sand was blown inland. When sea levels rose in the early and mid-Holocene, coastal features were eroded and redistributed to form dunes (Bird, 1997; Muckersie & Shepherd, 1995; Hugenholtz & Wolfe, 2005). The main source of sand supply to coastal dunes today comes from fluvial processes, whereby rivers at the coast discharge sediment from erosion, floods and weathering in the upper part of catchments, and cliff erosion (Bird, 1997). Tephra and loess also add significant sediment loads to rivers where volcanic eruptions occur (Bird, 1997).

Geomorphic processes

Marine and aeolian sediment transport processes deposit sand on coastal shorelines through the geomorphic agents of currents, wind and wave action (Davidson-Arnott, 2010; Maun, 2009; Psuty, 2004), which are primarily driven by past and present climate forcing (Hugenholtz et al., 2012). Marine processes of longshore drift and the prevailing currents carry sand along the coast, and wave energy deposits it on low-gradient shorelines (Shepherd & Hesp, 2003). Fine to medium-grained sand is blown up and deposited on the beach where wind speeds are sufficiently high to mobilise sand granules (Davidson-Arnott, 2010; Hesp, 2011; Short & Hesp, 1982). Wind and wave action control the formation of dune landforms in two main ways: 1) ambient conditions with consistent wind and wave energy build foredunes gradually through accretion, creating regular foredunes and dune swales (da Silva et al., 2008; Eisma, 1997; Miller et al., 2010). 2.) Natural disturbance (disturbance) from high wind and wave energy during storms, combined with flooding from rain and overwash, causes erosion and sand movement (Eisma, 1997; Maun, 2009; Miller et al., 2010) (Fig.1). Sand movement from disturbance caused by destructive wind and waves, particularly burial, is a major selective force for dune vegetation and an internal control on dune landforms (Davidson-Arnott, 2010; Maun, 2009). Besides the destructive mechanistic effects of burial and erosion (Grime, 2001), disturbance affects nutrient and moisture availability and decreases diversity since few species can tolerate burial conditions (Maun, 2009; Moreno-Casasola, 1986) (Fig. 1).

1.1.1.2 Biotic processes

Plant functional roles

Few plants can survive in the harsh coastal dune environment (Maun, 2009), and their characteristics play a functional role in dune morphology by interacting with and contributing to active dune ecosystem processes (Biel et al. 2017, 2019; Hacker et al., 2012; 2019; Jay et al., 2022; McGuirk et al., 2022; Olsen & Van der Maarel, 1997; Reijers et al., 2020; Ruggiero et al., 2018; Stallins et al., 2005; Zinnert et al., 2016; Zarnetske et al., 2012, 2015), since plant architecture controls the movement and accumulation of sand around vegetation (Biel et al. 2017, 2019; Davidson-Arnott, 2010; Gao et al., 2023; Jay et al., 2022; Johnson, 1993; Hacker et al. 2012, 2019; Hesp, 1989; McGuirk et al., 2022; Maun, 2009; Miller et al., 2010; Olsen & Van der Maarel, 1997; Ruggiero et al., 2018; Zarnetske et al. 2012, 2015). Groups of dune plant species are variously adapted to disturbance from sand movement; the so-called “dune-builders”, also known as sandbinders, are specialised plants that have a positive growth response to burial (Stallins et al., 2005); burial-tolerant stabilisers often comprise the vegetation of dune swales and salt marshes and are adapted to overwash during storms (Stallins, 2005; Zinnert et al., 2016); and burial-intolerant species have vertical root structures that cannot sufficiently spread the strain of sand burial (Stallins et al., 2005; Zinnert et al., 2016).

Colonisation

Sandbinders colonise bare sand and survive on the most active parts of dunes (Hesp, 1989; Hesp, 2021; Maun 2009). Specific adaptions of sandbinders include lateral growth habits and rapid growth and reproduction (Hacker et al. 2012, 2019; McGuirk, 2022; Olson & van der Maarel, 1997). While germination from seeds is possible for many sandbinders, resulting in shadow, incipient or hummocky dunes, rapid vegetative reproduction from rhizomes or stolons is common (Maun, 2009; McGuirk et al., 2022).

Rhizomes are spreading root systems with nodes that produce roots and shoots that grow upwards through sand; examples of sandbinders with rhizomatous root systems are *Ammophila arenaria* (L.) Link, a European weed species established in temperate climates around the world (International Union for the Conservation of Nature [IUCN], 2024a), *Ammophila breviligulata* (Fern), a beach grass that is widespread on the Atlantic and Great Lakes coasts of the USA (United States Department of Agriculture, 2006) and the sedge *Ficinia spiralis* (A.Rich.) Muasya et de Lange, endemic to Aotearoa New Zealand (Bergin., 2011a). Stolons are horizontal stems, or runners, that produce shoots and roots from nodes that send up new shoots when buried (Hacker et al. 2012, 2019; McGuirk, 2022). An example of a stoloniferous sandbinder is *Spinifex sericeus* R.Br., a grass native to Australasia (Bergin, 2011b). Rhizomes and stolons of sandbinding plants can form dense belowground structures, and recent studies have shown that this belowground biomass is important for stabilising dune environments and reducing wave-induced erosion, with different species having different effects (Charbonneau et al., 2017; Walker & Zinnert, 2022).

Other functional groups of plants lack traits to survive the disturbance events on active dunes, or other stressors such as regular exposure to sea spray, saltwater inundation, strong winds, waves, rain, and rapidly changing temperatures (Hacker et al., 2012; 2019; Maun, 2009). Thus, diversity has been observed to be low or absent in these environments (Castillo & Moreno-Casasola, 1996; Ciccarelli & Bacaro, 2016; Da Silva et al., 2008; Forey et al., 2008; García-Mora et al., 1999; García-Novo et al., 1997; Hacker et al. 2012, 2019; Martínez et al., 2001; Miller et al., 2010; Moreno-Casasola, 1986; Stallins et al., 2005) (Fig. 1).

Facilitation

Sandbinders act as ecosystem engineers through the key biotic process of facilitation. Sandbinders create ecosystem structure by stabilising local areas of sand creating geomorphic diversity and generating microclimates such as shelter and shade. This in turn provides key ecosystem function of habitat provision that supports the establishment of a wide range of other plant and animal species

(Charbonneau et al., 2022; Holdaway et al., 2013), often with high levels of species endemism (Martínez et al., 2013; Van der Maarel & Van der Maarel-Versluys, 1997). In 1997, a conservative estimate of c. 9000 taxa were recorded in coastal dunes globally, albeit with incomplete descriptions in many countries (Van der Maarel & Van der Maarel-Versluys, 1997). The portion of these c.9000 coastal dune taxa that fall within active dunes has not been estimated (Van der Maarel & Van der Maarel-Versluys, 1997) (Fig. 1).

Zonation

Over time, the composition and abundance of plant communities reflect the local disturbance regime, for example, the type, frequency, duration and magnitude of disturbance (Castillo & Moreno-Casasola, 1986; Miller et al., 2010), both spatially and temporally. By building dunes, sandbinders modify the topography between storms (Hovenga et al., 2021; Miller et al., 2010), creating local stabilising effects until the next disturbance disrupts the stabilising process (Hovenga et al., 2021; Miller et al., 2010). The distribution and evenness of the functional groups of dune builders, burial-tolerant species and burial-intolerant functional groups will vary along the shore, in tandem with the prevailing disturbance regime (Stallins et al., 2005; Zinnert et al., 2016) and a spatial pattern called “zonation” occurs, whereby a mosaic of plant communities occurs within zones that reflect gradients of disturbance with distance inland from the beach (Doing, 1985; García-Novo et al., 2004; Maun, 2009; Miller et al., 2010; Moreno-Casasola, 1986; Psuty, 2004; Van de Maarel, 1997).

1.1.1.3 Dune landforms

A gradient of sand mobility occurs in most dune systems from the seaward to the most landward dunes due to decreasing exposure to coastal processes (Davidson-Arnott, 2010). This gradient can be broadly represented by three main landforms: primary dunes (the most active), secondary (semi-stable) dunes and stable dunes (Davidson-Arnott, 2010). Primary dunes are fed by sand supply from the beach and comprise incipient dunes and foredunes (Davidson-Arnott, 2010). Incipient dunes form near the mean high-water mark when grains of sand are blown up the beach and accumulate around an obstacle such as vegetation, driftwood or shells (Biel et al., 2019; Durán & Moore, 2013; Davidson-Arnott, 2010; Hacker et al., 2012; Hesp, 2002; Jay et al., 2022; Ruggiero et al., 2018; Zarnetske et al. 2012, 2015.). Over time, established foredunes evolve from incipient dunes, taking a variety of forms dependent on reciprocal geomorphic and ecological processes (Corenblit et al., 2011, 2015; Durán & Moore, 2013; Miller et al., 2010; Zarnetske et al., 2015), but they are generally parallel with the shore, convex, symmetrical to asymmetrical dune ridges and perpendicular to the prevailing winds (Davidson-Arnott, 2010).

On accreting coasts, shorelines prograde and new incipient dunes form in front of established foredunes (Davidson-Arnott, 2010) which becomes increasingly sheltered from coastal processes and sand supply from the beach (Davies 1980; Davidson-Arnott, 2010). These modified primary dunes become secondary dunes and are more stable, but they can still be subject to substantial sand movement by wind and overwash during storms (Davidson-Arnott, 2010). The main secondary dune landforms include blowouts, parabolic dunes, and transgressive sand sheets and dunefields, although the latter three can also be primary (active) dunes when they occur adjacent to the beach (Davidson-Arnott, 2010; Hesp 2002, 2011).

Blowouts are formed through wind-driven erosional processes and are typically trough, bowl, or saucer-shaped depressions that form in the absence of vegetation (Hesp, 2002, 2011). Parabolic, or U-shaped or V-shaped dunes often form as a result of continued sand transport from winds through blowouts (Hesp 2002, 2011). Transgressive sand sheets and dune fields are wind-driven sand deposits and form downwind or alongshore of the beach, over vegetated to semi-vegetated land (Hesp, 2011). Transgressive sheets are often flat to undulating, while transgressive dune fields comprise a variety of dune forms (Davidson-Arnott, 2010; Hesp et al., 2022).

1.1.2 The relative importance of abiotic and biotic components of ecosystem structure and function

The relative contribution of external controls, such as geomorphic drivers and agents, compared to internal controls, such as vegetation type or beach morphology on active dune ecosystem processes is poorly known (Biel et al., 2019; Davidson-Arnott, 2010; Durán & Moore, 2013; Hovenga et al., 2021, 2023; Jay et al., 2022; Keijsers et al., 2016; Ruggiero et al., 2019; Zarnetske et al., 2015). It is widely considered that sand supply is the most important driver of active dune ecosystems (Biel et al., 2019; Hacker et al., 2012; Hesp, 2011; Jay et al., 2022). However, the relative influence of aeolian sediment transport, vegetation and sand supply has been discussed in the literature: Keijsers et al. (2015) suggest that foredune vegetation patterns influence sand deposition and accretion, but since foredune vegetation grows in both eroding and accreting dunes, the reciprocal does not conclusively occur. However, the authors suggest that the effect of sedimentation on nutrients and other factors that limit vegetation may be relevant (Keijsers et al., 2016); Moore et al. (2016) suggest that on prograding coasts, vegetation controls create conditions for the formation of multiple dunes due to feedbacks internal to the dune system, and that this may be more important than the role of external factors such as sediment supply; Konlechner and Hilton (2022) make the case that wind speed, wind direction and vegetation characteristics are the most critical influences on dune evolution following disturbance, reinforcing or dampening the feedbacks between vegetation and geomorphic processes through time, similar to Lancaster (1988); and González-Villanueva et al. (2023) found that wind-stilling caused a net decrease in foredune recovery following storms over 105 years on the Iberian coast, Spain.

In recent years, a suite of process-based models have been developed to test boundary conditions and dune responses, to understand the relative importance of the constituent inputs of biotic and abiotic active dune processes (Davidson-Arnott et al.,2018; Durán & Moore, 2013; Hovenga et al., 2021, 2023; Keijsers et al., 2016; Roelvink & Costas, 2019; Ruggiero et al., 2019). However, the models are still immature and require in-situ data for a range of spatial and temporal scenarios to robustly estimate dune responses to different inputs.

The traits of different sandbinder species have also been shown to be highly influential in determining dune shape and mobility (Biel et al., 2019; Hacker et al., 2012, 2019; Hesp et al., 1989; Jay et al., 2022; Reijers et al., 2019; Zarnetske et al., 2015). Hacker et al. (2012) compared active dune profiles in two congener species (members of the same taxonomic genus) on the Pacific coast of the USA. The authors suggested that differences in dune morphology were likely due to a combination of variable sand supply and species-specific traits, affecting foredune size and shape. They showed that at many of the beaches where *A. breviligulata* dominated, foredunes were shorter and wider and had a shallower slope at these beaches compared to sites where *A. arenaria* dominated, which had taller, thinner and steeper foredunes. They also noted that where *A. breviligulata* dominated, beaches had ample sand supply. They suggested that because *A. breviligulata* produces more lateral rhizomes with larger tillers than *A. arenaria*, it can more efficiently colonise the newly created bare substrate associated with disturbance from bare substrates, compared to the tussock-like growth form of *A. arenaria*, which has more vertical rhizome growth (Hacker et al., 2012).

Other studies have suggested that vegetation plays an important role at different temporal scales. Biel et al. (2019) and Jay et al. (2022), found that species specific plant density had a large influence on dune morphology at the interannual scale. However, Zarnetske et al. (2015) found that on prograding shorelines on the pacific coastline of the USA at the decadal scale, the characteristic dune-building effects of invasive *A. breviligulata* explained more of the variation in dune height.

1.2 Ecosystem services

Ecosystem services are the benefits humans obtain from ecosystem functions (Lovett et al., 2006). Active dunes provide important ecosystem services; they attenuate storm-driven wave energy (Barbier et al., 2008; Hacker et al., 2012; Ma et al., 2024; Ruggiero et al., 2001; Walker & Zinnert, 2022) protecting human settlements and infrastructure from flooding (Barbier et al., 2011; Hesp, 1989; Stockdon, 2007; Walker et al., 2013; Walker & Zinnert, 2022). Active dunes also sustain cultural practices, recreation and tourism (Barbier et al., 2011).

Sandbinder plant traits affect dune height, width and plant density (Hacker et al., 2012; Reijers et al., 2019; Seabloom et al., 2012; Zarnetske et al. 2012, 2015), thus, the dominant sandbinder species could affect the extent to which dunes can provide coastal protection. Many authors suggest that since *A. arenaria* builds tall dunes, it is better suited to provide coastal protection services compared to *A. breviligulata*, which builds low-elevation, wide dunes (Hacker et al., 2012; Reijers et al., 2019; Seabloom et al., 2012; Zarnetske et al. 2012, 2015). However, Itzken et al. (2020) suggest that the width of beaches and dunes must also be considered. They found that low and wide dunes associated with wide beaches lose less volume than tall and narrow dunes during longer and more intense storms, thus proving less vulnerable and more resilient to extreme weather (Itzken et al., 2020). Similarly, Zinnert et al. (2016) suggest that tall, narrow dune systems are vulnerable to erosion and that the interplay of a range of dune landforms and functional plant roles can create a system on a continuum of resilience and resistance to overwash associated with storms and sea level rise scenarios (Zinnert et al., 2016).

1.3. Threats

Despite their importance, the extent of active dunes is declining worldwide (Gao et al., 2020; Jackson et al., 2019; Martínez et al., 2004), and the composition of dune flora is rapidly changing (McGuirk, 2022). Legal protection for active dunes is weak in many countries (Heslenfeld et al., 2004), and significant pressure from farming, forestry, human settlement, tourism, recreation and extractive industries has resulted in net losses in extent (Gao et al., 2020; Martínez et al., 2004). For example, the removal of native dune vegetation for coastal development has altered geomorphic and ecological processes, causing imbalances in the sediment budgets and exacerbating erosion and sand movement, leading to a decline in extent (Martínez et al., 2004). Large-scale intentional planting and invasions by stabilising plant species have led to further losses in extent (Biel et al., 2019; Gao et al., 2020; Hacker et al., 2012; Ruggiero et al., 2018; Wiedemann, 1984; Zarnetske et al., 2012). In an example from the Netherlands, the deterioration of many characteristic animal species of dunes is related to dune stabilisation due to increased grass and woody vegetation cover, and high nitrogen deposition (Lof et al., 2019). Browsing mammals such as rabbits and hares graze on the shoots of sandbinder restricting the natural replacement of populations (Norbury, 1996).

1.3.1 Climate change related threats

Coastal dune ecosystems are vulnerable to the effects of climate change, since ecosystem processes are in part driven by disturbances that are caused by local and regional climate and ocean processes and altered by a changing climate (Walker & Zinnert, 2022). Higher sea levels will cause a continuously eroding front on sandy shorelines as sea levels creep up the beach (Bruun, 1988). A widely used estimated range of shoreline retreat is that for every 1 m of sea level rise, a landward migration of 20 - 50 m will occur (Bruun, 1988). However, shoreline retreat distances will vary depending on site-specific factors, and uncertainty remains around climate feedbacks (Bryan et al., 2020; Coco et al., 2020).

Reduction in extent is a major problem in light of accelerating climate change. Where natural active dune processes are operating, coastal erosion is balanced by inward dune migration (Maun, 2009; Psuty, 2004). In contrast, where impervious surfaces or stabilising plant species occur, their resilience is threatened by so-called “coastal squeeze”, whereby erosion from storms and sea level rise on the seaward side of foredunes causes erosion, but the dunes are unable to migrate inland due to stabilisation (Martínez & Psuty, 2004). Although local variation will occur, beaches and dunes are expected to become progressively narrower, and rising sea levels are likely to increase erosion on most coasts (Davidson-Arnott, 2010). Where well-established dune systems occur, dunes could provide a sufficient buffer to delay erosion and beaches with an ample sand supply may continue to accrete at a slower rate or stabilise (Intergovernmental Panel on Climate Change [IPCC], 2019).

Foredune community shifts due to the effects of climate change are also likely to affect the balance of functional roles in active dune vegetation communities, as feedbacks from altered disturbance patterns and sea level rise start to impact, affecting dune building and stabilisation processes (Charbonneau et al., 2017; 2023; Charbonneau & Wootton, 2023; Hacker et al., 2019). Equally, new woody and herbaceous species can have better chances of establishing in warmer temperatures as latitudinal gradients move poleward (Davidson-Arnott, 2010), facilitating shifts to woody and exotic communities (Charbonneau et al., 2022). In general, there is high confidence that dune, beach and saltmarsh habitats will fragment, restricting landward migration further and coastal ecosystems will progressively lose their ability to adapt to climate-induced changes and provide ecosystem services such as acting as protective barriers (IPCC, 2020), and they will also not be able to provide quality habitat for many coastal species (IPCC, 2020).

1.3.2 Invasive plant species

Invading plant species can disrupt ecosystem processes, structure, function, and services (Lázaro-Lobo, 2023). Invading species are indigenous to some regions or ecosystems other than the area being invaded, and their adaptive traits allow them to colonise and reproduce successfully in the location they are invading (Castillo & Moreno–Casasola, 1996). Disturbed ecosystems, such as active dunes, are more often subject to invasions than intact communities (Castillo & Moreno–Casasola, 1996) since openings for the recruitment of juveniles occur more frequently. For example, the aggressive exotic sandbinder, *A. arenaria*, has invaded active dunes in temperate climates worldwide [IUCN, 2024a], often following widespread intentional planting for stabilisation in adjacent areas (Gao et al., 2023; Hacker et al. 2012; Hilton et al., 2000; Wardle, 1991; Wiedemann, 1998; Wiedemann & Pickart 1996). *A. arenaria* can invade so successfully due to its faster growth rate, reproduction and height than other sandbinders (Gadgil, 2001; Verhoeven et al., 2013; Wardle, 1991; Zarnetske et al., 2012). *A. arenaria* also forms a denser cover compared to many other sandbinders, thus excluding the establishment of natives (Baye, 1990; Biel & Hacker, 2021; Hacker et al. 2012, 2019; Verhoeven et al., 2013; Wardle, 1991; Zarnetske et al., 2013), threatening the balance of functional roles and dune building processes in dune ecosystems (Zinnert et al., 2016).

1.4 Aotearoa New Zealand as a case study

Active dunes in Aotearoa New Zealand mirror the international context in that they have an endangered status (Department of Conservation [DoC], 2020; Holdaway et al., 2012), their extent is declining (Stats New Zealand, 2015), and few studies of active dune condition exist (Ryan et al., 2023). They also provide important ecosystem services such as coastline protection, habitat for biodiversity and recreation and have cultural value to Māori, the indigenous people of Aotearoa (DoC, 2020).

1.4.1 Context

Aotearoa is an archipelago in the south–west Pacific Ocean with over 15,000 km of coastline (Ministry for the Environment, 2019) spanning 13 degrees of latitude and a climate ranging from warm subtropical in the north to cool temperate in the south (Macara, 2018). Much of Aotearoa is exposed to south-west and southerly swells, refracted by the north-south orientation of the two main islands, resulting in oblique, high-energy waves carrying sediment in longshore currents (Shepherd & Hesp, 2003). When the waves are combined with prevailing westerly winds, there is a net northward longshore drift along the east and west coasts, depositing sediment on beach shores, thus making it available for dune formation (Shepherd & Hesp, 2003). The sediment primarily originates from rivers, eroding cliffs and past volcanism (Shepherd & Hesp, 2003). Tectonic uplift of the seabed has also resulted in net onshore sediment transport to equalise the sediment budget (Shepherd & Hesp, 2003).

Active dunes are widespread around the coast of Aotearoa (Shepherd & Hesp, 2003), although they have always comprised less than 0.5 % of the mainland landmass, making them naturally uncommon (Williams et al., 2007). The extent of an ecosystem is relevant to its condition in that it is a predictor of the trajectory of ecosystem decline and collapse (Bland et al., 2016). In Aotearoa, the extent has been estimated only three times in the last 35 years. The most recent estimate of 25,208 ha in 2008 was by the New Zealand Department of Conservation (Stats New Zealand, 2015) and represents a decrease of 80.5% from the predicted pre-human extent of 129,402 ha (Stats New Zealand, 2015). Given the highly dynamic nature of active dunes and increasing pressure from human activities, these assessments of condition and extent are likely to be outdated.

1.4.2 Ecosystem structure and function

The characteristic active dune landform of Aotearoa is that of low, rolling dunes (Hilton et al., 2000), although a wide diversity of dune morphologies and habitats occur due to a dynamic geomorphic history (Hilton et al., 2000) which support a unique biotic community (Cockayne, 1911; Johnson, 1993). The processes of disturbance from sand movement (erosion and deposition) and subsequent colonisation by native sandbinders, facilitation and zonation are key processes related to active dune ecosystem condition (Holdaway et al., 2013). Natural on-going sand disturbance prevents the establishment of native tall woody vegetation (Holdaway et al., 2013).

Sandbinding plants such as the endemic *F. spiralis* and native *S. sericeus* occur in the most active areas, providing ecosystem engineer functions, building foredunes and habitats for other species through colonisation and facilitation (Cockayne, 1911; Holdaway et al., 2013). *F. spiralis* and *S. sericeus* have tolerances to moving sand, burial, drought, high temperatures, salt winds, and low nutrients (Cockayne, 1911; Gadgil, 2001) and can quickly colonise disturbed substrates through vegetative reproduction and lateral growth (Cockayne, 1911; Wardle, 1991). *F. spiralis* was once common around the coast of Aotearoa (Herbert & Oliphant, 1991; Wardle, 1991) but now only occurs in remnant populations, whereas *S. sericeus* occurs from Northland to the upper South Island (Bergin, 2011a). Distributions of these characteristic plant species have not been quantified, although *F. spiralis* populations have been designated a conservation status of "At Risk – Declining" based on analysis of existing records and expert knowledge (de Lange et al., 2017, p. 35; Herbert & Oliphant, 1991). Remnant populations of native species provide genetic diversity and resilience (Eriksson, 2000; Hoban, 2021; Polley et al., 2005; UN, 2021), thus remnant *F. spiralis* and *S. sericeus* populations have high conservation value.

A diversity of small herbs and other ephemeral specialists can be found in dune slacks with damp hollows and fluctuating water tables (Rapson et al., 2016). Semi-stable dune flora comprises a small number of characteristic prostrate, divaricating, spreading, and many-branched woody shrub species that accrete sand (Cockayne, 1911; Hilton et al., 2006; Johnson, 1992; Newsome, 1987; Partridge, 1992; Wardle, 1991). The shrubs, *Pimelia villosa* Sol. ex Sm and *Pimelea prostrata* (J.R. Forst. et G. Forst.) Willd. Subsp. *prostrata*, form mounds and can grow at the pace of moderate accretion; *Coprosma acerosa* A. Cunn roots adventitiously with sand burial. Characteristic woody species of stable dunes include the liana *Muehlenbeckia complexa* (A. Cunn.) Meisn. var. *complexa*; *Coprosma propinqua* A. Cunn. var. *propinqua*), *Ozothamnus leptophyllus* (G. Forst.) Breitw. et J.M. Ward and a range of other liana and shrub species (Wardle, 1991).

1.4.3 Threats

Human activity has heavily modified the dunelands of Aotearoa, including fire, farming, forestry, roading, urban expansion, holiday homes, vehicles, grazing, deforestation, and browsing from hares and rabbits (Hilton et al., 2000; Holdaway et al., 2013; Johnson, 1992; Johnson, 1993; Norbury, 1996; Partridge, 1992). Large-scale dune stabilisation schemes have also taken a toll. From 1915 to 1987, 115,000 ha of active dunes were stabilised using a method Cockayne (1911) recommended to promote rapid succession to woody species (Gadgil, 2001). Firstly, the exotic sandbinder *A.*

arenaria was planted to arrest major surface sand movement and increase soil organic matter; secondly, exotic tree lupin (*Lupinus arboreus* Sims) seed was sown for its nitrogen-fixing qualities to build soil structure and to achieve ground cover; lastly, exotic Monterey pine (*Pinus radiata* D. Don) trees were planted as permanent forests rather than production pine forests (Gadgil, 2001). The approach successfully modified sparsely vegetated active dune beaches to become densely vegetated grasslands and woodlands (Johnson, 1993). Moreover, *A. arenaria*, *L. arborea, P. radiata* and other woody species have invaded well beyond the stabilisation scheme areas (Gadgil, 2001). Examples of other exotic woody foredune invaders that lock up the substrate and increasing the stability of active dunes include boxthorn (*Lycium ferocissimum* Miers), gorse *Ulex europaeus* L. and common broom *Cytisus scoparius* (L.) Link (Wardle, 1991).

Some reasons for the success of *A. arenaria* invasions into active dunes in Aotearoa are that it grows faster and can withstand higher rates of burial than *F. spiralis* and *S. sericeus* (Konlechner et al., 2015) except on seaward dunes (Gadgil, 2001). Hilton et al. (2005) suggests that *A. arenaria* can displace *F. spiralis* and other native species on active dune sites. However, Gadgil (2006) suggests *A. arenaria* does not compete well with *S. sericeus* on the foredunes. *Carprobrotus edulis* (L.) N.E. Brown is another major herbaceous foredune invader (Holdaway et al., 2013). In a study comparing the traits of a range of native and exotic sandbinding species, Verhoeven et al. (2013) suggested that *A*. *arenaria* and *C. edulis* successfully outcompete the native sandbinders *F. spiralis, S. sericeus* and *Poa billardierei* (Spreng.) St.-Yves due to their more aggressive traits (Murphy et al., 2012; Verhoeven et al., 2013). Specifically, they are more vigorous and have greater leaf area (Specific Leaf Area), a trait that aids light capture for photosynthesis and growth (Violle et al., 2009). *A. arenaria* was taller than the native sandbinders, a trait related to competitive vigour (Gaudet & Keddy, 1995) *C. edulis* also showed trait plasticity in response to competition from other plant species across the dune environment, growing taller with the increased per cent cover of other species and having an increase in Specific Leaf Area size towards the back dunes (Murphy et al., 2012; Verhoeven et al., 2013).

Similar to sandy shorelines globally, climate change threatens active dune ecosystems in Aotearoa by increasing erosion risk (Bryan et al., 2020). El Niño events are predicted to become more extreme and La Niña events more frequent, causing an increase in coastal erosion (Cai et al., 2014; 2015), and wave height is also projected to increase in the Southern Ocean (Meucci et al., 2020). Further erosion, or shoreline retreat, will also occur due to sea level rise (Bryan et al., 2020), although there are uncertainties about shoreline retreat distances (Bryan et al., 2020; Coco et al., 2020). There is evidence that the coasts of Aotearoa are already experiencing increasing energy in the form of extreme storms or clusters of storms based on a 45-year-long wave hindcast from NIWA (Godoi et al., 2016; 2017; 2018).

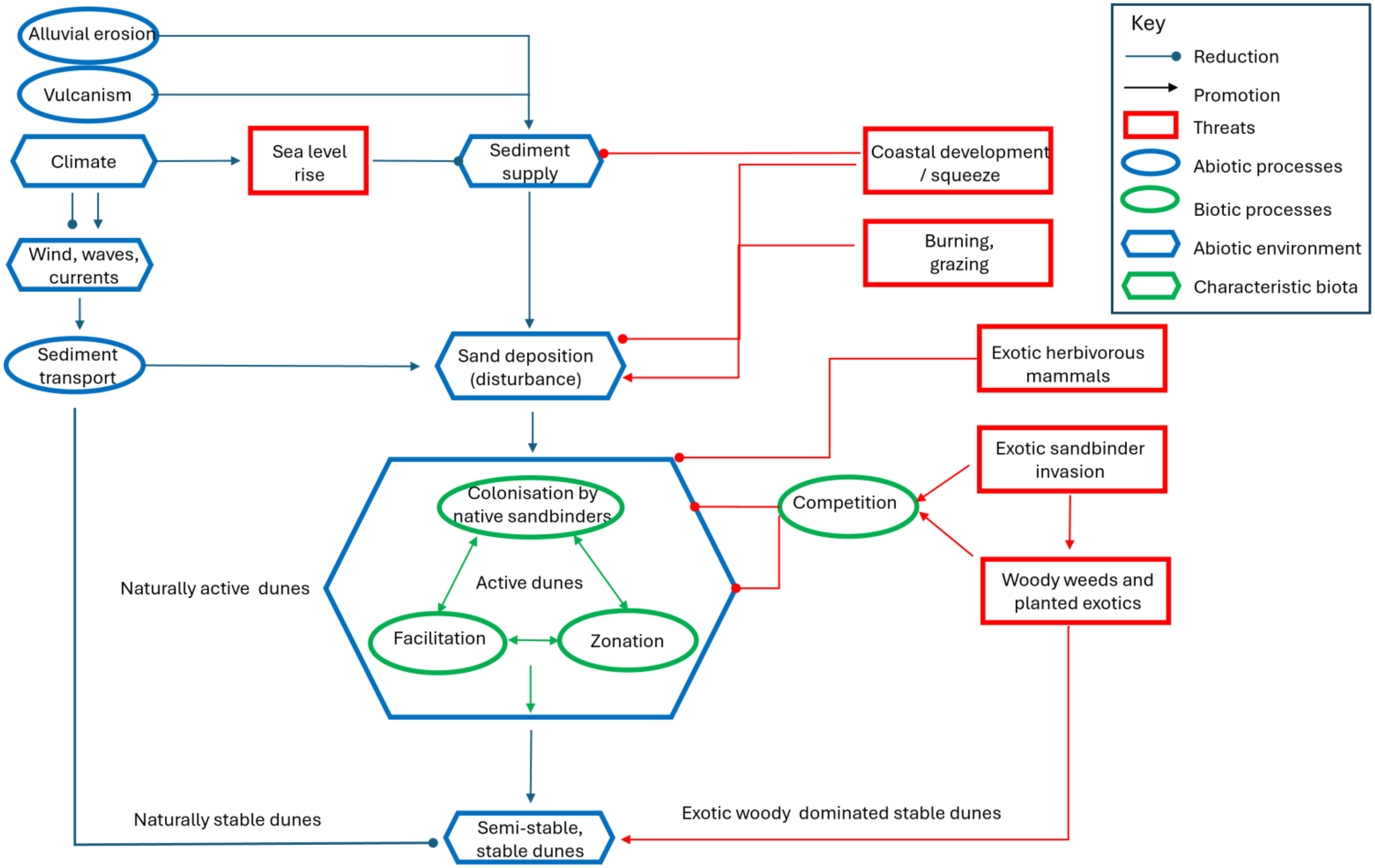


Fig. 1. Conceptual model of active dune geomorphic (abiotic) processes, ecological (biotic) processes and threats and the feedbacks between them. The symbology used in the conceptual model follows the Red List Ecosystem guidelines (Bland et al., 2016).

1.5 Indicators of active dune ecosystem condition

Given the importance of active dunes, effective indicators of their condition are essential to track progress towards restoration goals and biodiversity targets. Several national and international risk assessment frameworks exist globally, to assess and monitor ecosystem condition (IUCN, 2016; Rendon et al., 2019; United Nations Statistics Division [UNSD], 2021), many of which are based on the concept of ecosystem integrity. Ecosystem integrity refers to the ability of an ecosystem to be resilient to natural or anthropogenic changes, and to maintain characteristic composition, structure (including extent), functioning and self-organisation over time within a natural range of variability (Holling, 1973; Karr, 1993). In this thesis, ecosystem integrity and condition are used interchangeably. Examples of ecosystem integrity-based risk assessments include the Red List of Ecosystems (Bland et al., 2016); the IUCN Green Status of Species (Grace et al., 2021); Ecosystem Biodiversity Variables (Pereira et al., 2013); the System of Environmental-Economic Accounting – Ecosystem Accounting (United Nations et al., 2021) and the New Zealand Department of Conservation Outcome Monitoring Framework (Lee et al., 2005).

A precursor to using risk assessments necessarily entails the identification of the indicators of an ecosystem's characteristic composition, structure and functioning. However, identifying effective indicators of active dune condition is challenging due to the many causal drivers of their structure and function noted here. Moreover, a major barrier to the identification and monitoring of active dune indicators in many places is a lack of quantitative data on a national scale (Farrell et al., 2021; Ryan et al., 2023).

1.6 National mapping and condition assessments globally

National scale studies of active dune extent and, in some cases, condition have been carried out in a limited number of countries. The distribution of dunes was mapped for twenty-five countries in Western Europe (Doody, ed. 1991). It included maps of main dune sites based on topographic maps and aerial imagery, with selected field survey sites in selected countries (Doody, ed., 1991).

Descriptions of dune vegetation and conservation issues accompanied the maps (Doody et al., 2005). Detailed mapping was produced for Great Britain, Finland, Denmark, Ireland, and the Atlantic coast of France (Doody, 2005). Mapped extents became the basis for the current Natura 2000 network (Mücher et al., 2023), for which ecosystem mapping and an assessment of species, threats and pressures is currently mandated every six years by the 1992 European Habitats Directive. The Natura 2000 database currently records 1,478 active dune sites (European Environment Agency, 2024), with vegetation classifications created from the European Nature Information System (EUNIS), which is based on field sampling and phytosociological analysis (EUNIS, 2021; Mücher et al., 2023). Other national-scale studies of active dunes were not located outside Western Europe.

In recent years, remote sensing and modelling methods have been developed for mapping in selected European Union countries to improve the frequency of mapping ecosystem extent and condition and assess the representativeness of the European conservation network through predictive modelling (Mücher et al., 2023). Additionally, a spatially explicit classified map of the extent and distribution of all habitats in England, including dunes, was published in 2022 based on Sentinel 2 satellite imagery, achieving an average habitat classification accuracy of 88% (Natural England, 2024). However, ecosystem condition was not part of the assessment. In some cases, national-scale data has been collected for reasons other than ecosystem assessments, but the data could potentially provide information about active dune extent or condition. For example, the United States Geological Society (USGS) has recently (2024) developed a tool for repeat mapping of coastal vegetation from satellite imagery for the assessment of hurricane impacts on the U.S. East Coast and the Gulf of Mexico, for use with 3 m spatial resolution satellite imagery (USGS, 2024a). The tool could be useful for mapping the extent of active dunes. The USGS has also collated LiDAR-derived beach and dune morphology (dune crest, dune toe, and shoreline), or ecosystem structure, data for locations on U.S. sandy coastlines that are subject to extreme storms and shoreline change (USGS, 2024b).

The only national survey of active dune condition for Aotearoa (Johnson, 1992; Partridge, 1992) is now over 30 years old. The authors conducted a qualitative assessment of representativeness primarily based on botanical values and threats for prioritising areas for conservation management (Johnson, 1992; Partridge, 1992), although abiotic processes were not a focus (Hilton et al., 2000). The quantification of sediment supply to beaches in Aotearoa is poorly known and sediment budget cells that define sediment flows, direction and deposition have not yet been fully described (Stephens, S. personal communication 2024). The geomorphic agents of wind and waves are better known abiotic variables and have been modelled from buoy data over 45 years (Bryan, et al., 2020). Examples of national scale assessments or data collections for other countries were not found in the literature searches undertaken for this research. However, description and commentary on ecosystem structure of active dunes is available for nearly all other countries Van der Maarel (1993a, 1993b, 1997).

Information about the extent and condition of ecosystems is useful at the scale of policy, regulation and reporting to protect ecosystem integrity and therefore, ecosystem services (Maes et al., 2012, 2016; UN, 2021), thus, it is typically mandated at the national scale (Maes et al., 2012, 2016; UN, 2021). For example, the United Nations requires member countries to report on ecosystem extent and condition using the System of Environmental and Economic Accounting – Ecosystem Assessment (SEEA-EA) to evaluate ecosystem services (UN, 2021). The Global Biodiversity Framework (2022) requires countries to report on the Red List of Ecosystem assessment of ecosystem condition (United Nations Environment Program – World Conservation Monitoring Centre [UNEP-WCMC] (2024). To complete such assessments, methods are needed to define the extent and ecosystem typologies nationally and then derive indicators of ecosystem condition related to characteristic ecosystem processes, structure and function that can be monitored (Maes et al., 2012; 2016). Long-term botanical datasets collected from field surveys over extensive spatial extents can be used as input data to describe typologies, for example, studies of coastal dunes in the European Union (Doody, ed., 1991) and coastal dunes in Mexico (Moreno-Casasola, date). In many countries, long-term, spatially extensive datasets have not been collected and alternative approaches to field surveys are needed.

1.7 Utility of remotely sensed imagery to assess and monitor condition

Traditional approaches to measuring the geomorphic change in beach-dune systems include time and labour-intensive field surveys of vegetative and topographic features (see Biel et al. 2017, 2019; Hacker et al., 2012; Hovenga et al. 2019, 2021, 2022; Jay et al., 2022; Sykes & Wilson, 1991; Ruggiero et al., 2018); erosion and deposition pins; visual interpretation of aerial photography and maps (Walker et al., 2013) and qualitative surveys (Partridge, 1992; Johnson, 1992). Data are often used to estimate changes in morphometrics, such as dune volume and vegetation composition (Biel et al., 2019; Hacker et al., 2012; Jay et al., 2022; Hovenga et al., 2019, 2021, 2022; Ruggiero et al., 2018; 2015; Walker et al., 2013; Zarnetske et al. 2012, 2015).

In recent decades, remote sensing techniques have been developed to complement and replace various field survey methods (Morgan et al., 2010; Turner et al., 2015). Additional and major advantages of remote sensing approaches include quantitative datasets at increased spatial coverage and temporal frequency (Hugenholtz et al., 2012; Murray et al., 2018), thus allowing for shorter intervals between sampling events.

Available options for remotely sensed imagery that provide information about ecosystem structure and function require tradeoffs between spatial resolution (size of pixels), spectral resolution (number of electromagnetic bands) and temporal resolution (frequency of imagery collection) (Ustin & Gamon, 2010). The time between imagery capture for satellite–based sensors varies between 1 – 30 days (Zhao et al., 2022), enabling daily and seasonal changes to be captured, for example, due to phenology or disturbance from storms. However, very high spatial resolution imagery (≤ 0.5 m) is often required to discriminate between individual plants, plant species or vegetation types for training samples in classification (Carrión-Klier, 2022; Morgan et al., 2010; Oddi et al., 2021), precluding the current generation of public and privately owned satellite-based sensors which have a spatial resolution of *c.* 1 – 2m for multispectral imagery (Blacksky, 2024; European Space Agency, 2024). Encouragingly, the next generation of satellite-based sensors will likely be sub 1 m2 spatial resolution, such as Blacksky Images at 35 cm2 (Blacksky, 2024), offering a promising direction for discriminating vegetation and capturing changes in ecosystem structure and function. The high spatial resolution of aerial imagery (< 1 m) is well-suited for detecting ecological patterns compared to satellite imagery (Oddi et al., 2021).

Aerial imagery also offers continuous spatial coverage of ecosystems, allowing for data collection across larger areas than traditional field survey (Morgan et al., 2010), or than is possible with imagery captured by Unoccupied Aerial Vehicles (UAV) due to the higher elevation at which aircraft are flown. Within the recent active dune literature, aerial imagery is increasingly used as input datasets to studies of ecosystem structure. A common approach has been to classify dune cover types into vegetated or unvegetated classes using pixel-based classification techniques (Gao et al., 2021; Keisjers et al., 2015; Konlechner et al., 201; Ryu & Sherman, 2014). For example, in a multiyear study of an island coast in the Netherlands, Keisjers et al. (2015) assessed the extent of feedbacks between sedimentation and vegetation patterns through the analysis of vegetated and unvegetated cover classes. Input datasets included aerial imagery classified into cover classes, elevation data from Light Detection Imagery (LiDAR) and ground truthing with field survey of 2.5 m circular plots on transects 1 km apart. The study concluded that vegetation patterns affected sedimentation, but there was limited evidence for the reverse. Gao et al. (2021) analysed bare and vegetated surfaces from historical and current imagery of dune fields in Victoria, Australia, concluding that variation in dune field vegetation and mobility was due to interactions between topography climate and human influences. Ryu & Sherman (2014) created a binary classification of 22 beaches on the east and west coasts of the USA and quantified spatial patterns and climate variables, observing that the per cent cover of bare sand was a significant indicator of geomorphic activity. Konlechner et al. (2015) used similar methods throughout a dune restoration sequence at a beach in Aotearoa, compared to a reference site. These studies offer insight into the potential of remotely sensed aerial imagery to investigate the structure, function and condition of active dunes.

1.8 Aims, research questions and objectives

In Aotearoa, high-resolution, national coverage aerial imagery from the 1950s onwards is available to the public at low or no cost. It has been utilised for just one national scale study to delineate the extent of active dunes, with data collection ~ 30 years ago (Hilton et al., 2000). To our knowledge, the aerial imagery record has not been utilised to discern and classify dune functional plant types, to develop vegetation cover types or derive indicators of their condition from the spatial patterns of the vegetation types at a national scale. This thesis aims to develop a method for detecting relative differences in the condition of active dunes nationally by answering the question: Can the condition of active dune ecosystems be discerned from remotely sensed imagery, specifically aerial imagery, at a national scale? This question will be addressed by investigating the following research questions and objectives:

Question 1. How can remotely sensed imagery be best utilised to determine active dune condition?

- Objective 1: Construct an index to compare and communicate a gradient of condition of different beaches based on ecosystem integrity / condition.
- Objective 2: Identify the potential extent of active dunes in Aotearoa and a nationally representative sample of active dunes.
- Objective 3: Evaluate the ecosystem integrity / condition of active dunes using existing datasets and identify gaps in the data that remote sensing could fill.

Question 2. What are the capabilities and limitations of aerial imagery for characterising the vegetation of active dunes?

- Objective 1: Discriminate and characterise the vegetation of active dunes using image classification procedures.

- Objective 2: Identify national vegetation types from the classification data
- Objective 3: Identify the potential drivers of variation in vegetation cover types across Aotearoa.

Question 3: What are the indicators of active dune structure and function that can be derived from remotely sensed imagery?

- Objective 1: Relate the vegetation types and spatial pattern of active dunes to ecosystem processes, structure and function to identify indicators of ecosystem condition.
- Objective 2: Identify the potential drivers of variation in spatial pattern across the different cover types across Aotearoa.

1.9 Thesis structure

The thesis research comprises this introductory chapter (Chapter 1), three research data chapters (Chapters 2-4), and a final discussion chapter (Chapter 5). In Chapter 2, a method to construct an index, or gradient of condition, was developed to identify where remotely sensed imagery could be of most use within a framework of ecosystem integrity. A novel workflow was also proposed to 1) identify the maximum potential extent of active dunes from large-scale geospatial datasets, 2) select a nationally representative stratified sample of active dune beaches, and 3) estimate the extent of the active dune ecosystems at the selected beaches from high-resolution, RGB (colour) aerial imagery.

In Chapter 3, a novel workflow was developed to 1) process imagery, 2) subsample the imagery in plots on beaches, and 3) classify imagery using object-based image analysis, and 4) collect and process environmental data for the sample. Vegetation types were derived from the classified imagery using hierarchical agglomerative clustering and Principal Coordinate Analysis to create a national vegetation typology for active dunes. The relationship between the vegetation types and environmental variables was examined through classification tree analysis.

In Chapter 4, variation in the spatial pattern of the different national vegetation types was identified using landscape metrics from landscape ecology and Principal Components Analysis. The impact of invasives on the spatial pattern of active cover types was also analysed using Principal Components Analysis. Candidate indicators of active dune condition were identified from spatial patterns and the literature on the ecological processes of active dunes. The relationships between the spatial pattern of the different vegetation types and environmental variables were examined with conditional inference tree models.

In Chapter 5, the thesis results are synthesised to highlight the main advances in knowledge that this research contributes and limitations, for each data chapter. Recommendations for operationalising results are put forward, and future research questions are posed.

2 Ecosystem integrity of active dunes: a case study to implement and test the SEEA-EA global standard, from Aotearoa New Zealand

2.1. Introduction

To halt the decline in global biodiversity (Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services [IPBES], 2019), 196 countries have committed to the United Nations (UN) Convention on Biological Diversity which requires reporting on the changing status of ecosystems (UN, 1992). Effective monitoring is essential to track progress towards targets, including a fit for purpose indicator framework (Nicholson et al., 2021, Tittensor et al., 2014), but, as yet there is no harmonized, universal framework that delivers timely data on biodiversity and ecosystem change (Hansen et al., 2021; Pereira et al., 2013).

Ecosystem integrity is a unifying concept, referring to the capacity of an ecosystem to be resilient to natural or anthropogenic perturbations, and to maintain characteristic composition, structure, functioning and self-organisation over time within a natural range of variability (Holling, 1973; Karr 1993; McGlone et al., 2020). Measures of integrity include characteristics of abiotic state (physical and chemical), biotic state (composition, structure and function), and landscape state (e.g., connectivity and fragmentation) in relation to a reference state (Andreason, 2001; Czúcz et al., 2021; Keith et al., 2020; Noss, 1990). Another important measure is ecosystem areal extent which can reflect how many species an ecosystem can support (Bellingham et al., 2021; Cieraad et al., 2015; Dengler, 2009), and the rate of decline in extent can be used as an indicator of the trajectory towards ecosystem collapse (Bland, et al. 2016).

The UN Statistics Division has developed the System of Environmental – Economic Accounting, Ecosystem Accounting (SEEA-EA) which assesses physical, chemical, biotic and landscape characteristics to quantify ecosystem condition (ecosystem integrity), as part of a new method to account for the provision of ecosystem services (UN, 2021). To quantify ecosystem integrity specifically, indicators are derived from a range of variables in relation to reference states (UN, 2021). Areal extent is also measured to compare temporal change (Keith, et al., 2020). The identification and monitoring of ecosystem integrity indicators is critical for monitoring all ecosystems, especially ones that are highly dynamic whether due to natural or human-caused perturbations.

SEEA-EA accounts have been compiled in 41 countries across a range of ecosystems (United Nations Statistics Division [UNSD], 2023). The most commonly compiled ecosystem accounts include ecosystem extent, condition, and physical ecosystem services accounts (UNSD, 2023). While some countries are in a position to repurpose their existing reporting to meet the SEEA-EA requirement (for example, reporting on the European Habitats Directive in European Union countries) (Bogaart et al., 2023; Hein et al., 2020), many countries do not currently undertake such reporting and baseline data, and analysis is required. Issues encountered in SEEA-EA assessments at different scales and ecosystems include: a high degree of diversity in the quality and quantity of relevant data across countries (Farrell et al., 2021; Lof et al., 2019); defining ecosystem boundaries and an appropriate spatial scale poses challenges (Bogaart et al., 2023); uncertainty associated with models, or lack of models to monetarily account for ecosystem services (Hein et al., 2020); and there is also the inherent uncertainly of accounting for the impacts of ecosystem collapse given the lack of scientific data on collapse thresholds (Hein et al., 2020). To date, no countries have applied the SEEA-EA to active dune ecosystems (UNSD, 2023).

A number of alternative methods to account for ecosystem integrity exist. Two candidate international methods include the IUCN Red List of Ecosystems (IUCN, 2016), which is required under the 2022 Global Biodiversity Framework (UNEP, WCMC, 2022) or the Ecosystem Biodiversity Variables (Pereira et al., 2013). However, both of these require developed, quantifiable ecosystem indicators to assess ecosystem condition which currently do not exist for active dunes based on the criteria in these frameworks (Group on Earth Observation and Biodiversity Management [GEOBON], 2024; IUCN, 2024b). Meanwhile, the SEEA-EA has a robust approach to develop qualitative indicators, while also illuminating which elements of ecosystem integrity are data deficient.

Coastal active dune ecosystems occur globally and are dynamic dune systems where the physical, ecological and landscape characteristics result from continuously moving aeolian sands (Hesp & Walker, 2021; Psuty, 2008). They are highly mobile with bare to sparse, scattered vegetation (Hesp & Walker, 2021; Hilton et al., 2000). Active dunes also provide vital ecosystem services, such as acting as natural barriers to storm surge and protection from coastal erosion and sea level rise (Biel et al., 2017; Ruggiero, Seabloom et al., 2013; Stockdon, 2007; Walker et al., 2013). They are a habitat for indigenous biodiversity, sustain culturally important practices and provide opportunities for recreation and tourism (Bergin, 2011a; Hare et al., 2019; Hesp, 2000; Martínez et al., 2013; Nordstrom & Jackson, 2021). Despite their economic, ecological, cultural and social importance, coastal dune systems remain among the most threatened ecosystems in the world (Muñoz-Vallés & Cambrollé, 2014), primarily because of human activities.

Globally, the drivers of active dune ecosystem integrity are relatively ubiquitous. The key factors for contemporary dune system formation and development are an available supply of sandy sediment, sufficiently high wind speeds to mobilise the sediment, the presence of coloniser, sand-binding vegetation, and adequate space inland from the shoreline to accommodate coastal processes (Biel et al. 2017, 2019; Delgado-Fernandez & Davidson-Arnott, 2011; Gao et al., 2020; Hacker et al., 2012; Hesp, 2011; Hovenga et al. 2019, 2021, 2022; Jay et al., 2022; Psuty & Silveira, 2010; Ruggiero et al., 2018; Zarnetske et al. 2012, 2015). Storms also affect dune mobility through shifts in sediment supply and impact plant community dynamics (Miller et al., 2010). Changes in these factors over time create a continuum of dune mobility from highly active dunes comprising bare sand to entirely vegetated or ‘stable’ dunes (Gao et al., 2020; Levin et al., 2008; Tsoar, 2005). Gao et al. (2020) presented a global trend towards stabilisation of active dunes in the last century largely due to land use change, practices such as afforestation, overgrazing, urbanisation, and intentional stabilisation projects, and sediment supply decline due to hydrological works and human-made sea defences (Gao et al., 2020).

As a large island nation in the South Pacific, Aotearoa has c.15,000 km of coastline (Bell & Gibb, 1996) that contain a subset of active dunes with a wide variability of ages and morphological diversity, creating a range of habitat types (Hilton et al., 2000). Active dunes in Aotearoa are characterised by a high frequency of natural disturbance and, therefore, sparse, low-growing herbaceous vegetation (Hilton et al., 2000). Pioneer sand binding plants such as native *Ficinia spiralis* (A.Rich.) Muasya et de Lange and *Spinifex sericeus* R.Br. are adapted to the stressful conditions of coastal dunes in Aotearoa and act as ecosystem engineers (Hesp, 2000). *F. spiralis* and *S. sericeus* cause accretion, building foredunes and other geomorphic habitats for other species (Hesp, 2000; Maun, 2009), increasing their ecological diversity and resilience (Nordstrom, 2008; Walker et al., 2013). Remnant populations of native species, for example, *F. spiralis* and spinifex, also provide greater genetic diversity and resilience (Eriksson, 2000; Hoban et al., 2021; Polley et al., 2005; United Nations, 2021) compared to planted populations and non-native populations (Polley et al., 2005). The presence of exotic woody species represents a change in active dune vegetation structure to a more stable state (Gadgil & Ede, 1998; Gao et al., 2020; Pegman & Rapson, 2005), with lower geomorphic and ecological diversity (Hugenholtz et al., 2012; Nordstrom, 2008; Walker et al., 2013). The presence of exotic forbs and graminoids also increases the stability of active dunes to varying extents, depending on the species (Provoost et al., 2011; Pegman & Rapson, 2005). However, the presence of woody species indicates increased moisture and accumulated organic matter, and the ecosystem is more stable than active dunes (Maun, 2009).

Despite their widespread importance for coastline protection, recreation and cultural value to the indigenous Māori people of Aotearoa (Coastal Restoration Trust, 2011), there have been relatively few attempts to systematically assess the ecosystem integrity of active dunes. The only national survey, published as Johnson (1992) and Partridge (1992), ranked nearly all dune systems in Aotearoa (Hilton et al., 2000) and is now 30 years old. It ranks sand dune sites along a gradient of conservation values, with the main aim of identifying priority areas for conservation protection. The authors ranked 606 sites in all 25 local government regions by assessing a mix of state and pressure factors using qualitative field surveys. They used a scoring matrix comprising four criteria: "diversity of vegetation communities and diversity of dune landforms; the number or proportion of native sand species, or good representation of characteristic or rare dune species; the degree of invasion by weed species; and the degree of modification from human or animal interference" (Johnson, 1992; Partridge, 1992, p. 12). In addition, Shepherd and Hesp (2003) provide a geomorphologic overview of large-scale sandy barriers and coastal dune systems of Aotearoa, which are mapped nationally at a high level, alongside detailed descriptions and mapping for a handful of sites.

A few active dune surveys exist for local government regions in Aotearoa. Uys and Crisp (2019) used field survey methods to quantify several variables for the Greater Wellington Region, including indigenous dominance, species richness, areal cover, proportion of bare ground, canopy height of vegetation, alongside a number of pressures such as rabbit browse and human traffic. Wildlands (2008) used a field survey to map sand dunes in the Tauranga Ecological District based on Atkinson (1985), which is standardized physiognomic method for describing vegetation that incorporates both compositional and structural information. Their method also included scoring for both positive and negative impacts of humans and pest animals. Between 1980 and 2000, a handful of regional Protected Natural Area Programme surveys (Bellingham, 2001), which are used to assess and prioritise areas for conservation protection, have described active dune ecosystem integrity at local and regional scales (Hilton et al., 2000).

As a key indicator of ecosystem integrity, extent has been estimated three times for Aotearoa's active sand dunes in the last 35 years. Newsome (1987) identified potential active dunes from the New Zealand Land Resource Inventory at a scale of 1:1,000,000 based on aerial imagery from the 1960s and 1970s, but because of the low resolution of the maps significant dunelands were omitted (Hilton et al., 2000). Active dune extent was mapped by hand from regional aerial imagery from the 1950s, 70s, 80s and 90s by Hilton et al. (2000). In 2008, the New Zealand Department of Conservation assessed extent from moderate resolution satellite imagery and lithology maps (Stats New Zealand, 2015), and estimated active dunes extent has decreased 80.5%, to 25,208 ha, from their predicted pre-human extent of 129,402 ha (Stats New Zealand, 2015). Given the highly dynamic nature of active sand dunes, and the spread and intensification of human development around Aotearoa, these estimates are likely to be out of date. The current extent of active dune ecosystems is therefore unknown.

In sum, the information about the ecosystem integrity and extent of active dunes in Aotearoa is sparse, disparate, and in need of updating. Notably, a recent attempt to register active dunes in Aotearoa as a Red List Ecosystem was not successful due to data deficiencies (Holdaway, R. pers. comm, 2020). Similarly, an attempt to use the International Union for the Conservation of Nature (IUCN) Green List framework to assess the impact of conservation efforts on the threatened species *F. spiralis* was also deemed data deficient (Grace et al., 2021). Clearly, the dearth of up-to-date, quantitative information about the integrity and extent of active dunes in Aotearoa means there is no reliable indication of how, and how far, these ecosystems have diverged from their baseline extent, or their capacity to adequately provide critical conservation and ecosystem service values. Thus, there is a clear need for an up-to-date assessment of integrity and extent across Aotearoa's active sand dune ecosystems using a standard methodology that incorporates mapped extent and all the characteristics of ecosystem integrity: physical, chemical, biotic (plant community composition, structure and function), and landscape configuration. Moreover, the assessment should be repeatable, to enable comparison of the extent and ecological integrity of dunelands today with future states, and compatible with national and international environmental monitoring frameworks.

The aim of this study is to implement and test the UN SEEA-EA method to assess and rank ecosystem integrity and extent, using active dune systems in Aotearoa as a case study. Specifically, this study: (1) Assesses the change in areal extent of a representative sample of active sand dunes since the first mapped inventory (2) Identifies indicators and a method to assess and compare and rank active dune ecosystem integrity.

2.2. Methods

2.2.1 Estimating the change in areal extent of active dunes

A representative sample of active dunes was identified by firstly assessing the maximum potential areal extent of active dunes in Aotearoa, then applying a filter of location and condition. Maximum potential extent was defined here as potential areas where active sand dunes could form based on lithology and cover types. This was estimated by following a two-step process adapted from a method used by the New Zealand Department of Conservation to estimate active dune extent in 2008 (Burlace, D. and Brown, D., personal communication, 2020). First, surface lithology types on which active dunes could potentially form were identified from a classification provided as part of the New Zealand Land Resource Inventory (LRI) third edition dataset (Newsome, 1987; Newsome et al. 2008, Chapter 2, Appendix 1). This polygon dataset delineates physiographic areas of relatively homogenous surface and near-surface lithology (Newsome et al., 2008) and is derived from stereo aerial photograph interpretation, field verification and measurement at the 1:50,000 scale carried out over the 1970s and 1980s (Newsome et al., 2008). Lithology types identified as potentially forming active dunes were: ‘Wind blown sand’, ‘Unconsolidated sand’, ‘Estuary’, ‘River’, ‘Alluvium’ and ‘Towns’. The lithology type ‘Towns’, which often occur on duneland at the coast, was included so that the impact of towns on the natural processes of active dunes could be quantified and recorded.

Second, selected lithologies were overlaid in ArcGIS Pro, version 2.8 (Esri, 2020), with land cover types that could also potentially form active dunes from the New Zealand Land Cover Database v 5.0 (Landcare Research / Manaaki Whenua, 2020), which is a 1: 50,000 scale digital map and multi-temporal thematic classification of Aotearoa's land cover derived from satellite imagery (Chapter 2, Appendix 2). Land cover types that were identified as potentially forming active dunes were: 'Coastal Sand and Gravel', ' Herbaceous Saline Vegetation', ' Low Producing Grassland' and 'Urban Parkland/ Open Space'. The cover type 'Urban Parkland/ Open Space', which often occurs on duneland at the coast was also included because it is useful to assess the impact of this cover type on active dune processes (Chapter 2, Appendix 4).

To identify areas where both suitable lithology types and land cover types occurred, as well as the 'Towns' and 'Urban Parkland / Open Space' areas that were likely to occur on active dunes, the layers of suitable lithology and cover types were overlaid with the most recent digitised extent of active dunes by Hilton et. al (2000). It is important to note that this inventory of 'active' dunes' is therefore ~30 years old and may not truly reflect current (2023) conditions. The Hilton et al. (2000) layers were derived from 1990s aerial photography, at a scale of between 1:10,000 and 1:63,000. This overlay result offers a coarse approximation of maximum potential sand dune extent, returning a total area of 50,938 ha on the mainland islands and Rakiura / Stewart Island (Fig. 2). An overlap of more than 15% between the selected landcover (Manaaki Whenua / Landcare Research, 2020) and lithology types (Newsome, 2008) and the 1990s mapped extent of active dunes was taken as an indication that the dune system still existed but may have been modified by natural or human processes (Burlace, D., Browne, D. personal communication, 2021).

Fig. 2. Maximum potential extent of active sand dunes in Aotearoa. Input datasets include an overlay of surface lithology types that could potentially form active dunes from the New Zealand Land Resource Inventory, third edition (Landcare Research / Manaaki Whenua, 2021; Chapter 2, Appendix 1), with land cover types that could also potentially form active dunes from the New Zealand Land Cover Database v. 5.0 (Landcare Research / Manaaki Whenua, 2020, Chapter 2,

Appendix 2); and the most recent digitised extent of active dunes by Hilton et. al (2000), derived from 1990s aerial photography.

2.2.2. Site selection

Sites from the maximum potential extent were stratified by the two main coastlines (east and west), and then by different levels of condition from the most recent national survey of active dune condition (Johnson, 1992; Partridge, 1992) in three steps (see Chapter 2, Appendix 4). The first step was to filter the Partridge (1992) and Johnson (1992) sites (n = 606) based on their condition scores, into three groups for each of the 25 regions used in their study: Highest, mid - range (based on the median, not the mean) and lowest, but where these sites also had sandy substrate, and where characteristic, native sand binding species *F. spiralis* and/ or spinifex were present on the highest and mid - range scoring sites; these species were rarely recorded as occurring on the lowest scoring sites. This provided a total of 75 locations for further consideration as study sites. Second, the filtered sites were plotted on a map including geographic coordinates of sand dunes given in Johnson (1992) and Partridge (1992). Where the 75 sites occurred on an area that was also part of the maximum potential extent, it was assumed that these sites still existed and, therefore, they were retained on the list (n = 18) (Chapter 2, Appendix 4). The eighteen sites retained occurred on both the east and west coasts, and four more sites were added from the Auckland Region due to availability of additional imagery, totalling 22 sites that were used for the analysis presented in this paper (Fig. 3).

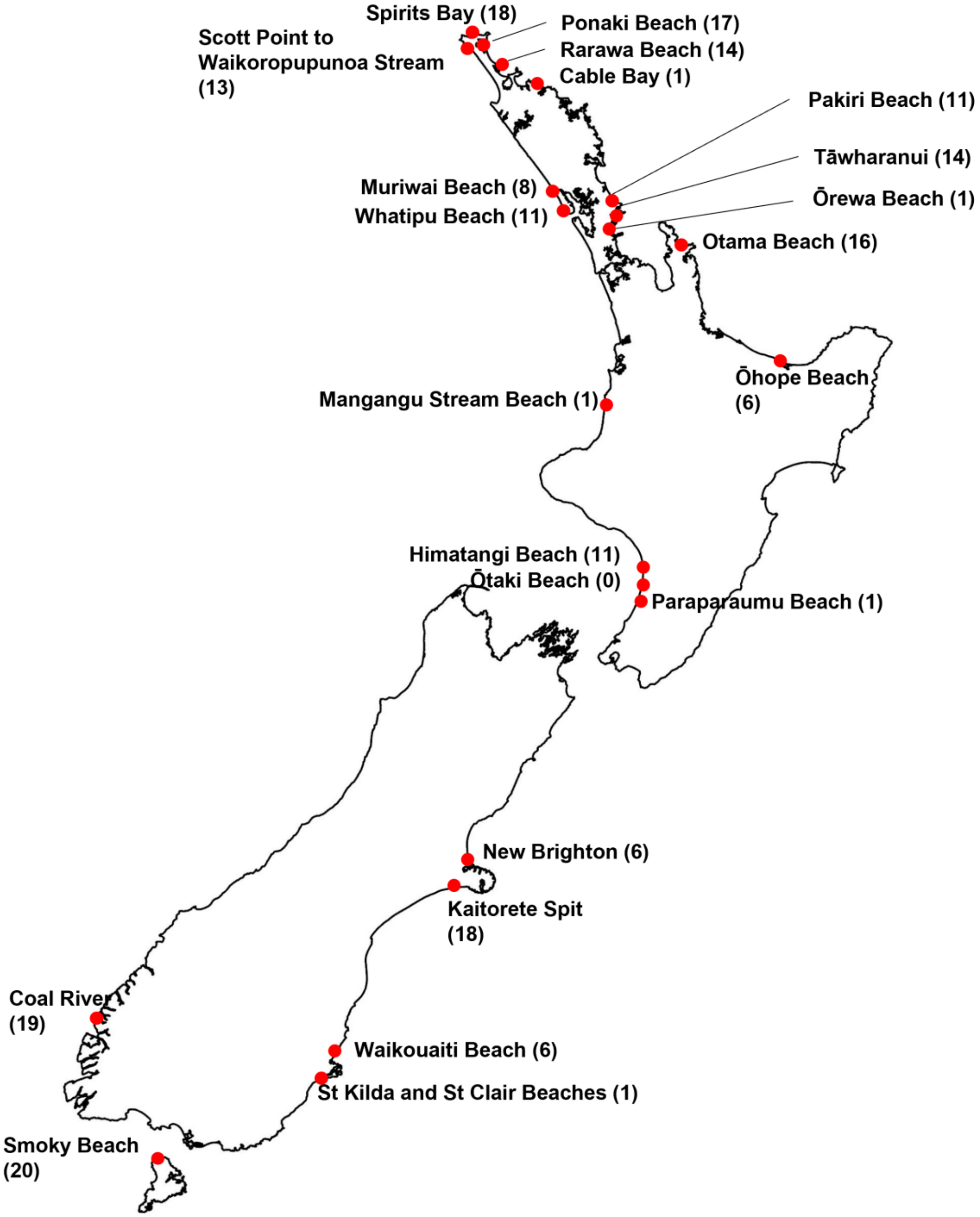


Fig. 3. Sites selected as a representative sample of active dunes in Aotearoa (n = 22). Dune integrity rank scores from Partridge (1992) and Johnson (1992) out of 20, given in parentheses. Higher scores indicate dune sites with greater integrity.

Finally, the ecosystem extent of the 22 selected sites was visually identified from aerial imagery and hand mapped in ArcGIS Pro, verison 2.8 (Esri, 2020). Aerial imagery comprised the most recent and publicly available, high resolution, ortho-rectified aerial imagery covering sample sites, and was acquired from Land Information New Zealand (LINZ) Data Service under a Creative Commons open licence (CC BY 4.0). Snapshot imagery was taken by local and regional councils over the period 2007- 2021 and comprised red, green and blue colour bands. The original resolution ranged from 0.075 m to 7.5 m and came in New Zealand Transverse Mercator projection.

2.2.3 Indicator development process to assess variation in ecosystem integrity

Indicators for ecosystem integrity were developed following the process described in the United Nations SEEA-EA framework (Fig. 4; UN, 2021): Seven characteristics of active dune ecological processes were identified from the literature, incorporating abiotic (physical), biotic (composition, structure, function), and landscape characteristics (Table 1) and following SEEA-EA guidelines (Czúcz et al., 2021; UN, 2021). Specific variables that quantified these characteristics were selected, and the scoring rationale for each variable was compiled based on information gleaned from peer-reviewed and grey literature regarding how ecosystem integrity was putatively related to changing levels of each variable (Table 1; Chapter 2 Appendix 3). The study was limited to assessing ecosystem extent and condition. Valuation of ecosystem assets and services was not undertaken due to the inherent uncertainty in existing models, thresholds and impacts of collapse (Hein et al., 2020).

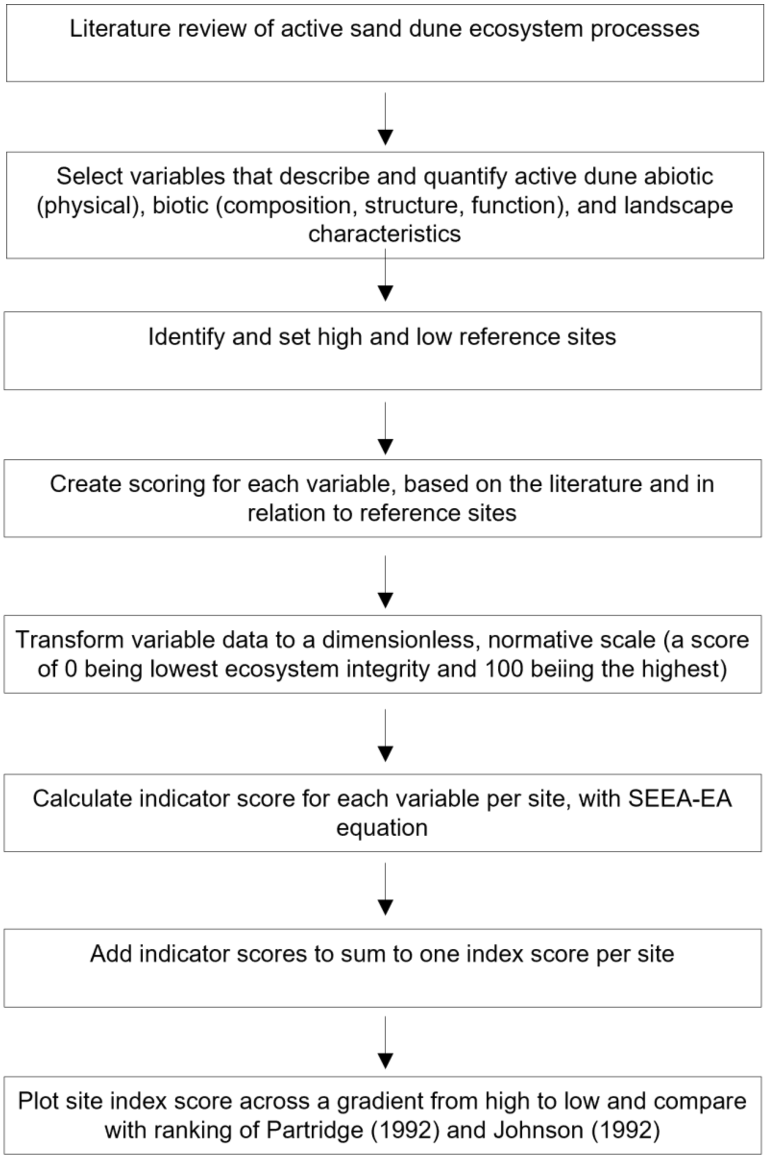


Fig. 4. Indicator development process for the ecosystem integrity of active dunes, based on the System for Economic and Environmental Accounting – Ecosystem Accounting (SEEA-EA) process (U.N, 2021).

Table 1. Characteristics, variables and rationale used to assess ecosystem integrity of active dunes in Aotearoa

Characteristics and variable description	**Scoring rationale**	**Aotearoa data source**
Abiotic characteristics		
Physical: Geomorphic alignment of shoreline	Greater exposure to incoming swell wave energy enables greater transport of sediment for dune building phases in normal weather conditions (Hesp, 2011; Miller et al., 2010).	Coastal Sensitivity Index (Goodhue et al., 2012, for NZ National Institute of Water and Air (NIWA))
Physical: Number of extreme wave events (greater than 4 m and longer than 12 hours).	Greater numbers of days exposed to extreme wave events means more disturbance, sediment transport and restarting of vegetation succession sequences. This creates the characteristic early-stage vegetation of active dunes (Hesp, 2011; Miller et al. 2010; Pegman & Rapson, 2005).	Extreme wave events (Gorman, 2016, for the NZ Ministry for the Environment, MfE)
Chemical	Data deficient at scale of active dunes. However, organic carbon and pH are strongly linked to biotic value and can serve as a proxy here.	Newsome et al. (2008).
Biotic characteristics		
Composition: Proportion of native species to exotic plant species	Greater proportion of native species represents lower ecological degradation (Lee et al., 2005).	National datasets for plant community composition of active dunes are deficient in Aotearoa but can be gathered from site specific data in the grey literature, consultancy reports and peer reviewed papers and datasets (Chapter 2, Appendix 3).
Vegetation structure: Presence of exotic woody species	The presence of exotic woody species is interpreted here as an increase in biomass at a site, since sites of high ecosystem integrity will have no or very few native woody species, therefore the presence of exotic woody species is used as a proxy for increased biomass given the lack data about cover type or density at the requisite scale. Woody species represent a change in ecosystem structure from a mobile to more stable state with less geomorphic and ecological diversity (Maun, 2009; Nordstrom, 2008; Walker et al., 2013 Pegman & Rapson, 2005)	

Table 1. (cont.)

Characteristics and variable description	**Scoring rationale**	**Aotearoa data source**
Biotic characteristics		
Function (characteristic flora): Presence of remnant and / or planted populations of native sand binders (treated as two separate indicators).	Native *Ficinia spiralis* and *Spinifex sericeus* populations increase ecological diversity and resilience (Hesp, 2000; Nordstrom, 2008; Walker et al., 2013). Remnant populations of native species provide genetic diversity and resilience (Eriksson, 2000; Hoban, 2021; Polley et al., 2005; UN, 2021).	
Landscape characteristics		
Fragmentation: Distance from roads	Road networks can cause habitat fragmentation and loss (Bennett, 2017; Gao et al., 2020). Increasing distance from road networks reduces negative impacts on ecosystems (Bennet, 2017; Benítez-López, 2010).	LINZ (2016). NZ Roads Addressing

Variable data were then transformed into a common, dimensionless scale by allocating a score along a scale of zero to 100%, with 100% representing with the highest ecosystem integrity. Reference sites were identified and set from the 22 sample sites. The purpose of these sites was to reflect the highest and lowest quality active duneland sites and provide normative data to compare other sites against. The top scoring site from Partridge (1992) and Johnson (1992) was Smoky Beach on Rakiura, Stewart Island. Smoky Beach was selected to use as the highest quality reference site since it remains largely unmodified since the last survey in 1987 (DoC, 2012; Hilton and Konlechner, 2021). However, the bottom scoring site from 1992, Ōtaki Beach, had improved due to natural accretion and restoration activities (Todd et al., 2022), so the next lowest scoring site (Ōrewa Beach) was assessed and used as the low reference site. This site was selected due to ongoing erosion from natural processes, relatively sheltered geomorphology, extensive modification from sand mining, extensive sea defences, residential development and lack of established native vegetation over much of the beach (Mead et al., 2009; Roberts et al., 2020). Transformed data were then converted to ecosystem indicators calculated by a linear transformation function using the formula:

$I = (V - V_L) / (V_H - V_L)$,

where I is the value of the indicator, V is the value of the variable, V_H is the high integrity score and V_L is the low integrity score (UN, 2021). Note that the V_H score is required to be higher than the V_L score. Indicator scores for each characteristic are then summed to give one score per site, which allows for comparison between sites (see Table 2 for scoring criteria and Table 3 for a worked example).

The process was then iterated to ensure a balance between abiotic, biotic and landscape characteristics, yet remain sensitive to environmental changes. For example, iterations of functional characteristics tested the sensitivity of different treatments of the two main species of sand binders, *F. spiralis* and *S. sericeus*. Initially, the presence or absence of *F. spiralis* and *S. sericeus* were treated separately; however, because there is a latitudinal limit to *S. sericeus* distribution (Coastal Restoration Trust NZ, 2011), the gradient was loaded in favour of functional state characteristics, due to the nationwide extent of *F. spiralis*. To mitigate this loading of the gradient, both species were incorporated under one indicator. Further refinements were made to reflect the importance of remnant populations of these two species, since remnants (i.e., un-planted sites) have more value due to their genetic diversity and resilience (Eriksson, 2000; Hoban, 2021; Polley, 2005; UN, 2021). The final variable included a scenario where, if only remnant populations occurred at a site, it was scored equally with planted populations to ensure remnant populations did not falsely score poorly or with null values due to divisions by zero, should no planted populations occur at the VH reference site. Finally, site index scores were converted to a per cent rank, plotted on a gradient from low to high, and compared with the per cent score ranking of Partridge (1992) and Johnson (1992).

Table 2. Variables and scoring criteria for active dunes in Aotearoa using the System for Environmental and Economic Accounting – Ecosystem Accounting (SEEA-EA).

Characteristic	**Variable descriptor**	**Unit**	**Variable values**	**Transformation score (out of 100)**
Abiotic characteristics	Exposure to weather and currents	Ordinal category	Exposed (e)	100
			Exposed - pocket beach	50
Physical	Number of extreme wave events (greater than 4m of a 4 hr duration, 2008-2015)	Integer	More than 40	100
			30 to 39	80
			20 to 29	60
			10 to 19	40
			0 to 9	20
Chemical	Data deficient	Data deficient	N/A	N/A
Biotic characteristics **Composition**	Proportion of native to exotic species	Index of native to exotic plant species (-1 to 1)	0.7 to 1.0 (mostly- all native)	100
			0.5 to 0.7 (mostly native)	80
			0.01 to 0.5 (about equal)	60
			-0.5 to 0 (mostly exotic)	40
			-1 to - 0.5 (very low)	20
Structure	Vegetation structure	Presence/ absence exotic woody species	Absence of exotic woody species	100
			Presence of exotic woody species	0
Function	Characteristic flora: Native sand binders – remnant	Presence / absence	Present	100
			Absent	0
	Characteristic flora: Native sand binders – planted	Ordinal category	Planted plus remnant, or remnant only	100
			Planted only	50
			No planted or remnant	0

Table 2. (cont.)

Characteristic	Variable descriptor	Unit	Variable values	Transformation score (out of 100)
Landscape characteristics	Fragmentation	Geographic distance (km) from site to nearest road	5 - 35	100
			1 - 5	80
			0.1 - 1	60
			0.02 - 0.1	40
			0 - 0.02	20

Table 3. Worked example of how variables were transformed into indicator scores. Raw data for active sand dune characteristics (column A) is transformed to a score (column B) on a common scale where 5 is good and 1 is poor, based on scoring in Table 2. The transformed score is then compared to the high reference site score (column C) and the low reference site score (column D) using the formula ((B-D)/(C-D)). The resulting score is the indicator score for that characteristic, at that site (column E). Note, because some characteristics have a transformed score which is the same as the reference score, a division with zero as the numerator is required, returning an indicator score of zero.

Characteristics	**Unit**	**A**	**B**	**C**	**D**	**E**
		Cable Bay (raw variable data)	**Cable Bay (variable transformed)**	**Reference site value (V_H)**	**Reference site value (V_L)**	**Cable Bay (indicator score)**
Exposure to weather and currents	Ordinal category	E (exposed)	100	100	50	1
Average annual number of extreme wave events greater than 4m of a 4 hr duration, 2008-2015	Integer	10	40	100	20	0.25
Proportion of natives to exotics	Ratio -1 to 1	-0.25	40	100	20	0.25
Vegetation type	Presence/ absence exotic woody species	0	0	100	0	0
Characteristic flora: Native sand binders - remnant	Presence / absence	0	0	100	0	0
Characteristic flora: Native sand binders - planted	Ordinal category	0	0	100	50	-1
Landscape (fragmentation)	Km from site	0.02	20	100	20	0
Total			**200.00**	**700.00**	**160.00**	**0.50**

2.3. Results

2.3.1 Change in extent of active dunes in Aotearoa

The combined total extent for the 22 sample beaches (i.e., the extent of active dune ecosystems) reduced by 59% between the 1950s and 2021. Eighty percent of this reduction occurred between the 1950s and 1970s - equating to 52% of the 1950s area (Fig.5). Over the 1970s, 80s and 90s, active dune extent remained largely constant, but then decreased a further 20% of the1950s area, between the 1990s and the current assessment (Fig. 5). The average annual rate of decline in active dune ecosystem extent since the 1950s is 85 ha per year.

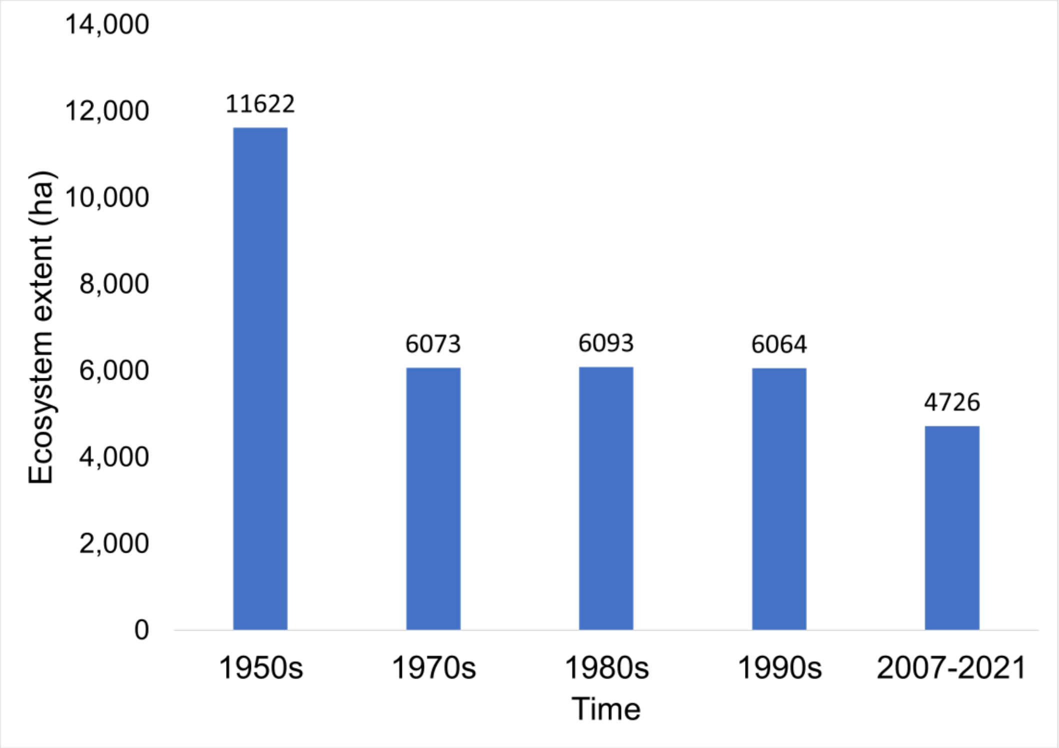


Fig. 5. Total estimated active sand dune ecosystem extent in hectares for all sample sites in this study over the last 70 years, excluding Mangangu Stream Beach and Ōrewa Beach, which were not mapped by Hilton et al. (2000). Sample sites were identified from polygon map datasets by Hilton et al. (2000), and ecosystem extent was calculated per site using ArcGIS Pro v.8.0 (Esri, 2020) and summed for the 1950s, 70s, 80s, and 90s. Extent for sites 2007 – 2021 was identified and hand digitised in ArcGIS Pro, verison 2.8 (Esri, 2020) from aerial imagery, comprising the most recent

and publicly available, high resolution, ortho-rectified aerial imagery from Land Information New Zealand (LINZ) Data Service under a Creative Commons open licence (CC BY 4.0). Snapshot imagery was taken by local and regional councils over the period 2007- 2021 and comprised red, green and blue colour bands. The original resolution ranged from 0.075 m to 7.5 m and came in New Zealand Transverse Mercator projection. The years 2000-2007 were not included in this analysis since only the most recent imagery per site was used, resulting in snapshot imagery of beaches between 2007 and 2021.

Of the 22 active sand dune sites, 14 individual sites had decreased in extent between 1950 and 2021, seven had increased, and three were data deficient (Table 4). On average, individual sites decreased by 33% between the 1950s and 2021.The largest decreases were at Muriwai Beach and Pakiri Beach (95%) and the largest increase was at Whatipu Beach (107%).

Table 4. Change in ecosystem extent for sampled sites, 1950s-2021. Area (ha) was derived from hand digitised maps from aerial imagery in different decades. DD indicates a lack of data for that site (data deficient). The years 2000-2007 were not included in this analysis since only the most recent imagery per site was used, resulting in snapshot imagery of beaches between 2007 and 2021.

	Data from Hilton et al. (2000)				**Hand drawn (this research)**	
Beach	**1950s area (ha)**	**1970s area (ha)**	**1980s area (ha)**	**1990s area (ha)**	**2007/2021 area (ha)**	**% change since 1950s**
Muriwai Beach	2798	779	779	779	137	-95
Pakiri Beach	1703	241	241	241	88	-95
Waikouaiti Beach	80	37	37	31	17	-78
Cable Bay	3	3	3	3	1	-76
Himatangi Beach	3648	1476	1476	1477	919	-75
Paraparaumu Beach	73	73	94	94	21	-72
Ōtaki Beach	49	14	14	14	15	-70
New Brighton	200	48	48	48	73	-64
St Kilda and St Clair Beaches	13	13	13	13	5	-62
Rarawa Beach	47	47	47	47	20	-58
Ōtama Beach	17	18	18	18	12	-28
Spirits Bay	100	100	100	100	76	-24
Smoky Beach	62	62	62	51	50	-18
Scott Point to Waikoropupunoa Stream	1895	1894	1894	1880	1929	2
Tāwharanui	25	25	25	25	25	0
Coal River	14	14	14	14	16	14
Ponaki Beach	152	152	152	152	181	19
Kaitorete Spit	535	535	535	535	682	28
Whatipu Beach	200	499	499	499	414	107
Ōhope Beach*	DD	32	32	32	37	DD
Mangangu Stream Beach	DD	DD	DD	DD	11	DD
Ōrewa Beach	DD	DD	DD	DD	2	DD
Total area (ha)	**11619**	**6073**	**6093**	**6064**	**4739**	**-59**

2.3.2 Gradient of active dune ecosystem integrity

Most sites (n = 17) had relatively higher rankings on the new gradient compared to the Johnson and Partridge gradient. Differences in rank between the two gradients represents change in the status of sites over time and/ or the difference between the two ranking systems. Better condition sites with high rankings on both gradients were more similar in their rankings (Figs. 6 and 7). At the lower end of the Partridge (1992) and Johnson (1992) gradient, there was little distinction among sites; in contrast, these same sites were well distinguished on the new gradient (Figs. 6 and 7). Conversely, sites with mid-range scores, between 60 and 80 percent rank on the 2022 gradient, were spread across a wider range of values on the 1992 gradient (Figs. 6 and 7).

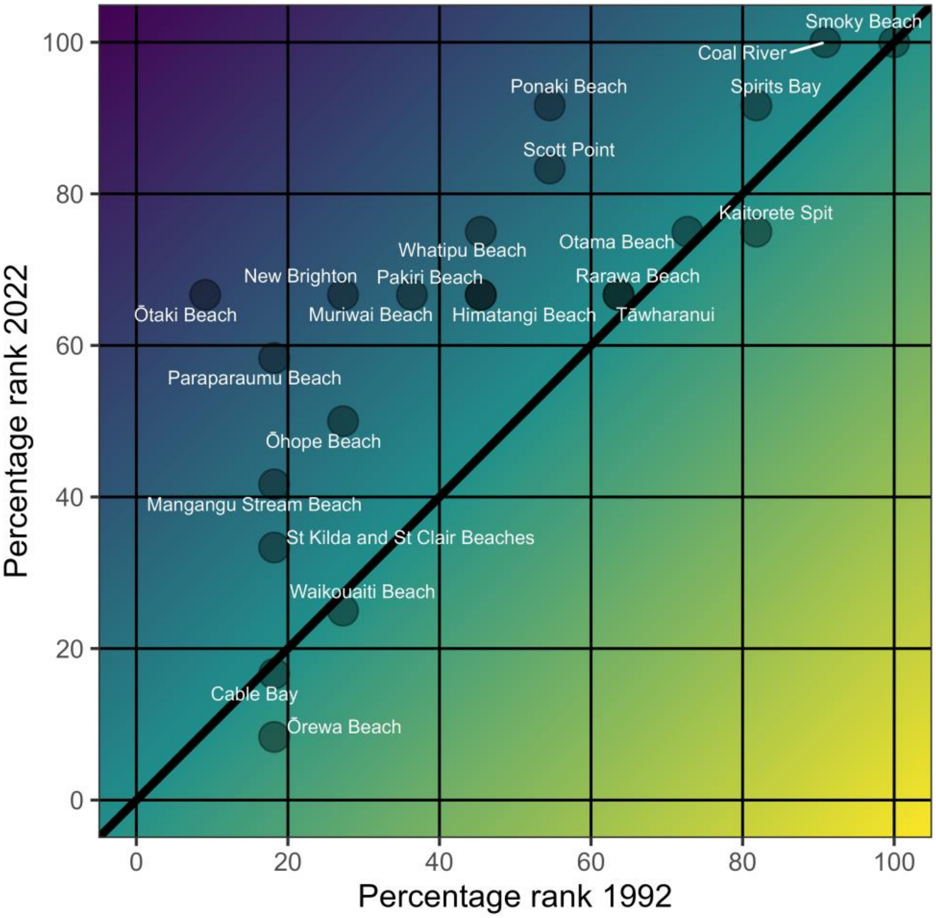


Fig. 6. Comparison of two gradients of active dune ecosystem integrity using 22 selected active sand dune sites selected to represent a range of active dune condition and geographic representation of sites around Aotearoa. The thick, black line is the 1:1 relationship; points above this line were ranked relatively higher on the new gradient presented in this paper compared to the ranking by Johnson (1992) and Partridge (1992). The background colour scale represents the relative degree of match (teal) or mismatch, as either improved rank over time (blue) or decreased rank over time (yellow). Two pairs of points completely overlap: Rarawa Beach and Tāwharanui, and Himatangi Beach and Pakiri Beach. ‘Scott Point’ represents the site ‘Scott Point to Waikoropupunoa Stream’.

Fig. 7. Selection of sites and their rankings (high, mid - range or low) from the new 2022 ranking (this study). From left to right and top to bottom: a.) Smoky Beach, high (Alistair Hay); b.) Spirits Bay, high (Thomas Buckley); c.) Pakiri Beach, mid - range (Cate Ryan); d.) Whatipu Beach, mid - range (Graham Hinchliffe); e.) New Brighton, low (Cate Ryan); f.). Ōrewa Beach, low (Graham Hinchliffe, unpublished data).

2.4. Discussion

Using a case study of active sand dunes in Aotearoa, the UN SEEA-EA method has been successfully implemented and tested to create a narrative of the state of ecosystem integrity at a national scale. Given the ubiquity of the drivers of active dune condition globally, it is possible the method could also be successfully used on active sand dune ecosystems in other countries. Our research shows that dune extent in Aotearoa continues to decline, in line with trends globally (Gao et al., 2020; Jackson et al., 2019). The new factors incorporated to assess ecosystem integrity correctly identify sites at the high end of the gradient, produces finer scale scoring for sites at the lower end of the gradient, but mid-range sites are less well distinguished. Reasons for discrepancies in results with previous studies are most likely due to the long interval between surveys, given active sand dunes are dynamic systems subject to both natural processes and human impacts. Despite data gaps and constraints (described in the next sections), the assessment of ecosystem extent and this gradient of ecosystem integrity is both feasible and highly useful in building a narrative on the state of active sand dunes in Aotearoa, which can be of use for coastal management. Moreover, the method can be repeated to show how these systems are changing.

Our estimated decline of 59% of active dune extent between the 1950s and 2021 differs from Hilton et al. (2000), who estimated the historical decline at 70% nationwide between the 1950s and the 1990s. Holdaway et al. (2012) estimated the decline at 50-80% since the 1750s (based on Hilton et al. 2000), and Stats New Zealand estimated the decline at 80.5% between the 1950s and 2008 (Stats New Zealand, 2015). Our estimated decline of 59% in Aotearoa's active dunes is equivalent to a status of an 'Endangered Ecosystem' in the IUCN Red List Ecosystem Risk Assessment framework (IUCN, 2016), which concurs with the status given by Holdaway et al. (2012).

The differing timescales between the studies are likely to be an important reason for discrepancies between Hilton et al. (2000) and this study, since the Hilton dataset would have been collected more than 30 years ago, and the Stats New Zealand dataset at least 14 years ago. Given that active dunes are highly dynamic systems, we can expect significant variation in extent over time due to natural processes and human impacts. For example, the extent of Muriwai Beach has decreased significantly since Hilton et al. (2000) recorded 779 ha in 1990, compared to the 137 ha in this study. Over that time, there has been a natural reduction in sediment supply and erosion on this stretch of coast (McKelvey, 1999; Boyle et al., 2016) as well as extensive planting to stabilise the dunes with marram and the growing of Monterey Pine (*Pinus radiata* D. Don) as a commercial crop (McKelvey, 1999).

Different methods will likely have a bearing on the different results among studies. Both Hilton et al. (2000) and the Stats NZ survey used spatially explicit methods to estimate the extent compared to the representative sample used in this study. Additionally, different approaches were used: Stats NZ analysed large scale GIS datasets and satellite imagery, whereas Hilton et al. (2000) hand mapped sites from aerial imagery, as in this study, albeit in the 1990s. Different data sources will also have an effect since image quality and resolution have improved since the 1990s. Such issues of incompatibility and lack of standardisation are inherent when comparing data and results from different studies (Krebs, 2014) and are a limitation of this study that has arisen due to a lack of existing national datasets on the spatial distribution of active dunes in Aotearoa.

The new gradient of ecosystem integrity tended towards higher scores compared to the 1992 ranking; most sites (n = 17) scored higher (Fig. 6). This could be explained by the method used to generate the ranking comparison and ecological changes at the sites over time. The Partridge (1992) and Johnson (1992) sites were selected for this study using a stratified sample of high, mid - range and low scoring sites, and because many of the sites had the same score, when new scoring criteria was applied many sites were differentiated. This was especially true for the lower scoring 1992 sites. Ecological changes, caused by either natural or human processes, could further distinguish the sites. For example, Ōrewa Beach, Cable Bay, St Kilda and St Clair Beaches, Paraparaumu Beach and Mangangu Stream Beach all scored equally in the 1992 rank, resulting in these sites occurring just below 20% on the 1992 *x*-axis, but these sites had a much wider spread on the corresponding 2022 *y*-axis (Fig. 6). These sites all scored very differently from each other across the new indicators for abiotic processes (physical: disturbance) and biotic processes (function: planted and remnant sand binders). Conversely, sites with a wide spread of low to mid - range scores on the 1992 axis, such as Himatangi Beach, Muriwai Beach, New Brighton, Ōtaki Beach, Pākiri Beach, Rarawa Beach, and Tāwharanui (Fig.6), were spread across a narrower range of values in the 2022 gradient (between 60 and 80 percent rank). This result is likely due to these sites having the same score for vegetation structure (exotic woody vegetation) and function (planted and remnant sand binders), reflecting the limited, coarse data available at the requisite scale, whereas continuous, vegetation cover type data at site level could better draw out differences. Sites with high rankings on both gradients, such as Smoky Beach, Coal River and Spirits Bay were more similar in the new gradient (Fig. 6). This is likely due to criteria converging, for example in both gradients top scores go to sites with few or no exotics species and populations of native sand binders. Moreover, these sites are unlikely to have changed much over time due to human impacts since they are in high conservation protection areas. Overall, the spread of scores in the new gradient suggests the new indicators are effective at identifying high quality sites, drawing out differences between sites at the lower end of the gradient, but it also suggests that data for structure and function characteristics is too coarse to clearly draw out the differences between mid - range sites.

A few sites decreased in score compared to the 1992 gradient. New indicators that had the most influence on these sites included species composition, vegetation structure and distances to the road (fragmentation). For example, Kaitorete Spit scored extremely high on the 1992 gradient but dropped several places on 2022 gradient due to a relatively high proportion of exotics to natives, presence of woody exotics and close proximity to roads. Moreover, the datasets available to use with this new gradient do not include abundance estimates, such as percent cover of characteristic plants, or botanical rarity. These are both likely to have heavily influenced the higher score awarded to Kaitorete Spit by Johnson (1992), since it has populations of rare endemic plants (*Carmichaelia appressa* G. Simpson), rare, vegetated dune deflation hollows, and the largest, most continuous population of *F. spiralis* in Aotearoa (Johnson, 1992).

The gradient could be refined since disparate datasets were aligned and overlayed in this research, but gaps in the data remain. A better understanding of the contribution of different species to ecosystem structure and function, and decline in ecosystem processes, can be obtained from estimates of the relative abundance of different vegetation types (Holdaway et al., 2012), through vegetation cover type analysis of remotely sensed imagery. It follows that, spatial patterns of different vegetation types could be identified, related to ecosystem integrity, and monitored for changes at a national scale. Other factors that could be relevant to include are the presence and abundance of characteristic native fauna, such as shore dwelling and nesting birds, lizards and invertebrates; the presence and abundance of rare plants; the presence and abundance of exotic fauna; and a more multi-faceted, finer-scale assessment of human impacts. Mātauranga Māori, or indigenous knowledge and perspectives, are highly relevant in the context of conservation ecology in Aotearoa (Bellingham et al., 2021; Lyver et al., 2018) and globally (IPBES, 2019). The inclusion of *F. spiralis* as an indicator, a species important in Māori culture and practices (Hare et al., 2019), can provide a starting point for the development of a cultural assessment of the condition of active dunes that is relevant and meaningful to Māori in Aotearoa (Lyver, 2018; Normyle, 2022). In addition, the linking and valuing of the science of natural processes to human use and impacts is in its infancy in Aotearoa and the rest of world, and validation of assumptions, methods and results are required over a range of spatial and temporal scales.

From a management perspective, an indexed rank of active dune ecosystem integrity and extent is useful to both prioritise areas for conservation protection and monitor sites for direction of dune recovery. In addition, the framework could be used to create a baseline for a joined-up view of active dune ecosystems through indicators, that can be applied to a range of issues such as biodiversity conservation and coastal hazard management. Due to the similarities between active dune system drivers globally (Gao et al., 2020; Jackson et al., 2019), and the ability to select characteristics to suit different contexts, this framework can be translated to active dune ecosystems around the world.

Since the method used is grounded in the ecosystem integrity concept it is broadly compatible with national and international frameworks, such as the New Zealand Department of Conservation Biodiversity Assessment Framework (McGlone et al, 2020; Lee et al., 2005), the UN SEEA-EA accounts, Essential Biodiversity Variables, draft Goal A of the Post - 2020 Global Biodiversity Framework, and the Sustainable Development Goals (UN, 2021).

Long-term or nationally representative surveys of active dune extent and condition are rare globally (Schlacher et al., 2008, Holdaway et al., 2012). Our new gradient is the first since the 1990s to assess ecosystem extent and integrity at a site and national level in Aotearoa. It has built on previous studies and improves our understanding of active dunes in Aotearoa. The challenges identified reflect those identified in similar studies attempting to implement the SEEA-EA and other frameworks that assess ecosystem extent and integrity around the world, namely, data gaps and lack of sufficiently regular temporal or spatial data (Holdaway et al., 2012; Farrell et al., 2021). Since our method relies on a representative sample of active dunes rather than all active dune sites, it is suitable for countries of any size. Nonetheless, we conclude that this first attempt provides an important steppingstone in the development of an effective, globally relevant monitoring framework specific to active dunes, with ecosystem integrity at the core of its purpose. Our methods are repeatable, and the new gradient can be continued to be refined and improved over time with the incorporation of new data.

3 Quantifying vegetation cover on coastal active dunes using nationwide aerial image analysis

3.1. Introduction

Coastal active dunes (active dunes) occur on dynamic sandy shorelines at all latitudes (Martínez et al., 2004) as a result of aeolian and marine sediment transport and ecological processes (Biel et al., 2017, 2019; Durán & Moore, 2013; Jay et al., 2022; Hacker et al., 2012; Hesp & Walker, 2021; Hovenga et al. 2019, 2021, 2022; Martínez et al., 2004; Miller et al., 2010; Ruggiero et al., 2018; Zarnetske et al. 2012, 2015). Active dunes occur where sand is highly mobile, such as incipient dunes and the most seaward dune or foredune, and have sparse, low-growing herbaceous vegetation (Durán & Moore, 2013; Miller et al., 2010; Psuty, 2004). Behind the foredune, semi-stable dunes are active to a lesser extent as geomorphic activity decreases and the diversity of plant species, including woody species, increases (Maun, 2009; Psuty, 2004; Wardle, 1991). Active dunes provide important ecosystem services; they are a barrier to storm surge and sea level rise (Biel et al., 2017; Seabloom et al., 2013; Walker et al., 2013), sustain recreation, tourism and cultural practices (Barbier et al., 2011), and are a habitat for biodiversity (Martínez et al., 2013). Despite their importance, active dunes are under significant pressure from non-native species invasions, human settlement, recreation, farming, forestry, and extractive industries (Gao et al., 2020). Faced with these threats, the extent of active dunes is declining globally (Gao et al., 2020; Jackson et al., 2019), and their floristic composition is undergoing rapid change (Gao et al., 2020; McGuirk et al., 2022).

Active dunes in Aotearoa, New Zealand, provide a case study that mirrors the international situation. Aotearoa has *c*.15,000 km of coastline (Bell & Gibb, 1996) containing a subset of active dunes characterised by frequent natural disturbance and sparse, low-growing vegetation (Hilton et al., 2000). Sandbinding plants, such as the endemic *Ficinia spiralis* (A. Rich.) Muasya et de Lange and native *Spinifex sericeus* R.Br. occur on highly active dunes and act as ecosystem engineers, building foredunes and a range of geomorphic habitats for other species (Hesp, 2000). Semi-stable mid and rear dune vegetation comprises a small number of prostrate, divaricating, spreading woody shrub and liana species that trap and accrete sand (Cockayne, 1911; Johnson, 1993; Newsome, 1987; Wardle, 1991). In addition, the dune slacks and hollows associated with active dunes have unique, endemic and threatened floral assemblages (Holdaway et al., 2012). Active dunes in Aotearoa provide a range of important cultural, recreation, and coastal protection services, yet they are declining in extent (Stats NZ, 2015), are endangered (Department of Conservation 2020; Holdaway et al., 2012) and are threatened by forestry, farming, introduced invasive weeds, coastal development and recreation (Hilton, 2006). Data about active dunes are generally sparse, qualitative, and insufficiently representative to infer national-scale condition and extent (Ryan et al., 2023).

In recent decades, techniques have been developed for use with remotely sensed imagery to complement and replace a range of time and labour-intensive field survey methods (Turner et al., 2015), providing quantitative datasets at increased spatial coverage and temporal frequency (Hugenholtz et al., 2012; Murray et al., 2018). Current options for remotely sensed imagery that provides information about ecosystem structure and function require tradeoffs between spatial and temporal resolution (Morgan et al., 2010) and cost. For example, current and historic, freely available Landsat or Sentinel 2 satellite imagery offers freely available imagery at revisit rates of every 30 or 10 days (respectively). However, the spatial resolution of 30 m^2 and 10 m^2 is often too coarse to discriminate vegetation species or types for algorithm training purposes (Carrión-Klier, 2022; Morgan et al., 2010; Oddi et al., 2021), depending on the target species or type. Even the commercially available imagery supplied from current generation of satellite-based sensors, which can capture large swathes of the earth's surface at very high resolution, are relatively coarse for discrimination of vegetation in training. For example, the Pleiades multispectral sensor at 2 m^2 spatial resolution (European Space Agency, 2024), or the highest spatial resolution multispectral imagery from private providers such as Blacksky Images at *c.*1 m2 (Blacksky, 2024). The next generation of satellite-based sensors will likely offer even higher, sub 1m^2, spatial resolution, such as Blacksky at 35 cm^2 (Blacksky, 2024) providing a promising direction for analysis and high-frequency monitoring of ecosystem structure and function.

Aerial imagery offers a sub 1 m^2, high spatial resolution alternative to satellite imagery (Morgan et al., 2010). Aerial imagery is a vastly underused remotely sensed dataset for describing baseline ecosystem condition and long-term monitoring (Morgan et al., 2010). Typically comprised of the electromagnetic spectrum's red, green and blue (RGB) bands, aerial imagery offers spatially continuous images at sub-metre resolutions, allowing for vegetation mapping across larger areas than traditional field surveys (Morgan et al., 2010). Aerial imagery, with a much higher spatial resolution, is well-suited for detecting ecological patterns using supervised classification compared to satellite imagery (Morgan et al., 2010; Turner et al., 2015). Additionally, it offers greater spatial coverage than imagery captured by Unoccupied Aerial Vehicles (UAV) due to the higher elevation at which aircraft are flown.

Within the active dune literature, aerial imagery has been increasingly used as input datasets to discern dune vegetative features, spatial pattern or extent using traditional pixel-based, supervised classification techniques (for example, see Gao et al., 2021; Konlechner et al., 2015; Moulton et al., 2021; Ryu & Sherman, 2014; Smyth et al., 2020). Object-Based Image Analysis (OBIA) is an image classification technique that has recently gained popularity over pixel-based methods due to perceived improvements in accuracy (Blaksche et al., 2010; Ye et al., 2018). OBIA involves segmenting images into discrete objects based on the homogeneity of spectral and textural qualities of pixels, followed by the classification of these objects (Lillesand et al., 2008). Improvements over pixel-based methods are attributed to minimising spectral variability across classes and integrating spatial and contextual relationships (Ma et al., 2017; Ye et al., 2018). OBIA is well suited for classifying active dune vegetation compared to pixel-based techniques because it identifies boundaries between classes, accommodating the fuzzy, spatially variable nature of vegetation in ambiguous backgrounds, such as natural landscapes (Blaksche, 2010). To our knowledge, no studies of active dune vegetation have used OBIA techniques with aerial imagery and the few studies that do use OBIA in dune environments used ultra high-resolution (≤ 0.1 m^2) imagery (Agrillo et al., 2023; Belcore et al., 2024; Innangi et al., 2023; Marzialetti et al., 2021) or used Light Detection and Ranging (LiDAR) and hyperspectral imagery (Brownett & Mills, 2017).

Given the clear need to characterise and quantify the vegetation cover of active dunes, we evaluated the application of OBIA techniques to RGB aerial imagery for this purpose across a selection of nationally representative active dunes. Our research objectives were to: (1) determine the cover classes on dunes that could be discriminated from aerial imagery using OBIA image classification procedures, (2) identify a national active dune cover typology from the classification data, and (3) identify the potential drivers of variation in the percent cover of active dune vegetation cover types in 90 × 120 m plots across Aotearoa. This study provides a better understanding of the capabilities and limitations of aerial imagery for characterising the vegetation of active dunes. Further, this work contributes an initial, national, quantitative typology of the vegetation of active dunes in Aotearoa and demonstrates a low-cost approach for assessing the influence of human and geomorphic variables on active dune vegetation cover, of use to conservation and land managers.

3.2. Material and Methods

Pre-processing, classification and post-processing of aerial imagery (Fig. 8) were carried out on a stratified random subsample of 21 representative beaches (Fig. 9) drawn from a maximum potential extent of active dunes in Aotearoa (Ryan et al., 2023). Publicly available, high-resolution (0.075-0.75 m) RGB ortho-imagery collected by local and regional government agencies for the period 2007 to 2021 (Chapter 3, Appendix 1) was compiled for this research. All imagery was processed separately for each beach in ArcGIS Pro version 2.9 (Esri Inc., 2021).

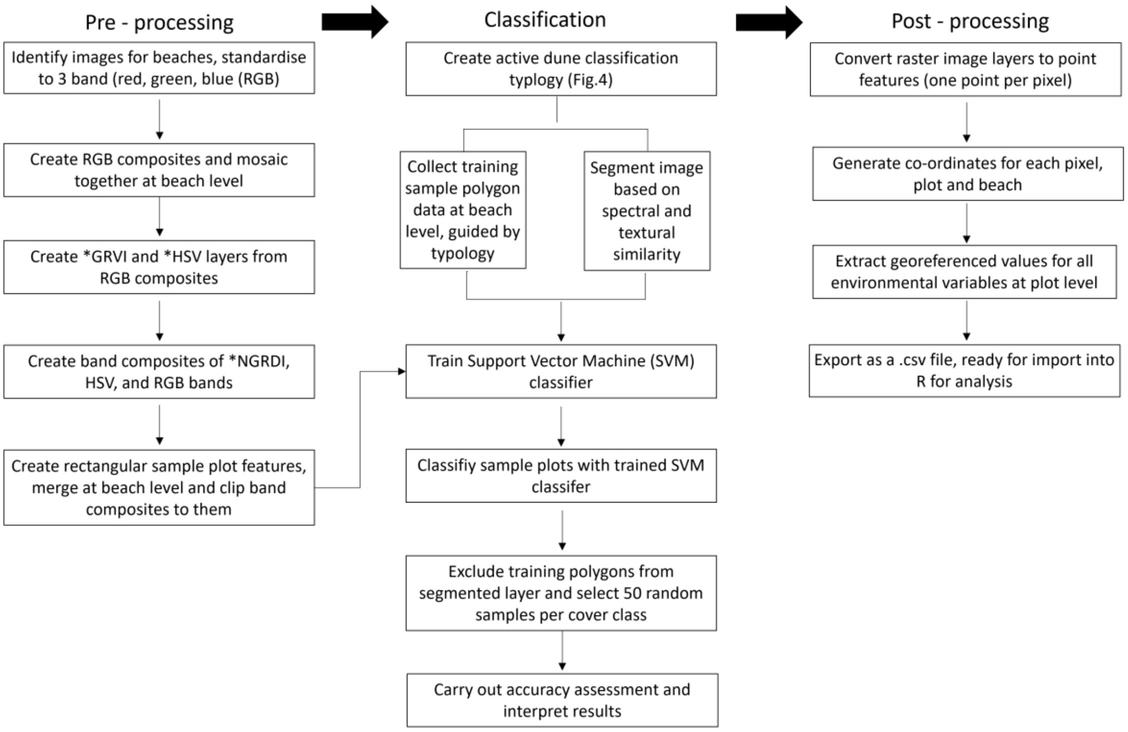


Fig. 8. Steps taken to process and classify imagery. All image processing was carried out in ArcGIS v.2.9. Imagery datasets are given in Chapter 3, Appendix 1. * NGRDI is the Normalised Green – Red vegetation index; HSV is Hue, Saturation, Value (*sensu* Smith, 1978).

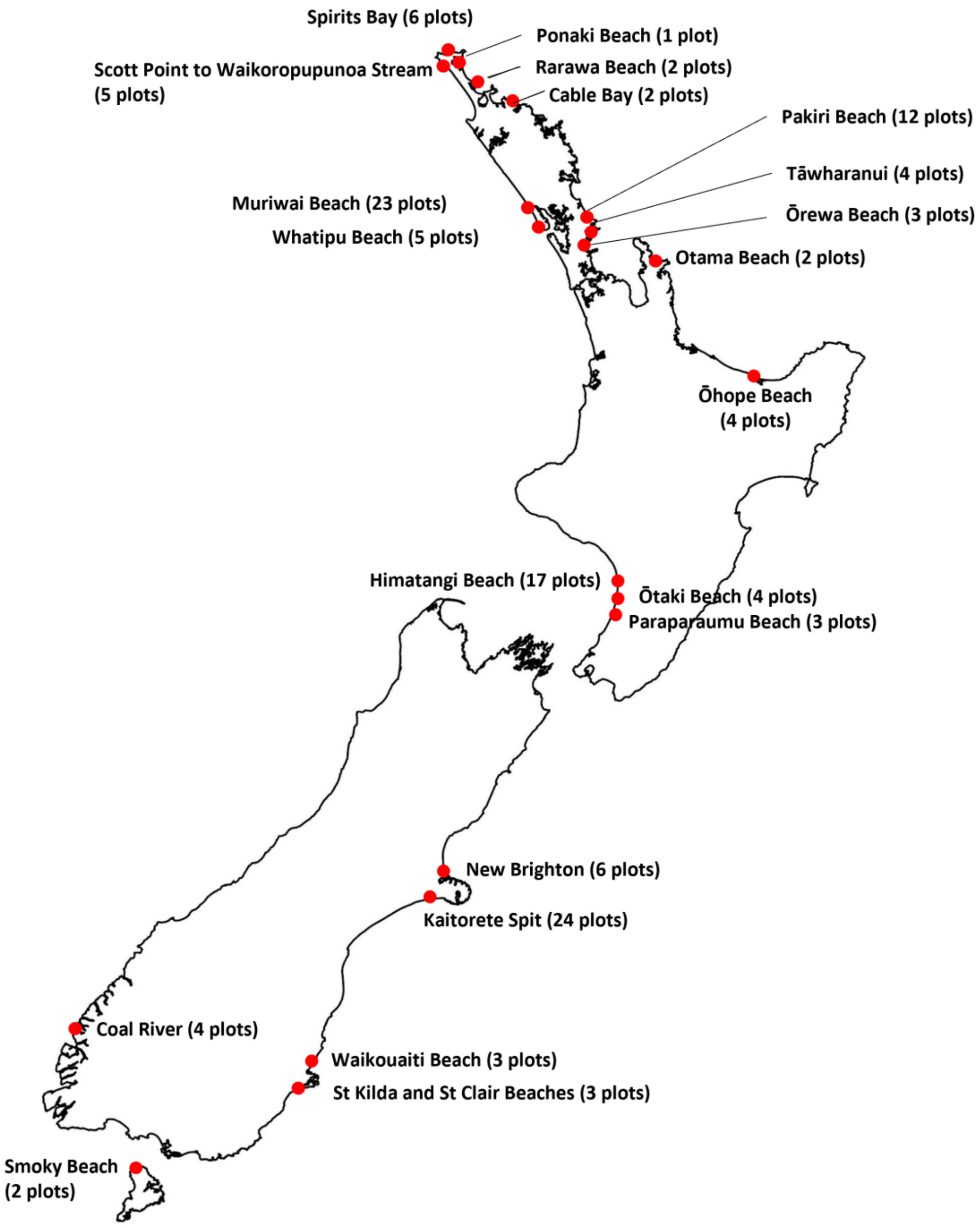


Fig. 9. Beaches selected as a representative sample of active dunes in Aotearoa (*n* = 21). Numbers in parenthesis represent the number of sample plots per beach. The approach to select beaches is described in Ryan et al. (2023). Note, the sample of beaches from Chapter 2 includes Mangangu Beach (Fig. 3) but it is excluded from the Chapter 3 analysis since plots did not meet the threshold of containing more than 5% vegetation cover and held insufficient information about active dune vegetation processes to warrant inclusion.

3.2.1. Pre-processing

To enhance the contrast between vegetation, sand, and soil in the classification, the RGB colour bands were used to calculate values for the Normalised Green - Red Difference Index (NGRDI; Chapter 3, Appendix 2; Hunt et al., 2005) for each beach. NGRDI can also reduce variation in illumination in imagery due to differences in light and shade (Hamadu et al., 2016). RGB colour bands were also transformed to the alternative colour space "Hue Saturation and Value" (HSV) (*sensu* Smith, 1978). NGRDI, HSV, red, green, and blue band layers were then brought back together into one composite band raster for each beach.

Dunes were then subsampled at each beach to identify spectrally distinct cover classes. Virtual sample plots that were 90 × 120 m rectangular features were created in the GIS and then positioned within beach images, with the shorter side aligned perpendicular to the shoreline, running landward from the seaward toe of the incipient dune (Fig.10). This plot size was selected to contain sufficient information to characterise the vegetation of foredunes, consistent with other research internationally on large foredune systems (Hesp, 2002; Jay et al., 2022; Ryu & Sherman, 2014) and incorporating previous descriptions of large dune ecosystems in Aotearoa (Johnson, 1993; Wardle, 1991). Selecting the width of the largest foredune allowed us to accommodate variation in width among beaches and capture vegetation of semi-stable and stable dunes where they occurred. Three beaches were sampled using a smaller plot of 20 × 27 m since foredunes were foreshortened due to coastal development. Plots were placed along foredunes in the imagery every 1 km along the length of the beach, starting from a random location. Highly modified areas, such as housing, roads or car parks, were excluded from the analysis. Plots that were primarily sand, i.e., those that did not contain at least 5% vegetation cover, were also excluded from the analysis since they would not yield sufficient information about vegetation.

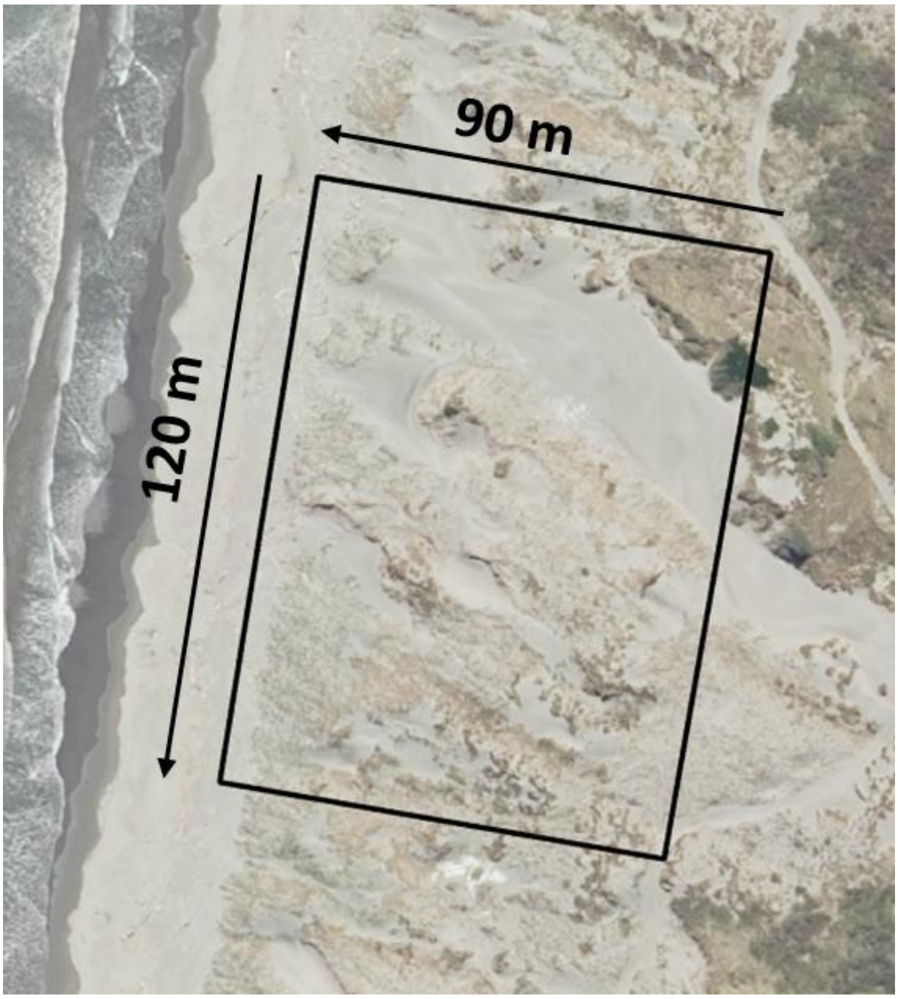


Fig. 10. Example placement of one 90 × 120 m sample plot within a foredune at Himatangi Beach. Plots were placed every 1 km along each beach, starting from a random point at the seaward toe of incipient dunes, landward. Sample plots were created as polygon features in ArcGIS Pro v.2.9. Imagery is from the aerial photograph tile Manawatu Whanganui 0.3m Rural Aerial Photos (2015-2016), from Land Information New Zealand Data Service (CC BY 4.0).

All 21 beaches contained foredunes, with the smallest beaches containing one plot (*n* = 1) and the largest beach containing 24 plots, resulting in a total of 135 plots at beaches across the country (Fig. 9). The NGRDI, HSV, red, green, and blue band composites were clipped to the plot polygon features for each beach. Nearest neighbour resampling to 1 m^2 was then carried out within plots for consistency across all images and analyses.

3.2.2. Classification

Classification comprised the collection of training samples, segmentation, training of the classifier algorithm and then classification. To guide the collection of training samples for use in the discrimination among cover classes during classification, a vegetation cover typology (Fig.11) was developed based on botanical records, photographs, imagery, expert knowledge, and beach visits. This typology, comprising 22 cover classes, was specifically created to differentiate stable and active cover classes and, within the active cover classes, among different types of sandbinder species (Fig.11). Examples of species or objects that could occur in each class are given in Chapter 3, Appendix 6. Based on this typology for each beach, a minimum of 30 training samples (groups of pixels within hand digitised training sample polygons) from within all plots were taken of each cover class present, comprising red, green, blue, NGRDI and HSV data. Beaches were sampled independently due to variation in weather and illumination conditions across beaches when the imagery was captured.

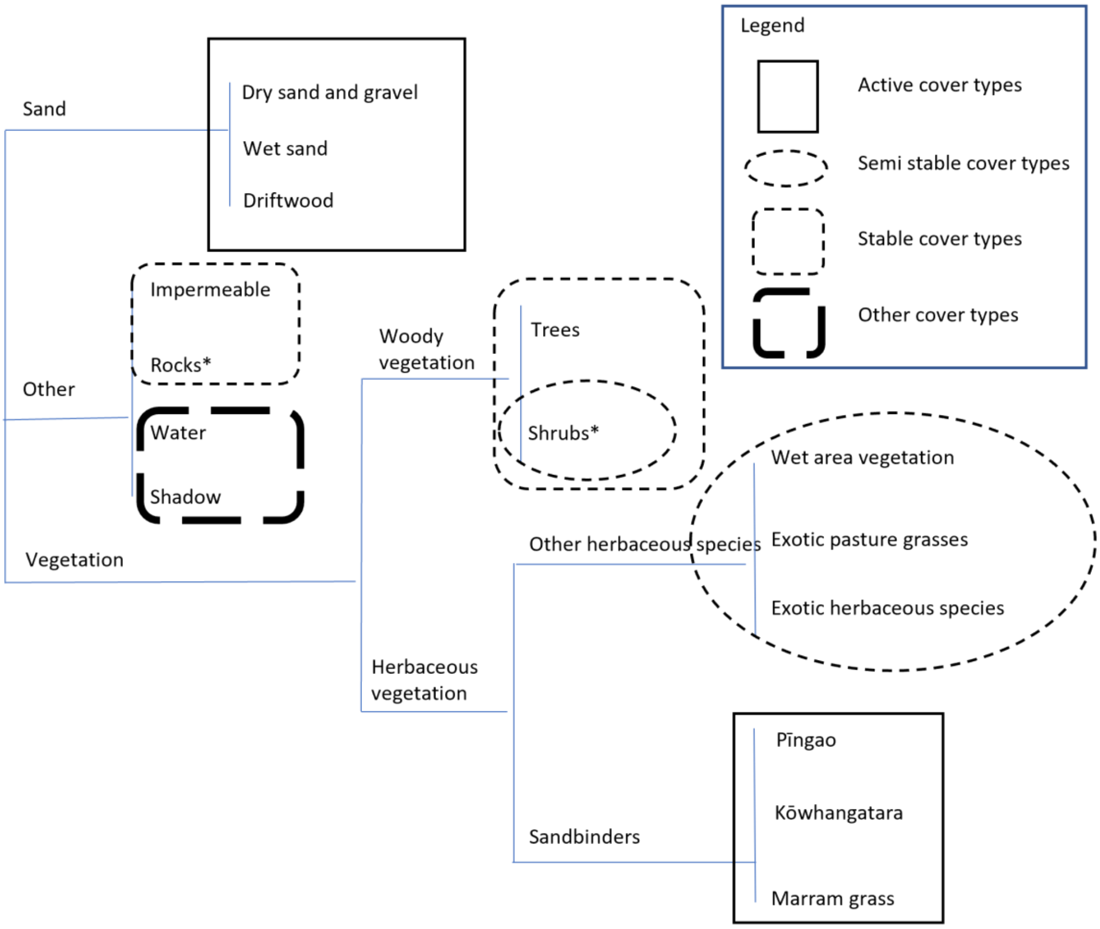


Fig. 11. Initial cover class typology, developed to guide the collection of training samples. Cover classes were based on botanical records, photographs, imagery, expert knowledge and beach visits and distinguished between stable, semi-stable and active cover classes. Wet area vegetation refers to areas with freshwater or brackish water seepages, such as dune slacks. Where the term "Pīngao" is used, it is equivalent to *Ficinia spiralis*. Where the term "Kōwhangatara" is used, it is equivalent to *Spinifex sericeus* and "Marram grass" is used, it is equivalent to *Ammophila arenaria.* Examples of species or objects that could occur in each class are given in Chapter 3, Appendix 6.

Next, band composites were segmented using the Mean Shift method, whereby neighbouring pixels with similar spectral and spatial characteristics were grouped. The Mean Shift algorithm was configured for the highest level of sensitivity (20) for fine-grained discrimination of features. Images were segmented with a minimum segment size of 1 m^2, with the expectation that it would capture the qualities of the target native sandbinder plants, which are typically 1 – 4 m in minimum diameter, with narrow rhizomes or stolons and a lateral growth habit (Wardle, 1991). Given the narrowness of native sandbinders, we expected to capture mixed pixels of sand and native sandbinders within segments and training samples.

A limited range of object-based classifiers were informally trialled to determine an optimal classifier solution that balanced ecological interpretation and classifier training optimisation. The supervised classification techniques, Random Forest (RF) and Support Vector Machine (SVM), and the unsupervised classification techniques *k* – means and ISO Cluster were used at a selection of plots on beaches and reviewed for accuracy of classification. The SVM classifier was selected since visual inspection of results mapped well to features in the imagery compared to the other classifiers trialled. SVM is well suited to defining and maximising boundaries between fuzzy classes, such as in vegetation (Blaksche, 2010; Lillesand et al., 2015; Müller & Guido, 2016; Richards, 2022). The SVM classifier has advantages over other classifiers in that it does not require samples to be normally distributed and performs at a high level of accuracy (Lillesand et al., 2015; Ma et al., 2018; Melgani & Bruzzoni, 2004; Richards, 2022). Moreover, SVM is effective when, as in this case, the sandbinder target features are smaller than the pixel size since SVM maximises the margin between different classes by using a kernel trick to transform data into a higher-dimensional space, where it can identify optimal hyperplanes that separate classes, capturing subtle patterns that correspond to small target areas within a larger pixel (Lillesand et al., 2015; Müller & Guido, 2016; Richards, 2022).

In contrast, RF creates an ensemble of decision trees, each trained on a random subset of the data and features, and then makes a classification decision based on the majority vote of the trees (Lillesand et al., 2015; Richards, 2022). RF can struggle when objects smaller than a pixel are underrepresented because RF may favour more dominant classes and inaccurately classify them. Conversely, SVMs focus on maximizing the margin between classes and can be fine-tuned to ensure that minority classes are not ignored (Lillesand et al., 2015; Richards, 2022). RFs are also less effective in high-dimensional feature spaces compared to SVMs. OBIA extracts a range of features (e.g., spectral, texture, and shape) and. SVMs can map the data into a higher-dimensional space through kernel functions. This is highly relevant when trying to capture complex relationships and subtle patterns when classifying objects that are smaller than a pixel. Additionally, RF's average decision boundaries across multiple trees, which can lead to less precise classifications when objects are small, while SVMs are designed to find the optimal boundary by maximizing the margin between classes and identify fine-scale distinctions between object classes (Lillesand et al., 2015; Richards, 2022).

Visual inspection of the results of the unsupervised classification classifiers k means and ISO Cluster showed a poor mapping to features in the imagery, potentially because unsupervised techniques rely largely on spectral data and don’t consider spatial or contextual details (Lillesand et al., 2015;Richards, 2022). When the target of classification is smaller than the pixel, these methods can miss information about small objects, reducing classification accuracy (Richards, 2022). Further research into optimising the classifier choice was not deemed to add sufficient value to the project to warrant the additional time.

A Support Vector Machine (SVM) classifier was then trained using the segments and the training sample polygons; then, the trained SVM algorithm was used to classify the image layers for plots at each beach. Input data to the classifier included values for each segment for red, green and blue bands, NGRDI and HSV. Classification accuracy was assessed for a stratified random selection of validation image segments within plots across the beaches, independent of the training dataset (Radoux & Bogaert, 2017). Stratification was based on cover class, and 50 segments per cover class per beach were selected, or as many as possible if fewer than 50 instances occurred. Reference materials for ground truthing included high-resolution satellite imagery in Google Earth Pro (Google Earth, 2024) and UAV imagery where available, and the original aerial image datasets. Botanical and research records were also checked to see if species occurred at sample beaches. Accuracy measures were calculated using a confusion matrix for each beach, that compared the cover classes assigned to each validation segment by the classification process to those assigned visually using reference sources. Computed accuracy metrics comprised overall accuracy, Cohen’s Kappa, user’s accuracy, and producer’s accuracy (Ye et al., 2018). All confusion matrices were aggregated into one matrix for all beaches and then summarised for key classes.

3.2.3 Analysis

Classified cover class data for plots were then manipulated and analysed. Cover classes that were not of interest, e.g., dry sand *vs*. wet sand, or those that were very similar to each other, e.g. trees *vs*. shrubs, or a stand of one type of tree or shrub, were aggregated with similar classes to ensure consistency across all plots and beaches and maximise the sample size of each cover class at the plot-level. As a result, the 21 cover classes discerned from the training dataset (see results) were reduced to nine in the rationalised classified dataset (Table 6). The percentage cover for each cover class in each plot was then calculated.

A Principal Coordinate Analysis (PCoA) was used to visualise the differences among plots in their per cent cover of the nine cover classes of pairwise Bray-Curtis distances. Since the "Sand" class occurred in every plot, the contribution of different vegetation cover types was overshadowed. Thus, it was removed from the analysis. The "Other" and "Water" classes (Table 6) were also excluded since they were not of interest to this research. The data were first standardised to account for a right skew using a square root transformation. They were relativised to the maximum across the dataset to down-weight the effects of uncommon cover classes on the analysis and to ensure negative values were not introduced to ensure compatibility with the Bray-Curtis distance calculation (McCune & Grace, 2002). Next, the same distance matrix was used in a hierarchical, agglomerative, polythetic cluster analysis using the Average Group Linkage method to organise plots into spectrally similar groups, which were then named as different "plot vegetation types" by the greatest percentage of vegetation type per plot, across all plots in the cluster (Van der Maarel, 2005). The PCoA bi-plot points for plots were then labelled according to the cluster (vegetation type) they were clustered into.

To explore the relative influence of anthropogenic and geomorphic agents in explaining the variation among the different plot vegetation types, a range of spatial environmental data was compiled at the plot level from available spatial data layers (Table 5; Chapter 3, Appendix 3). Measures of sand supply or sediment deposition, beach and dune morphology do not exist for Aotearoa at a national scale. However, geomorphic agents or the so-called "met ocean" or climate variables of wind and waves are also highly influential in active dune processes (da Silva et al., 2008;

Eisma, 1997; Miller et al., 2010; Keijsers et al., 2015; Konlechner & Hilton, 2022; Shepherd & Hesp, 2003; Short and Hesp, 1982) and national datasets are available. The geomorphic agents used in analysis were: the modelled number of days per annum with mean wind gusts above 24 knots for the local area; the number of days per annum with coastal waves over 4 m for at least 12 hours; mean daily wind run for the local area (km) and total rainfall (mm) for the local area (Table 5; Chapter 3, Appendix 3). The anthropogenic variables were: the distance of the plot from the nearest road (m); the type of land cover adjacent to each plot; the ecosystem extent (ha), i.e., the area covered by active dunes that the sampled plots occurred in; the median of personal income bands (NZD) in a region; and usually resident human population in a local area (Table 5; Chapter 3, Appendix 3). Pairwise Pearson's correlations were computed for the environmental data for each plot to quantify the strength of any intercorrelations (Chapter 3, Appendix 4). Results indicated weak to moderate correlations among most variables, with r values between 0.25 and 0.54. However, the mean daily wind run and days of strong wind gusts were highly correlated, with an r-value of 0.98 (Chapter 3, Appendix 4). We kept both variables to explore the potential contrasting impacts of ambient and extreme wind gust conditions on vegetation types.

Table 5. Environmental variables selected for the CART model analysis and the mechanisms of each that could potentially drive vegetation cover type. The large wave event data used was for the coastal region associated with each sample beach (Chapter 3, Appendix 3). This data was generated using the 12 km resolution New Zealand National Institute of Water and Air (NIWA) operational wave forecasting model (NZWAVE-12), which models wave heights using wind from NIWA's NZLAM-12 weather forecast model and swell from NIWA's global wave forecast model (Gorman, 2016). Rain and wind data were collected from climate stations, which were selected based on whether the requisite data was recorded; proximity to study beaches (Chapter 3, Appendix 3); similarity of coast, and where possible, where there was the same degree of exposure to prevailing winds.

Environmental variable	Mechanism
Adjacent land cover type	Bordering land used for human activities, such as agriculture, forestry and settlement, contain weed species and are connected by corridors that facilitate their spread through the landscape (Castillo & Moreno-Casasola, 1996).
Coast	Much of Aotearoa is exposed to south-west and southerly swells and prevailing westerly winds (Shepherd & Hesp, 2003). Given that wind and wave energy are drivers of dune formation (Psuty, 2004), geomorphic processes are expected to be a strong driver of cover type on western and southern coasts, compared to eastern coasts.
Distance from nearest road (m)	Road networks can cause habitat fragmentation, loss and the rapid spread of weed species (Bennett, 2017; Castillo & Moreno-Casasola, 1996; Gao et al., 2020). Increasing distance from road networks reduces negative impacts on ecosystems (Benítez-López, 2010; Bennett, 2017).
Ecosystem extent (ha)	Species - area relationships suggest that smaller areas support fewer species (Begon et al., 2006), thus smaller extents are expected to support fewer species affecting geomorphic processes.
Days (per annum) of large wave events (greater than 4 m and over longer than 12 hours).	Large wave events cause disturbance and the restarting of primary succession sequences, creating the characteristic vegetation of active dunes (Hesp, 2011; Miller et al. 2010; Pegman & Rapson, 2005). The threshold of 4 m was selected since it was the lowest value of the three bands in this dataset, thus ensuring variation across climatic regions was captured.

Table 5. (cont.)

Days of wind gusts ≥ 24 knots (per annum)	Disturbance from high winds during storms (and associated wave energy, rain and overwash) catalyse geomorphic processes of erosion and sand movement (Eisma, 1997; Maun, 2009; Miller et al., 2010). The threshold of ≥ 24 knots was selected since it was the lowest value of the bands in this dataset, thus ensuring variation across climatic regions was captured.
Mean daily wind run (km)	Consistent wind and wave energy associated with ambient conditions build foredunes gradually through accretion, creating regular foredunes and dune slacks (da Silva et al., 2008; Eisma, 1997; Miller et al., 2010).
Median of personal income bands ($ gross NZD)	Environmental degradation of coastal dunes is associated with urbanisation (Lansu et al., 2014; Malavasi et al., 2016; Salgado et al., 2021), in Aotearoa the most populous areas are associated with the highest personal income bands (Statistics New Zealand, 2014).
Total rainfall (mm)	Higher rainfall creates more favourable conditions for vegetation establishment, soil development (Laporte-Fauret)
Usually resident population	Environmental degradation of coastal dunes is associated with residential development and other human influence (Lansu et al., 2014; Malavasi et al., 2016; Salgado et al., 2021).

Next, Classification and Regression Tree (CART) analysis (Breiman et al., 1984) was used to assess the relative influence of the ten different environmental variables on the vegetation types derived through cluster analysis. A 75:25 split between test and training data was used, and a 10-fold cross-validation method was used to assess the model's performance. Cohen's Kappa statistic was used to select the final model; the maximum depth was set to five, and the minimum split for terminal nodes was set to five plots. The global importance of variables was calculated across the whole tree to understand their overall influence.

Data analyses were carried out in R (R Core Team, 2023) and R Studio v.2022.07.2 (RStudio Team, 2020), including data manipulation with base R and the tidyverse (Wickham et al., 2019); PCoA in vegan (Oksanen et al., 2022); Pearson correlation in stats (R Core Team, 2023); CART analysis in rpart (Therneau & Atkinson, 2022) with caret for cross validation (Kuhn, 2008) and visualisation in ggplot2 (Wickham, 2016). Hierarchical Agglomerative Cluster Analysis was carried out in PC-ORD v.7.10 (McCune & Mefford, 2018).

3.3 Results

In total, 21 cover classes were discerned from the training samples based on the initial vegetation cover typology (Fig.11; Table 6; Chapter 3, Appendix 6) and the segmentation layer resulting in the classification of 137.6 ha of active dune area in the 135 plots on 21 beaches. The three target sandbinders could be discerned to species, but woody vegetation could only be identified at a structural level (trees and shrubs), as were exotic pasture grasses. Vegetation in wet areas, such as dune slacks, could not be distinguished to species level. Classification was based on an average of 64 (median = 40) groups of training pixels per cover class across all beaches (Fig. 12). Each training sample group comprised, on average, 40 pixels and ranged from one pixel to 7,200 pixels in size. Groups of more than 100 pixels typically represented homogenous areas of sand or continuous woody vegetation.

Table 6. Description of the 21 cover classes used to train the Support Vector Machine (SVM) classification algorithm. The cover classes were aggregated into nine classes for analysis (right-hand column). Examples of species or objects that could occur in each class are given in Chapter 3, Appendix 6.

Cover class	Description	Aggregate cover class
Ficinia spiralis (A.Rich.)	An endemic and endangered sand binding sedge growing on active dunes. It has a natural distribution the length and breadth of Aotearoa.	*Ficinia spiralis*
Spinifex sericeus R.Br.	A native sand binding grass growing on active dunes with a natural distribution from Cape Reinga to Nelson.	*Spinifex sericeus*
Ammophila arenaria (L.) Link	A vigorous exotic sandbinder from Northern Europe and North Africa, that commonly occurs throughout Aotearoa.	*Ammophila arenaria*
Mixed native and exotic trees	A mix of native and exotic trees which are often not identifiable, or distinguishable in imagery.	Woody vegetation
Mixed native and exotic shrubs	Mix of native and exotic shrubs often not identifiable, or distinguishable, from each other in imagery.	Woody vegetation

Table 6. (cont.)

Cover class	Description	Aggregate cover class
Mixed native and exotic shrubs	Mix of native and exotic shrubs often not identifiable, or distinguishable, from each other in imagery.	Woody vegetation
Exotic trees and shrubs	Exotic trees and shrubs that were identifiable as exotic but often not able to be distinguished individually. Examples are given in the tree and shrub cover classes above.	Woody vegetation
Shadow from trees and shrubs	Shadow from trees and shrubs when clearly falling within an area of woody vegetation.	Woody vegetation
Exotic pasture grasses	A mixture of exotic pasture grasses, typically rank and found adjacent to areas used by people or agriculture.	Exotic pasture grasses
Exotic herbaceous species	A mixture of low-growing herbs and forb species (excluding exotic pasture grasses), that are not discernible individually.	Other herbaceous species
Mixed native and exotic herbaceous species	A mixture of typically taller rushes, sedges, grasses - native or exotic, that were not necessarily discernible individually.	Other herbaceous species
Shadow	Shadows from dunes, vegetation, or impermeable structures or any other feature (except for trees and shrubs, included above).	Other
Buildings	Houses, sheds or other similar structures.	Other
Bare earth	Patches of exposed non-sand sediment.	Other
Other impermeable	Small impermeable features such as cars or signs.	Other
Dry sand	Areas of dry sand without tidal or estuarine water.	Sand
Water over sand	Sand areas with shallow tidal or estuarine water present.	Sand
Wet sand	Sand areas still wet and darker in colour from tidal water.	Sand
Gravel	Weathered, rounded rock fragments, larger than coarse sand and smaller than pebbles.	Sand
Driftwood	Trees and branches washed up on the beach, generally around the high tide mark.	Sand
Rocks	Stones, rocks, boulders and bedrock.	Sand
Water	Estuaries, rivers or ponded water (this class is deeper than ‘Water over sand’)	Water

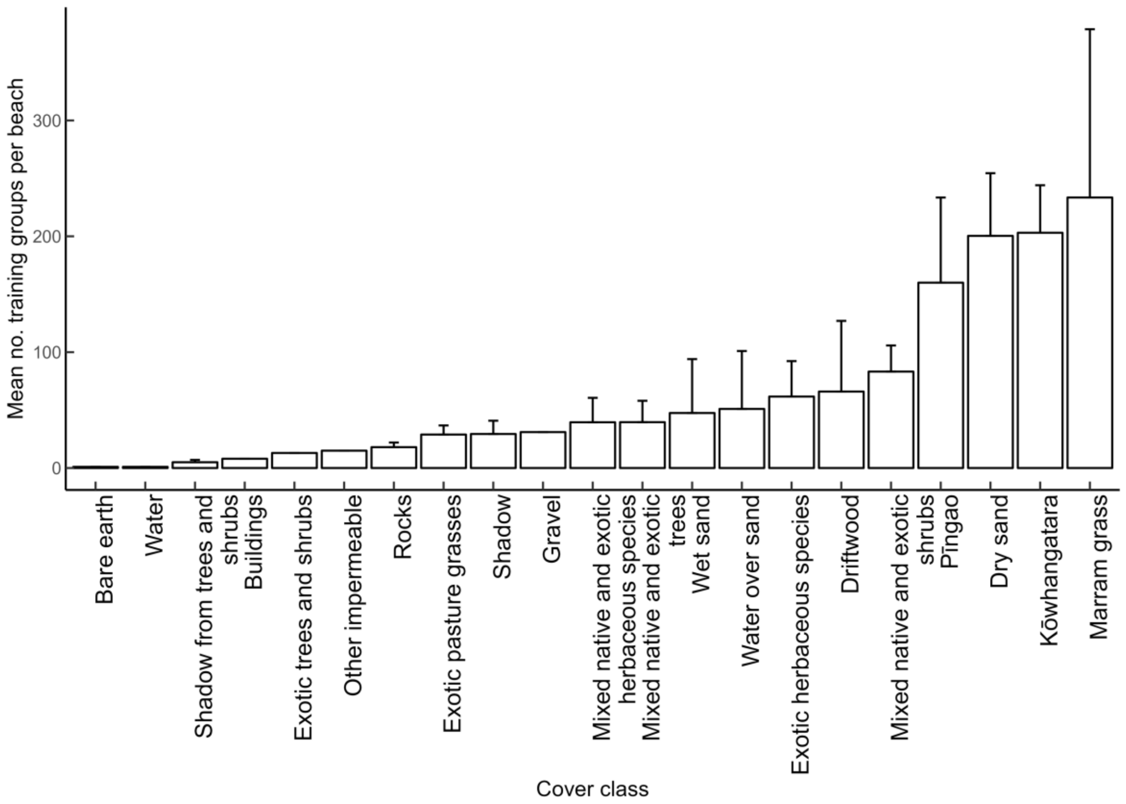


Fig. 12. Mean (± 1 standard error) number of training samples for each cover class manually discerned from the imagery across all beaches. Training samples comprised hand-digitised polygons encompassing groups of pixels of the same cover class, selected from all plots across each beach. The number of training samples per beach was proportional to the number of cover classes present. Where the term "Pīngao" is used it is equivalent to *Ficinia spiralis*. Where the term "Kōwhangatara" is used it is equivalent to *Spinifex sericeus* and "Marram grass" is used it is equivalent to *Ammophila arenaria*.

The mean overall accuracy across all beaches for the nine aggregate cover classes was 0.76. The mean Kappa score, or estimated accuracy of classification across all beaches, was 0.70 (Fig.13). The mean user's accuracy was 0.66, and the mean producer's accuracy was 0.79 (Fig.13). Selected examples of the classified output for aggregated cover classes are given in Chapter 3 Appendix 7. The sand cover class exhibited the highest and most balanced user's and producer's accuracy.

In contrast, the vegetation classes had reasonable user's and producer's accuracy scores of between 65 and 84 (Fig. 13). *F. spiralis* and *S. sericeus* cover classes were most often confused with sand, followed by woody vegetation (Fig. 13). *A. arenaria* was the most misclassified cover class, often confused with woody vegetation, sand or exotic pasture grasses (Fig. 13). Cover classes with fewer training samples (e.g. “water” and “other”) had lower accuracy (Fig. 13; Fig.14). The overall accuracy for the non-aggregated cover classes was 0.70 across all beaches, with a Kappa score of 0.66 (Chapter 3, Appendix 8).

Cover class	Sand	Woody vegetation	Water	Other	Exotic pasture grasses	*Ficinia spiralis*	*Spinifex sericeus*	*Ammophila arenaria*	Other herbaceous species	Total	User's accuracy
Sand	1376	73	0	6	30	24	78	62	25	1674	0.82
Woody vegetation	60	1352	0	26	36	5	20	76	33	1608	0.84
Water	9	27	11	0	3	0	0	0	0	50	0.22
Other	56	120	0	200	1	4	12	8	1	402	0.50
Exotic pasture grasses	17	21	0	1	343	0	2	37	0	421	0.81
Ficinia spiralis	50	27	0	6	0	163	4	0	0	250	0.65
Spinifex sericeus	111	50	0	4	19	2	462	224	16	688	0.67
Ammophila arenaria	10	35	0	0	9	0	9	334	2	399	0.84
Other herbaceous species	80	43	0	0	3	0	7	13	240	386	0.62
Total	1769	1748	11	243	444	198	594	554	317	5878	0.00
Producer's accuracy	0.78	0.77	1.00	0.82	0.77	0.82	0.78	0.60	0.76		
Overall accuracy											0.76
Kappa											0.70

Fig. 13. Aggregated confusion matrix for the nine aggregate cover classes across all beaches. The cover classes assigned to each validation segment through image classification were compared to those assigned visually using reference sources. Diagonal shaded cells represent the number of correctly classified segments for each cover class. The off-diagonal cells indicate the misclassifications between different cover classes. Overall accuracy is the number of correctly classified segments divided by the total segments in the sample. Kappa isa another misclassification measure that compares overall accuracy to a random classification.

Sand was the most common cover class, followed by woody vegetation, *A. arenaria*, *F. spiralis*, *S. sericeus*, exotic pasture grasses, other herbaceous species, other (comprising shadow and impermeable cover classes) and water (Fig.14). The cluster analysis resulted in seven plot vegetation types (Fig. 15; Chapter 3, Appendix 9), with the largest cluster comprising 49 plots and the smallest, five plots; chaining of clusters cluster was low (4.12 %), suggesting a relatively high level of confidence in the generated clusters. In the PCoA of the plot cover class data, most of the variation among plots was associated with the first two axes, whereby the cumulative variance was 60%, comprising 42% and 18 % for the first and second axes, respectively (Fig.16). Plot vegetation types formed relatively distinct groups on the PCoA (Fig.16). Plots classified as the *F. spiralis* plot vegetation type formed an isolated cluster in the PCoA, indicating that *F. spiralis* dominated plots were more similar to each other than plots classified as other plot vegetation types. Plots classified as dominated by woody vegetation were also tightly clustered but were closer to plots classified as several other plot vegetation types. Only plots classified as dominated by either *S. sericeus* or as other herbaceous species, overlapped substantially with other plot vegetation types on the PCoA.

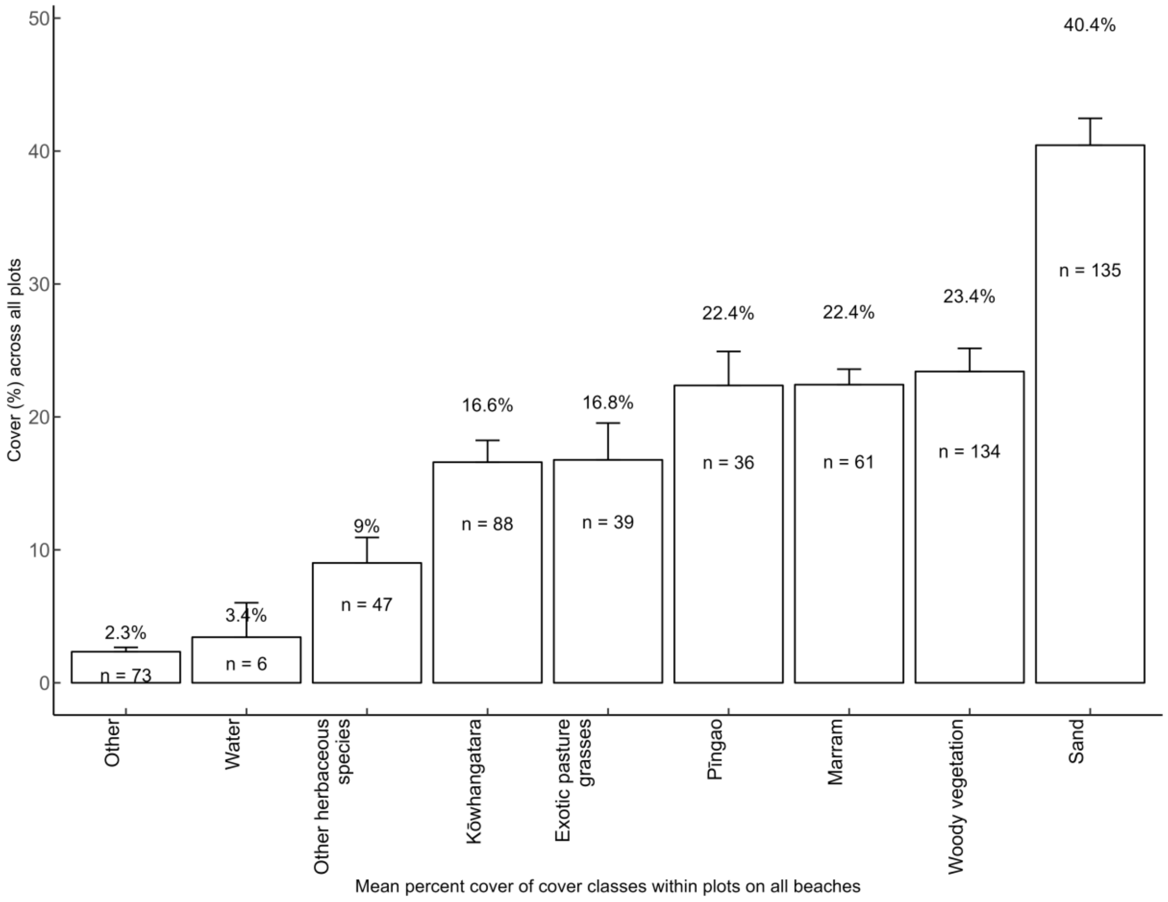
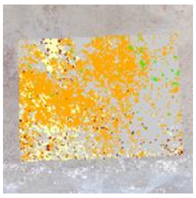
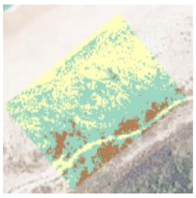
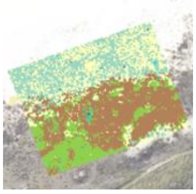
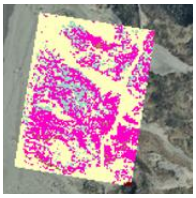
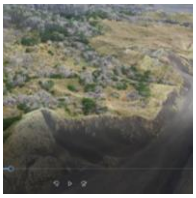
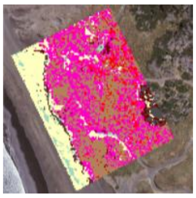
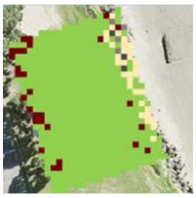
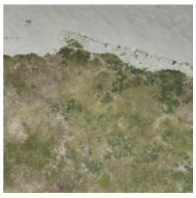
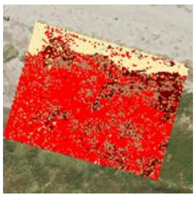


Fig. 14. Mean percent cover (± 1 standard error) within plots of the nine cover classes across all plots (n = 135). The “Other” cover class contained shadows, building and small, impermeable objects, such as signs or vehicles. The cover class termed “Pīngao” is equivalent to *Ficinia spiralis,* “Kōwhangatara” is *Spinifex sericeus* and “Marram” is *Ammophila arenaria*.

Vegetation type and aerial image resolution	Ground level	Aerial image	Classified image
1. *Ficinia spiralis* (0.3 m, scale 1:1000)			
2. *Spinifex sericeus* (0.3 m, scale 1:1000)			
3. *Spinifex sericeus* – woody vegetation (0.3 m, scale 1:1000)			
4. *Ammophila arenaria* (0.4 m, scale 1:1000)			
5. Woody vegetation – *A. arenaria* (0.4 m, scale 1:1000)			
6. Exotic pasture grasses (0.3 m, scale 1:200)			
7. Other herbaceous species (0.1 m, scale 1:1000)			

🟨 Sand
⬜ Gravel
🟧 *F. spiralis*
🟩 *S. sericeus*
🟪 *A. arenaria*
🟫 Shrubs
🟥 Trees
⬛ Shadow
🟦 Water
🟪 *A. arenaria* – woody vegetation
🟦 *S. sericeus* – woody vegetation
🟩 Exotic pasture grasses
🟥 Other herbaceous species

Fig. 15. Examples of the seven dominant plot vegetation types identified using hierarchical agglomerative cluster analysis. Aerial imagery is shown at a scale of 1:1000, apart from the "Exotic pasture grasses" plot vegetation type, which is shown at 1:200, due to this being a smaller plot (540 m^2), which accommodated a foreshortened foredune. Locations of the "Ground level" photos and photo credits: 1. *F. spiralis*: Kaitorete Spit, credit: Hannah Buckley; 2. *S. sericeus*: Spirits Bay / Kapowairua, credit: Thomas Buckley; 3. *S. sericeus* – woody vegetation: Pakiri Beach; 4. *A. arenaria*: Himatangi Beach, credit: Elizabeth Bargh; 5. Woody vegetation – *A. arenaria*: Muriwai Beach, credit: Graham Hinchliffe (UAV footage taken for this research); 6. Exotic pasture grasses: Ōrewa Beach; 7. Other herbaceous species: Ōhope Beach, credit: Sarah Beadel.

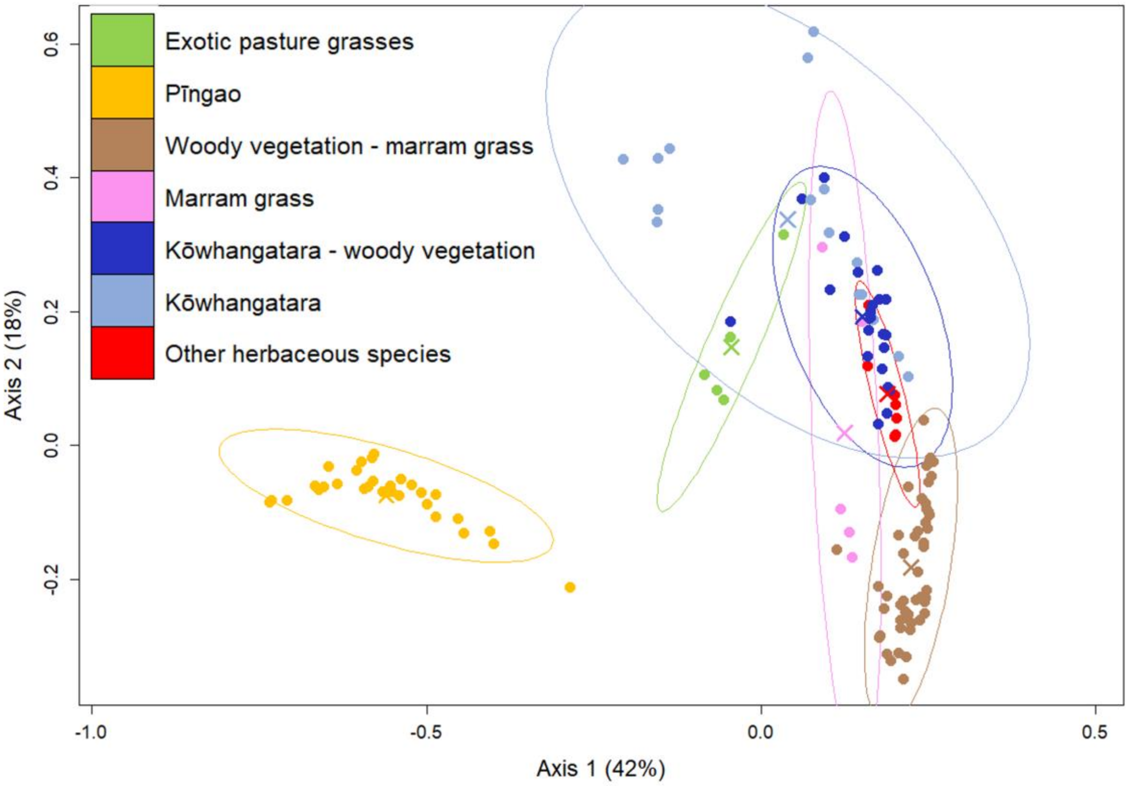


Fig. 16. Principal Coordinates Analysis (PCoA) of the square-root transformed per cent cover values for foredune cover classes within plots, relativised to maximum percent cover values. The plot shows the first two ordination axes, which is the unrotated solution. The plot symbols are coloured by the seven plot vegetation types resulting from the cluster analysis of the cover class dataset for the 135 plots, i.e., the dominant vegetation type. The centroids of each plot vegetation type cluster are marked with an '×' symbol on the diagram. Ellipses show the 95% confidence intervals around each plot vegetation type. Where the term "Pīngao" occurs it is equivalent to *Ficinia spiralis*. Where the term "Kōwhangatara" occurs it is equivalent to *Spinifex sericeus* and the term "Marram grass" is equivalent to *Ammophila arenaria*.

The CART analysis, which explored relationships between the vegetation types and environmental factors, produced a classification tree model with a Cohen's Kappa statistic of 0.75 (Fig.17). The tree was pruned at a depth of five division levels, which produced ten nodes using four of the ten environmental explanatory variables to characterise the branches (Fig.17). The vegetation types *F. spiralis*, and other herbaceous species formed pure nodes at this depth, while *A. arenaria* did not appear in a terminal node (Fig.17). The most important global variable was distance from road (m), population followed by days of high wind gust (Table 7).

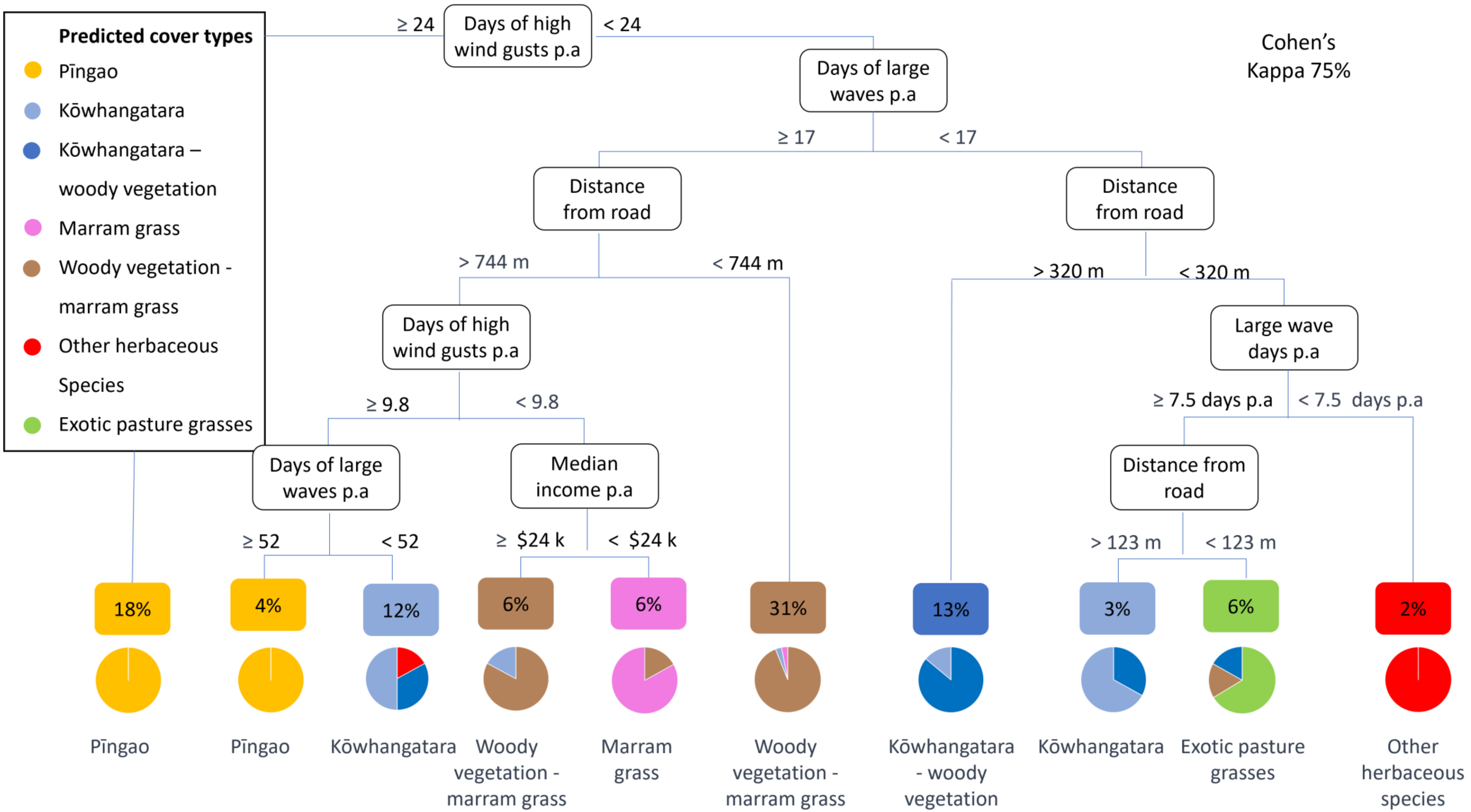

Predicted cover types
Pīngao
Kōwhangatara
Kōwhangatara – woody vegetation
Marram grass
Woody vegetation - marram grass
Other herbaceous Species
Exotic pasture grasses
Days of high wind gusts p.a
≥ 24
< 24
Days of large waves p.a
≥ 17
< 17
Cohen's Kappa 75%
Distance from road
> 744 m
< 744 m
Distance from road
> 320 m
< 320 m
Days of high wind gusts p.a
≥ 9.8
< 9.8
Large wave days p.a
≥ 7.5 days p.a
< 7.5 days p.a
Days of large waves p.a
≥ 52
< 52
Median income p.a
≥ $24 k
< $24 k
Distance from road
> 123 m
< 123 m
18%
4%
12%
6%
6%
31%
13%
3%
6%
2%
Pīngao
Pīngao
Kōwhangatara
Woody vegetation - marram grass
Marram grass
Woody vegetation - marram grass
Kōwhangatara - woody vegetation
Kōwhangatara
Exotic pasture grasses
Other herbaceous species

Fig. 17. CART Classification Tree model relating plot vegetation type to selected, recursively partitioned, human and geomorphic variables for 135 plots on active dunes from 21 beaches around Aotearoa New Zealand. Branches show the thresholds for each variable that characterised groups of plots by each important explanatory variable. Pie graphs show the relative correct and incorrect percentage of plots included in that terminal group of plots. The length of branches is aesthetic rather than indicative of the amount of variation explained by the model. Where the term "Pīngao" occurs, it is equivalent to *Ficinia spiralis*. The term "Kōwhangatara" is equivalent to *Spinifex sericeus*, and the term "Marram grass" is equivalent to *Ammophila arenaria*.

Table 7. Importance values (%) for explanatory variables from the CART analysis relating the classification of plot vegetation type (seven dominant types) to environmental variables.

Environmental variable	Importance (%)
Distance from road (m)	17
Population	16.8
Days of high wind gust per annum	16
Mean daily wind run (knots)	14.1
Median income (NZD)	11
Number of days of large coastal waves per annum	10.4
Total rain (mm)	9.6
Adjacent land cover (low producing grassland)	2.5
Ecosystem extent (ha)	1.8
Adjacent land cover (sand)	0.8
Coast	0

3.4 Discussion

This study shows how the cover classes on coastal active dunes in Aotearoa can be identified from aerial imagery at the scale of 90 × 120 m plots. It shows that these cover classes, at this scale, can be used to characterise distinct vegetation types that geomorphic and anthropogenic variables can predict. The key sandbinder and other cover classes on incipient dunes and active foredunes were accurately discerned; however, differentiation among woody vegetation of semi-stable and stable dunes posed challenges. Our CART model suggests that where systems were heavily used by people and geomorphic agents were less strong, exotic, herbaceous and woody vegetation types dominated. Our study shows that extracting key vegetation features from aerial imagery is feasible, and this methodology will be useful for creating inputs to conservation management in Aotearoa and other countries. Given the ubiquity of the drivers of active dune condition globally (Hesp & Walker, 2021; Psuty, 2004), it is possible that the method could also be successfully used on active dune ecosystems in other countries. Moreover, our method could easily be repeated through time to quantify how these systems change where historical aerial imagery exists.

3.4.1 Discriminating among sand dune vegetation components in aerial imagery

Our accuracy assessment showed good agreement between the image classification and reference sources, with an overall accuracy of 0.76 and a mean Cohen's Kappa score of 0.70 (Richards, 2013), which is a strong result considering the small sizes of target sandbinder plants and imagery resolution (Ye et al., 2018). *F. spiralis* and *S. sericeus* could be discerned because they are often dominant, with few other species present (Johnson, 1993), resulting in fewer pixels mixed with other species in training samples (Hugenholtz et al., 2012). Native sandbinders occur on highly active parts of dunes (Hesp, 2000); thus, the area occupied by them is a critical indicator of how active these dune ecosystems are. Our results suggest that this method can derive these areas from aerial imagery. However, given that *F. spiralis* and *S. sericeus* plants are narrower in width than the minimum segment size and occur in a background of sand, some mixed segments with sand would have occurred and, thus, were often confused with sand. Although this is not ideal, we argue that distinction from other species is the more important level of discrimination, so this does not detract from the applicability of this method to these cover classes. In addition, the differentiation between native sandbinders and the exotic *A. arenaria* was clear and is highly useful for indicating indigenous dominance and ongoing natural ecosystem processes on dunes in Aotearoa (Johnson, 1993).

The collection of training samples for woody plant species was more challenging due to the complexity of woody vegetation relative to the ability of the method to resolve individual species with the imagery selected consistently. This necessitated aggregation of all woody species into two broad groups for image classification: (1) mixed native and exotic trees and (2) mixed native and exotic shrubs. Although the accuracy of these classes was high, false positives and false negatives occurred (Chapter 3, Appendix 8). In contrast, other studies have achieved identification at species level either with very high-resolution multispectral imagery (< 5 cm) (Case et al., 2019; Bilkey, 2022; Laporte-Fauret et al., 2020), or by combining multispectral imagery with Light Detection and Ranging (LiDAR) data (Hantson et al., 2012).

Identification of exotic pasture grasses and other herbaceous plants to the species level in training samples was also challenging, again necessitating broad groupings. Given the sub-pixel size of many herbaceous plants, mixed pixels in training samples and segments were likely to be relatively common, contributing to the lower accuracy and higher standard error of these cover classes. Given that several rare and threatened herbaceous plants occur in dune slacks and hollows (Rapson et al., 2016; Wardle, 1991), further research is warranted to determine methods to more accurately identify ecologically important species in these environments; very high-resolution imagery RGB and multispectral UAV imagery could be considered (Bakacsy et al., 2023; Laporte-Fauret et al., 2020; Wolffe et al., 2023).

Although the SVM classifier is perceived to work well with high dimensional data (Ma et al., 2018), a potential improvement in discrimination among active dune vegetation types could be to train the classifier on the Hue, Saturation and NGRDI bands only to avoid potential correlation between RBG and HSV bands. Removing the Value and RGB bands from the classifier could have additional benefits, such as minimising variation in light and shade (Cheng et al., 2001).

3.4.2. Emergent sand dune vegetation types from image classification data

Our study revealed clear patterns in the composition of vegetation types across the different plots through PCoA ordinations and cluster analysis. Notably, *F. spiralis*-dominant plots were distinctly separate from those dominated by all other vegetation types, potentially due to their locations in remote, southern areas with high conservation protection. Consequently, these areas have likely mainly remained untouched by human activity and invasion from exotic plant species, thus providing refuge for *F. spiralis*, which was once widespread around Aotearoa (Johnson, 1993; Wardle, 1991). In contrast, *S. sericeus*-dominant plots occurred with plots dominated by all other vegetation types except for *F. spiralis*, reflecting its widespread distribution within its geographical range, which lies to the northern tip of the South Island (Wardle, 1991). It may also reflect the prevalence of this species on the seaward side of foredunes, an area typically uninhabited by other species, except for *F. spiralis* (Esler, 1978; Wardle, 1991).

The *A. arenaria* cover class had the third highest mean percent cover within plots, after the sand and woody vegetation cover classes (Fig.14); however, only seven plots were classified as marram-dominant by the cluster analysis (Chapter 3, Appendix 8). Accuracy assessment results showed that some beaches with relatively high *A. arenaria* cover had high misclassification rates for the marram cover class with the woody vegetation and exotic pasture grass cover classes, suggesting spectral similarities. Potentially this means *A. arenaria* had greater dominance in more plots and methods should be developed to improve discrimination of this aggressive species.

The woody vegetation cover class had the highest mean percent cover after sand (Fig.14), and cluster analysis showed it was a co-dominant in plots with either *S. sericeus* or *A. arenaria*, rather than a dominant (Chapter 3, Appendix 8). However, the co-dominant woody vegetation-*A. arenaria* vegetation type was the most common dominant plot vegetation type. These results suggest stable, woody-dominated vegetation had increased on active dunes since the last survey of extent (Hilton, 2000), consistent with international trends (Gao et al., 2020; Jackson et al., 2019). Extensive, historical co-planting of *P. radiata* and *A. arenaria* may explain the high percentage of these co-occurring cover classes within plots (Gadgil & Ede, 1998). However, we could not distinguish between invasive woody species and characteristic native species of semi-stable and stable mid and rear dunes.

3.4.3 Potential drivers of sand dune vegetation type variation across Aotearoa

The results suggest that human activities could have an influence on plot vegetation type in areas of Aotearoa where human influence was strongest, specifically where distances from the nearest road were the lowest, resulting in mainly woody-dominated plot vegetation types on beaches. This pattern may be explained by road access to facilitate planting for forestry or farming, which, historically, would have catalysed dune stabilisation, a trend that also occurs internationally (Gao et al., 2020). Where wind and waves were calmest, and beaches were close to the road, plots were predicted to be dominated by exotic pasture grasses or other exotic herbaceous species vegetation types, which aligns with a common impact of coastal development, the loss of characteristic native species (Cicarelli et al., 2014; Pintó et al., 2023; Salgado et al., 2021). However, given the lack of available data about geomorphic drivers such as sand supply, and because discrepancies in the scale of the datasets exists, the correlations identified through the CART analysis are useful for preliminary analysis but require further testing to quantify and validate the relationships, specifically historical land use and land use intensity, and geomorphic drivers such as sand supply.

3.4.4 Conclusions

This study has shown that the key sandbinder cover classes of active dunes in Aotearoa can be successfully discerned from other cover classes in aerial imagery. Thus, plots can be adequately classified from RGB, high-resolution aerial imagery, and derived NGRDI and HSV layers when these vegetation cover classes dominate. The resolution of the imagery (0.075-0.75 m) meant that native woody species within plots on semi-stable and stable dune areas could not be differentiated as a separate cover class from exotic woody species, which is an important indicator for assessing active dunes' changing stability and condition. Techniques and datasets to identify woody vegetation to the species level require further research; for instance, incorporating higher resolution imagery (< 5 cm), multispectral imagery, and LiDAR data. Additionally, our results suggest relationships exist between human activities and vegetation type where human influence is strongest, but these require validation. From the perspective of conservation management of active dunes in Aotearoa, the classification of dominant vegetation types at the plot scale through image analysis provides a practical approach to habitat mapping and monitoring at different scales of interest, from the beach to national levels. Such quantitative and spatially explicit data are critical for addressing conservation aims regarding ecosystem representation and the dominance of indigenous vegetation. This is especially important for the sandbinder *F. spiralis* since, although it has been identified as dominant in some plots and beaches in this research, nationally it has a status of "At Risk – Declining" (de Lange et al., 2017, p. 35), thus its identification is important for conservation aims which include the conservation of rare species (Bowman & Hacker, 2021). These methods can now be used on longer-term and more frequent survey datasets that will give greater insight into the temporal impacts of environmental variables, particularly coastal development and disturbance from storms.

4 Characterising the Spatial Pattern of Vegetation on Active Dunes

4.1. Introduction

Coastal active dune ecosystems (active dunes) are dynamic habitats with complex environmental gradients and high spatial heterogeneity (Acosta et al., 2009; Carboni et al., 2009; Cicarelli & Barcoro, 2016). They occur where sand is highly mobile, and vegetation comprises sparse, low-growing disturbance tolerant species (Durán & Moore, 2013; Miller et al., 2010; Psuty, 2004). Complex interactions between aeolian and marine sediment transport and ecological processes drive active dune ecosystem structure and function (Biel et al., 2017; Durán & Moore, 2013; Hacker et al. 2012, 2019; Keisjers et al., 2015; Hesp & Walker, 2021; Jay et al., 2022; Martínez et al., 2004; Miller et al., 2010; Ruggiero et al., 2018; Zarnetske et al. 2012, 2015).

Despite providing vital ecosystem services, such as coastal protection of human settlements (Barbier et al., 2011; Hacker et al., 2012; 2019; Itzken et al., 2021; Stockdon, 2007) and a habitat for biodiversity (Martínez et al., 2013; Van der Maarel & Van der Maarel-Versluys, 1997), human activities have disrupted active dune processes (Martinez et al., 2004) and they are some of the most vulnerable and degraded habitats worldwide (Barbier et al., 2011; Gao et al., 2021; Jackson et al., 2019; Martínez et al., 2004; McGuirk et al., 2022; Sperandii et al., 2020). Surprisingly, there are relatively few studies on the condition of active dunes (Chapter 1, 2; Hovenga et al., 2023). Where national studies exist, they are largely concerned with extent and range, rather than condition, or criteria is inconsistency applied across sites and timeframes (Maes et al., 2021; Sperandii et al., 2020). Similar to active dunes globally, the active dunes of Aotearoa New Zealand are at risk from human activities such as forestry, farming, fire, invasive species and coastal development (Holdaway et al., 2013), and the few studies of the extent or condition are local, qualitative, or out of date (Chapter 2). Due to similarities with the international situation, a study of the condition of the active dunes of Aotearoa can provide a case study for active dunes globally.

Ecological processes underpin ecosystem conditions (Chapin III, 2011) and patterns of spatial heterogeneity (spatial pattern) are both an outcome and driver of many ecologically important processes (Turner, 1989; Uuemaa, 2009; Watt, 1947). Spatial pattern is a measure of ecosystem structure, for example landscape configuration, or the mosaic of different habitat patches and their connectivity (Mücher et al., 2023). Structure is fundamentally linked to ecosystem processes and functions through feedback loops (Forman, 1999). Thus, indicators of process can also provide information about structure and function; for example, bare soil can provide information about processes of erosion and sedimentation (Mücher et al., 2023).

Within active dune systems, key overlapping and reciprocal ecosystem processes include natural disturbance, colonisation of disturbed substrates by plant species, facilitation of habitat provision by plant species (Biel et al., 2017; Durán & Moore, 2013; Hacker et al., 2012, 2019; Hesp & Walker, 2021; Jay et al., 2022; Keisjers et al., 2015; Martínez et al., 2004; Miller et al., 2010; Ruggiero et al., 2018; Zarnetske et al., 2012, 2015), and invasion by exotic plant species disrupts these ecosystem processes (Hacker et al., 2012, 2019; Charbonneau et al., 2017, 2023; Charbonneau & Wootton, 2020).

Disturbances are events that remove plant biomass, either wholly or partially killing plants (Grime, 2001) and are key drivers of spatial and temporal heterogeneity in ecosystems (García-Mora et al., 1999; García-Novo et al., 2004; Kumar et al., 2006). In active dune ecosystems, disturbance is caused by the interaction of geomorphic drivers of climate and sand supply and the geomorphic agents of wind, waves and currents causing marine and aeolian sediment transport (Davidson-Arnott, 2010; Hugenholtz et al., 2012; Maun, 2009; Psuty, 2004). Disturbances primarily comprise local winter and severe storms with high wind and wave energy, resulting in sand burial, sand movement, ablation from high winds, saltwater inundation and destruction from wave energy (Laporte Fauret et al., 2021; Maun et al., 2009; Miller et al., 2010). These disturbances cause repeated extinction of, or damage to, plant communities and the creation of patchy mosaics of surviving, recovering or colonising communities (Corenblit et al., 2015; Kumar et al., 2006; Garcia-Mora et al., 1999; Miller et al., 2010). Disturbance can also be the repetitive actions of the winds and tides in the absence of storms, primarily in the summer months, which leads to accretion and dune building (Da Silva et al., 2018; Miller et al., 2010), which create patchiness and spatial heterogeneity (Miller et al., 2010).

4.1.1 Colonisation

Sandbinders are a dune-building, functional group of plants (Chapter 1) that colonise bare sand and survive on the most active parts of dunes (Hesp, 1989; Hesp, 2021; Maun, 2009). Tolerance to burial by sand movement due to lateral growth habits, rapid growth and clonal reproduction from rhizomes or stolons allow sandbinder populations to expand and dominate in the disturbance-prone active dune environment (Hacker et al., 2012, 2019; Maun, 2009; McGuirk, 2022; Olson & van der Maarel, 1997; Reijers et al., 2019).

Recent literature suggests that plant species that expand clonally in disturbance-prone environments have a different spatial pattern of expansion that affects dune morphology (Reijers et al., 2019; Silliman et al., 2015). Sandbinders have been shown to take up a patchy shoot organisation and growth pattern during colonisation to increase their capacity to engineer dune systems in newly disturbed substrates, to secure their survival better and balance the need to increase their populations (Reijers et al., 2019). This pattern has been found to occur in the congeneric species *Ammophila arenaria* (L.) Link and *Ammophila brevigulata* (Fernald) (Reijers et al., 2019), although they have different strategies and patterns. *A. arenaria* maximises sand-trapping efficiency by accreting sediment within multiple dense shoot patches in a “Lévy” random walk strategy (Reijers et al., 2019), whereby the plant grows in small increments in one direction and then large steps in another (Reijers et al., 2019; Routledge, 1990). This strategy builds tall, narrow dunes and potentially explains why *A. arenaria* is successful in environments with limited sand supply (Reijers et al., 2019). On the other hand, the more dispersed shoots of *A. breviligulata* maximise total sand capture over a larger area, building-wide, shorter dunes, potentially preventing excessive sand burial in any one place (Reijers et al., 2019).

Patterns of sandbinder plant density could also be a useful indicator of ecosystem structure. A positive relationship has been observed between the density of sandbinder plants irrespective of species and wider yet taller dune morphologies (Biel et al., 2019; Jay et al., 2022), and a change in annual plant densities has also been found to explain a greater proportion of variance in dune morphologies (Biel et al., 2019; Jay et al., 2022). Additionally, Hacker et al. (2012) observed that the relatively low-density *A. brevigulata* was better at colonising beaches with a greater sand supply than the relatively high-density *A. arenaria.*

Other functional groups of plants lack traits to colonise and survive on active dunes (Hacker et al., 2012; 2019; Maun, 2009); for example, many herbaceous and woody plants require a soil structure for nutrients, and woody plants require a soil structure that can support growth forms with vertical root structures (MacGuirk et al., 2022), unlike the shifting sands of active dunes. Thus, diversity has been observed to be low or absent in these environments (Castillo & Moreno-Casasola, 1996; Ciccarelli & Bacaro, 2016; Da Silva et al., 2008; Forey et al., 2008; García-Mora et al., 1999; García-Novo et al., 1997; Hacker et al. 2012, 2019; Martínez et al., 2001; Miller et al., 2010; Moreno-Casasola, 1986; Stallins et al., 2005).

4.1.2 Facilitation

Sandbinders act as ecosystem engineers by creating geomorphic diversity through colonisation of bare substrates after disturbance and the subsequent stabilisation of local areas of the beach. This, in turn, creates favourable microclimates for other species to establish such as shelter and shade (Charbonneau et al., 2022; Holdaway et al., 2013).

4.1.3 Zonation

The interaction and feedbacks between processes of disturbance, colonisation and facilitation create a predictable spatial patterns in vegetation along a disturbance gradient from the shoreline moving inland (García-Novo et al., 2004; Miller et al., 2010) called zonation (Doing, 1985; García-Novo et al., 2004; Maun, 2009; Miller, 2010; Moreno-Casasola, 1986; Psuty, 2004; Van de Maarel, 1997). For example, Miller et al. (2010) established that a relationship occurs between plant communities, disturbance and recovery in different dune habitat types in Florida over nine years, demonstrating that processes of vegetation succession were interrupted by disturbance from storm events, creating mosaics of different communities along coastal environmental gradients.

4.1.4 Invasion

Invasive or planted species on active dunes can disrupt natural processes, increasing competition and diversity (Biel et al., 2017; Hacker et al., 2012; Lázaro-Lobo, 2023; Martínez & García-Franco, 2004; Zarnetske et al., 2012). For example, in Aotearoa the dense cover of *A. arenaria* accumulates more sand than native sandbinders and, subsequently, more organic matter, creating more nutrients and stabilising the substrate (Gadgil, 2001), paving the way for woody species (Gadgil, 2001). In a study comparing the traits of a range of native and exotic sandbinders, Verhoeven et al. (2013) found that *A. arenaria* and the sandbinding liana *Carpobrotus edulis* (L.) N.E. Brown outcompeted the native sandbinders *Ficinia spiralis* (A.Rich.) Muasya et de Lange*, Spinifex sericeus* R.Br. and *Poa billardierei* (Spreng.) St.-Yves due to a range of more aggressive traits (Murphy et al., 2012; Verhoeven et al., 2013).

4.1.5 Landscape metrics

Processes of disturbance, colonisation and invasion occur on active dunes and give rise to patterns of spatial heterogeneity in vegetation (Biel et al., 2019; Corenblit et al., 2015; Kumar et al., 2006; Garcia-Mora et al., 1999; Hacker et al., 2012; Jay et al., 2022; Miller et al., 2010; Reijers et al., 2019). The relationship between spatial pattern and ecosystem processes can be explored through analysis of remotely sensed imagery with landscape metrics (Forman, 1999; Kumar et al., 2006; Mucher et al., 2023; Petorelli et al., 2016; Turner et al., 2015; Uemaa et al., 2009). Landscape metrics are algorithms that quantify the composition (type and amount) and configuration (spatial arrangement) of landscape components and the spatial relationships among them (Kumar et al., 2006; McGarigal & Marks, 1995; Turner et al., 1989; Turner et al., 2015; Ueema et al., 2009), in turn, revealing information about components of ecosystem structure and function and therefore processes and condition (Chapin III, 2011; Chapter 1, 2; Forman, 1999; Holling, 1973; Karr, 1993; Mücher et al., 2023; Turner, 1989; Uuemaa, 2009; Watt, 1947).

Alternative methods to landscape metrics detect spatial pattern were considered. For example, surface metrics (McGarigal et al., 2009) measure continuous variables such as elevation (McGarigal et al., 2009) were investigated, but ultimately not selected, since the metrics originate from mechanical engineering and do not relate to ecosystem structure and function (McGarigal et al., 2009). Spatial point pattern analysis requires the conversion of objects, in this case, patches of vegetation (Chapter 3), to points (Bivand et al., 2013), which can lead to a loss of information about patch size, shape and connectivity (Baddeley et al., 2016). Spatial autocorrelation techniques such as Moran's I and Geary's C, retain the spatial context and relationships between objects but requisite assumptions of stationarity in the data (Dale & Fortin, 2014) were violated in this dataset. In contrast, landscape metrics capture detailed information about patch characteristics with categorical spatial data, providing a more nuanced understanding of spatial structure without many of the assumptions required for spatial autocorrelation (Turner et al., 2015).

To our knowledge, few studies have used landscape metrics to assess the condition of active dunes (Carranza et al., 2010; Konlechner et al., 2015; Malavasi et al. 2013, 2014, 2016; Marzialetti et al., 2024; Pinto et al., 2023; Ryu & Sherman, 2014). The relationship between climatic drivers and dune mobility in the USA has been demonstrated by quantifying the pattern of bare sand compared to vegetated patches and climate variables with landscape metrics (Ryu & Sherman, 2014). Disturbance from high wind and low moisture was shown to increase the mobility of active dunes (Ryu & Sherman, 2014) in line with the literature on dune mobility indices (Lancaster, 1988). The pattern of bare sand compared to vegetated patches has also been shown to demonstrate progress towards restoring active dunes at a beach on Rakiura, Aotearoa (Konlechner et al., 2015) and the authors observed that the percentage of sand cover was greater in *F. spiralis* dominated vegetation compared to *A. arenaria* dominated vegetation, and that *A. arenaria* patches were more aggregated (Konlechner et al., 2015).

Although studies quantifying colonisation, facilitation and zonation processes in active dunes using landscape metrics were not found in this literature review, relationships have been demonstrated between species richness and the Simpson's Diversity metric when applied to cover classes in grasslands, forest, and tundra landscapes at varying spatial scales (Kumar et al., 2006). However, field-based measures show that species richness is higher in stable dunes than in accreting or eroding dunes (Cicarelli & Barcoro, 2016).

Relationships have been shown to exist between spatial pattern and populations of invasive *Carpobrotus* spp. in coastal dunes in Italy with landscape metrics, whereby patches of mature invasive *Carpobrotus* spp. had elongated, irregular shapes which were not significantly different from natural herbaceous vegetation shapes, although they were larger and occurred in proximity to urban areas (Carranza et al., 2010; Marzialetti et al., 2024). To our knowledge, there have been no studies of the effects of encroachment from woody invasive species on the spatial pattern of active dunes.

Additionally, a small number of landscape metrics studies have investigated the effect of human activities on active dune vegetation influences (Malavasi et al. 2013, 2014, 2016; Pinto et al., 2023). Coastal development has been shown to alter incipient and foredune vegetation patch sizes and shapes to become more rectangular, uniform, fragmented, and larger with exposure to human influences (Malavasi et al. 2013, 2014, 2016; Pinto et al., 2023), with a corresponding decrease in plant species richness (Malavasi et al., 2016).

4.1.6 Aim and hypotheses

Given the decline in active dune ecosystems in Aotearoa and worldwide (Gao et al., 2021; Hilton et al., 2000), finding methods to assess ecosystem condition and effectively monitor change is critical. To our knowledge, globally, there are no studies that investigate the relationship between active dune spatial pattern, vegetation types and vegetation processes using landscape metrics. Such studies would provide information about ecosystem structure, function and condition at the landscape and national scale. Based on our sample of beaches across a gradient of condition (Chapter 2) and the active dune literature, we knew that a range of ecogeomorphic processes would occur across the gradient, therefore we sought to identify spatial patterns across the range of condition in relation to their vegetation types (Chapter 3). In contrast, previous studies have investigated binary vegetated and unvegetated classes only (Carranza et al., 2010; Konlechner et al., 2015; Malavasi et al. 2013, 2014, 2016; Marzialetti et al., 2024; Pinto et al., 2023; Ryu & Sherman, 2014). Therefore, the originality of this study and its aim, is to explore the utility of landscape metrics at the scale of foredunes to characterise the spatial patterns of a nationally representative sample of foredune vegetation cover types, and key environmental variables that could drive those patterns. This information could provide valuable information about the current state of the active dunes of Aotearoa to inform the management and monitoring of these important coastal habitats. We hypothesised that:

(1) the spatial pattern of foredune vegetation would vary across the seven nationally representative active dune vegetation types identified in Chapter 3;
(2) vegetation types dominated by native sandbinders would exhibit greater patchiness and edginess, suggesting interactions between processes of natural disturbance and colonisation;
(3) processes of disturbance could be further indicated from the percentages of bare sand in vegetation dominated by native sandbinders, compared to vegetation types dominated by other growth forms, thus also indicating the mobility of active dunes;
(4) vegetation types dominated by sandbinders would have greater dominance and lower evenness of spectral cover classes in the classified imagery compared to vegetation types dominated by other growth forms, suggesting they have adaptations to colonise loose substrates and greater tolerance to stresses of the harsh foredune environment compared to other species;
(5) a pattern of aggregation and high per cent cover of invasive cover classes would exist where invasive species co-occur with native sandbinders, which could suggest that invasion processes are occurring;
(6) relationships exist between human activities and the spatial pattern of vegetation cover classes in dune systems heavily used by people, suggesting that human activities affect eco-geomorphic processes.

4.2. Methods

Due to redundancy in many landscape metrics (McGarigal et al., 2002), we selected a range of landscape metrics (Table 8) to test our hypotheses. Principal Components Analysis was used to determine which metrics provided the most information about spatial patterns concerning our hypotheses. Landscape metrics were computed on a raster dataset of vegetation classified from high resolution (0.075– 0.75 m), publicly available red-green-blue (RGB) aerial ortho-imagery containing a stratified random national sample of 135 plots (90 × 120 m) over 21 beaches where active dunes occur in a range of conditions (see chapters 2 and 3 of this thesis). Seven plots were smaller sizes (20 × 27 m) due to foreshortening of foredunes by coastal development. The landscape metrics were computed on the cover classes defined in Chapter 3 in two raster layers:(1) a sand–vegetation binary raster layer (sand plus other cover classes = 0, vegetation = 1) derived from a raster layer of all cover classes, and (2) a raster layer of all cover classes. The cover classes were: (1) *F. spiralis*, *S. sericeus*, *A. arenaria*, woody vegetation, other herbaceous species and exotic pasture grasses (Chapter 3).

The spatial pattern of vegetation was quantified using fourteen different landscape metrics for the sand-vegetation binary raster: an aggregation index; clumpiness; contagion; euclidean nearest neighbour; edge density; fractal dimension; largest patch index; mean core patch area; mean patch area; mean radius of gyration; mean shape; patch area coefficient of variation; patch density; percentage of landscape (see Table 8 for definitions). Five additional metrics were computed for the cover class raster layer: cohesion; the interspersion and juxtaposition index; the percentage of like adjacencies; Simpson's diversity index and Simpson's evenness index (Table 8). The latter two metrics assess the diversity of cover classes in the classified imagery. These metrics were selected because we considered they would reveal patterns of vegetation composition and configuration that related to disturbance, colonisation, facilitation, zonation and invasion processes.

Table 8. Descriptions of selected landscape metrics. Metrics were chosen because they pertained to the hypothesised composition and configuration of vegetation patterns under processes of disturbance, colonisation, facilitation, zonation and invasion. McGarigal et al. (2002) give additional information, including equations.

Metric	**Class of metric**	**Brief description**
Aggregation index (AI)	Aggregation	A measure of how clustered or dispersed patches of the same cover type are in a landscape, with a higher value indicating the patches are more tightly clustered. Aggregation is based on cell adjacencies identified through a matrix of calculated cell adjacencies.
Clumpiness (CLUMPY)	Aggregation	How much the grouping of similar patches in a landscape deviates from a random distribution. Clumpy is calculated based on the cell adjacencies of the same cover class and compares the actual number of adjacent cells to what would be expected under a random distribution of cells. A higher value indicates a more clumped, compared to a random distribution.
Patch cohesion index (Cohesion)	Aggregation	The degree of connectedness of patches within a particular class in a landscape. Cohesion compares how much edge length there is in a cover class relative to the total area of the patches in the same cover classThe result is scaled to a percentage and higher values indicate greater cohesion. Below a certain density threshold (the percolation threshold), Cohesion increases as patches become more aggregated. Above the threshold, the metric becomes less responsive, suggesting a landscape where the patches are already highly connected.
Contagion (CONTAG)	Aggregation	How mixed or intermingled patches of different cover types are within a landscape, indicating the likelihood that any two adjacent cells will be of the same type. Contagion is based on cell adjacencies identified through a matrix of calculated cell adjacencies. High contagion values suggest the landscape comprises large, contiguous patches of the same cover type, while low values point to a landscape with many small, dispersed patches.

Table 8. (cont.)

Metric	Class of metric	Brief description
Interspersion and juxtaposition index (IJI)	Aggregation	The degree of intermixing of different patch classes, based on patch adjacencies (rather than cell adjacencies in, for example, Contagion or Aggregation). The interspersion and juxtaposition index requires more than three cover classes, so it is not suitable for rasters with binary cover classes. Low numbers represent low interspersion and juxtaposition, and higher values represent high interspersion and juxtaposition.
Mean euclidean nearest neighbour (ENN_MN)	Aggregation	The average distance from each patch to its nearest neighbouring patch of the same cover type in a landscape, effectively quantifying how scattered, or dispersed, the patches are within the landscape.
Patch density (PD)	Aggregation	The number of patches in a landscape per unit area, effectively a measure of how dispersed patches are within the landscape. A higher value indicates more dispersed (more “patchy”), but possibly smaller patches.
Percentage of like adjacencies (PLADJ)	Aggregation	The proportion of cell adjacencies of the same class, reflecting the level of aggregation of that class. The percentages of like adjacencies is based on cell adjacencies identified through a matrix of calculated cell adjacencies and the counting the frequency of adjacencies. Low values represent greater disaggregation (i.e. every cell is a different patch) and high values represent greater aggregation.
Largest Patch Index (LPI)	Area and edge	The dominance of the largest patch of a specific cover type within a landscape, expressed as a percentage of the total landscape area. Higher values indicate greater dominance by the largest patch.
Mean core patch area (CORE_MN)	Area and edge	The portion of a patch's area that is not adjacent to any different patch, effectively capturing the size and compactness of a patch's 'core' or central region. High values indicate that patches have large, compact cores undisturbed by edge effects.
Mean patch area (AREA_MN)	Area and edge	The mean area of patches for a cover type. Larger values indicate that patches tend to be larger on average than other patches of the same cover type.

Table 8. (cont.)

Metric	**Class of metric**	**Brief description**
Mean radius of gyration (GYRATE)	Area and edge	The spatial extent of a patch or how far the patch extends from its centroid. A higher value suggests that the patch is either larger in size or irregularly shaped / widely spread out.
Patch area coefficient of variation (AREA_CV)	Area and edge	The relative variation in the sizes of patches within a cover type. A higher value suggests greater variation in patch sizes, implying a more heterogeneous landscape.
Percentage of landscape (PLAND)	Area and edge	The proportion of the landscape occupied by a specific cover type, signifying its dominance or scarcity in the landscape. A higher value indicates a larger proportion of the landscape occupied by that cover type.
Edge density (ED)	Shape	The total length of the edges of a patch in relation to the total area of the landscape, serving as an indicator of how fragmented or dispersed the patches are. A higher value signifies more fragmented or dispersed patches, while a lower value suggests that the patches are more aggregated or clumped together, having less edge per unit area.
Fractal dimension (FRAC_MN)	Shape	The average complexity of patch shapes in a cover type, taking into account area and perimeter ratio of each patch. Higher values indicating more irregular and complex shapes.
Mean shape (SHAPE_MN)	Shape	The average ratio between the actual perimeter of each patch and its hypothetical minimum perimeter if the patch was perfectly compact, providing a measure of patch shape complexity in a cover type. Higher values indicate more complex shapes.

Table 8. (cont.)

Metric	**Class of metric**	**Brief description**
Simpson's diversity index (SIDI / dominance and evenness)	Diversity	A measure of spectral dominance that calculates the proportion of each class, then subtracts the sum of the squared proportions of all classes from 1. This index gives the likelihood that two randomly chosen cells from the landscape will be of the same class. SIDI is not overly influenced by the presence of rare classes. Lower values indicate a greater dominance.
Simpson's evenness index (SIEI / evenness)	Diversity	A measure of how spectrally evenly classes are distributed across a landscape. SIEI is the ratio between the actual SIDI and the maximum possible value of SIDI. This ratio indicates how close the landscape is to having a perfectly even distribution of classes. Higher values of SIEI suggest a more even distribution of classes.

We were also interested in understanding the spatial pattern differences between the seven nationally representative vegetation types (vegetation types) identified in Chapter 3. These vegetation types were derived from the plot-level cover classes through Hierarchical Agglomerative Clustering and Principal Coordinate Analysis in Chapter 3, and each of the 135 plots was classified as one of the seven vegetation types. The plot-level vegetation types were named after the dominant cover class within each of the seven vegetation types: 1) *F. spiralis*, 2) *S. sericeus*, 3) *A. arenaria*, 4) *S. sericeus* – woody vegetation, 5) woody vegetation – *A. arenaria*, 6) other herbaceous species, 7) exotic pasture grasses (Chapter 3).

Thus, three sets of vegetation data are used in this chapter, which will be consistently referred to as: 1) The sand-vegetation binary raster, comprising a sand class and a vegetation class; 2) The cover class dataset, comprising the six vegetation cover classes within plots, created in chapter 3, and 3) the seven nationally representative, dominant vegetation types derived from the cover classes in

Chapter 3. The landscape metrics were computed on 1 and 2, and the resulting landscape metrics datasets were selectively overlaid with 3 to infer patterns.

To provide the national context for our research, each beach was classified as one of the nationally-representative vegetation types by aggregating the percentages of each vegetation type at each beach and then classifying the beach as being dominated by the most abundant vegetation type. We then used box plots to visualise the variation in percentages of all vegetation compared to the percentage cover of sand on these classified beaches using the sand-vegetation binary dataset. To understand differences in diversity across the different beaches, we analysed the cover class dataset using the spectral diversity measures calculated using the per cent cover of cover classes within plots. Specifically, we used the landscape metrics dominance (the landscape metric SIDI) and evenness (the landscape metric SIEI), whereby a lower score means greater dominance or less-even distribution of cover classes within plots, respectively. We then used boxplots classified by dominant vegetation type, aggregated across all beaches, to show the plot-level percentages of cover classes, dominance, and evenness.

Principal component analysis (PCA) on the landscape metrics computed on the sand-vegetation binary raster for each of the 135 plots was used to visualise variation in the plot-level spatial pattern of vegetation. Data were centred and standardised as Z-scores prior to carrying out the PCA to ensure all variables were on the same scale and to minimise the influence of outliers on the PCA results. PCA bi-plots were plotted for components that explained at least 10% of the variation in spatial pattern. The 135 plot site scores on the biplots were overlayed with their dominant vegetation type to identify which metrics were associated with different vegetation types.

PCA was also used on a subset of the cover class raster for plots where *F. spiralis* occurred and a subset of the data for plots where *S. sericeus* occurred to explore the impact of encroachment of invasive *A. arenaria* and woody vegetation on the spatial pattern. These results were visualised with bi-plots, and the presence of *A. arenaria* or woody vegetation symbolised for components that explained at least 10% of the variation in spatial pattern.

To explore the relative influence of anthropogenic and geomorphic agents in explaining the different spatial patterns, a range of spatial environmental data was compiled in ArcGIS Pro (v.3.03) at the plot, beach, local area, region or coastal regional levels, depending on data availability (Chapter 4, Appendix 1). The geomorphic agents were: the number of days per annum with mean wind gusts above 24 knots for the local area; the number of days per annum with coastal waves over 4 m over 12 hours for the coastal region; mean daily wind run for the local area (km) and total rainfall (mm) for the local area (Chapter 4, Appendix 1). The anthropogenic variables were: the distance of the plot from the nearest road (m); the type of land cover adjacent to each plot; the ecosystem extent (ha), i.e. the area covered by active dunes that the sampled plots occurred in; median personal income (NZD) in the region; and usually resident human population in the local area (Chapter 4, Appendix 1). Regression analysis, using conditional inference trees (Hothorn et al., 2006), was then carried out on the variables as predictors against the principal components that explained more than 10% of the variation in each PCA model to understand the relative influence of different environmental variables on the spatial pattern of vegetation. A conditional inference tree approach was selected to elucidate the differences between beaches yet minimise the effects of pseudoreplication given the one-to-many relationships of beaches to plots (Hothorn et al., 2015). A 95% confidence interval was set, and a ten k-fold cross-validation method was used to assess the model's performance. Cohen's Kappa statistic, a misclassification measure, was used to select final models (Richards, 2013). The importance of variables across the whole model (global variables) and trees for each principal component were then extracted.

All analyses were repeated on two subsets of the 135 plots: (1) for plots where *F. spiralis* was present (n = 36) and (2) for plots where *S. sericeus* was present (n = 88). Data analyses were carried out in R (R Core Team, 2023, version 4.2.3) and RStudio v.2022.07.2 + 576 (RStudio Team, 2020), including data manipulation using base R and the tidyverse (Wickham et al., 2019); landscape metrics were computed using the landscapemetrics package (Hesselbarth et al., 2019); PCA was conducted using the prcomp package (R Core Team (2023); classification tree analysis was conducted using the ctree and party (Hothorn et al., 2006) packages, and the caret package (version 6.0-94 Kuhn, 2008) was used for cross-validation. Visualisation was carried out using the ggplot2 package (Wickham, 2016).

4.3. Results

4.3.1. National context

The nationally-aggregated analysis of the dominant vegetation types at beaches showed that the percentage of sand to vegetation was greater for beaches dominated by the two native sandbinder vegetation types. Where it was dominant, *S. sericeus* had the highest percent sand cover (Whatipu Beach had a sand cover of 59%), followed by *F. spiralis*-dominated beaches (56%; SD = 12). All beaches dominated by other vegetation types had a mean sand percent cover of 18 – 42% (Fig 18; Table 9).

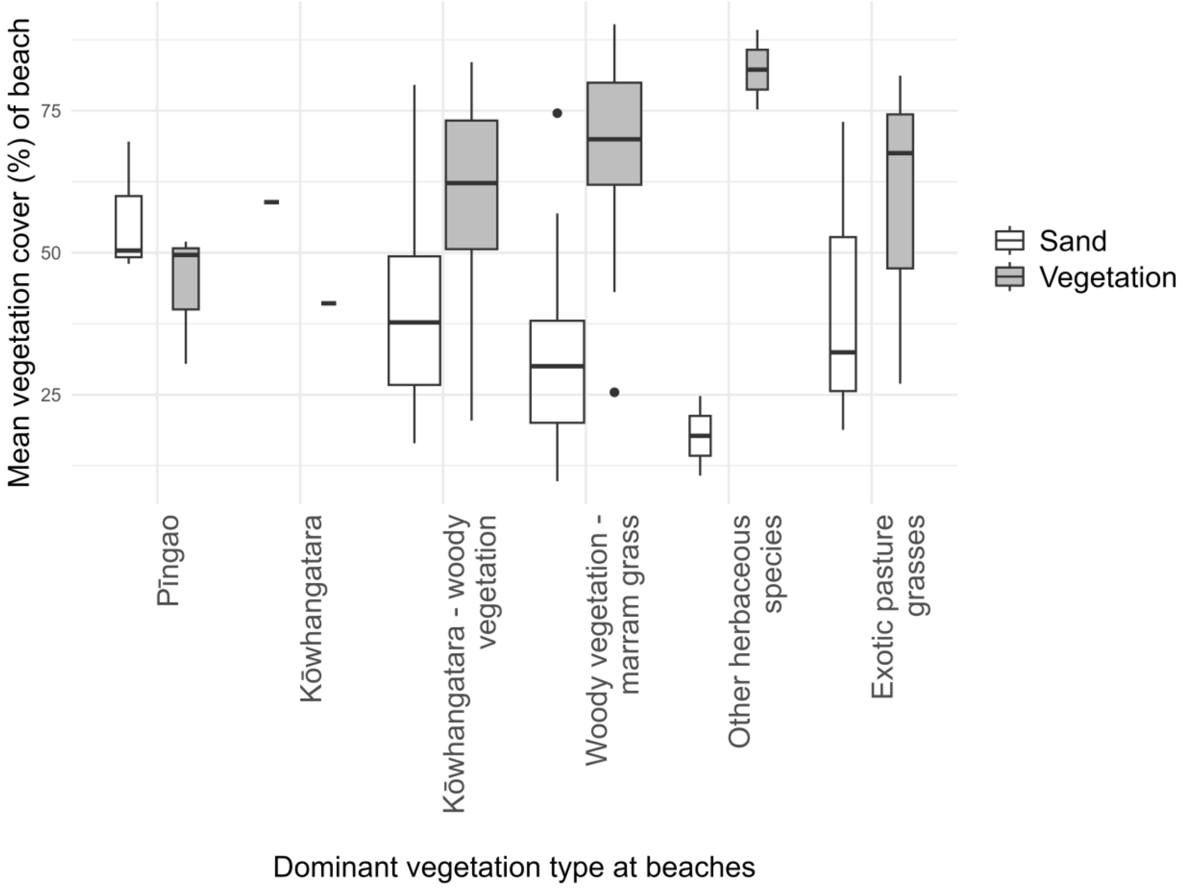


Fig. 18. Mean percent cover of sand and vegetation within plots at beaches (*n* = 21), which were classified as being dominated by one of seven vegetation types. These vegetation types were derived

from classified aerial imagery at the plot scale using Hierarchical Agglomerative Clustering and Principal Coordinate Analysis (Chapter 3 of this thesis). To classify each beach as one of the seven vegetation types, the total percent cover of each vegetation type was summed and the beach was classified as being dominated by the most abundant vegetation type. Boxplot width is proportional to the number of beaches where each of the vegetation types were dominant: *F. spiralis* (*n* = 3); *S. sericeus* (*n* = 1); *S. sericeus* – woody vegetation (*n* = 6); woody vegetation – *A. arenaria* (*n* = 8); other herbaceous species (*n* = 1) exotic pasture grasses (*n* = 2). Note that *A. arenaria* is not shown because it did not dominate at any beaches. Where the term "Pīngao" occurs it is equivalent to *Ficinia spiralis*. Where the term "Kōwhangatara" occurs it is equivalent to *Spinifex sericeus* and the term "Marram grass" is equivalent to *Ammophila arenaria*.

Table 9. Mean sand and vegetation percent cover within plots on beaches dominated by different vegetation types.

Dominant vegetation type	No. beaches	Vegetation			Sand		
		Mean	Median	SD	Mean	Median	SD
Ficinia spiralis	3	44.00	49.61	11.81	56.00	50.39	11.81
Spinifex sericeus	1	41.09	41.09	NA	58.91	58.91	NA
S. sericeus - woody vegetation	6	59.37	62.27	20.10	40.63	37.73	20.10
Ammophila arenaria	0	n/a	n/a	n/a	n/a	n/a	n/a
Woody vegetation - *A. arenaria*	8	65.73	69.99	18.58	34.27	30.01	18.58
Other herbaceous species	1	82.24	82.24	9.91	17.76	17.76	9.91
Exotic pasture grasses	2	58.56	67.54	28.21	41.44	32.46	28.21

The vegetation type that dominated the most beaches was woody vegetation - *A. arenaria* (*n* = 8), followed by *S. sericeus* - woody vegetation (*n* = 6); *F. spiralis* (n = 3); exotic pasture grasses (*n* = 2); *S. sericeus* (*n* = 1), and other herbaceous species (*n* = 1) (Table 9). The *A. arenaria* vegetation type did not dominate at any beach, and there was relatively high variation in the mean plot percent cover of the exotic pasture grasses vegetation type (Fig 18; Table 9).

The relative percent cover of the different cover classes within plots at a national level (Fig 19; Table 10) showed that where native sandbinders dominated plots, there was a low percentage cover of other cover classes and high percent cover of sand. Where other vegetation types dominated, there was a greater mix of, and percent cover, of all cover classes and lower percent sand.

When *F. spiralis* was the dominant vegetation type, plots had the highest relative percent cover of sand (65.6%, SD = 17.3), followed by *A. arenaria* dominated plots (59.4, SD = 11.8), although *A. arenaria* was dominant in only seven plots. Plots dominated by exotic pasture grasses also had a relatively high percent cover of sand, albeit with the highest standard deviation (52.6%, SD = 33), followed by *S. sericeus* (45.4%; SD = 14.6); *S. sericeus* – woody vegetation (28.8%, SD =19.3), then lastly woody vegetation – *A. arenaria* (27%, SD = 15.8) (Fig. 19, Table 10).

Landscape metric spectral diversity measures calculated using the per cent cover of the six cover classes within plots dominated by different cover types showed that lower values for dominance (the landscape metric SIDI) and evenness (the landscape metric SIEI) were observed for plots dominated by *F. spiralis* and *S. sericeus* vegetation types (Fig 20, Table 11), whereby a lower score means greater dominance or less-even distribution of cover classes within plots, respectively (Fig. 20; Table 11). However, plots dominated by the exotic pasture grasses had the lowest dominance and evenness of all types, albeit with the greatest standard deviation, and *A. arenaria* had similar scores to *S. sericeus* (Fig 20, Table 11).

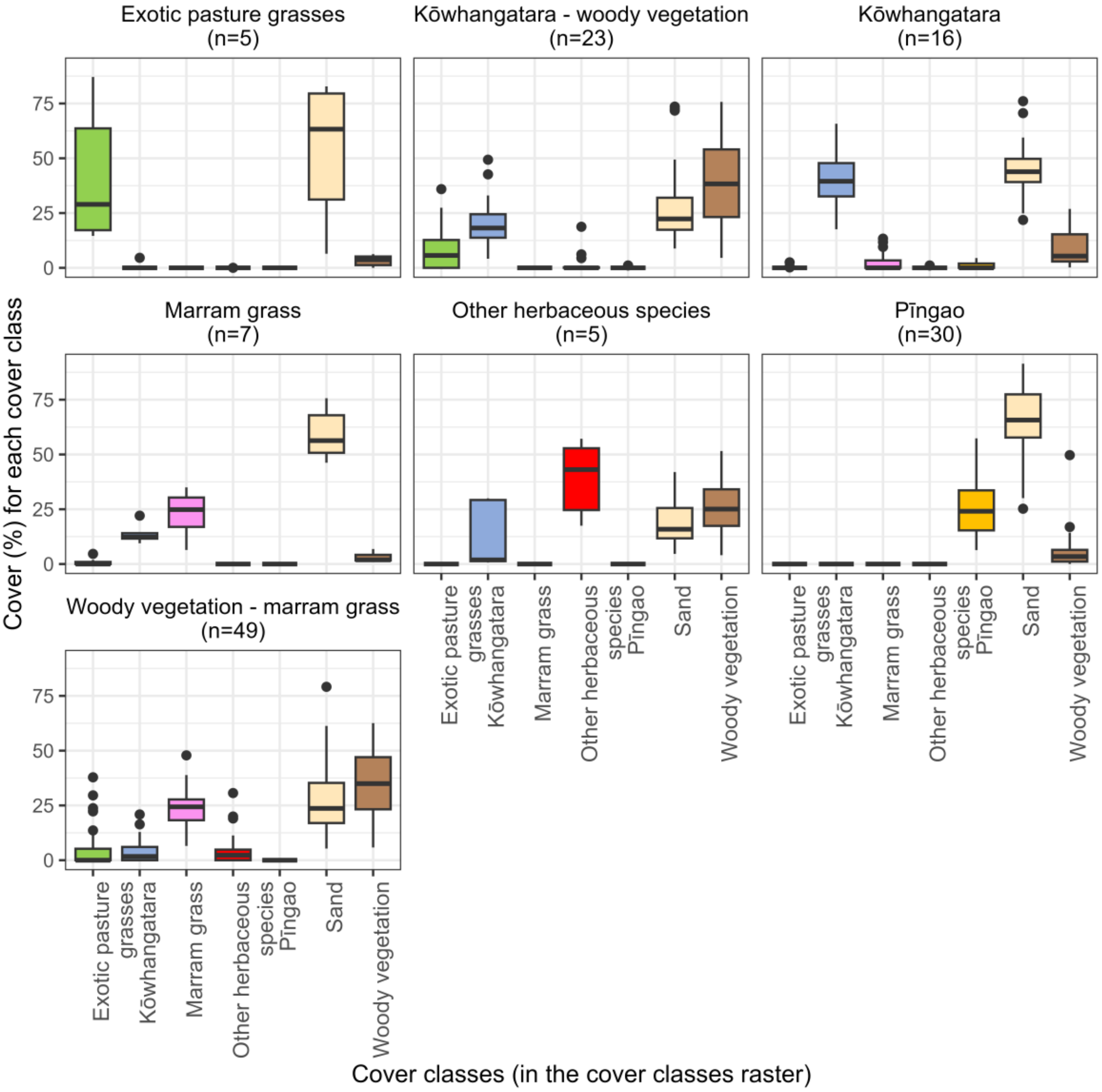


Fig. 19. Per cent cover within plots of different cover classes in 135 plots on 21 beaches, by dominant vegetation type of the plot (shown in the titles of the graphs). Where the term "Pīngao" occurs it is equivalent to *Ficinia spiralis*. Where the term "Kōwhangatara" occurs, it is equivalent to *Spinifex sericeus* and the term "Marram grass" is equivalent to *Ammophila arenaria*.

Table 10. Mean per cent cover of seven cover classes within plots dominated by different vegetation types.

Dominant vegetation type	No. plots	*Ficinia spiralis*		*Spinifex sericeus*		*Ammophila arenaria*		Woody vegetation		Other herbaceous species		Exotic pasture grasses		Sand	
		Mean	SD	Mean	SD	Mean	SD	Mean	SD	Mean	SD	Mean	SD	Mean	SD
Ficinia spiralis	30	26.36	13.62	0.00	0.00	0.00	0.00	5.90	9.39	0.00	0.00	0.00	0.00	65.58	17.25
Spinifex sericeus	16	0.84	1.39	39.82	12.57	2.76	4.80	9.35	8.93	0.07	0.27	0.21	0.64	45.36	14.62
S. sericeus - woody vegetation	23	0.05	0.23	20.58	10.46	0.00	0.00	39.41	20.40	1.36	4.09	8.69	10.39	28.84	19.36
A. arenaria	7	0.00	0.00	13.60	4.06	22.97	10.95	3.13	2.37	0.00	0.00	0.93	1.78	59.36	11.79
Woody vegetation - *A. arenaria*	49	0.00	0.00	3.82	4.83	23.74	8.12	35.40	14.46	4.00	6.00	4.75	9.23	27.03	15.27
Other herbaceous species	5	0.00	0.00	12.62	15.51	0.00	0.00	26.44	17.85	39.05	17.37	0.00	0.00	19.96	14.45
Exotic pasture grasses	5	0.00	0.00	0.92	2.05	0.00	0.00	3.27	2.62	0.00	0.01	42.30	31.78	52.66	32.95

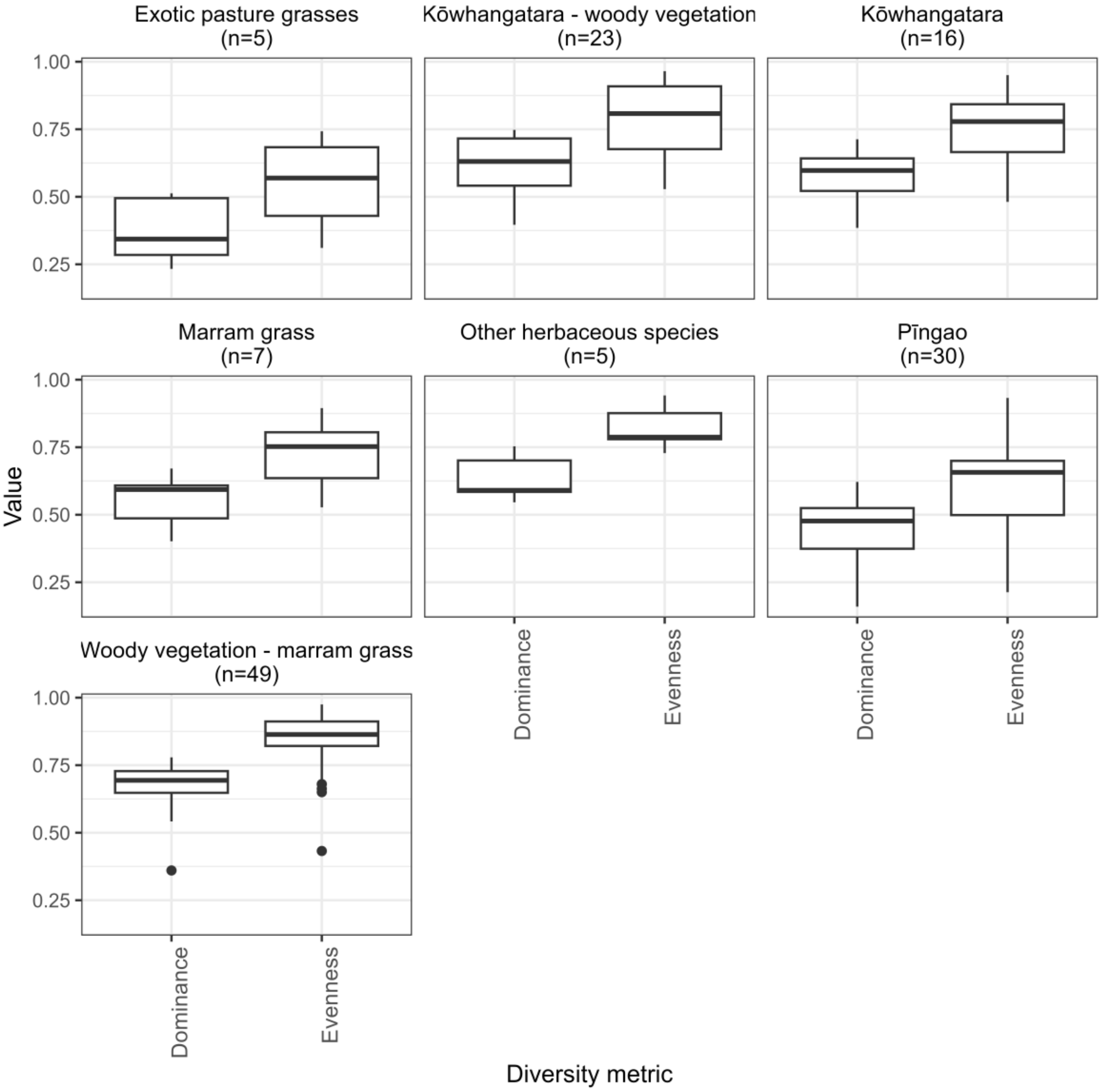


Fig. 20. Dominance (SIDI) and evenness (SIEI) scores calculated for the spectral cover classes within 135 plots on 21 beaches, by dominant vegetation type of the plot (shown in the titles of the graphs). A lower score means greater dominance or less-even distribution of cover classes within plots, respectively. Where the term “Pīngao” occurs, it is equivalent to *Ficinia spiralis*. Where the term “Kōwhangatara” occurs, it is equivalent to *Spinifex sericeus* and the term “Marram grass” is equivalent to *Ammophila arenaria*.

Table 11. Landscape metric spectral diversity statistics calculated using percent cover of the six cover classes within plots dominated by different vegetation types. A lower score means greater dominance or less-even distribution of cover classes within plots, respectively.

Dominant vegetation type	Number of plots	Dominance (SIDI)		Evenness (SIEI)	
		Mean	Standard deviation	Mean	Standard deviation
Woody vegetation - *A. arenaria*	49	0.68	0.08	0.85	0.10
Ficinia spiralis	30	0.44	0.14	0.59	0.19
Spinifex sericeus - woody vegetation	23	0.61	0.11	0.78	0.14
S. sericeus	16	0.58	0.09	0.75	0.13
A. arenaria	7	0.55	0.10	0.72	0.14
Other herbaceous species	5	0.64	0.09	0.82	0.09
Exotic pasture grasses	5	0.37	0.13	0.55	0.18

In comparing dominance with presence at plot level, woody vegetation was a co-dominant in 72 plots, while it was present as a cover class within plots in 134 of 135 plots (Table 12). *F. spiralis* was the dominant vegetation type in 30 of the 36 plots where it was present; *S. sericeus* in 16 out of 88 plots, although it was co-dominant with woody vegetation in 23 other plots; *A. arenaria* was the dominant vegetation type in only seven of the 61 plots where it occurred, and other herbaceous species and exotic pasture grasses had similarly low rates of dominance compared to its presence (5 out of 47, and 5 out of 38, respectively) (Table 12).

Table 12. Number of plots (n = 135) where the cover class was present and the number of plots dominated by each vegetation type.

Cover class	Number of plots where the cover class was present	Number of plots dominated by each vegetation type *
Ficinia spiralis	36	30
Spinifex sericeus	88	16 (23)

Ammophila arenaria	61	7 (49)
Woody vegetation	134	(72)
Other herbaceous species	47	5
Exotic pasture grasses	38	5

4.3.2. Spatial pattern of vegetation in plots based on the sand – vegetation binary raster

Approximately 76% of the variation in the landscape spatial pattern metrics across all plots in the sand-vegetation binary dataset (sand and other cover classes = 0, vegetation = 1) was explained by the first three principal components (Figs. 21 and 22), which explained 47%, 16.2%, and 13.1%, respectively (Figs. 21 and 22).

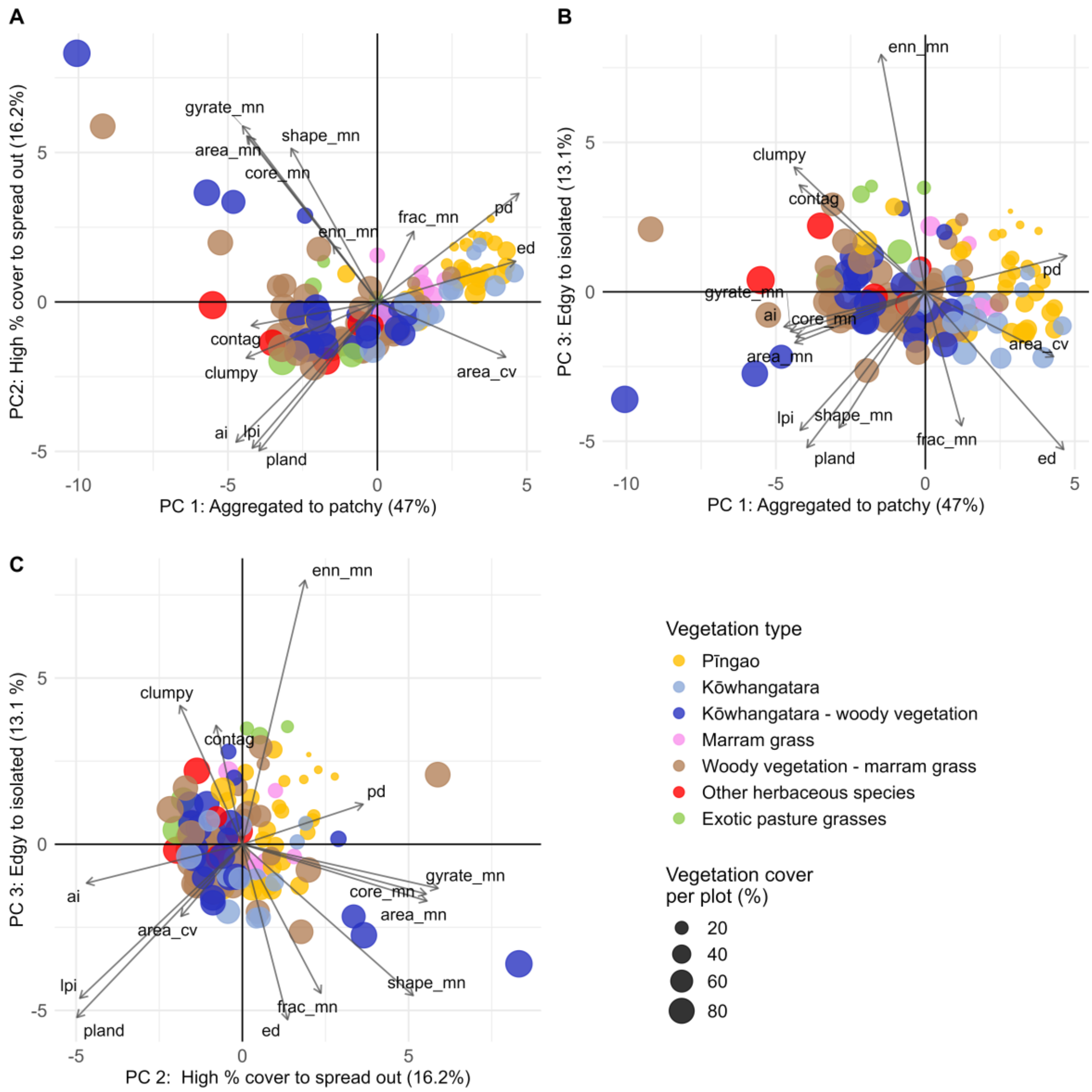
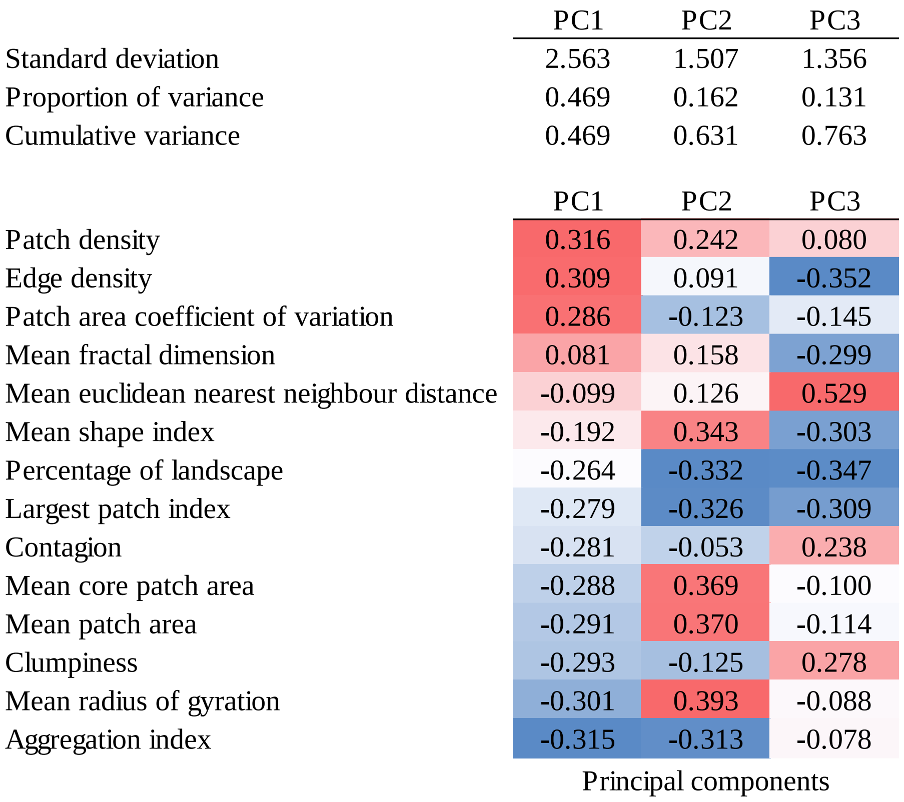


Fig. 21. Bi-plots for the first three principal components of landscape metrics for the vegetation cover class from the sand and vegetation binary layer, which collectively explained approximately 76% of the variation in the data. Dots represent the positions of the 135 plots in ordination space, coloured by the dominant vegetation type of each plot. PCA eigenvectors are labelled with an abbreviated version of the name of the variables; the full names are given in Table 8 and loadings are given in Fig. 22. Where the term “Pīngao” occurs it is equivalent to *Ficinia spiralis*. Where the term “Kōwhangatara” occurs it is equivalent to *Spinifex sericeus* and the term “Marram grass” is equivalent to *Ammophila arenaria*.



	PC1	PC2	PC3
Standard deviation	2.563	1.507	1.356
Proportion of variance	0.469	0.162	0.131
Cumulative variance	0.469	0.631	0.763

	PC1	PC2	PC3
Patch density	0.316	0.242	0.080
Edge density	0.309	0.091	-0.352
Patch area coefficient of variation	0.286	-0.123	-0.145
Mean fractal dimension	0.081	0.158	-0.299
Mean euclidean nearest neighbour distance	-0.099	0.126	0.529
Mean shape index	-0.192	0.343	-0.303
Percentage of landscape	-0.264	-0.332	-0.347
Largest patch index	-0.279	-0.326	-0.309
Contagion	-0.281	-0.053	0.238
Mean core patch area	-0.288	0.369	-0.100
Mean patch area	-0.291	0.370	-0.114
Clumpiness	-0.293	-0.125	0.278
Mean radius of gyration	-0.301	0.393	-0.088
Aggregation index	-0.315	-0.313	-0.078

Fig. 22. PCA axis loadings for landscape metrics calculated for the sand-vegetation binary raster. Colour and hues represent the strength of the loading, grading from positive (red), to neutral (white), and to negative (blue).

The dominant vegetation type of plots was strongly related to the gradients in landscape metrics. For example, for the first principal component (Figs. 21 and 22), plots dominated by the native sand binder vegetation types, *F. spiralis* and *S. sericeus*, were strongly aligned with the positive end of this gradient, exhibiting high patch density and lower total percent vegetation cover. Conversely, plots dominated by *S. sericeus* - woody vegetation, woody vegetation – *A. arenaria*, exotic pasture grasses and other herbaceous species were most strongly aligned with the negative end. They had large, tightly clustered (aggregated) patches and higher total percent vegetation cover. Plots dominated by *A. arenaria* were scattered across the ordination space, showing a higher diversity of spatial patterns. The positive end of the second component (Figs. 21 and 22), had plots dominated by *S. sericeus* - woody vegetation and woody vegetation – *A. arenaria*, with large, spread out (high radius of gyration) patches. The negative end of the second component also had plots dominated by *S.*

sericeus - woody vegetation, woody vegetation – *A. arenaria* as well as exotic pasture grasses and other herbaceous species. These plots had a higher total percent cover of vegetation. The positive end of the third component (Figs. 21 and 22) was characterised by isolated patches of exotic pasture grasses, and the negative end was characterised by plots with large, edgy patches of *S. sericeus* - woody vegetation characterised plots at the negative end (Figs. 21 and 22).

4.3.3. Spatial pattern of vegetation cover classes for the subset of plots where Ficinia spiralis was present

Approximately 78% of the variation in the landscape metrics computed for the vegetation cover classes, for the subset of plots where the *F. spiralis* cover class was present, was explained by the first two principal components (Figs. 23 and 24), which explained 51% and 27%, respectively. The third component was not included in the results given here as it did not explain more than 10% of the variation in the data (Figs. 23 and 24).

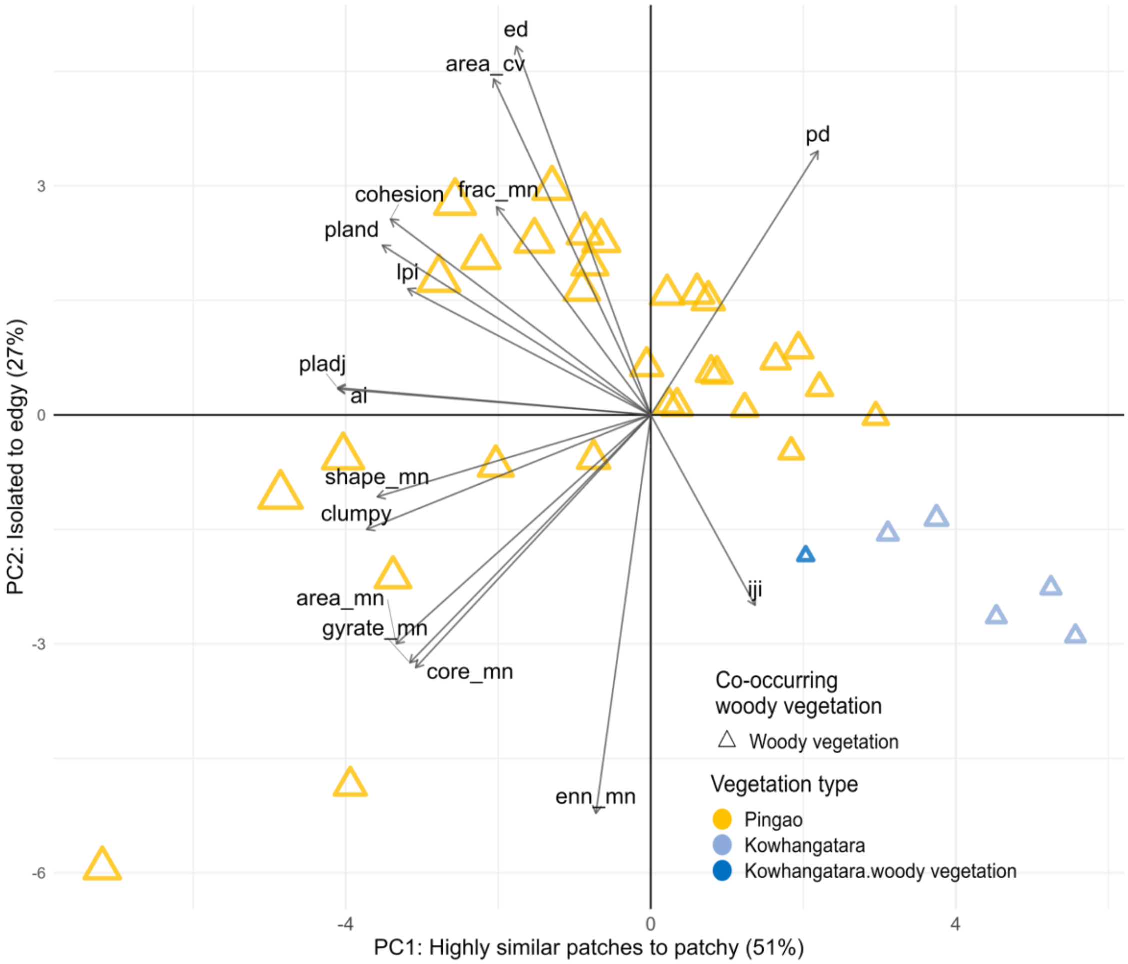
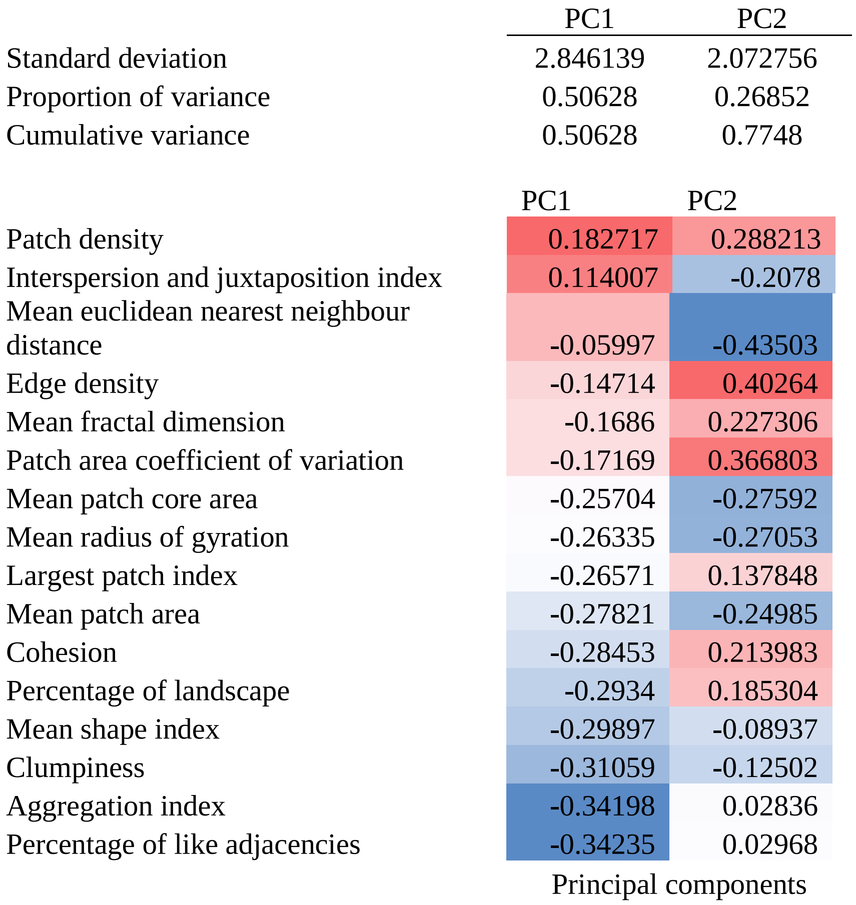


Fig. 23. Bi-plots for the principal component analysis (PCA) of landscape metrics computed on the cover class raster for the plots where the native sandbinder *F. spiralis* occurred. Bi-plots are shown for the first and second principal components, which collectively explained approximately 78% of the variation in the data. The loadings for each variable are given in Fig. 24. Where the term “Pīngao” occurs, it is equivalent to *Ficinia spiralis*. Where the term “Kōwhangatara” occurs, it is equivalent to *Spinifex sericeus.*



	PC1	PC2
Standard deviation	2.846139	2.072756
Proportion of variance	0.50628	0.26852
Cumulative variance	0.50628	0.7748

	PC1	PC2
Patch density	0.182717	0.288213
Interspersion and juxtaposition index	0.114007	-0.2078
Mean euclidean nearest neighbour distance	-0.05997	-0.43503
Edge density	-0.14714	0.40264
Mean fractal dimension	-0.1686	0.227306
Patch area coefficient of variation	-0.17169	0.366803
Mean patch core area	-0.25704	-0.27592
Mean radius of gyration	-0.26335	-0.27053
Largest patch index	-0.26571	0.137848
Mean patch area	-0.27821	-0.24985
Cohesion	-0.28453	0.213983
Percentage of landscape	-0.2934	0.185304
Mean shape index	-0.29897	-0.08937
Clumpiness	-0.31059	-0.12502
Aggregation index	-0.34198	0.02836
Percentage of like adjacencies	-0.34235	0.02968

Principal components

Fig. 24. Principal component loadings for the PCA of landscape metrics computed for the cover class raster for the plots where the native sandbinder *F. spiralis* occurred. Colour and hue represent the strength of the loading, grading from positive (red), to neutral (white), and to negative (blue).

Nearly all of the 36 plots where *F. spiralis* was present were dominated by *F. spiralis* (Fig. 23, Table 12). Woody vegetation occurred in all plots, and *A. arenaria* was not present in any plots (Fig. 23). The percent cover of *F. spiralis* in plots was positively related to the first principal component (Figs. 23 and 24). At the negative end of the gradient plots exhibited a high similarity of cover classes and aggregated cover classes, as well as greater total percent cover of *F. spiralis* in plots. The positive end was characterised by higher patch density and lower total percent cover of *F. spiralis* in plots. The second component had plots with relatively greater edginess and variation in size at the positive end of the gradient, while the negative end had higher amounts of isolated patches of *F. spiralis* (Figs. 23 and 24).

4.3.4. Spatial pattern of vegetation cover classes for the subset of plots where S. sericeus was present

Approximately 67% of the variation in landscape metrics computed for the vegetation cover classes in the subset of plots where *S. sericeus* was present was explained by the first two principal components (Figs. 25 and 26), which explained 45% and 22%, respectively. The third component is not included in the results given here as it did not explain more than 10% of the variation in the data (Figs. 25 and 26).

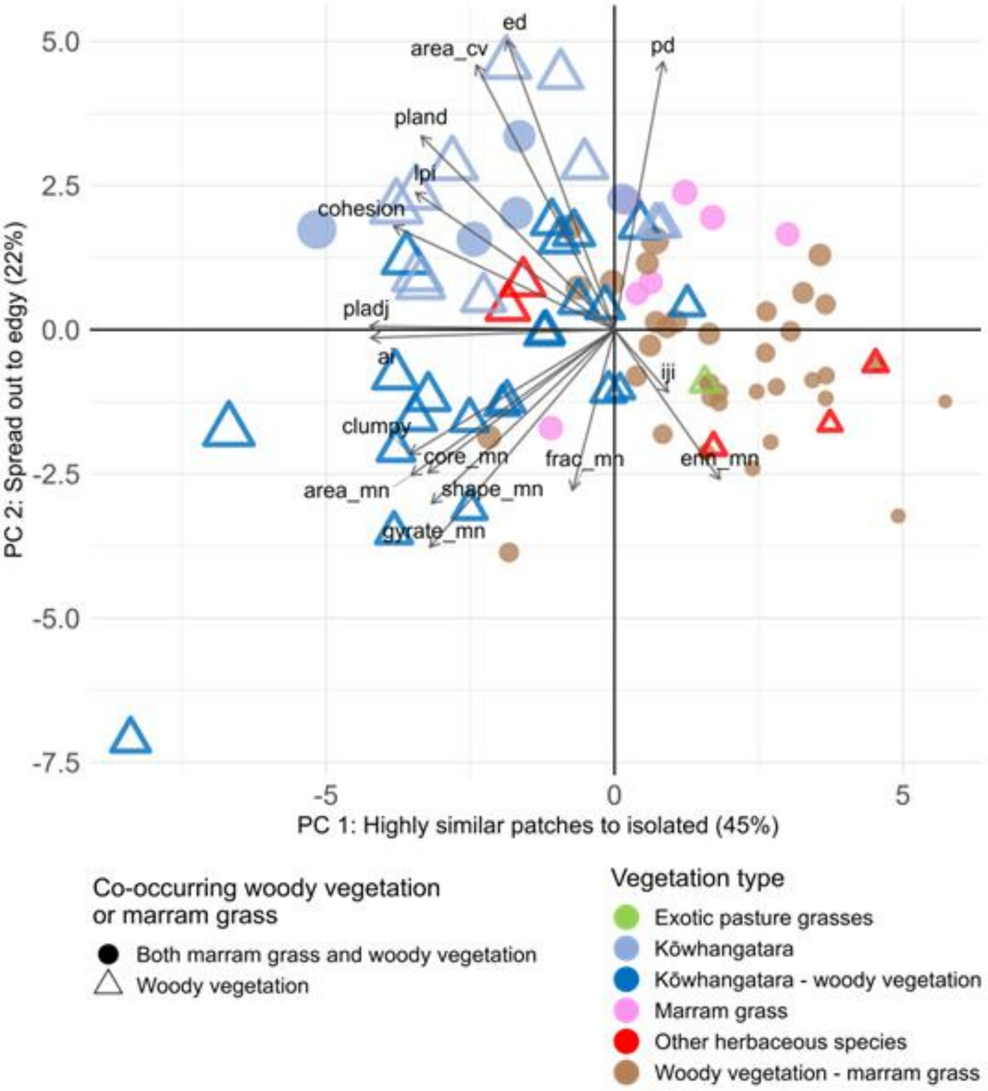
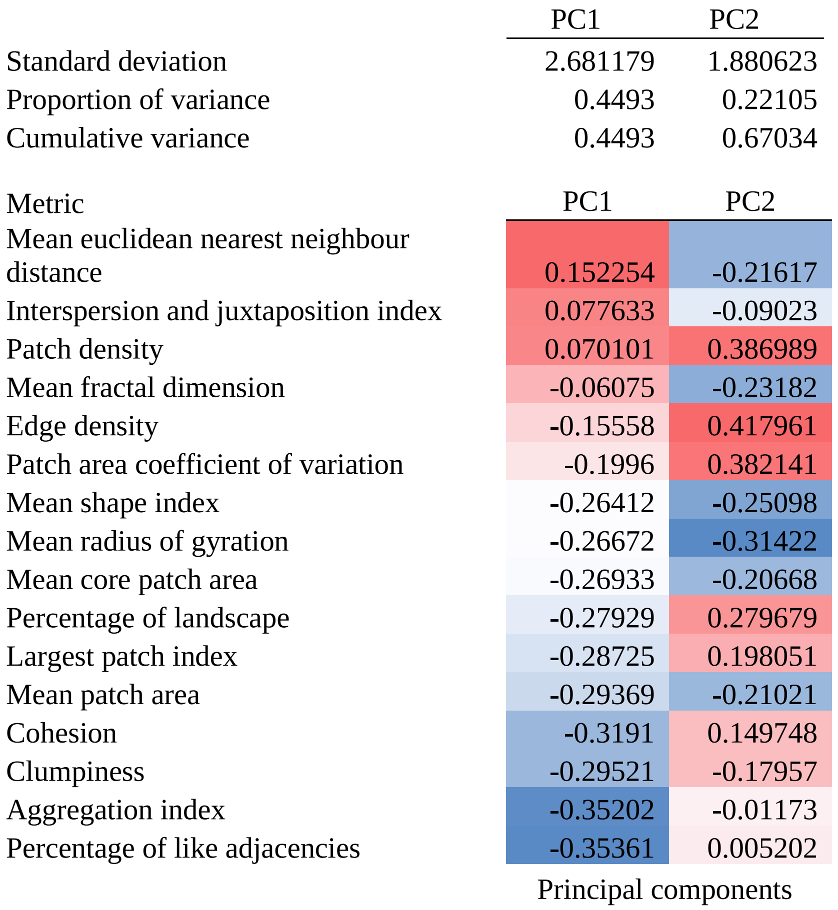


Fig. 25. Bi-plots for the principal components analysis (PCA) of landscape metrics computed on the cover class raster for plots where *S. sericeus* occurred. Bi-plots are shown for the first and second principal components, which collectively explained approximately 67% of the variation in the data. The loadings for each variable are given in Fig. 26. Symbols on the plot represent the positions of the 88 plots in ordination space. Points are coloured by the dominant vegetation type of each plot, with the presence of *A. arenaria* and / or woody vegetation shown by filled circles or triangles, respectively. The size of symbols reflects is proportional to the percent cover of *S. sericeus* in the plot. PCA eigenvectors are labelled with shortened names; full names and definitions are given in Table 6, and PCA loadings are given in Fig. 26. Where the term “Kōwhangatara” occurs, it is equivalent to *Spinifex sericeus* and the term “Marram grass” is equivalent to *Ammophila arenaria*.



	PC1	PC2
Standard deviation	2.681179	1.880623
Proportion of variance	0.4493	0.22105
Cumulative variance	0.4493	0.67034

Metric	PC1	PC2
Mean euclidean nearest neighbour distance	0.152254	-0.21617
Interspersion and juxtaposition index	0.077633	-0.09023
Patch density	0.070101	0.386989
Mean fractal dimension	-0.06075	-0.23182
Edge density	-0.15558	0.417961
Patch area coefficient of variation	-0.1996	0.382141
Mean shape index	-0.26412	-0.25098
Mean radius of gyration	-0.26672	-0.31422
Mean core patch area	-0.26933	-0.20668
Percentage of landscape	-0.27929	0.279679
Largest patch index	-0.28725	0.198051
Mean patch area	-0.29369	-0.21021
Cohesion	-0.3191	0.149748
Clumpiness	-0.29521	-0.17957
Aggregation index	-0.35202	-0.01173
Percentage of like adjacencies	-0.35361	0.005202

Principal components

Fig. 26. Principal component loadings for the PCA of landscape metrics computed for the cover class raster for the plots where the native sandbinder *S. sericeus* occurred. Colour and hue represent the strength of the loading, grading from positive (red), to neutral (white), and to negative (blue).

Dominant vegetation type and the percent cover of *S. sericeus* in the plot were strongly aligned with the spatial pattern of vegetation described by the principal components (Figs. 25 and 26). Woody vegetation occurred in 134 of 135 plots, and *A. arenaria* occurred in 49 plots (Table 12); thus, there were no plots in the dataset where *A. arenaria* occurred without woody vegetation. For the first principal component, at the negative end of the gradient there was a high similarity and aggregation of cover classes and a high percent cover of *S. sericeus*. In contrast, at the positive end of the gradient plots were strongly associated with isolated patches, low percent cover of *S. sericeus* and domination by woody vegetation and *A. arenaria*. The second component was characterised by co-dominant *S.*

sericeus-woody vegetation plots with spread-out patches at the negative end of the gradient, while patches dominated by *S. sericeus* with greater edginess and variation in size were associated with the positive end of the gradient (Figs. 25 and 26).

4.3.5. Effect of environmental variables on spatial pattern

Sand – vegetation binary raster

The conditional inference tree model for the first principal component of landscape metrics (response variable) for the sand-vegetation binary raster explained 49% of the variation in the data, with a root mean squared error (RMSE) of 1.9 (Table 13). Subsequent models for the second and third principal components (response variables) explained 28% (RMSE = 1.31) and 10.8% (RMSE = 1.35) of the variance, respectively (Table 13). Consistently, the top predictor variable of global importance (global variable) for explaining all three principal components was ecosystem extent. High wind gusts or mean daily wind run was either the second or third most important global predictor variables, and total rainfall was the third most important for the third principal component (Table 14).

Table 13. Accuracy scores for conditional inference tree analysis with the principal components as the response variable and environmental variables as the predictors for the sand-vegetation binary raster (Figs. 21 and 22); cover class raster for the plots where the native sandbinder *F. spiralis* occurred (Figs. 23 and 24), and the cover class raster for the plots where the native sandbinder *S. sericeus* occurred (Figs. 25 and 26). The minimum criterion for accuracy was set at 95%.

Principal Component	R squared	RMSE
1. Sand - vegetation binary raster		
PC1	0.49	1.91
PC2	0.28	1.31
PC3	0.11	1.35
2. Cover classes subset - *F. spiralis*		
PC1	0.74	1.59
PC2	0.76	1.20
3. Cover classes subset - *S. sericeus*		
PC1	0.36	2.33
PC2	0.27	1.71

Table 14. Importance (%) of global variables for the conditional inference tree model for the first three principal components of the PCA calculated using the landscape metrics for the sand-vegetation binary raster (*n* = 135 plots).

Variable	Normalised importance (%)		
	PC1	PC2	PC3
Days of coastal waves greater than 4 m (p.a)	12.7	12.6	6.2
Ecosystem extent (ha)	15.5	18.8	31.2
Days of wind gusts greater than 24 knots (p.a)	14.3	14.6	7.1
Mean daily wind run (km)	14.3	14.6	7.1
Mean total rainfall (mm)	14.3	14.6	7.1
Median income (NZD)	9.6	7.8	5.1
Adjacent land cover type	7.4	8.9	1.5
Human population	6.7	2.9	1.6
Distance from road (m)	5.1	5.4	4.8

The individual tree model (tree) for the first component of landscape metrics (response variable) had six terminal nodes and a depth of three nodes (Fig. 27). The initial split in the tree, or the strongest predictor of spatial pattern, was made on whether or not the plot had an adjacent land cover of sand. Beaches lacking adjacent sand cover exhibited an aggregated spatial pattern in vegetation and were associated with woody vegetation at the negative end of the PC1 axis (Fig. 21). Conversely, beaches with adjacent sand cover had higher patch and edge density and were associated with a dominant cover of *F. spiralis* and *S. sericeus* at the positive end of the PC1 axis (Fig. 21).

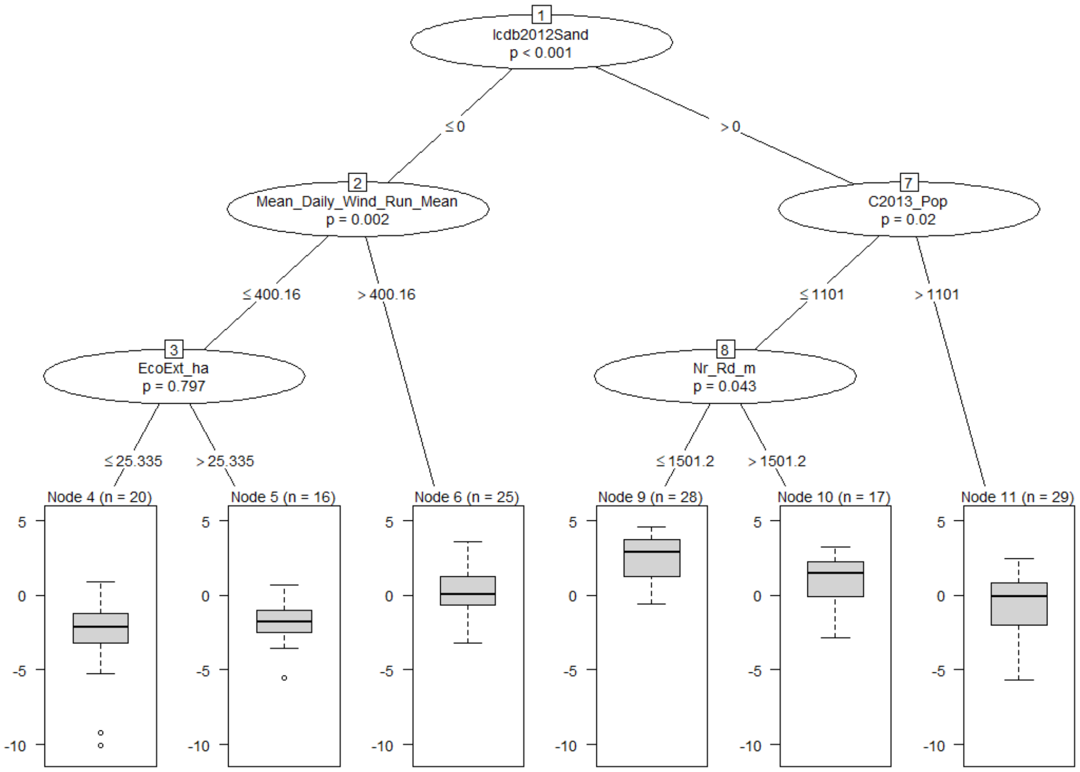


Fig. 27. Conditional inference tree for the first principal component of the landscape metrics calculated for the sand-vegetation binary raster for 135 plots, showing the environmental variables that distinguished among plots. Numbers on the branches represent thresholds for environmental variables. The y-axes of the boxplots of nodes at the base of the tree show the first principal component values for each group of plots. Low values on PC1 represent an aggregated spatial pattern and high values represent high patch density.

The tree for the second principal component (response variable) had three terminal nodes and a depth of two. The initial split was made on the effect of the size of the usually resident human population (Fig. 28). Both ends of PC2 were associated with woody vegetation as the dominant vegetation type (Fig. 21). The negative end of the PC2 gradient was aligned with a high population, lack of adjacent farmland, greater percentage of landscape in vegetation, aggregated and larger patches. The positive end of PC2 was aligned with a lower population, higher mean daily wind run, and a spatial pattern of more spread-out woody vegetation patches with higher mean and core patch area (Fig. 21). The tree plot for the third component could not be computed due to weak correlations between the PC3 response variable and the predictors (Chapter 4, Appendix 2).

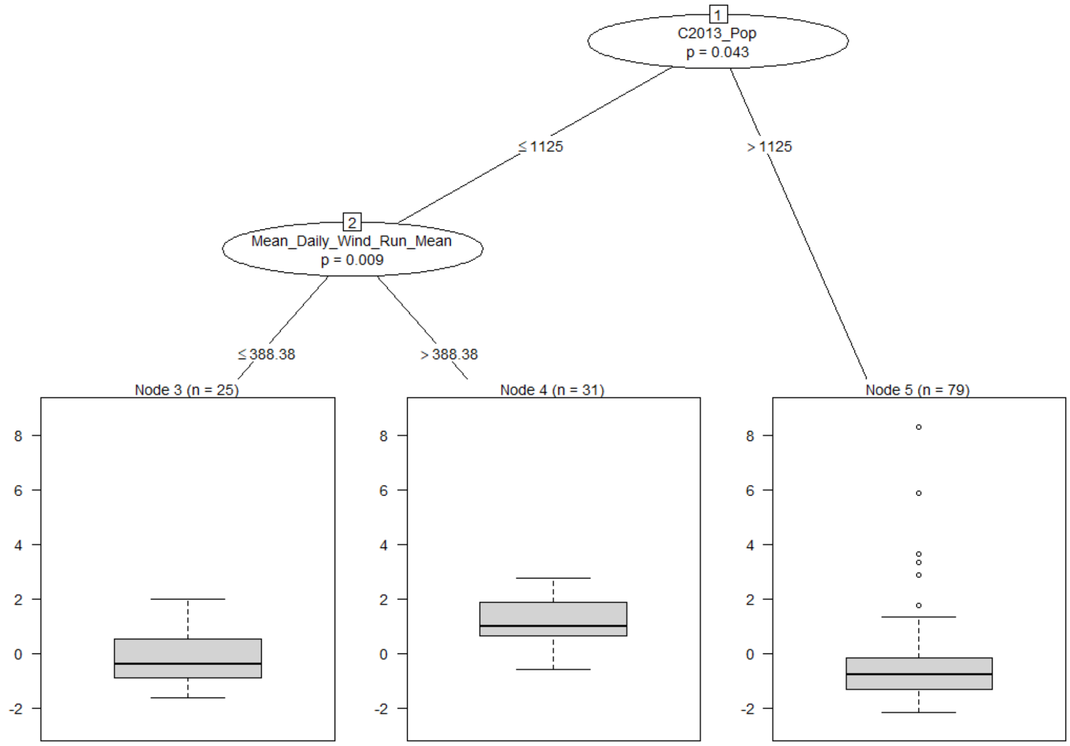


Fig. 28. Conditional inference tree for the second principal component of the landscape metrics calculated for the sand-vegetation binary raster for 135 plots, showing the environmental variables that distinguished among plots. Numbers on the branches represent thresholds for environmental variables. The y-axes of the boxplots of nodes at the base of the tree show the second principal component values for each group of plots. Low values on PC2 represent high percent cover of vegetation and high values represent a spatial pattern of spread-out shaped patches.

Environmental variable relationships with the spatial pattern of vegetation cover classes for the subset plots where F. spiralis was present

The conditional inference tree model for the first principal component of landscape metrics (response variable) computed for the vegetation cover classes within plots where *F. spiralis* co-occurred with woody vegetation explained 74% of the variation in the data (RMSE = 1.59), and 76% for the second component (response variable) (RMSE = 1.2). The top two predictor global variables explaining the first and second principal components were human population, followed by days of high wind gusts. The third most important global predictor variables for explaining PC1 was either mean daily wind run or days of large waves in PC2 (Table 15).

Table 15. Importance (%) of global variables for the conditional inference tree model for the first two principal components of the PCA calculated using the landscape metrics for the cover class raster for the plots where the native sandbinder *F. spiralis* occurred (*n* = 36).

Variable	Normalised importance (%)	
	PC1	PC2
Population	13.6	13.5
Days of wind gusts greater than 24 knots (p.a)	13.2	13.5
Ecosystem extent (ha)	13.2	13.5
Mean total rain (mm)	13.2	13.5
Mean daily wind run (km)	13.2	13.5
Days of coastal waves greater than 4 m (p.a)	13.2	13.5
Median income	12.9	10.7
Distance from road (m)	5.6	6.5
Land cover type	1.8	2.0

The tree for the first component of landscape metrics (response variable) had three terminal nodes, a depth of two, and the initial split was made on the predictor of mean number of days of wind gusts over 24 km / hr. p.a. (Fig. 29). Where the number of days was less than or equal to 11.43 p.a., the plots at terminal nodes occurred at the positive end of the PCA gradient (Fig. 23), which was characterised by plots with high patch density and low percent cover of the *F. spiralis* cover class. They were largely dominated by *F. spiralis*, although occasionally were dominated by *S. sericeus*. Where days of wind gusts were greater than 11.43 p.a., and the number of days of large waves exceeded 19 p.a., beaches occurred at the negative end of the gradient, where the *F. spiralis* cover class was aggregated, clumpy, and had a higher percent cover within plots (Fig. 23).

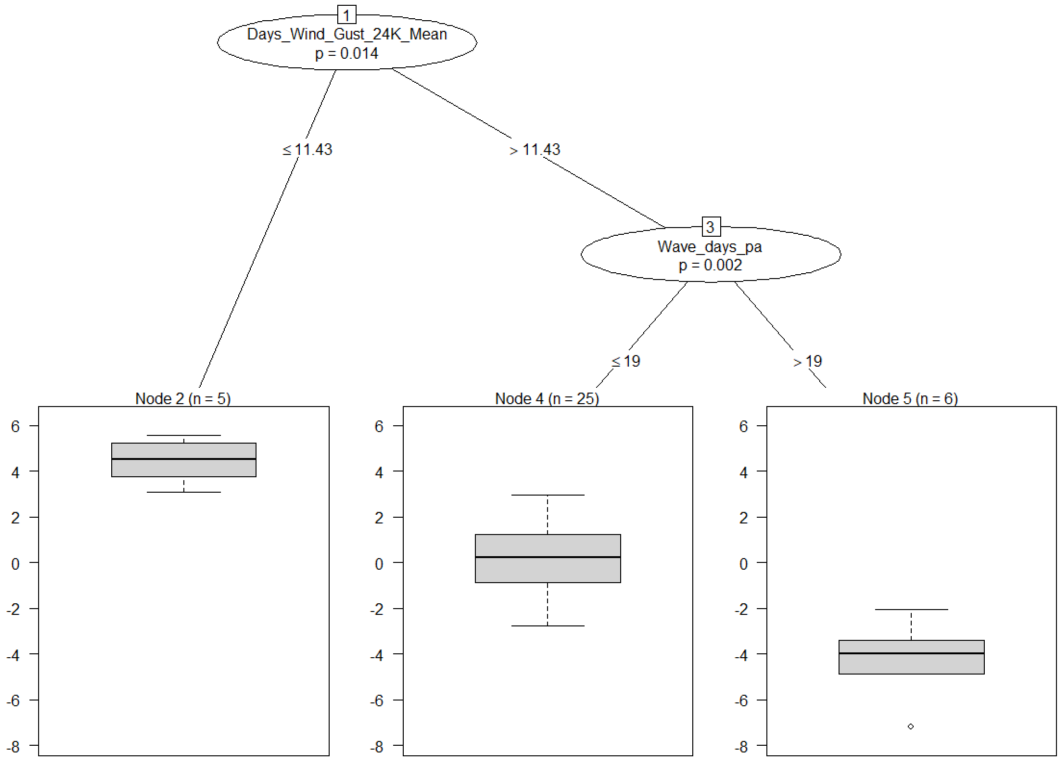


Fig. 29. Conditional inference tree for the first principal component of the landscape metrics calculated for the cover class raster for 36 plots where *F. spiralis* occurred, showing the environmental variables that distinguished among plots. Numbers on the branches represent thresholds for environmental variables. The y-axes of the boxplots of nodes at the base of the tree show the first principal component values for each group of plots. Low values on PC1 represent a spatial pattern of highly similar patches and high values represent a high patch density

The tree for the second component of landscape metrics had five terminal nodes and a depth of three (Fig. 30). Plots associated with this axis were dominated by *F. spiralis* at both ends of the axis, with *S. sericeus* in the lower mid-range (Fig. 23). The initial split was on the predictor of ecosystem extent, with nodes at the negative end of the axis characterised by small beaches that were farthest from the road and associated with isolated patches (Fig. 30). The positive end of the axis was associated with plots on large beaches that were closer to the road, but still further than other beaches. The positive end of the gradient was associated with plots that had edgy cover classes (Fig. 23).

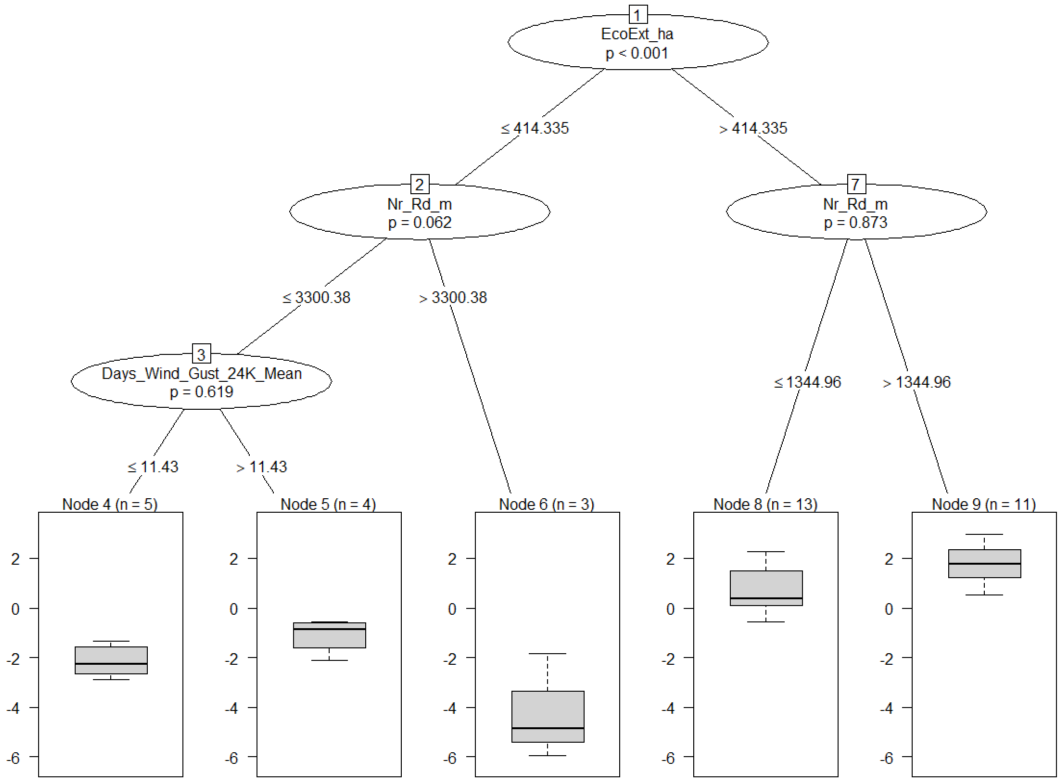


Fig. 30. Conditional inference tree for the second principal component of the landscape metrics calculated for the cover class raster for 36 plots where *F. spiralis* occurred, showing the environmental variables that distinguished among plots. Numbers on the branches represent thresholds for environmental variables. The y-axes of the boxplots of nodes at the base of the tree show the second principal component values for each group of plots. Low values on PC2 represent isolated patches and high values represent high edge density.

Environmental variable relationships with the spatial pattern of vegetation cover classes for the subset of plots where Spinifex sericeus was present

The conditional inference tree model for the first principal component (response variable) of landscape metrics computed for the vegetation cover classes within the subset of plots where *S.*

sericeus was present explained 36% of the variation in the data (RMSE = 2.32) and 26% for the second component (response variable) (RMSE = 1.71). Consistently, the top predictor variable of global importance in the first and second principal components was ecosystem extent. The second and third most important predictor variables were either strong wind gusts, mean daily wind run, or total rain (Table 16).

Table 16. Importance (%) of global variables for the conditional inference tree model for the first two principal components of the PCA calculated using the landscape metrics for the cover class raster for the plots where the native sandbinder *S. sericeus* occurred (*n* = 88).

Variable	Normalised importance (%)	
	PC1	PC2
Ecosystem extent (ha)	16.1	17.3
Mean total rain (mm)	15.4	15.4
Mean daily wind run (km)	15.4	15.4
Days of wind gusts greater than 24 knots (p.a)	15.4	15.4
Population	14.6	12.4
Days of coastal waves greater than 4 m (p.a)	7.9	6.6
Distance from road (m)	5.9	4.3
Land cover type	5.3	3.4
Median income	4.0	9.8

The tree for the first component of landscape metrics (response variable) had six terminal nodes and a depth of four (Fig. 31). The initial split was made on the predictor of mean number of days p.a. with coastal waves over four metres. Where the number of days was less than or equal to 16, plots at terminal nodes occurred at the negative end of the PCA gradient (Fig. 25), where the spatial pattern of vegetation was characterised by high similarity of cover classes comprising either woody vegetation or *S. sericeus* dominated plots (Fig. 25). Where the number of days exceeded 16, further splits were made on the predictor's median income, the mean number of days p.a when wind gusts exceeded 24 km / hr., distance from the road and an adjacent land cover of sand. Plots in terminal nodes at the positive end of the gradient had predictors of high median incomes, a high number of days with wind gusts over 24 knots, proximity to the road, and an adjacent land cover of sand and were associated with *A. arenaria* and *S. sericeus* - woody vegetation (Fig. 25). Where plots did not have a predictor of an adjacent land cover of sand, they had the most positive scores on the PCA gradient and comprised largely of other herbaceous species (Fig. 25).

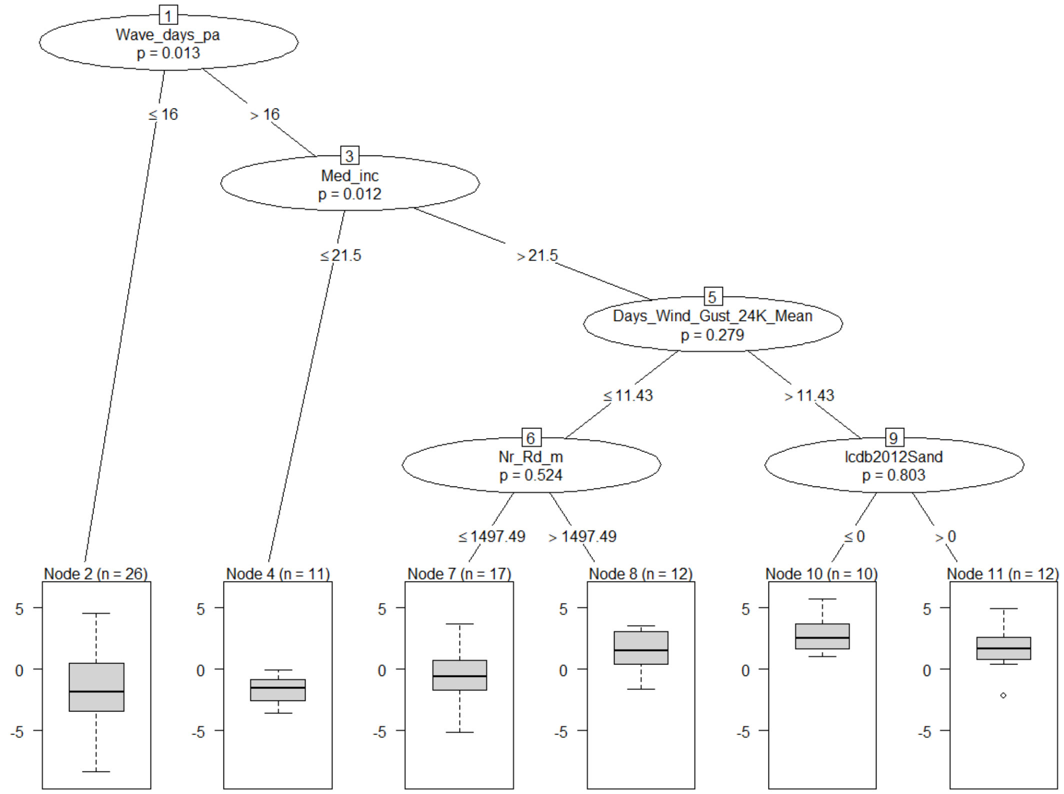


Fig. 31. Conditional inference tree for the first principal component of the landscape metrics calculated for the cover class raster for 88 plots where *S. sericeus* occurred, showing the environmental variables that distinguished among plots. Numbers on the branches represent thresholds for environmental variables. The y-axes of the boxplots of nodes at the base of the tree show the first principal component values for each group of plots. Low values on PC1 represent highly similar patches and high values represent isolated patches.

The tree for the second component of landscape metrics (response variable) also had six terminal nodes and a depth of three (Fig. 32). The initial split was made on the predictor of distance from the road. The negative end of the gradient had predictors of proximity to the road, with low rainfall and both high and low numbers of days of wind gusts over 24 knots (Fig. 32) and was associated with spread out vegetation patches comprised largely of woody dominated vegetation (Fig. 25). The positive end of the gradient had plots that were further from the road, adjacent sand cover, low population (Fig. 32) and was associated with edgy vegetation patches dominated by *S. sericeus* (Fig. 25).

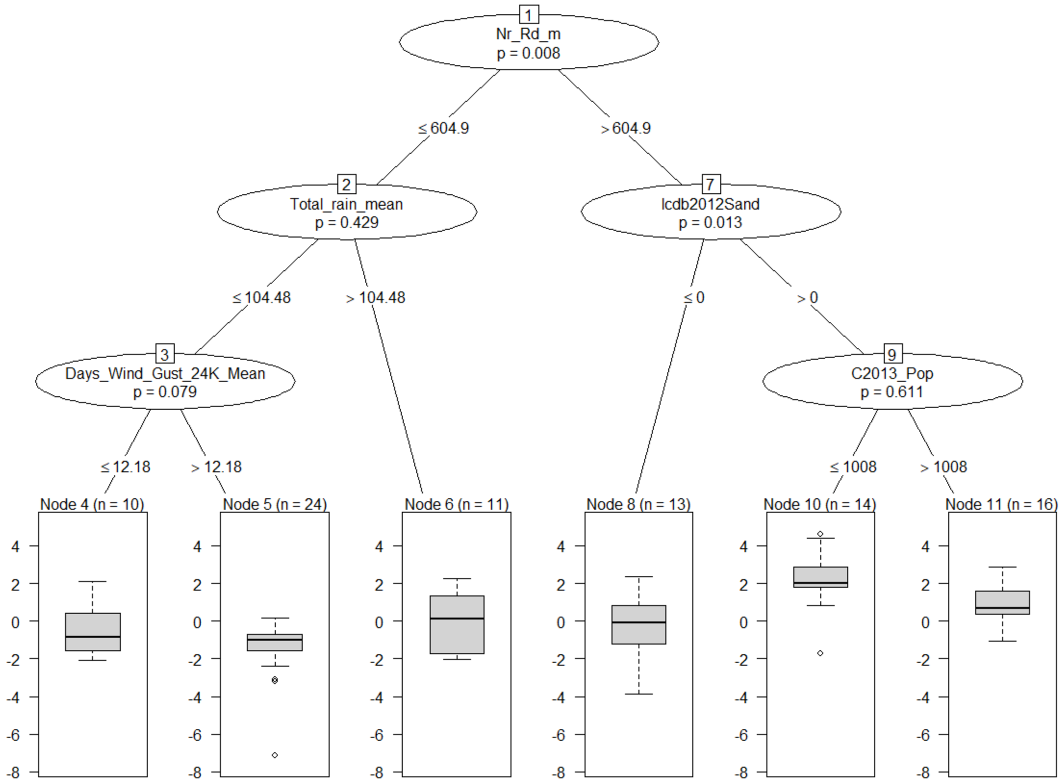


Fig. 32. Conditional inference tree for the second principal component of the landscape metrics calculated for the cover class raster for 88 plots where *S. sericeus* occurred, showing the environmental variables that distinguished among plots. Numbers on the branches represent thresholds for environmental variables. The y-axes of the boxplots of nodes at the base of the tree show the second principal component values for each group of plots. Low values on PC2 represent a spatial pattern of spread-out patches and high values represent a pattern of patches with high edge density.

4.4. Discussion

This study describes the spatial pattern of vegetation cover within 135 plots (90 × 120 m, except for seven plots that were 20 × 27 m) in coastal active dune vegetation in Aotearoa derived from classified, high-resolution RGB aerial imagery. Distinct spatial patterns were identified for plots classified as being dominated by seven nationally representative vegetation types. Plots dominated by native sandbinder vegetation types exhibited higher patch density, edge density, and bare sand than plots dominated by other growth forms, possibly reflecting the effect of natural disturbance. Plots dominated by sandbinder vegetation types, native and exotic, had greater dominance and less evenness of classified cover classes than plots dominated by other growth forms, possibly demonstrating that the specialised traits of sandbinders allow them to colonise and dominate disturbed substrates. Invasion was potentially characterised by a pattern of greater aggregation of patches and higher per cent cover of exotic and woody cover classes, where they co-occurred with *S. sericeus*. Potential reference sites for non-invaded *F. spiralis* dominant beaches were identified at Smoky Beach and Coal River, which were known to have native woody vegetation and low per cent cover of *A. arenaria* present. Conditional inference tree model results suggested that a relationship exists between human activities and spatial patterns of vegetation within plots. A common theme was that where human variables were important, for example smaller ecosystem extents with modified land covers, vegetation patches within plots were more aggregated suggesting eco-geomorphological processes of disturbance and colonisation are affected by human activities. Measures of geomorphic drivers such as the quantification of the sand supply, and geomorphic processes of sand deposition, coastal erosion and flooding at beaches are required to confirm our results, should they become available. Overall, this research suggests the potential for coarse indicators of complex active dune ecomorphological processes and their interactions with human activity to be detected from spatial patterns of vegetation, derived from low-cost RGB aerial imagery. The information can be used in conservation management for baseline and ongoing monitoring of active dunes.

4.4.1. A spatial pattern of patchy mosaics of vegetation and a high per cent cover of bare sand indicate the presence of active dunes processes

Wide variation in the spatial pattern of different vegetation types was evident in the PCA of landscape metrics for the sand–vegetation binary raster (Fig. 21). A large amount of variation in the spatial pattern was explained in the first three principal components (76%), meaning this pattern is a reasonable basis for an indicator of the spatial variation within plots dominated by a range of different vegetation types. The high patch density and edge density observed in plots dominated by *F. spiralis* and *S. sericeus* vegetation types is consistent with spatial heterogeneity caused by disturbance, colonisation and zonation in dune environments (Corenblit et al., 2015; Kumar et al., 2006; García-Mora et al., 1999; García-Novo et al., 2004; Reijers, 2019): patchy vegetation forms during colonisation on disturbed substrate (García-Novo et al., 2004; Reijers et al., 2019) creating mosaics of vegetation patches in a matrix of sand along gradients of disturbance (García-Novo et al., 2004; Moreno-Casasola, 1986). Sandbinders are in a strong position to take advantage of disturbed conditions due to their ability to rapidly vegetatively reproduce through patchy clonal expansion (Reijers et al., 2019), and laterally spreading rhizomes or stolons (Maun, 2009). Conversely, the more aggregated spatial pattern of other growth forms, such as woody plants which formed a range of large, spread out, and irregularly shaped patches suggesting the substrate where these forms occur is relatively undisturbed and soil has had sufficient time to build and support growth forms with vertical root structures (MacGuirk et al., 2022).

The mean per cent cover of sand to vegetation was higher for plots dominated by native sandbinders than other growth forms. These results are consistent with Konlechner et al. (2015), where the per cent cover of vegetation on a pristine, reference, *F. spiralis*-dominated beach was 50% sand cover over two sample periods, while our national study had a mean of 56% sand cover within *F. spiralis* dominant plots on beaches (SD = 11.8%; Table 9). Our *S. sericeus* dominant beaches fell within a similar range, whereby Whatipu Beach had an average of 59 ± SD% sand cover in plots (Table 9). Moreover, results from beaches with plots dominated by either *F. spiralis* or *S. sericeus* are aligned with conclusions from Ryu & Sherman (2014), who showed that dunes were active or had a composite of active and aggregated patches, where sand comprised more than 50% cover. However, Ryu and Sherman (2014) do not investigate the effect of different vegetative cover. In contrast, plots on beaches dominated by exotic, other herbaceous, or woody cover classes had less than 40% sand cover, suggesting they fall within the 'aggregated' or 'inactive' categories of Ryu and Sherman (2014).

Interactions between native sandbinder plant traits and disturbance processes could explain the greater sand cover in native sandbinder-dominant beaches compared to beaches dominated by other vegetation types. The widely spaced nodes and lateral growth forms of *F. spiralis* and *S. sericeus* catalyse the loose accumulation of sand (McGuirk et al., 2022; Wardle, 1991). Conversely, exotic sandbinders, herbaceous species, and woody species occurring on active dunes have more clearly vertical root structures that, once established, promote sand stabilisation and soil formation (MacGuirk et al., 2022). The differences in the spatial pattern of vegetation patches among plots dominated by different plant functional groups align with the international literature on plant functional traits as a driver of dune morphology (Biel et al., 2017, 2019, Gao et al., 2023; Johnson 1993; Hacker et al., 2012; 2019; Hesp, 1989; Jay et al., 2022; MacGuirk et al., 2022; Maun, 2009; Ruggiero et al., 2018; Van der Maarel et al., 1997; Zarnetske et al. 2012, 2015).

Measures of sand deposition on the beaches studied, dune geomorphology, coastal erosion and flooding are needed to confirm that the patterns found in this study are caused by the interaction of processes of disturbance and colonisation and facilitation, creating zones of patchy mosaics of vegetation of different ages and functional roles (zonation) (Walker & Zinnert, 2016), but this data does not currently exist nationally for Aotearoa. An alternative hypothesis to explain the patterns found is that *F. spiralis* and *S. sericeus* lack a functional morphology to capture sand effectively. While the traits of sandbinders in Aotearoa are not well known (Verhoeven et al., 2013), similarities exist in the morphologies of dunes built by *F. spiralis*, *S.sericeus* and the American beachgrass *Ammophila brevigulata* (Fern), which effectively captures sand in high sand deposition environments (Hacker, 2012; 2019). Since sandbinder traits have been shown to affect morphology (Biel et al., 2019; Hacker et al., 2012, 2019; Jay et al., 2022), some degree of similarity in the functional morphology of *A. brevigulata, F. spiralis,* and *S. sericeus* may exist.

Contrary to our expectations, the exotic sandbinder *A. arenaria* exhibited a range of spatial patterns, potentially due to interspersion with other cover classes. Moreover, sand cover was relatively high in plots dominated by *A. arenaria* (59%; SD = 11.79; Table 10). *A. arenaria* had a lower percent sand cover than *F. spiralis* dominated plots but higher than *S. sericeus* dominated plots. This result was surprising given the high density of tillers and vertical growth of *A. arenaria*, which typically produces denser cover (McGuirk, et al., 2022; Wardle, 1991). A reason for this could be because a small number of plots where *A. arenaria* dominated (*n* = 7 out of 135 plots) occur at an actively prograding beach (Himatangi Beach). It is possible that a wider variety of sand cover percentages could have been observed with a larger sample size. Alternatively, this pattern could be an example of a Lévy random walk strategy (Reijers et al., 2019). *A. arenaria* has been shown to use this strategy during early colonisation following disturbance, whereby it reproduces vegetatively in large steps away from its clonal base in order to maximise its sand trapping efficiency by accreting sand in multiple dense shoot patches, forming a patchy spatial pattern (Reijers et al., 2019). Patchy clonal expansion patterns may also be occurring at beaches where *F. spiralis* and *S. sericeus* occur reflected in the patchy spatial pattern of these vegetation types, but the traits of these species are understudied.

Our results lend support to our hypothesis that 1) the spatial pattern of foredune vegetation would vary across the seven nationally representative active dune vegetation types identified in Chapter 3, 2) vegetation in plots dominated by native sandbinders would exhibit greater patchiness and edginess, suggesting interactions between processes of colonisation and natural disturbance; and 3) that processes of disturbance could be further indicated from the percentages of bare sand in vegetation dominated by native sandbinders, compared to vegetation types dominated by other growth forms, thus also indicating the mobility of active dunes. However, the conclusions that can be made about 2) and 3) are limited and measures of abiotic variables such as supply to beaches, dune geomorphology, coastal erosion and flooding are needed to confirm these hypotheses.

4.4.2. Dominance of spectral cover classes was highest and evenness was lowest in plots dominated by native sandbinder vegetation types

Where sandbinder cover classes prevailed within plots, we observed lower scores for the dominance metric, SIDI, reflecting the dominance of sandbinder cover classes. Similarly, low scores for the evenness metric, SIEI, reflected a less even distribution of all cover classes (Fig. 20; Table 11). Conversely, where non-sandbinder vegetation types prevailed, we observed lower dominance (higher scores) and greater evenness of cover classes (higher scores), potentially suggesting the plots occurred on foredunes with greater substrate stability where plants with vertical rooting systems can establish. However, plots dominated by the exotic pasture grasses vegetation type exhibited the greatest dominance and lowest evenness scores overall, likely due to the plots occurring within planted grass sward monocultures. However, the high variation across plots shows several outliers fell within this plot vegetation type.

The resolution of imagery selected in this study precluded the inclusion of small herbaceous species, such as those occurring in dune slacks and deflation hollows in this analysis. The ability to detect these species may have increased diversity values, and the ability to show evidence of facilitation processes, whereby increasing numbers of other species occur due to the geomorphic diversity caused through dune-building by sandbinders, for example, in dune slacks and hollows. Equally, this could have shown evidence of zonation, whereby mosaics of different functional vegetation types occur in predictable patterns aligned with the local disturbance regime (Castillo & Moreno-Casasola, 1986; Miller et al., 2010; Stallins et al., 2005; Zinnert et al., 2016). Thus, we recommend coupling aerial imagery with finer-scale fieldwork or UAV-based analysis with very high-resolution imagery (< 5cm) to better gauge the role of these plant species in dune vegetation diversity analyses.

These results lend support to our fourth hypothesis that vegetation types dominated by sandbinders would have greater dominance and lower evenness of cover classes in the classified imagery compared to vegetation types dominated by other growth forms, suggesting they have adaptations to colonise loose substrates and greater tolerance to stresses of the harsh foredune environment compared to other species. However, measures of species composition are needed to confirm results. Field campaigns following disturbance events would also verify if sandbinders are being destroyed by sand deposition, overwash, and then the removal of bare substrates.

4.4.3. Ficinia spiralis reference sites for restoration / conservation

The PCA on landscape metrics for the subset of plots where the *F. spiralis* cover class occurred (Fig. 23) showed that plots with a high per cent cover of *F. spiralis* had an aggregated, clumpy pattern. In contrast, plots with a low percentage cover of *F. spiralis* had a high patch and edge density. This could partly be explained by the contribution of woody vegetation at the plots with high per cent *F. spiralis* cover. The woody vegetation in these plots is considered native (Ryan et al., 2023); thus, the pattern at these beaches could serve as a reference to track progress towards restoration goals for sequences of native vegetation from incipient dunes to rear dunes in coastal forest in Aotearoa. Moreover, the high amounts of variation (78%) explained in the first two components of this PCA (Fig. 23) means this pattern provides a useful basis for an indicator of sites that have not been invaded.

4.4.4. Invasives modify the spatial pattern of S. sericeus within plots

In the PCA on landscape metrics for the subset of plots where *S. sericeus* occurred (Fig.25), vegetation in *S. sericeus* dominant plots varied more in size and had higher edge density, in line with the overall pattern for native sandbinders in the sand-vegetation binary raster. In contrast, where *S. sericeus* vegetation was present in plots dominated by invasive *A. arenaria* or co-dominant woody vegetation, *S. sericeus* had a low per cent cover and was more isolated with relatively higher patch density. This pattern suggests that competition from other species is limiting the spread of *S. sericeus* and reducing its abundance in these areas, which is consistent with invasion processes. The high amount of variation explained in the first two axes (67%) of this PCA (Fig. 25) means this pattern is a reasonable basis for an indicator of a pattern of invasion. These results support our fifth hypothesis that a pattern of aggregation and high per cent cover of invasive woody and exotic cover classes would exist where invasive species co-occur with native sandbinders, suggesting invasion processes are occurring.

However, where *S. sericeus*-woody vegetation dominated, vegetation had a wider variety of spatial patterns, including aggregated, spread out, large, irregular shaped patches. Because this vegetation type comprised native and exotic woody cover, native-dominated plots were not distinguishable from plots dominated by exotic woody invasives. This is relevant because the presence of native woody species could be encroachment, natural succession, or the presence of woody species of semi-stable dunes (Chapter1 & 3). Previous studies from the international literature have shown that beaches less modified by human activity contain elongated and irregular shaped vegetation patches, in contrast to compact and rectangular shapes of environments with greater anthropogenic disturbance (Carboni et al., 2009; Cicarelli & Barcoro, 2016; Malavasi et al., 2013; 2016). Further work is warranted to differentiate between woody species in classified imagery as a precursor to input and calculation of landscape metrics to distinguish patterns of native and exotic woody invasion.

4.4.5. Relationships exist between human activity and spatial pattern

Our conditional inference tree model results suggest that the global predictor variables of ecosystem extent (the area covered by active dunes that the sampled plots occurred in), followed by the human population, wind and rain, had the largest overall influence on the spatial pattern of vegetation within plots, across all principal components modelled. The international literature on dune mobility indices (Lancaster, 1988) and geomorphology supports the importance of climatic influences (Lancaster, 1988; Laporte-Fauret, 2021; Ryu & Sherman, 2014). Given the extent (80.5%) to which active dunes have declined due to human activities in Aotearoa (Hilton et al., 2000), the importance of ecosystem extent and human population suggested by the model seems likely. However, quantification of geomorphic processes and historical land management is also needed to confirm the relative importance of these variables.

The individual tree models (trees) showed that interactions among explanatory variables produced different results for each of the principal components. However, a common theme was that aggregated vegetation patterns occurred where human variables were strongest. This was particularly evident in interactions between modified adjacent land cover, higher human population and smaller ecosystem extent, producing more aggregated woody and exotic vegetation patches. In contrast, where geomorphic variables such as wind and waves were stronger, patchy, edgy or irregular shaped vegetation, spatial patterns occurred and were most commonly associated with plots dominated by native sandbinder vegetation types.

For example, in the tree for PC1 of the sand-vegetation binary landscape metrics (Fig. 27), the interaction between low population and an adjacent cover of sand (suggesting a large extent) resulted in a spatial pattern of high edge and patch density associated with *F. spiralis* and *S. sericeus* vegetation types (Fig. 27). An adjacent land cover of sand and native sandbinders in a patchy distribution suggests dune mobility and processes of disturbance and primary succession. In contrast, another branch of the tree shows that the interaction between an adjacent land cover other than sand (suggesting a smaller extent) with low mean ambient wind resulted in aggregated woody and exotic vegetation types.

In an example from the first component of the PCA of the subset of *S. sericeus* cover classes, the tree for PC1 (Fig. 31) showed that where median income was lowest, the percentage cover of *S. sericeus* was greater, and *S. sericeus* dominated. In this example, we considered median income as a proxy for resources available for coastal development; thus, a lower median income suggests lower levels of human activity, resulting in *S. sericeus* dominated vegetation types.

We observed that our conditional inference tree model results were more similar for plots that occurred on the same beach, which was expected given the inherent spatial autocorrelation in our design given the one-to-many relationship of plots to beaches. An alternative could have been to use a mixed model approach with the beach as the random effect; however, we wanted to draw out the differences between beaches and the different vegetation types or cover classes that occur on beaches. Additionally, although the amount of variation in the data explained by the models varied (26% - 76%), given the complex interactions between ecological variables, as well as a paucity of data, we argue that these results are a valuable steppingstone in quantifying the impact of human activities on active dune processes. Thus, our results broadly support our sixth hypothesis that relationships exist between human activities and the spatial pattern of vegetation cover classes in dune systems heavily used by people, whereby vegetation patches are more aggregated, suggesting human activities affect ecogeomorphic processes.

4.4.6. Conclusions

We conclude that the analysis of the spatial pattern of vegetation within plots on active dunes has high potential as an indicator of the ecomorphological processes that underpin ecosystem structure and function, showing variation in condition across beaches at a national scale. This research has shown that plots dominated by each of the seven nationally representative vegetation types on active dunes in Aotearoa had distinct spatial patterns of vegetation. We identified spatial patterns that are potentially indicative of interactions between natural disturbance and colonisation processes, whereby plots dominated by native sandbinder vegetation types exhibited greater vegetation patchiness, edginess, and percentages than plots dominated by other growth forms. Beaches with plots dominated by native sandbinder types also had greater percentages of bare sand (≥ 56%) compared to beaches with plots dominated by other growth forms (≤ 40%), which aligns with research by others on thresholds of mobility on active dunes (Konlechner et al., 2015; Ryu & Sherman, 2014), thus distinguishing native sandbinders as potential indicators of mobility in active dunes. Processes of colonisation by sandbinder species were potentially indicated from measures of dominance and evenness of spectral cover classes since plots dominated by sandbinder vegetation types had greater dominance and were less even than plots dominated by other growth forms, suggesting they could be colonisers of disturbed substrates. Invasion processes were also potentially indicated from an aggregated spatial pattern of vegetation within plots and a high per cent cover of exotic and woody cover classes, where they co-occurred with *S. sericeus*; in addition, potential reference sites for non–invaded *F. spiralis* dominant beaches were identified (Coal River and Smoky Beach). We also showed that relationships potentially exist between human activities and the spatial patterns of vegetation within plots, whereby vegetation patches are more aggregated, potentially inhibiting eco-geomorphological processes. Higher resolution input imagery datasets are likely to detect greater diversity values of spectral classes, bringing insights about facilitation and zonation processes. However, given the lack of datasets at a national scale for key geomorphological drivers such as sand supply, geomorphological processes such as sand deposition, coastal erosion and flooding, the conclusions that can be made are limited and further research should seek to quantify these variables. Additionally, the collection of species composition data would confirm colonisation and invasion processes and detailed information about land management history at study beaches could help quantity the extent of human influence at study beaches. Despite these limitations, overall, this research suggests that coarse indicators of complex ecogeomorphological processes and their interactions with human activity can be detected at large scale from spatial patterns derived from low-cost aerial imagery.

5 General Discussion

This thesis aimed to investigate the question: Can the ecosystem condition of active dunes be discerned from remotely sensed imagery, specifically aerial imagery at a national scale, and investigated how aerial imagery could be used to determine the components of ecosystem structure and function and therefore condition. In Chapter 2, the potential maximum extent and the extent of a representative sample of active dunes were estimated from large-scale geospatial datasets and high-resolution aerial imagery. A method to construct an index, or gradient of condition, was also developed to identify where remotely sensed imagery could be of most use within a framework of ecosystem integrity. In Chapter 3, vegetation was classified from the imagery, and a national vegetation typology for active dunes was created. In Chapter 4, indicators of active dune condition were derived from vegetation types and spatial pattern, grounded in the literature on ecological processes. This research is of national significance since it is the first estimate of the national extent of active dunes in Aotearoa in 15 years, the first national assessment of the condition in ~ 30 years (Ryan et al., 2023), and the only one assessing condition using quantitative methods through the development of indicators of ecosystem condition. A highly useful result was that the native sandbinders, *F. spiralis* and *S. sericeus*, could be discerned and classified from aerial imagery and were quantitative indicators of active dune condition. This work is of international significance because it provides a framework and method to identify indicators of active dune condition that can be trialled and refined in any country with sandy beaches, irrespective of size, since the drivers of active dune condition are ubiquitous globally. In this final chapter, the main advances in knowledge emerging from the body of work, as they pertain to the research questions in section 1.8, are drawn out, and the study's limitations are described (Section 5.1). Potential avenues for future research are then put forward (section 5.2), and conclusions are made (section 5.3).

5.1. Advances in knowledge and limitations of the research

Chapter 2

Main advances in scientific knowledge

Chapter 2 aimed to answer the question, “How can remotely sensed imagery be best utilised to determine active dune condition?” In response, the research identified a suitably representative sample of beaches from a large-scale ecological and geospatial analysis and constructed an index using the System of Economic and Environmental Accounting – Ecosystem Accounting (SEEA-EA) to compare the ecosystem integrity of active dunes. Gaps in the data that remote sensing techniques could fill were identified.

Prior to this research, the SEEA-EA method to create an index of ecosystem condition had not been trialled on active dune ecosystems. Despite data gaps and constraints, it was successfully implemented and tested to create a narrative of the state of ecosystem integrity of active dunes at a national scale. The SEEA-EA method, incorporating abiotic and landscape factors, correctly identified sites at the high end of the ecological integrity gradient and produced finer scale scoring for sites at the lower end of the gradient, but mid-range sites were less well distinguished, mainly due to a lack of information about ecosystem structure such as per cent cover of different vegetation types, or cover classes. Thus, a strength of this internationally accepted, albeit new SEEA-EA index construction method was that it identified a gap that remote sensing techniques are well placed to fill. Another strength was the development of a country-independent method to identify the potential maximum extent, a nationally representative sample and the extent of active dunes from large-scale geospatial and ecological datasets and high-resolution aerial imagery.

Limitations / critique

Limited existing national datasets

Long-term datasets related to active dune extent and condition are rare globally (Schlacher et al., 2008, Holdaway et al., 2012). Like many countries, datasets on human and environmental variables in Aotearoa are scarce at local (beach) and national scales (Farrell et al., 2021). The challenges identified in Chapter 2 are not unique and they are reflected in those identified in similar studies attempting to implement the SEEA-EA and other frameworks of ecosystem integrity and extent around the world, namely, data gaps and the lack of sufficiently regular temporal or spatial data (Holdaway et al., 2012; Farrell et al., 2021).

A lack of data limited the parameters that we could use in the SEEA-EA assessment. In some cases, proxy datasets were used in their place; for example, extreme wave event data, a geomorphic agent of geomorphic disturbance, was used as a proxy for sand deposition since foundational datasets relating to sediment transport do not exist for Aotearoa (for example sediment transport cells around the coast and deposition rates) (Stephens, S., personal communication). The use of proxies for geomorphic disturbance from sand deposition necessarily occurred across all of the data chapters. Nonetheless, results showed a coarse indication of condition that aligned with our expectations of ecosystem condition nationally. More direct measures of disturbance from sand deposition at the beach level are needed to confirm results.

Another limitation was that, due to the lack of existing national datasets, data from different studies and their methods were necessarily compared to each other, which can lead to issues around incompatibility and lack of standardisation (Krebs, 2014). For example, although the approach to digitisation methods used in Chapter 2 were the same as Hilton et al. (2000), image quality and resolution was different, and the details of individual decisions about the placement of boundaries were different. Although it is impossible to quantify the impact of different approaches in dynamic ecosystems, they will have a bearing on results.

Chapter 3

The main question to answer in Chapter 3 was, "What are the capabilities and limitations of aerial imagery for characterising the vegetation of active dunes?". To investigate this, vegetation across the representative sample beaches was classified, and vegetation types were developed from the classification data. The potential drivers of variation in vegetation cover types were also explored.

Main advances in scientific knowledge

A national typology of active dunes

An important result from this chapter was that native sandbinders could be accurately discerned and classified from RGB aerial imagery and were distinct from other cover classes. The discernment of native sandbinders was an ambitious target, similar to sandbinder plants the world over, they are narrow compared to the image resolution, although density does vary. A nationally representative typology was then derived using these classified cover classes and hierarchical agglomerative clustering techniques from phytosociology, comprising seven vegetation types, including two native sandbinder vegetation types. This was the first time a national typology of the vegetation of active dunes was undertaken using quantitative methods in Aotearoa and the first time from aerial imagery that we know of from the international literature. This part of the thesis laid the foundation for the subsequent characterisation of the spatial pattern of the vegetation of active dunes in Chapter 4.

Limitations / critique

National scale imagery selection

This project aimed to determine if RGB aerial imagery could be used to develop indicators of active dunes for monitoring. While this has proved feasible, and the sub-metre resolution of the aerial imagery used was necessary for typology and subsequent indicator development, the asynchronous nature of aerial imagery data capture across study sites meant that training datasets and supervised classification had to be done at an individual beach level, since conditions differ per aerial mission. Moreover, RGB band boundaries overlap (Lillesand et al., 2015), meaning precise spectral profiles could not be collected. Combined, these things mean a spectral library of active dune vegetation signatures could not be collected and used for future monitoring. In addition, the most recent imagery ranged between two and thirteen years old; thus, results for some sites with older imagery are already out of date, given the dynamic nature of active dunes.

High-resolution, high-frequency satellite imagery that can capture large swathes of the earth's surface is an alternative data source that was considered at the early stages of the thesis since it can provide synchronous, multispectral national areas coverage and remedy some of these issues. Sentinel 2 satellite imagery (European Space Agency) has a resolution of 10 m^2 and was trialled. However, it was found too coarse to discern sandbinder vegetation types in training due to their sub-metre size. Thus, the value added from additional bands and derived indices could not be leveraged for classification. The current generation of satellites providing imagery at 1 m^2 or 2 m^2 resolution (e.g. Pleiades Imagery from Airbus Defence and Space or BlackSky imagery from BlackSky Global) was also considered but found to be too expensive while also being too coarse for indicator development for the same reason. However, once a spectral signature is known, current high-resolution imagery could enable the extrapolation of typologies and indicators nationally. It would also allow for national coverage and application to new sites of results and an up-to-date record of vegetation types at sites, including the discovery of new active dune sites not captured using the geospatial datasets listed in Chapter 2. The resolution of satellite imagery will increasingly improve, and going forward, access to high-resolution satellite imagery should be considered for testing and monitoring purposes. Bands such as shortwave infrared could identify water held in dune slacks, an endangered ecosystem, and enable insights into how these related ecosystems interact with active dunes.

Classification of higher density species and growth forms

While sandbinders native to Aotearoa were able to be discerned and classified amongst a background of sand, this may be more difficult than target sandbinder species that grow in higher densities, for example, *Ammophila brevigulata* (Fern) in the Pacific Northwest U.S.A, or *Ammophila arenaria* (L.) Link inside its natural range of Northern Europe. Surprisingly, *A. arenaria* dominated only seven of the 135 plots, given its comparatively aggressive traits (Murphy et al., 2012; Verhoeven et al., 2013). This could be due to the sample of beaches, which were stratified for a range of conditions and geographic contexts, rather than invasion by *A. arenaria* (Chapter 2). Additionally, *A. arenaria* was often locally misclassified with exotic pasture grasses and was interspersed with other cover types at the plot scale. It was also surprising that *F. spiralis* and *A. arenaria* did not occur together in any of the plots. While *F. spiralis* dominant vegetation occurred at beaches in high conservation protection, we know from other studies (Case et al., 2019) that *A. arenaria* occurred on at least one beach (Kaitorete Spit), although not in high amounts due to weed control. The resolution of the imagery was potentially insufficient to discriminate between *F. spiralis* and *A. arenaria*. In contrast, there was more distinction between marram and *S. sericeus* where these two species co-occurred. Future efforts could consider a range of *A. arenaria* distributions to explain the interactions between *A. arenaria* and other plant species that occur on active dunes better. Equally, it was not possible to identify individual woody shrub and tree species, including woody species that loosely accrete sand, for example, *Pimelia villosa* Sol. ex Sm. (Wardle, 1991), which are features of semi-stable dunes, or the continuum of stability in natural dunelands (Cockayne, 1911; Wardle, 1991), rather than stable dunes. As the resolution of aerial and even satellite imagery increases, this issue can be expected to decrease.

Choice of classifier

The Support Vector Machine (SVM) classifier is a mature object-based classical learning classifier well suited to defining boundaries between fuzzy classes, such as vegetation in backgrounds of natural landscapes (Blaksche, 2010). While a limited range of classifiers were investigated to determine an optimal solution that balanced ecological interpretation and classifier optimisation needs, further research into the classifier choice was not deemed to add sufficient value to the project to warrant additional time on this. Future projects could consider the benefits of artificial neural networks to improve extrapolation to new imagery, an exciting development in the ability to monitor active dune landscapes at a large scale (Pettorelli et al., 2024).

Visualisation / communication of output

Due to national scale sampling and a plot-based approach, the geospatial outputs of research were classified imagery for 1 ha plots on beaches around a more than 15,000 km coastline. The classified plots are too small to be seen when viewed in the context of the whole country or even large beaches. While the visual communication of results was not a focus of this research, enhancing it could improve the dissemination of the results to a broader audience. One approach could be to create an animation of the classified plots and beaches or an interactive web-based product like an ArcGIS Story Map (Esri, 2024).

Chapter 4

The main question to answer in Chapter 4 was, “What are the indicators of active dune structure and function that can be derived from remotely sensed imagery?”. The research in this Chapter characterised the spatial pattern of the seven national vegetation types developed in Chapter 3. It related them to the literature on ecological processes, structure and function to identify indicators of ecosystem conditions. The drivers of variation in spatial pattern across the different vegetation types were also explored.

Main advances in scientific knowledge

Spatial pattern of different vegetation types

Spatial pattern, or the spatial structure of an ecosystem, has a reciprocal relationship with ecological processes and function (Forman, 1995; Turner, 1989; Uuemaa, 2009; Watt, 1947) and underpins ecosystem condition (Chapin III, 2011). Thus, the spatial patterns in vegetation observed on active dunes in this thesis could contribute to knowledge about active dunes' structure, function and condition. In Chapter 4, vegetation on dunes dominated by the seven national vegetation types was shown to have distinct spatial patterns, including patterns known to occur in ecosystems with high levels of natural disturbance and subject to anthropogenic influence (Carranza et al., 2010; Cicarelli & Barcoro, 2016; Corenblit et al., 2015; Kumar et al., 2006; Garcia-Mora et al., 1999; Malavasi et al. 2013, 2014, 2016; Marzialetti et al., 2024; Miller et al., 2010; Pinto et al., 2023). For example, the spatial pattern of vegetation dominated by native sandbinders had higher patch density and edge density, which suggests interacting processes of disturbance and colonisation. Beaches dominated by sandbinder vegetation types corresponded with beaches at the high (better condition) end of the gradient of condition (Chapter 2). In contrast, the spatial pattern of vegetation dominated by exotic and woody vegetation was more aggregated, and this pattern was related to anthropogenic influences. Beaches dominated by these vegetation types corresponded with beaches at the lower (degraded) end of the gradient of condition (Chapter 2).

Indicators of active dunes

Vegetation type

Results showed that for dunes dominated by native sandbinders, the mean per cent cover of sand (56% in *F. spiralis* and 59% in *S. sericeus*) was consistent with patterns reported in the literature on thresholds for active cover types on active dunes (Konlechner et al., 2015; Ryu & Sherman, 2014), suggesting that dominance by native sandbinders was an indicator of active dune processes. The significance of these results is that there is potential to estimate the extent of active dunes based on vegetation type at a large scale using the methods in this thesis. Cost savings from semi-automated classification and analysis of low-cost aerial imagery with landscape metrics make this method viable compared to manual, field-based baseline measurement and ongoing monitoring. Practically, monitoring of ecosystem extent is required under obligations of the 2022 Global Biodiversity Framework, of which 188 countries are signatories (United Nations Environment Programme [UNEP], 2022a). Additionally, ecosystem extent provides the basis for calculating species occupancy and habitat mapping indicators. Moreover, the per cent cover of bare sand within vegetation dominated by sandbinders is an intuitive metric easily communicated to the public and other stakeholders. However, measures of geomorphic drivers of active dunes, such as the supply of sand and sediment deposition, are required to confirm that active dune processes are also occurring at beaches where these vegetation types and thresholds of bare sand occur.

Pattern of invasives

Invasive *A. arenaria* was observed to modify the spatial pattern of the *S. sericeus* cover class, with a lower per cent cover of *S. sericeus* where they co-occurred within plots, suggesting competition from this invasive species. The spatial pattern of plots dominated by woody vegetation also showed a lower per cent cover of *S. sericeus* where these cover classes co-occurred and showed greater cover class diversity, again suggesting competition. Where woody vegetation was co-dominant with *S. sericeus*, a range of vegetation spatial patterns were observed that were distinct from sandbinder vegetation. Encroachment by woody species suggests dune stabilisation is occurring (Charbonneau et al., 2017, 2023; Charbonneau & Wootton, 2020; Gadgil, 2001). Practically, this spatial pattern could be monitored at a large scale with remotely sensed imagery using the methods in this thesis, facilitating prioritisation of sites for weed control. However, further work is needed to distinguish between native and exotic woody vegetation cover classes during imagery classification, likely requiring imagery of less than 5 cm (Chapter 3).

Limitations / critique

Dominance and evenness of spectral cover classes

Vegetation dominated by sandbinders had higher spectral dominance and lower spectral evenness in cover class composition than other vegetation types studied, potentially reflecting processes of colonisation by sandbinders. However, the resolution of the imagery selected was too coarse to differentiate among species with more aggregated growth forms and small herbaceous species, limiting what can reasonably be concluded about diversity where these growth forms occur. Future research could include strategic, repeat field sampling or using very high-resolution imagery captured by UAVs to better understand the extent to which non-sandbinder species are changing composition and, therefore, ecosystem processes, such as colonisation, facilitation and zonation in active dunes. Field campaigns at the beach level could also gather data about whether rare species occur.

Abiotic processes

Limited data about geomorphological disturbance, such as sand deposition on beaches, constrains the conclusions about the relationship between spatial patterns and ecosystem processes, specifically interactions with abiotic processes. Due to the national scale of the research, field data collection was outside the scope of this thesis since data about sand supply to beaches must be collected at beach scale over repeated field campaigns in a range of conditions with specialised equipment. It would not have been possible to collect this data and also complete the other parts of the thesis. Morphometric data, such as beach width and dune profiles (Hacker et al., 2012, 2019; Konlechner & Hilton, 2022; Rose & Martano, 2023), could also provide data about sand supply and must be collected at the individual beaches through field campaigns. Alternatively, it could be derived from analysing high-resolution LiDAR or UAV imagery collected over several aerial campaigns to build a picture of the sand supply to sites. While LiDAR data is now publicly available for most of Aotearoa, there is typically only one collection date for each region.

Shoreline change rate data indicates sand supply to beaches (Biel et al., 2019; Jay et al., 2021) and can be assessed by analysing historical aerial or high-resolution satellite imagery (Moore et al., 2000). Attempts were made to do this at the site level in the early stages of this thesis, although the processes feeding into shoreline change are not well understood in Aotearoa. Similarly, coastal hazard information such as total water level and erosion at the coast was unavailable for the beaches and periods studied.

While the absence of data about sand supply, sand deposition and species composition limits the conclusions that could be made about spatial patterns in Chapter 4, numerous studies have demonstrated close alignment between the results of landscape metric studies and ecological field campaigns internationally (Ueema et al., 2009), albeit not in active dune ecosystems. Further field research to verify results should be completed so these indicators can be operationalised in a low-cost approach to providing data on active dune processes and condition. University research and Citizen Science data could be considered for this purpose. Thus, the aerial imagery analysis presented in this thesis has identified indicators of the condition of active dunes by characterising the vegetation type and characteristic spatial patterns of vegetation on active dunes that can potentially be related to ecological and anthropogenic processes. These results have laid the foundations for and given rise to new questions for future research, for which this research is a stepping stone.

5.2. Next steps and future research

The indicators of active dune condition identified in this research could be deployed immediately for testing and refining, for example, within the New Zealand Department of Conservation's Tier 1 broad scale monitoring for the national context, to identify baseline condition and set the stage for future research questions about temporal change. The results could inform the strategic design of sampling, for example, given active dunes are naturally rare ecosystems (Williams et al., 2007), monitoring design could be a stratified sample within the potential national extent identified in Chapter 2. Sampling could be further stratified by the seven national vegetation types and related dune landforms identified in Chapter 3, to give spatial definition for spatial monitoring, which in turn, aligns with the emphasis on spatial planning in the 2022 Global Biodiversity Framework (UN

Environment Programme, 2022a). Historical imagery could be used to test the approach. Potentially, these methods are extensible to other rare ecosystems with similar, sparse, low-growing vegetation and that are vulnerable to encroachment by invasive species, for example alpine and estuarine ecosystems.

New questions that can be investigated include the rate of decline in ecosystem extent since the rate of decline is a predictor of the trajectory of ecosystem collapse (Keith et al., 2013). Monitoring of extent is required to identify the frequency of change in these dynamic systems, causes, and how often they should be monitored to detect catastrophic declines or progress towards restoration goals and inform policy interventions. For example, the rate and quantity of invasion of *A. arenaria* and encroachment by woody vegetation could be compared against changes in the spatial pattern of native sandbinder dominant vegetation. Other related questions could be the impact of increasing frequency of disturbance from storms on the spatial pattern of vegetation due to climate change, and progress towards restoration goals due to conservation effort. In this scenario, potential restoration goals could be the recreation of the natural spatial and temporal heterogeneity of active dunes under natural disturbance and primary succession (for example through dune notching', see Laporte-Fauret et al., 2021) and monitoring with the indicators given in this thesis. Complementary research to build upon the results of Verhoeven et. al (2012) into the plant functional traits of native and exotic active dune species could enhance understanding of the resilience of indigenous active dune species to climate change, invasions, and human pressures for conservation outcomes.

Another area of future research is the extent of indigenous dominance in active dunes. Whilst the key native active cover types of *F. spiralis* and *S. sericeus* could be discerned to species in imagery classification, more aggregated growth forms, such as trees, shrubs, *A. arenaria* and small herbaceous species in dune slacks, posed challenges due to the resolution of the imagery selected; thus, dominance by indigenous or exotic species could not be quantified. Examples where identification to species level in more aggregated growth forms has been achieved involve the use of very high-resolution imagery (≤ 5 cm) in conjunction with multispectral or LiDAR data and field sampling (Bakacsy et al., 2023; Bilkey, 2022; Case et al., 2019; Hantson et al., 2012; Laporte-Fauret et al., 2020; Wolff et al., 2023). Therefore, strategic deployment of UAV and plot sampling within a Tier 1 framework is recommended to develop a method to discern and classify these groups to species level.

Finally, there is an opportunity to deploy the indicators within a monitoring design that is interoperable with international risk assessment reporting frameworks, such as the IUCN Red List of Ecosystems, which is required under the 2022 Global Biodiversity Framework (UNEP, 2022b); the IUCN Green status of Species (Grace et al., 2021); and the Ecosystem Biodiversity Variables (Pereira et al., 2013). Chapter 2 gives an example of how the international SEEA-EA framework (United Nations et al., 2021) could be applied to active dune ecosystems and an index was also constructed to compare beaches along a gradient of condition based on ecosystem integrity and the SEEA-EA method (United Nations et al., 2021). The tool successfully teased out beaches at the high and low ends of the gradient; however, the central part was clustered, mainly due to the lack of data on ecosystem structure, hence providing the rationale for using remote sensing in this thesis. The gradient of condition is a useful tool for conveying ecosystem condition across beaches and over time and could be repurposed for future reporting needs across the different international ecosystem risk assessments.

5.3. Conclusion

This research has contributed towards understanding how to, and the utility of, assessing the condition of active dunes from remotely sensed imagery. Methods were successfully developed to identify indicators of active dune condition grounded in ecological theory and scientific principles, from publicly available, RGB high-resolution aerial imagery. Methodological challenges were also identified, solutions were suggested to address the issues, and recommendations were made to confirm results of this research and implement the results within a framework for broad-scale monitoring of the national context. This research provides a foundation and stepping stone to answer new questions about exciting future research directions outlined here, with the goal to improve understanding about how to be better kaitiaki (guardians) of active dunes in an increasingly complex and uncertain world.

References

Acosta, A., Carranza, M., Izzi, C. (2009). Are there habitats that contribute best to plant species diversity in coastal dunes? *Biodiversity and Conservation*. *18*. 1087-1098. 10.1007/s10531-008-9454-9.

Andreasen, J.K., O’Neill, V.O., Noss, R., Nicholas C Slosser, N.C. (2001). Considerations for the development of a terrestrial index of ecological integrity. *Ecological Indicators*, *Volume 1, Issue 1*, Pages 21-35.

Atkinson I.A.E. (1985). Derivation of vegetation mapping for an ecological survey of Tongariro National Park, North Island, New Zealand. *New Zealand Journal of Botany*, *23*: 361-378.

Auckland Botanical Society. (2005). Field Trip: Coromandel Peninsula, Auckland Anniversary Weekend 28/01/05 to 1/02/05. *Auckland Botanical Society*. Retrieved October 28th, 2022, from https://bts.nzpcn.org.nz/articles/field-trip-coromandel-peninsula-auckland-anniversary-weekend-28-01-05-to-1-02-05/

Auckland Botanical Society. (2013). *A visit to the Whatipu Sands, West Auckland*. Auckland Botanical Society. Retrieved October 28th, 2022, from https://bts.nzpcn.org.nz/articles/a-visit-to-the-whatipu-sands-west-auckland/

Baddeley, A., Rubak, E., & Turner, R. (2016). *Spatial point patterns: Methodology and applications with R / Adrian Baddeley, Ege Rubak, Rolf Turner*. CRC Press.

Bakacsy, L., Tobak, Z., van Leeuwen, B., Szilassi, P., Biró, C., & Szatmári, J. (2023). Drone-Based Identification and Monitoring of Two Invasive Alien Plant Species in Open Sand Grasslands by Six RGB Vegetation Indices. *Drones*, *7(3), 207*. MDPI AG.

Barbier, E.B., Hacker, S.D., Kennedy, C.J., Koch, E.W., Stier, A.C., Silliman, B.R. (2011). The Value of Estuarine and Coastal Ecosystem Services. *Ecological Monographs*, *Vol. 81, No. 2*, pp. 169-193, 2011.

Baye, P. R. (1990). *Comparative Growth Responses and Population Ecology Of European And American Beachgrasses* (*Ammophila* spp.) *In Relation To Sand Accretion And Salinity*. Doctoral Thesis, Western University. Digitised Thesis. 1986. https://ir.lib.uwo.ca/digitizedtheses/1986

Belcore, E., Latella, M., Piras, M., & Camporeale, C. (2024). Enhancing precision in coastal dunes vegetation mapping: ultra-high resolution hierarchical classification at the individual plant level. *International Journal of Remote Sensing*, *45(13*), 4527–4552. https://doi.org/10.1080/01431161.2024.2354135.

Bell, S., Gibb, J.G. (1996). *Public access to the New Zealand coast: guidelines for determining legal and physical constraints*. New Zealand Department of Conservation.

Bellingham, B., Richardson, S., Burge, O., Wiser, S., Fitzgerald, N., Clarkson, B. (2021). *Standardised methods to report changes in the ecological integrity of sites managed by regional councils*. Contract Report: LC3903 prepared by Landcare Research / Manaaki Whenua, and Kevin Collins, Collins Consulting for Hawke's Bay Regional Council.

Bellingham, P. (2001). *Evaluating methods for the Protected Natural Areas Programme.* Science and Research Internal Report 190. Science, 32.

Benítez-López, A., Alkemade, R., & Verweij, P. A. (2010, June). The impacts of roads and other infrastructure on mammal and bird populations: A meta-analysis. *Biological Conservation*. https://doi.org/10.1016/j.biocon.2010.02.009

Bennett, V. J. (2017). Effects of Road Density and Pattern on the Conservation of Species and Biodiversity. *Curr Landscape Ecol Rep 2*, 1–11 https://doi.org/10.1007/s40823-017-0020-6

Bergin, D. (2005). *Establishment of spinifex planting trial Paraparaumu Beach, Kapiti Coast.* CSIRO and SCION, Ensis, Rotorua, New Zealand. Retrieved October 28th, 2022 from https://takutaikapiti.nz/wp-content/uploads/2020/11/Establishment-of-spinifex-planting-trial_Paraparaumu-Beach_Bergin_Ensis_2005.pdf

Bergin, D. (2011a). *F. spiralis, golden sand sedge - sustainable harvesting of F. spiralis for weaving.* Technical article no. 7.5. Dune Restoration Trust of New Zealand. Scion, Rotorua, New Zealand.

Bergin, D. (2011b*). Spinifex, Spinifex sericeus – ecology, habitat and growth. Technical article no. 7.1.* Dune Restoration Trust of New Zealand. Scion, Rotorua, New Zealand.

Biel, R. G., Hacker, S. D., Ruggiero, P., Cohn, N., Seabloom, E.W. (2017). Coastal protection and conservation on sandy beaches and dunes: context-dependent tradeoffs in ecosystem service supply. *Ecosphere 8(4)*: e01791.10.1002/ecs2.1791.

Biel, R., Hacker, S. Ruggiero, P. (2019). Elucidating Coastal Foredune Ecomorphodynamics in the U.S. Pacific Northwest via Bayesian Networks. *Journal of Geophysical Research: Earth Surface. 124*. 10.1029/2018JF004758.

Biel, R.G., Hacker, S.D. (2021). Warming alters the interaction of two invasive beachgrasses with implications for range shifts and coastal dune functions. *Oecologia* (2021) *197*:757–770.

Bilkey, J. 2022. *Sand Dune Vegetation Monitoring and Assessment Using UAV Remote Sensing: A Case Study for Karekare Beach Auckland Region New Zealand*. Master's dissertation, Auckland University of Technology, New Zealand. Tuwhera. https://openrepository.aut.ac.nz/items/09542e83-74a4-4d3b-b318-793813d4e1b1

Bird, E.C.F. (1997). General Features of the Coastline. In E. Van der Maarel, (Ed). *Dry coastal ecosystems: General aspects Volume C* (pp. 1-10). Elsevier, Amsterdam.

Bird, E. C. F. (2008). *Coastal geomorphology: An introduction* (Second edition.). Wiley. Chichester, England.

Bivand, R., Pebesma, E. J., & Gómez-Rubio, V. (2013). *Applied spatial data analysis with R* (Second edition.). Springer.

Blaschke, T. (2010). Object based image analysis for remote sensing. ISPRS J Photogramm Remote Sens. ISPRS *Journal of Photogrammetry and Remote Sensing*.

Bland, L.M., Keith, D.A., Miller, R.M., Murray, N.J. and Rodríguez, J.P. (eds.) (2016). *Guidelines for the application of IUCN Red List of Ecosystems Categories and Criteria*, Version 1.0. Gland, Switzerland: IUCN. ix + 94pp

Bowman, W. D., Hacker, S. D. (2021). *Ecology*. 5th edition. New York, Sinauer Associates, Oxford University Press. ISBN 9781605359236.

Boyle, J. J. (2016). *Beach change in the Auckland region: current state and trends.* Auckland Council technical report, TR2016/04. Auckland Council, Auckland, New Zealand.

Breiman, L., Friedman, J. H., Olshen, R. A., & Stone, C. J. (1984). *Classification and Regression Trees.* Chapman and Hall.

Brown, E. A. (1978). Vegetation patterns across the Farewell Spit dune system. *Tane 24*:9-19.

Brown NZ Ltd. (2011). *Coromandel Peninsula Landscape Assessment, draft.* [see Landscape Unit 38, Ōtama Beach]. Prepared for the Thames Coromandel District Council. Retrieved October 28th, 2022, from https://www.tcdc.govt.nz/Global/1_Your%20Council/Council%20Projects/Current%20Projects/District%20Plan%20Review/Introduction.pdf

Brownett, J., Mills, R. (2017). The development and application of remote sensing to monitor sand dune habitats. *J Coast Conserv 21*, 643–656.

Bruun, P. (1988). The Bruun Rule of Erosion by Sea-Level Rise: A Discussion on Large-Scale Two- and Three-Dimensional Usages. *Journal of Coastal Research*, *4*(4), 627–648. http://www.jstor.org/stable/4297466.

Bryan, K., Coco, G. (2020). The response of sandy coastal systems to changes associated with sea level rise. In Hendtlass, C., Morgan, S., Neale, D. (Eds.) (2020). Coastal Systems & Sea Level Rise. What to look for in the future. *New Zealand Coastal Society*, 68 p.

Cai, W., Borlace, S., Lengaigne, M., van Rensch, P., Collins, M., Vecchi, G., Timmermann, A., Santoso, A., McPhaden, M. J., Wu, L., England, M.H., Wang, G., Guilyardi E., Jin, F. (2014). Increasing frequency of extreme El Nino events due to greenhouse warming. *Nature Climate Change 4*, 111-116 (2014).

Cai, W., Wang, G., Santoso, A., McPhaden, M.J., Wu, L., Jin, F., Timmermann, A., Collins, M., Vecchi, G., Lengaigne, M., England, M.H., Dommenget, D., Takahashi, K., Guilyardi, E. (2015). Increasing frequency of extreme La Nina events under greenhouse warming. *Nature Climate Change 5*, 132-137 (2015).

Cameron, E.K. (1989). Vegetation of the Whatipu Sands, north Manukau Heads. *Auckland Botanical Society Journal 44*: 3–10. Retrieved October 28th, 2022, from https://bts.nzpcn.org.nz/articles/vegetation-of-the-whatipu-sands-north-manukau-heads/

Carboni, M., Carranza, M., Acosta, A. (2009). Assessing conservation status on coastal dunes: A multiscale approach. *Landscape and Urban Planning. 91*. 17-25. 10.1016/j.landurbplan.2008.11.004.

Carrión-Klier, C., Moity, N., Sevilla, C., Rueda, D., Jäger, H. (2022). The Importance of Very-High-Resolution Imagery to Map Invasive Plant Species: Evidence from Galapagos. *Land 2022,* 11, 2026. https://doi.org/10.3390/land11112026.

Carranza, M. L., Carboni, M., Feola, S., & Acosta, A. T. R. (2010). Landscape-scale patterns of alien plant species on coastal dunes: The case of iceplant in central Italy. *Applied Vegetation Science, 13*(2), 135.

Case, B. S., Buckley, H. L., Fake, M., Bryan, S., Bilkey, J. (2019). Assessing the use of UAV-collected data for characterising the distributions and frequencies of sand dune vegetation cover types at Kaitorete Spit, Canterbury. *DOC Research & Development Series 359*, 25. Retrieved October 28th, 2022, from https://www.doc.govt.nz/globalassets/documents/science-and-technical/doc-research-and-development-series/drds359entire.pdf

Castillo, S. A., & Moreno-Casasola, P. (1996). Coastal Sand Dune Vegetation: An Extreme Case of Species Invasion. *Journal of Coastal Conservation, 2*(1), 13–22. http://www.jstor.org/stable/25098214.

Chapin III, F. S., Matson, P.A., Mooney, H. (2002). *Principles of Terrestrial Ecosystem Ecology*. 10.1007/978-1-4419-9504-9.

Charbonneau, B.R., L.S. Wootton, J.P. Wnek, J.A. Langley, and M.A. Posner (2017). A Species Effect on Storm Erosion: Invasive Sedge Stabilized Dunes More than Native Grass during Hurricane Sandy. *Journal of Applied Ecology 54*: 1385–94.

Charbonneau, B., Nicoletta, R., & Wootton, L. (2020). A decade of expansion of the invasive plant *Carex kobomugi* in a coastal foredune system. *Biological Invasions, 22*(6), 2099. https://doi.org/10.1007/s10530-020-02240-6.

Charbonneau, B., Swannack, T., & Piercy, C. (2023). The persistence and conversion of coastal foredune and swale vegetation community distributions 63 years later. *Ecosphere, 14*(1), https://doi.org/10.1002/ecs2.4329.

Cheng, H.D; Jiang, X.H, Sun, Y., Wang, J. (2001). Color image segmentation: advances and prospects, *Pattern Recognition, Volume 34*, Issue 12, 2001, Pages 2259-2281, ISSN 0031-3203.

Ciccarelli, D., & Bacaro, G. (2016). Quantifying plant species diversity in coastal dunes: a piece of help from spatially constrained rarefaction. *Folia Geobotanica, 51*(2), 129–141. http://www.jstor.org/stable/44983446.

Cieraad, E., Walker, S., Price, R., & Barringer, J. (2015). An updated assessment of indigenous cover remaining and legal protection in New Zealand's land environments. *New Zealand Journal of Ecology, 39* (2).

Coastal Restoration Trust. (2011). Restoration of Coastal Sand Dunes Using Native Plants. Technical Handbook. Practical guide for coastal communities adapting to climate change. *Coastal Restoration Trust of New Zealand.*

Coastcare Bay of Plenty. (n.d). *Dune Management in Ohope*. Coastcare Bay of Plenty. Retrieved October 28th, 2022, from https://cdn.boprc.govt.nz/media/30282/CoastCare-090527-RestorationOhope.pdf

Cockayne, L. (1911). *Report on the dune-areas of New Zealand: Their geology, botany and reclamation.* Department of Lands, Wellington.

Cockayne, L. (2011). The vegetation of New Zealand. *The Vegetation of New Zealand.* Cambridge University Press.

Coco, G., Bryan, K., Montaño, J., & Cagigal, L. (2020). Modelling coastal evolution for rising sea levels. In C. Hendtlass, S. Morgan, & D. Neale (Eds.), *Coastal systems & sea level rise: What to look for in the future* (pp. 1–68). New Zealand Coastal Society.

Corenblit, D., Baas, A., Bornette, G., Darrozes, J. Delmotte, S., Francis, R., Gurnell, A., Julien, F., Naiman, R. Steiger, J. (2011). Feedbacks between geomorphology and biota controlling Earth surface processes and landforms: A review of foundation concepts and current understandings. *Earth-Science Reviews. 106*. 10.1016/j.earscirev.2011.03.002.

Corenblit, D., Baas, A., Balke, T., Bouma, T., Fromard, F., Garófano-Gómez, V., . . . Walcker, R. (2015). Engineer pioneer plants respond to and affect geomorphic constraints similarly along water-terrestrial interfaces world-wide. *Global Ecology and Biogeography, 24*(11/12), 1363.

Czúcz, B., Keith, H., Maes, J., Driver, A., Jackson, B., Nicholson, E., Kiss, M., Obst, C. (2021). Selection criteria for ecosystem integrity indicators. *Ecological Indicators*. Elsevier B.V. https://doi.org/10.1016/j.ecolind.2021.108376

Dale, M. R. T., & Fortin, M. (2014). *Spatial analysis: A guide for ecologists.* Mark R.T. Dale, University of Northern British Columbia; Marie-José Fortin, Univesity of Toronto (Second edition.). Cambridge University Press.

David, A. S., Zarnetske, P. L., Hacker, S. D., Ruggiero, P., Biel, R. G., & Seabloom, E. W. (2015). Invasive congeners differ in successional impacts across space and time. *PLoS ONE, 10*(2), e0117283. https://doi.org/10.1371/journal.pone.0117283.

da Silva, G.M, Hesp, P., Peixoto, J., Dillenburg, S. (2008). Foredune vegetation patterns and alongshore environmental gradients: Moçambique Beach, Santa Catarina Island, Brazil. *Earth Surface Processes and Landforms. 33*. 1557 - 1573. 10.1002/esp.1633.

Dargie, T.C.D. (1995). Sand Dune Vegetation Survey of Great Britain: A national inventory. Parts 1 -3. JNCC, Peterborough.

Davidson-Arnott, R. (2010) *Introduction to Coastal Processes and Geomorphology*. Cambridge University Press. Cambridge, England.

de Lange, P.J, Rolfe, J.R., Barkla, J.W., Courtney, S.P., Champion, P.D., Perrie, L.R., Beadel, S., Ford, K.A., Breitwieser, I., Schönberger, I., Hindmarsh-Walls, R., Heenan, P.B., Ladley, K. (2017). *Conservation status of New Zealand indigenous vascular plants, 2018. NZ Threat Classification Series 22.* Department of Conservation, Wellington. NZ.

Delgado-Fernandez, I., & Davidson-Arnott, R. (2011). Meso-scale aeolian sediment input to coastal dunes: The nature of aeolian transport events. *Geomorphology, 126*(1–2), 217–232. https://doi.org/10.1016/j.geomorph.2010.11.005.

Dengler, J. (2009). Which function describes the species-area relationship best? A review and empirical evaluation. *Journal of Biogeography, 36*(4), 728–744. https://doi.org/10.1111/j.1365-2699.2008.02038.x

Department of Conservation. (2012). *Stewart Island / Rakiura Conservation Management Strategy and Rakiura National Park Management Plan* (pp. 1–316). Department of Conservation. Wellington, New Zealand.

Department of Conservation. (n.d). *Outcome Monitoring Framework (OMF).* Retrieved 20 November, 2023. https://www.doc.govt.nz/globalassets/documents/our-work/monitoring/omf-assessment-templates-intermediate-outcome-1.pdf

Department of Conservation. (2020). *Biodiversity Monitoring Aotearoa.* NZ Department of Conservation. Wellington. New Zealand.

Doing, H. (1985). Coastal Fore-Dune Zonation and Succession in Various Parts of the World. *Vegetatio, 61*(1/3), 65–75. http://www.jstor.org/stable/20146234.

Doody, J.P., ed. (1991). *Sand Dune Inventory of Europe*. Peterborough, Joint Nature Conservation Committee/European Union for Coastal Conservation.

Doody, J. (2005). Sand Dune Inventory of Europe. 45-54. Durán O., Moore, L.J. (2013). Vegetation controls on the maximum size of coastal dunes. *Proc Natl Acad Sci U S A*. doi: 10.1073/pnas.1307580110.

Eisma, D. (1997). Physiography of Beaches and Coastal Dunes. In E. Van der Maarel, (Ed). *Dry coastal ecosystems: General aspects Volume C* (pp. 11 - 28). Elsevier, Amsterdam.

Eriksson, O. (2000). Functional Roles of Remnant Plant Populations in Communities and Ecosystems. *Global Ecology and Biogeography, 9*(6), 443–449.

Esler, E. A. (1978). *Botany of the Manawatu*. Government Printer, Wellington, New Zealand. 206 pp.

Esri Inc. (2020). ArcGIS Pro (Version 2.8). Esri Inc. Retrieved October 28th, 2022, from https://www.esri.com/en-us/arcgis/products/arcgis-pro/overview.

Esri. Inc. (2021). *ArcGIS Pro* (Version 2.9). Retrieved from https://www.esri.com/en-us/arcgis/products/arcgis-pro/.

Esri. Inc. (2024). *ArcGIS Story Maps*. Esri. Inc. Retrieved October 2nd, 2024, from https://storymaps.arcgis.com/

European Environment Agency. (2024). *Natura 2000 Network Viewer*. Retrieved from https://natura2000.eea.europa.eu/expertviewer/#

European Nature Information System. (2021). *EUNIS habitat classification* (Version 2021). European Environment Agency. https://www.eea.europa.eu/themes/biodiversity/eunis/eunis-habitat-classification

European Space Agency. (2024). *Pleides*. Retrieved from https://earth.esa.int/eogateway/missions/pleiades

Farrell, C., Coleman, L., Kelly-Quinn, M., Obst, C., Eigenraam, M., Norton, D., O'Donoghue, C., Kinsella, S., Delargy, O., Stout, J. (2021). Applying the System of Environmental Economic

Accounting-Ecosystem Accounting (SEEA-EA) framework at catchment scale to develop ecosystem extent and condition accounts. *One Ecosystem. 6*. 10.3897/oneeco.6. e65582.

Forman, R.T.T. (1999). *Land Mosaics: The ecology of landscapes and regions*. Cambridge University Press, Cambridge.

Gadgil, R. L., Ede, F. J. (1998). Application of scientific principles to sand dune stabilization in New Zealand: Past progress and future needs. *Land Degradation and Development, 9*(2), 131–142. https://doi.org/10.1002/(SICI)1099-145X(199803/04)9:2<131:AID-LDR286>3.0.CO;2-H.

Gadgil., R.L. (2001). *A. arenaria* (*Ammophila arenaria) and coastal sand stability in New Zealand*. NZ Forest Research Institute, Rotorua, New Zealand.

Gao, J., Kennedy, D. M., & Konlechner, T. M. (2020). Coastal dune mobility over the past century: A global review. *Progress in Physical Geography, 44*(6), 814–836. https://doi.org/10.1177/0309133320919612.

Gao, J., Kennedy, D. M., Konlechner, T. M., McSweeney, S., Chiaradia, A., & McGuirk, M. (2022). Changes in the vegetation cover of transgressive dune fields: A case study in Cape Woolamai, Victoria. *Earth Surface Processes and Landforms, 47*(3), 778-792.

Gao, J., Kennedy, D.M. & McSweeney, S. (2023) Patterns of vegetation expansion during dune stabilization at the decadal scale. *Earth Surface Processes and Landforms, 1–15*. Available from: https://doi.org/10.1002/esp.5681.

García-Mora, M. R., Gallego-Fernández, J. B., & García-Novo, F. (1999). Plant Functional Types in Coastal Foredunes in Relation to Environmental Stress and Disturbance. *Journal of Vegetation Science, 10*(1), 27–34. https://doi.org/10.2307/3237157.

García-Novo, F., Díaz Barradas, M.C, Zunzunegui, M., García-Mora, M. R., Gallego- Fernández, J.B. (2004). Plant Functional Types in Coastal Dune Habitats. In: Martínez, M.L., Psuty, N.P. (eds) *Ecological Studies, Vol. 171*. *Coastal Dunes, Ecology and Conservation*. Springer - Verlag Berlin Heidelberg, 2004.

Gaudet, C. L., Keddy, P. A. (1995). Competitive Performance and Species Distribution in Shortline Plant Communities: A Comparative Approach. *Ecology*, *76*(1), 280–291. https://doi.org/10.2307/1940649.

Godoi, V., Bryan, K., Gorman, R. (2016). Regional influences of climate patterns on the wave climate of the southwestern Pacific: The New Zealand region. *Journal of Geophysical Research: Oceans,* 10.1002/2015JC011572.

Godoi, V., Bryan, K., Stephens, S, Gorman, R. (2017). Extreme waves in New Zealand waters, *Ocean modelling. 117*, 97 -110.

Godoi, V., Bryan, K., Gorman, R. (2018). Storm wave clustering around New Zealand and its connection to climatic patterns. *International Journal of Climatology. 38*, e401 - e417.

Goldwater, N. (2013). The botanical treasures of Middle Pakiri Beach. *Auckland Botanical Society*, 2013. Retrieved October 28th, 2022 from https://bts.nzpcn.org.nz/articles/the-botanical-treasures-of-middle-pakiri-beach-farm-pakiri/

González-Villanueva, R., Pastoriza, M., Hernández, A., Carballeira, R., Bao, R., & Sáez, A. (2023). Primary drivers of dune cover and shoreline dynamics: A conceptual model based on the Iberian Atlantic coast. *Geomorphology, 423*. https://doi.org/10.1016/j.geomorph.2022.108556.

Goodhue, N., Rouse, H., Ramsay, D., Bell, R., Hume, T., Hicks, T. (2012). *Coastal Adaptation to Climate Change: Mapping a New Zealand Coastal Sensitivity Index*. Prepared by NIWA for MBIE contract C01X0802.

Google Earth. (2024). *Google Earth*. https://earth.google.com/

Gorman, R. (2016). *Number of extreme wave events exceeding 4m in coastal regions, 2008–15. A report for the New Zealand Ministry for the Environment*. Dataset. Retrieved October 28th, 2022, from https://data.mfe.govt.nz/layer/53500-number-of-extreme-wave-events-exceeding-4m-in-coastal-regions-200815/

Grace, M. K., Akçakaya, H. R., Bennett, E. L., Brooks, T. M., Heath, A., Hedges, S., Hilton-Taylor, C., Hoffmann, M., Hochkirch, A., Jenkins, R., Keith, D. A., Long, B., Mallon, D. P., Meijaard, E., Milner-Gulland, E. J., Rodriguez, J. P., Stephenson, P. J., Stuart, S. N., Young, R. P., Acebes, P., … Young, S. (2021). Testing a global standard for quantifying species recovery and assessing conservation impact. *Conservation biology: the journal of the Society for Conservation Biology, 35*(6), 1833–1849. https://doi.org/10.1111/cobi.13756.

Grime, J.P. (2001). *Plant Strategies, Vegetation Processes, and Ecosystem Properties*. John Wiley & Sons, Chichester, UK.

Group on Earth Observation Biodiversity Network. (2024). *Essential Biodiversity Variables*. Retrieved October 7th, 2024. https://geobon.org/ebvs/what-are-ebvs/

Hacker, S. D., Zarnetske, P., Seabloom, E., Ruggiero, P., Mull, J., Gerrity, S., & Jones, C. (2012). Subtle differences in two non-native congeneric beach grasses significantly affect their colonization, spread, and impact. *Oikos, 121*(1), 138–148. http://www.jstor.org/stable/41409027

Hacker, S.D., Jay, K.R., Cohn, N., Goldstein E.B., Hovenga, P.A., Itzkin, M., Moore, L.J., Mostow, R.S., Mullins, E.V., Ruggiero, P. (2019). Species-Specific Functional Morphology of Four US Atlantic Coast Dune Grasses: Biogeographic Implications for Dune Shape and Coastal Protection. *Diversity. 2019; 11*(5):82. https://doi.org/10.3390/d11050082

Hamuda, E., Glavin, M., Jones, E. (2016). A survey of image processing techniques for plant extraction and segmentation in the field. *Computers and Electronics in Agriculture, Volume 125*, 2016, Pages 184-199.

Hansen, A., Noble, B., Veneros, J., East, A., Goetz, S., Supples, C., Watson, J., Jantz, P., Pillay, R., Jetz, W., Ferrier, S., Grantham, H., Evans, T., Ervin, J., Venter, O., Virnig, A. (2021). Towards monitoring ecosystem integrity within the Post-2020 Global Biodiversity Framework. *Conservation Letters*. John Wiley and Sons Inc. https://doi.org/10.1111/conl.12822

Hantson, W., Kooistra, L., Slim, P. (2012). Mapping invasive woody species in coastal dunes in the Netherlands: A remote sensing approach using LIDAR and high-resolution aerial photographs. *Applied Vegetation Science. 15*. 10.1111/j.1654-109X.2012.01194.x.

Hare, K. M., Borrelle, S. B., Buckley, H. L., Collier, K. J., Constantine, R., Perrott, J. K., … Towns, D. R. (2019). Intractable: species in New Zealand that continue to decline despite conservation efforts. *Journal of the Royal Society of New Zealand*. Taylor and Francis Asia Pacific. https://doi.org/10.1080/03036758.2019.1599967

Hayward, B. (1980). Changes to coastal landforms south of Kawerau, 1942 - 1978. *Tane, 26*, 1980.

Hein, L., Remme, R.P., Schenau, S., Bogaart, P.W., Lof, M.E., Horlings, E. (2020). Ecosystem accounting in the Netherlands. *Ecosystem Services, Volume 44*, 101118, ISSN 2212-0416.

Herbert, A., Oliphant, J. (1991). *Pingao: the golden sand sedge*. *Nga Puna Waihanga*. Rotorua [N.Z.]: Nga Puna Waihanga, 1991.

Heslenfeld, P, Jungerius, P.D., Klijn, J.A. (2004). European Coastal Dunes: Ecological Values, Threats, Opportunities and Policy Development. In: Martínez, M.L., Psuty, N.P. (eds) *Ecological Studies, Vol. 171.* Coastal Dunes, Ecology and Conservation. Springer.

Hesp, P. (1989). A review of biological and geomorphological processes involved in the initiation and development of incipient foredunes. Proceedings of the Royal Society of Edinburgh. Section B. *Biological Sciences*. 96. 10.1017/S0269727000010927.

Hesp, P. (2000). *Coastal sand dune form and function. CVDN Technical Bulletin*. Forest Research, Rotorua, New Zealand.

Hesp, P. (2002). Foredunes and blowouts: initiation, geomorphology and dynamics. *Geomorphology 48* (2002) 245-268.

Hesp, P. (2011). Dune Coasts. In: Wolanski E and McLusky DS (eds.) *Treatise on Estuarine and Coastal Science, Vol 3,* pp. 193–221. Waltham: Academic Press

Hesp, P. and Walker, I. J. (2021). Coastal Dunes V2. In: Shroder, J.J.F. (Ed.), *Treatise on Geomorphology vol 7*, 2nd Edn., Reference Module in Earth Systems and Environmental

Sciences, Elsevier, Academic Press: 540-591. ISBN 9780124095489, https://dx.doi.org/10.1016/B978-0-12-818234-5.00220-0.

Hesp, P., DaSilva, M., Miot da Silva, G., Bruce, D., Keane, R. (2022). Review and direct evidence of transgressive aeolian sand sheet and dunefield initiation. *Earth Surface Processes and Landforms, 47*(11), 2660. https://doi.org/10.1002/esp.5400.

Hesselbarth, M.H.K., Sciaini, M., With, K.A., Wiegand, K., Nowosad, J. (2019). landscapemetrics: an open-source R tool to calculate landscape metrics. *Ecography, 42*: 1648-1657 (ver. 1.5.4).

Hetherington, J., & Bastow Wilson, J. (2014). Spatial associations between invasive tree lupin and populations of two katipo spiders at Kaitorete Spit, New Zealand. *New Zealand Journal of Ecology*, 38(2), 279–287.

Hilton, H., Konlechner, T. (2021). *The Rakiura Dune Restoration Programme (1999-2021). Lessons Learned from 21 Years of Operations, Monitoring & Research*. School of Geography, University of Otago, PO Box 56, Dunedin, New Zealand.

Hilton, M., Macauley, U., & Henderson, R. (2000). Inventory of New Zealand's active dunelands. In *Science for Conservation* (Issue 157).

Hilton, M.J. (2006). The loss of New Zealand's active dunes and the spread of *A. arenaria* (Ammophila arenaria). *New Zealand Geographer. Vol.62*, Issue, 2.

Hoban, S., Campbell, C. D., da Silva, J. M., Ekblom, R., Funk, C.W., Garner, B.A., Godoy, J.A., Kershaw, F., MacDonald, A.J., Mergeay, J.A., Minter, M., O'Brien, D., Paz Vinas, I, Pearson, S.K., Pérez-Espona, S., Potter, K.M., Russo, R.M., Segelbacher, G., Vernesi, C., Hunter, M.E. (2021). Genetic diversity is considered important but interpreted narrowly in country reports to the Convention on Biological Diversity: Current actions and indicators are insufficient. *Biological Conservation, Volume 261*, 2021, 109233, ISSN 0006-3207, https://doi.org/10.1016/j.biocon.2021.109233.

Holdaway, R. J., Wiser, S. K., & Williams, P. A. (2012). Status Assessment of New Zealand's Naturally Uncommon Ecosystems. *Conservation Biology, 26*(4), 619–629. https://doi.org/10.1111/j.1523-1739.2012.01868.x

Holling, C.S. (1973). Resilience and Stability of Ecological Systems. *Annual Review of Ecology and Systematics. Vol. 4* (1973), pp. 1-23.

Hoosen, S. (2017). *Site Significance Statement, Kaitorete Spit. SES/E/2*. In, *Christchurch (N.Z.). City Council. (2017). Christchurch District Plan*. [Appendix 9.1.6.1 Schedule of Sites of Ecological Significance, site SES/E/2]. Christchurch, New Zealand: The Council, 2017. Retrieved October 28th, 2022, from https://districtplan.ccc.govt.nz/pages/plan/book.aspx?exhibit=DistrictPlan

Hothorn, T., Zeileis, A. (2015). "*partykit: A Modular Toolkit for Recursive Partytioning in R*." Journal of Machine Learning Research, 16, 3905-3909.

Hovenga, P.A., Ruggiero, P., Goldstein, E.B., Hacker, S.D., Moore, L.J. (2021). The relative role of constructive and destructive processes in dune evolution on Cape Lookout National Seashore, North Carolina, USA. *Earth Surfaces, Processes and Landforms*. *Volume 46,* Issue 14. Pages 2824-2840.

Hovenga, P.A., Ruggiero, P., Itzkin, P., Jay, K.R., Moore, L. Hacker, S.D. (2023). Quantifying the relative influence of coastal foredune growth factors on the U.S. Mid-Atlantic Coast using field observations and the process-based numerical model Windsurf. *Coastal Engineering*, *Volume 181.*

Hugenholtz, C. H., Levin, N., Barchyn, T. E., & Baddock, M. C. (2012). Remote sensing and spatial analysis of aeolian sand dunes: A review and outlook. In *Earth-Science Reviews* (*Vol. 111*, Issues 3–4, pp. 319–334). https://doi.org/10.1016/j.earscirev.2011.11.006.

Hugenholtz, C.H., Wolfe, S. A. (2005). Biogeomorphic model of dunefield activation and stabilization on the northern Great Plains. *Geomorphology*. 70 (2005) 53–70.

Huggett, R.J. (2007). Fundamentals of Geomorphology. Second Edition. Routledge. Fundamentals of Physical Geography. Taylor Francis Group. London.

Hunt, E.R., Jr., Cavigelli, M., Daughtry, C.S.T., McMurtrey J.E. III, Walthall, C.L.(2005). Evaluation of digital photography from model aircraft for remote sensing of crop biomass and nitrogen status. *Precision Agric., 6*:359-378.

Hutchison, M., Patrick, B. (2020). Canterbury Botanical Society report on Kaitorete Spit October field trip. Trilepidea, Newsletter of the *New Zealand Plant Conservation Network*. *No.134,* Jan 2015. Retrieved October 28th, 2022 from https://www.researchgate.net/publication/345753083_Canterbury_Botanical_Society_report_on_Kaitorete_Spit_October_field_trip

Innangi, M., Marzialetti, F., Di Febbraro, M., Acosta, A.T.R., De Simone, W., Frate, L., Finizio, M., Villalobos Perna, P., Carranza, M.L. (2023). Coastal Dune Invaders: Integrative Mapping of *Carpobrotus* sp. pl. (Aizoaceae) Using UAVs. *Remote Sens. 15*(503):1-19

Intergovernmental Panel on Climate Change. (2019). The Ocean and Cryosphere in a Changing Climate. In H.-O. Pörtner, D.C. Roberts, V. Masson-Delmotte, et al. (Eds.), *Special Report of the Intergovernmental Panel on Climate Change* (pp. 285-318). Cambridge University Press. https://www.ipcc.ch/srocc/

Intergovernmental Panel on Climate Change. (2022): *Climate Change 2022: Impacts, Adaptation, and Vulnerability.* Contribution of Working Group II to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change [H.-O. Pörtner, D.C. Roberts, M. Tignor, E.S. Poloczanska, K. Mintenbeck, A. Alegría, M. Craig, S. Langsdorf, S. Löschke, V. Möller, A. Okem, B. Rama (eds.)]. Cambridge University Press. Cambridge University Press, Cambridge, UK and New York, NY, USA, 3056 pp., doi:10.1017/9781009325844.

Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. (2019). *Global assessment report on biodiversity and ecosystem services of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services*. E. S. Brondizio, J. Settele, S. Díaz, and H. T. Ngo (editors). IPBES secretariat, Bonn, Germany. 1148 pages. https://doi.org/10.5281/zenodo.3831673.

Itzkin, M., Moore, L. J., Ruggiero, P., Hacker, S. D., Biel, R. G. (2021). The relative influence of dune aspect ratio and beach width on dune erosion as a function of storm duration and surge level, *Earth Surf. Dynam., 9*, 1223–1237.

International Union for the Conservation of Nature. (2016). *Guidelines for Using the IUCN Red List Categories and Criteria.* Version 13. Prepared by the Standards and Petitions Subcommittee. D. Retrieved October 28th, 2022, from http://www.iucnredlist.org/documents/RedListGuidelines.pdf.

International Union for the Conservation of Nature. (2024a). *Global invasive species database. Ammophila arenaria*. Retrieved October 4th, 2024, from https://www.iucngisd.org/gisd/species.php?sc=1518.

IUCN (2024b). *Red List Ecosystem Database*. Retrieved October 7th, 2024. https://iucnrle.org/rle-database.

Jackson, D.W., Costas, S., González-Villanueva, R., Cooper, A. (2019). A global 'greening' of coastal dunes: an integrated consequence of climate change? *Global Planet. Change 182*, 103026.

Jane, G., Donaghy, G. (2008). Rarawa Beach dunes (RARW). [plant species list]. New Zealand *Plant Conservation Network*. https://www.nzpcn.org.nz/publications/plant-lists/plant-lists-by-region/rarawa-beach-dunes-rarw/

Jay, K., Hacker, S., Hovenga, P., Moore, L., Ruggiero, P. (2022). Sand supply and dune grass species density affect foredune shape along the US Central Atlantic Coast. *Ecosphere. 13.* 10.1002/ecs2.4256.

Johnson, P. N. (1992). The sand dune and beach vegetation inventory of New *Zealand: 2. South Island and Stewart Island. DSIR Land Resources Scientific Report, 16.* Department of Science and Industrial Research. Wellington, New Zealand.

Johnson, P.N. (1993). Dry Coastal Ecosystems of New Zealand. In Van der Maarel, E. (Ed.), (1993). *Ecosystems of the World, Dry Coastal Ecosystems* (pp197 - 221).

Johnson, P.N. (1993). Dry Coastal Ecosystems of New Zealand. In Van der Maarel, E. (Ed.), *Ecosystems of the World, Dry Coastal Ecosystems* Volume 2B (pp197 - 221).

Kāpiti Coast District Council (n.d). *Caring for Sand Dunes in Kāpiti*. Retrieved October 28th, 2022, from https://waikawabeach.org.nz/wp-content/uploads/2019/02/1092-kap-dune-posterbrochure-reading-ff2-for-web.pdf

Karr, J. (1993). Defining and assessing ecological integrity: Beyond water quality. *Environmental Toxicology and Chemistry Volume: 12* Issue 9 (1993) ISSN: 0730-7268 Online ISSN: 1552-8618

Keith, D.A., Rodríguez, J.P., Rodríguez-Clark, K.M., Nicholson, E., Aapala, K., Alonso, A., et al. (2013). Scientific Foundations for an IUCN Red List of Ecosystems. *PLoS ONE 8*(5): e62111. https://doi.org/10.1371/journal.pone.0062111

Keith, H., Czúcz, B., Jackson, B., Driver, A., Nicholson, E., & Maes, J. (2020). A conceptual framework and practical structure for implementing ecosystem integrity accounts. *One Ecosystem, 5*, 1–54. https://doi.org/10.3897/oneeco.5.e58216

Keijsers, J.G.S, De Groot, A.V., Riksen, M.P.M. (2015). Vegetation and sedimentation on coastal foredunes. *Geomorphology 228*, 723-734.

Keijsers, J.G.S, De Groot, A.V., Riksen, M.P.M. (2016). Modeling the biogeomorphic evolution of coastal dunes in response to climate change. *J. Geophys. Res. Earth Surf. 121*, 1161-1181.

Konlechner, T.M., Ryu, W., Hilton, M.J., Sherman, D.J. (2015). Evolution of foredune texture following dynamic restoration, Doughboy Bay, Stewart Island, New Zealand, *Aeolian Research, Volume 19,* Part B, Pages 203-214, ISSN 1875-9637.

Krebs, C. J. (2014). *Ecological Methodology* (3rd ed.). Addison-Wesley Educational Publishers.

Kuhn, M. (2008). Building Predictive Models in R Using the caret Package. *Journal of Statistical Software, 28*(5), 1–26. doi:10.18637/jss.v028.i05.

Lancaster, N. (1988). Development of linear dunes in the southwestern Kalahari, southern Africa. *Journal of Arid Environments 14* (3): 233–44.

Land Information New Zealand. (2021). *New Zealand Roads (Addressing)*. New Zealand Government. New Zealand/layer/53382-New Zealand-roads-addressing. Retrieved from https://data.linz.govt.nz/layer/53382-nz-roads-addressing/

Land Information New Zealand. (n.d). *Aerial photos*. Retrieved from https://data.linz.govt.nz/data/category/aerial-photos/

Landcare Research / Manaaki Whenua. (2020). Mainland, New Zealand. *Land Cover Database of New Zealand 5.0*. Retrieved October 28th, 2022, from https://lris.scinfo.org.New Zealand/layer/104400-lcdb-v50-land-cover-database-version-50-mainland-new-zealand

Landcare Research / Manaaki Whenua. (2021). *New Zealand LRI Land Use Capability*, Third edition New Zealand LRI layers. Retrieved October 28th, 2022 from https://lris.scinfo.org.New Zealand/layer/48076-New Zealandlri-land-use-capability-2021/

Laporte-Fauret, Q., Castelle, B., Michalet, R., Marieu, V., Bujan, S., Rosebery, D. (2021). Morphological and ecological responses of a managed coastal sand dune to experimental notches. *Science of The Total Environment, Volume 782*, 2021, 146813, ISSN 0048-9697.

Laporte-Fauret, Q., Lubac, B., Castelle, B., Michalet, R., Marieu, V., Bombrun, L., Launeau, P., Giraud, M., Normandin, C., Rosebery, D. (2020). Classification of Atlantic Coastal Sand

Dune Vegetation Using In Situ, UAV, and Airborne Hyperspectral Data. *Remote Sens. 2020, 12*, 2222. https://doi.org/10.3390/rs12142222.

Lázaro-Lobo, A., Alonso, Á., Granda, E., Romero-Blanco, A., Saldaña-López, A., Castro-Díez, P., & Fernández, R. (2023). Impacts of Plant Invasions on Ecosystem Functionality: A Perspective for Ecosystem Health and Ecosystem Services. *Plant Invasions and Global Climate Change,* 31. https://doi.org/10.1007/978-981-99-5910-5_2.

Lee, W., McGlone, M., Wright, E. (2005). *Biodiversity inventory and monitoring. A review of national and international systems and a proposed framework for future biodiversity monitoring* by the Department of Conservation. Landcare Research Contract Report: LC0405/122, prepared for the New Zealand Department of Conservation.

Levin, N., Kidron, G. J., Ben-Dor, E. (2008). A field quantification of coastal dune perennial plants as indicators of surface stability, erosion or deposition. *Sedimentology, 55*(4), 751–772. https://doi.org/10.1111/j.1365-3091.2007.00920.x

Lux, J., Holland, W., Rate, S., Beadel, S. (2009). *Natural areas of Te Paki Ecological District: reconnaissance survey report for the Protected Natural Areas Programme, New Zealand Department of Conservation*. Retrieved October 28th, 2022, from https://www.doc.govt.nz/about-us/science-publications/conservation-publications/land-and-freshwater/land/northland-conservancy-ecological-districts-survey-reports/natural-areas-of-te-paki-ecological-district/

Lyver, P. O. B., Richardson, S. J., Gormley, A. M., Timoti, P., Jones, C. J., & Tahi, B. L. (2018). Complementarity of indigenous and western scientific approaches for monitoring forest state. *Ecological Applications, 28*(7), 1909–1923. https://doi.org/10.1002/eap.1787

Ma, L., Li, M., Ma, X., Cheng, L., Du, P., Liu, Y. (2017). A review of supervised object-based land-cover image classification. ISPRS *Journal of Photogrammetry and Remote Sensing. 130*. 10.1016/j.isprsjprs.2017.06.001.

Macara, G.R. (2018). *The climate and weather of New Zealand (1st ed.)* NIWA Science and Technology Series. Number 74. National Institute of Water and Air. Wellington, New Zealand.

Maes, J., Egoh, B., Willemen, L., Liquete, C., Vihervaara, P., Schägner, J. P., . . . Bidoglio, G. (2012). Mapping ecosystem services for policy support and decision making in the European Union. *Ecosystem Services, 1*(1), 31-39. https://doi.org/10.1016/j.ecoser.2012.06.004.

Maes, J., Liquete, C., Paracchini, M., Barredo, J., Grizzetti, B., Cardoso, A., . . . Czúcz, B. (2016). An indicator framework for assessing ecosystem services in support of the EU Biodiversity Strategy to 2020. *Ecosystem Services, 17*, 14. https://doi.org/10.1016/j.ecoser.2015.10.023.

Maes, J., Driver, A. Czúcz, B., Keith, H. Jackson, B., Nicholson, E., Dasoo, M. (2020). A review of ecosystem condition accounts: Lessons learned and options for further development. *One Ecosystem. 5*. 10.3897/oneeco.5.e53485.Martínez & Psuty, 2004.

Manaaki Whenua Landcare Research. (2019). *Mainland, New Zealand. Land Cover Database of New Zealand 5.0*. Retrieved October 28th, 2022, from https://lris.scinfo.org.New Zealand/layer/104400-lcdb-v50-land-cover-database-version-50-mainland-new-zealand

Manawatu District Council. (2019). *Coastal Reserves Management Plan*. Manawatu District Council. Retrieved October 28th, 2022, from mdc.govt.nz/documents/plans/reserve-management-plans

Marcenò, C., Guarino, R., Loidi, J., Herrera, M., Isermann, M., Knollova, I., Tichý, L., Tzonev, R., Acosta, A., Fitzpatrick, Ú., Iakushenko, D., Janssen, J., Jiménez-Alfaro, B., Kacki, Z., Keizer-Sedláková, I., Kolomiichuk, V., Rodwell, J., Schaminée, J., Šilc, U., Chytry, M. (2018). Classification of European and Mediterranean coastal dune vegetation. *Applied Vegetation Science. 21*. 10.1111/avsc.12379.

Martínez, M. L., Hesp, P. A., & Gallego-Fernández, J. B. (2013). *Coastal Dunes: Human Impact and Need for Restoration* (pp. 1–14). https://doi.org/10.1007/978-3-642-33445-0_1.

Martínez, M. L., Vázquez, G., & Colón, S. S. (2001). Spatial and Temporal Variability during Primary Succession on Tropical Coastal Sand Dunes. *Journal of Vegetation Science, 12*(3), 361–372. https://doi.org/10.2307/3236850

Martínez, M.L., García-Franco, J.F. (2004). Plant-plant interactions in coastal dunes. In: Martínez, M.L., Psuty, N.P. (eds) *Coastal Dunes, Ecology and Conservation* (pp.205 -220). Springer - Verlag Berlin Heidelberg, 2004.

Martínez, M.L., Maun, M.A., Psuty, N.P. (2004). The Coastal Dune Paradox: Conservation vs Exploitation? In Martínez, M. L. & Psuty, N. P. (Eds.), *Coastal Dunes, Ecology and Conservation* (pp. 335–351). Springer - Verlag Berlin Heidelberg, 2004.

Martínez, M.L., Psuty, N.P., Lubke, R.A. (2004). A perspective on coastal dunes. In: Martínez, M.L., Psuty, N.P. (eds) Ecological Studies, Vol. 171. *Coastal Dunes, Ecology and Conservation* (pp.3-10). Springer - Verlag Berlin Heidelberg, 2004.

Marzialetti, F., Frate, L. De Simone, W. Frattaroli, A.R., Acosta, A.T.R., Carranza, M.L. (2021) Unmanned Aerial Vehicle (UAV)-Based Mapping of *Acacia saligna* invasion in the Mediterranean Coast. *Remote Sens. 13*, 3361. https://doi.org/10.3390/rs13173361.

Marzialetti, F., Grosso, G., Acosta, A.T.R., Malavasi, M., Cao Pinna, L., Sternberg, M., Gupta, S.K., Brundu, G., Carranza, M.L. (2024). Dunes under attack: untangling the effects of landscape changes on Iceplant invasion (*Carpobrotus* spp., Aizoaceae) in *Mediterranean Coasts. ARPHA* Preprints. https://doi.org/10.3897/arphapreprints.e133102

Maun, M.A. (2009). *The Biology of Coastal Sand Dunes*. Oxford University Press, New York.

McCune, B., Grace, J. B. (2002). *Analysis of ecological communities*. Gleneden Beach, Oregon: MjM Software Design.

McCune, B., Mefford, M.J. (2018). PC-ORD (v.7.10). *Multivariate Analysis of Ecological Data.* MjM Software Design, Gleneden Beach, Oregon, USA.

McGarigal, K.S. & Cushman, S. Neel, M., Ene, E. (2002). *FRAGSTATS: Spatial pattern analysis program for categorical maps / Fragstats Help.* Retrieved 18 October, 2024, from https://www.researchgate.net/publication/259011515_FRAGSTATS_Spatial_pattern_analysis_program_for_categorical_maps

McGarigal, K., Tagil, S, Cushman, S.A. (2009). Surface metrics: An alternative to patch metrics for the quantification of landscape structure. *Landscape Ecology, 24*: 433-450.

McGlone, M., McNutt, K. Richardson, S., Bellingham, P., Wright, E. (2020). Biodiversity monitoring, ecological integrity, and the design of the New Zealand Biodiversity Assessment Framework. *New Zealand Journal of Ecology. 44*. 10.20417/nzjecol.44.17

McGuirk, M. T., Kennedy, D. M., & Konlechner, T. (2022). The Role of Vegetation in Incipient Dune and Foredune Development and Morphology: A Review. *Journal of Coastal Research, 38*(2), 414–428.

McKelvey, P. (1999). *Sand Forests. A historical perspective of the stabilisation and afforestation of coastal sands in New Zealand*. Canterbury University Press. New Zealand.

Mead, S., Borrero, J.C., Harrison, S. (2009). *Orewa Beach Reefs. Design and assessment of physical effects of multi-purpose reefs for beach sand retention at Orewa Beach.* ASR consulting, prepared for Ōrewa Beach Reef Charitable Trust / RDC. Report no. 2009 - 048/1.

Melgani, F., Bruzzone, L. (2004). Classification of hyperspectral remote sensing images with support vector machines. In *IEEE Transactions on Geoscience and Remote Sensing, vol. 42*, no. 8, pp. 1778-1790, Aug. 2004, doi: 10.1109/TGRS.2004.831865.

Meucci, A., Young., I.R., Kirezci, E., Ranasinghe, R. (2020). Projected 21st century changes in extreme wind-wave events. *Sci. Adv.6,* eaaz7295.

Miller, T.E., Gornish, E.S. & Buckley, H.L. (2010). Climate and coastal dune vegetation: disturbance, recovery, and succession. *Plant Ecol 206, 97* (2010). https://doi.org/10.1007/s11258-009-9626-z

Milne, R., Sawyer, J. (2002). *Coastal foredune vegetation in Wellington Conservancy, current status and future management.* Published by Department of Conservation, Wellington Conservancy. Retrieved October 28th, 2022, from https://www.doc.govt.nz/about-us/science-publications/conservation-publications/native-plants/coastal-foredune-vegetation-in-wellington-conservancy/

Ministry for the Environment & Stats NZ. (2019). *New Zealand's Environmental Reporting Series: Our marine environment* 2019. Ministry for the Environment and Stats NZ, publication number: ME 1468.

Moore, L. J. (2000). Shoreline Mapping Techniques. *Journal of Coastal Research, 16* (1), 111-124.

Moreno-Casasola, P. (1986). Sand Movement as a Factor in the Distribution of Plant Communities in a Coastal Dune System. *Vegetatio, 65*(2), 67–76. http://www.jstor.org/stable/20037269.

Morgan, J., Gergel, S., Coops, N. (2010). Aerial Photography: A Rapidly Evolving Tool for Ecological Management. Bioscience. 60. 47-59. 10.1525/bio.2010.60.1.9.Motohka, T., Kenlo, N., Oguma, H., Tsuchida, S. (2010). Applicability of Green-Red Vegetation Index for Remote Sensing of Vegetation Phenology. *Remote Sensing. 2*. 10.3390/rs2102369.

Moulton, M.A., Hesp, P.A., Miot da Silva, G., Bouchez, C., Lavy, M., Fernandez, G.B. (2019). Changes in vegetation cover on the Younghusband Peninsula transgressive dunefields (Australia) 1949-2017. *Earth Surf. Proc. Land. 44* (2).

Mücher, C.A., Los, S., Kramer, H., Janssen, J., Roerink, G.J. (2023). Remote sensing for nature monitoring. *Wageningen, Wageningen Environmental Research, Report 3271.* 70 pp.; 38 fig.; 9 tab.; 87 ref.

Muckersie, C., Shepherd, M.J. (1995). Dune Phases as time-transgressive phenomena, Manawatu, New Zealand. *Quaternary International, Vol. 26*, pp. 61-67.

Muñoz-Vallés, S., Cambrollé, J. (2014). Successes and failures in the management of coastal dunes of SW Spain: Status analysis nine years after management decisions, *Ecological Engineering, Volume 71*, 2014, Pages 415-425, ISSN 0925-8574.

Murray, N. J., Keith, D. A., Bland, L. M., Ferrari, R., Lyons, M. B., Lucas, R., Pettorelli, N., & Nicholson, E. (2018). The role of satellite remote sensing in structured ecosystem risk assessments. *In Science of the Total Environment (Vols. 619–620*, pp. 249–257). Elsevier B.V. https://doi.org/10.1016/j.scitotenv.2017.11.034.

Müller, A. C., Guido, S. (2016). *Introduction to Machine Learning with Python: A Guide for Data Scientists*, O'Reilly Media, Incorporated.

National Institute of Water and Atmospheric Research. (NIWA). (2023). *CliFlo, New Zealand's National Climate Database.* Retrieved from https://cliflo.niwa.co.nz/.

Natural England. (2022). *Living England: From Satellite Imagery to a National Scale Habitat Map.* Retrieved from https://naturalengland.blog.gov.uk/2022/04/05/living-england-from-satellite-imagery-to-a-national-scale-habitat-map/

New Zealand National Vegetation Survey Databank. (1991). *Foxton Protected Natural Area Programme / National Vegetation Survey 1989 – 91*. [Data sheets supplied by Manaaki Whenua]. https://www.nvs.landcareresearch.co.nz)

New Zealand National Vegetation Survey Databank. (2012). *Birdlings Flat Gravel Beach Survey 2012 - Recce inventory (National Vegetation Survey).* https://www.nvs.landcareresearch.co.nz.

Newsome, P. F. J., New Zealand. National Water and Soil Conservation Authority, New Zealand. Water and Soil Directorate. (1987). *The vegetative cover of New Zealand.* Wellington, N.Z., published for the National Water and Soil Conservation Authority by the Water and Soil Directorate, Ministry of Works and Development.

Newsome, P., Wilde, R., Willoughby, E. (2008). *Land Resource Information System Spatial Data Layers, Data Dictionary*. Published by Landcare Research New Zealand Ltd., 2008.

Nicholson, E., Watermeyer, K.E., Rowland, J.A. (2021). Scientific foundations for an ecosystem goal, milestones and indicators for the post-2020 global biodiversity framework. *Nat Ecol Evol 5*, 1338–1349. https://doi.org/10.1038/s41559-021-01538-5.

Norbury, D. (1996). The effect of rabbits on conservation values. *Science for Conservation 34*. Department of Conservation. Wellington, New Zealand.

Nordstrom, K. F. (2008). Beach and dune restoration. *In Beach and Dune Restoration*. Cambridge University Press. https://doi.org/10.1017/CBO9780511535925.

Normyle, A., Vardon, M., Doran, B. (2022). Ecosystem accounting and the need to recognise Indigenous perspectives. *Humanit Soc Sci Commun 9*, 133. https://doi.org/10.1057/s41599-022-01149-w.

Northland Age. (2021). *Ngataki school kids restore Rarawa Beach dunes with pingao*. NZ Herald. Retrieved October 28th, 2022, from https://www.nzherald.co.nz/northland-age/news/ngataki-school-kids-restore-rarawa-beach-dunes-with-pingao/PWM7D4GMKKUM7ZS6GI77YL5JTY/.

Northland Regional Council (2014). *Northland Regional Landscape Assessment*. [Unit name - Twilight Beach / Kahokawa Beach Dunefields, Wetlands and Bush]. Retrieved October 28th, 2022, from https://www.nrc.govt.nz/media/kiwe3v4y/twilightbeachkahokawabeachdunefieldandwetlands.pdf.

Noss, R. F. (1990). Indicators for Monitoring Biodiversity: A Hierarchical Approach. *Conservation Biology*. https://doi.org/10.1111/j.1523-1739.1990.tb00309.x.

Oddi, L., Cremonese, E., Ascari, L., Filippa, G., Galvagno, M., Serafino, D., di. Cella, U. M. (2021). Using UAV Imagery to Detect and Map Woody Species Encroachment in a Subalpine Grassland: Advantages and Limits. *Remote Sensing, 13*(7), 1239. MDPI AG. Retrieved from http://dx.doi.org/10.3390/rs13071239.

Ogle, C.C. (2008). *Vegetation and threatened and adventive plants of the Wanganui sand country*. Paper presented to the Dune Restoration Trust, March 2008. Coastal Restoration Trust. Retrieved October 28th, 2022, from https://ref.coastalrestorationtrust.org.nz/site/assets/files/5987/weeds_and_indigenous_biodiversity_of_the_fed_northern_half_2008.pdf.

Oksanen J., Simpson G., Blanchet .F, Kindt R., Legendre P, Minchin P., O'Hara R., Solymos P., Stevens M., Szoecs E., Wagner H., Barbour M., Bedward M., Bolker B., Borcard D., Carvalho G., Chirico M., De Caceres M., Durand S., Evangelista H., FitzJohn R., Friendly M., Furneaux B., Hannigan G., Hill M., Lahti L., McGlinn D., Ouellette M., Ribeiro Cunha E., Smith T., Stier A., Ter Braak C., Weedon J. (2022). *_vegan: Community Ecology Package_*. R package version 2.6-4, <https://CRAN.R-project.org/package=vegan>.

Olsen, J.S., van der Maarel, E. (1997). Organic and physical dune building. In E. van der Maarel (Ed.). *Dry Coastal Ecosystems: General Aspects*, Volume C. Elsevier, Amsterdam.

Ōtama Reserves Group (n.d). *History of Otama*. Retrieved October 28th, 2022, from https://www.otamareservesgroup.co.nz/history-of-otama/

Partridge, T. R. (1992). *The sand dune and beach vegetation inventory of New Zealand*: *1. North Island.* DSIR Land Resources scientific report (Vol. 15).

Patrício, J., Elliott, M., Mazik, K., Papadopoulou, K. N., & Smith, C. J. (2016). DPSIR-Two decades of trying to develop a unifying framework for marine environmental management? *Frontiers in Marine Science*. Frontiers Media S. A. https://doi.org/10.3389/fmars.2016.00177.

Pegman, A. P. M. K., Rapson, G. L. (2005). Plant succession and dune dynamics on actively prograding dunes, Whatipu Beach, northern New Zealand. *New Zealand Journal of Botany, 43*(1), 223–244. https://doi.org/10.1080/0028825X.2005.9512952

Pereira, H. M., Ferrier, S., Walters, M., Geller, G. N., Jongman, R. H., Scholes, R. J., Bruford, M. W., Brummitt, N., Butchart, S. H., Cardoso, A. C., Coops, N. C., Dulloo, E., Faith, D. P., Freyhof, J., Gregory, R. D., Heip, C., Höft, R., Hurtt, G., Jetz, W., Karp, D. S., … Wegmann, M. (2013). Ecology. Essential biodiversity variables. *Science* (New York, N.Y.), *339*(6117), 277–278. https://doi.org/10.1126/science.1229931.

Pettorelli, N., Böhm, M., Wegmann, M., Heiden, U., Mueller, A., Skidmore, A., . . . Geller, G. (2016). Framing the concept of satellite remote sensing essential biodiversity variables: Challenges and future directions. *Remote Sensing in Ecology and Conservation, 2*(3), 122. https://doi.org/10.1002/rse2.15.

Pettorelli, N., Williams, J., Schulte T., Buhne, H., Crowson, M. (2024). Deep learning and satellite remote sensing for biodiversity monitoring and conservation. *Remote Sensing in Ecology and Conservation*. https://doi.org/10.1002/rse2.415

Pintó, J., Garcia-Lozano, C., Varga, D. (2023). Using dune-restricted species to assess the degree of natural diversity of dune systems on Mediterranean tourist coasts. *Ecological Indicators, Volume 147*,2023,110004, ISSN 1470-160X.

Polley, H. W., Derner, J. D., & Wilsey, B. J. (2005). Patterns of plant species diversity in remnant and restored tallgrass prairies. *Restoration Ecology, 13* (3), 480–487. https://doi.org/10.1111/j.1526-100X.2005.00060.x

Provoost, S., Jones, M.L.M., Edmondson, S.E. (2011). Changes in landscape and vegetation of coastal dunes in northwest Europe: a review. *J Coast Conserv 15*, 207–226. doi-org.ezproxy.aut.ac.nz/10.1007/s11852-009-0068-5

Psuty, N. P., & Silveira, T. M. (2010). Global climate change: An opportunity for coastal dunes? *Journal of Coastal Conservation, 14*(2), 153–160. https://doi.org/10.1007/s11852-010-0089-0

Psuty, N.P. (2004). The Coastal Foredune: A Morphological Basis for Regional Coastal Dune Development. In M. L. Martínez, M. L., N.P. Psuty (Eds.), *Coastal Dunes, Ecology and Conservation* (pp. 335–351). Springer.

Psuty, N.P. (2008). *The Coastal Foredune: A Morphological Basis for Regional Coastal Dune Development*. In: Martínez, M.L., Psuty, N.P. (eds) *Coastal Dunes. Ecological Studies*, vol 171. Springer, Berlin, Heidelberg. https://doi.org/10.1007/978-3-540-74002-5_2

R Core Team. (2021). *R: A language and environment for statistical computing*. R Foundation for Statistical Computing, Vienna, Austria. https://www.R-project.org/.

R Core Team. (2023). _*R: A Language and Environment for Statistical Computing*_. R Foundation for Statistical Computing, Vienna, Austria.

Radoux, J., Bogaert, P. (2017). Good Practices for Object-Based Accuracy Assessment. *Remote Sensing, 9* (7), 646. MDPI AG.

Rapson, G.L., Smith, A., Murphy, A.L. (2016). *Sand-dune vegetation of the Foxtangi Region, Manawatu Coast, New Zealand.* Report to the Department of Conservation by the Ecology Group, Institute of Agriculture and Environment, Massey University, Palmerston North, New Zealand. Retrieved October 28th, 2022, from https://www.massey.ac.nz/massey/expertise/profile.cfm?stref=361030.

Rendon, P., Erhard, M., Maes, J., Burkhard, B. (2019). Analysis of trends in mapping and assessment of ecosystem condition in Europe, *Ecosystems and People, 15:*1, 156-172, DOI: 10.1080/26395916.2019.1609581.

Richards, J.A. (2013). *Remote Sensing Digital Image Analysis, An Introduction*. 5th Edition. Springer eBooks.ISBN 978-3-642-30062-2.

Roberts, R., Carpenter, N., Klinac, P. (2020). *Predicting Auckland's exposure to coastal instability and erosion, Auckland Council, technical report*, TR2020/021

Rouse, H.L., Nichol, S., Goff., J. (2003). *Introduction to the New Zealand Coast*. In: Nichol, N., Goff., J and Rouse (Eds.), The New Zealand Coast, Te Tai O Aotearoa. Dunmore Press, Wellington, New Zealand.

Routledge, R. D. (1990). Spatial Patterns Arising from Plant Dispersal as Modelled by a Correlated Random Walk. *Journal of Applied Probability, 27*(1), 1-13. https://doi.org/10.2307/3214591.

Ruggiero, P., Hacker, S., Seabloom, E., Zarnetske, P. (2018). The Role of Vegetation in Determining Dune Morphology, Exposure to Sea-Level Rise, and Storm-Induced Coastal Hazards: A U.S. Pacific Northwest Perspective. In: Moore, L., Murray, A. (eds) *Barrier Dynamics and*

Response to Changing Climate. Springer, Cham. https://doi.org/10.1007/978-3-319-68086-6_11.

RStudio Team. (2020). *RStudio: Integrated Development for R*. RStudio, PBC, Boston, MA. http://www.rstudio.com/.

Ryan, C., Case, S.B., Bishop, C.D, Buckley, H.L. (2023). Ecosystem integrity of active sand dunes: A case study to implement and test the SEEA-EA global standard, from Aotearoa New Zealand. *Ecological Indicators, Volume 149*. https://doi.org/10.1016/j.ecolind.2023.110172.

Ryu, W., Sherman, D.J. (2014). Foredune texture: landscape metrics and climate. Ann. Assoc. *Am. Geogr.* 104, 903–921.

Salgado, K., M., Martínez, L., Lucero, L., Álvarez-Molina, L., Hesp, P., Equihua M., Mariño-Tapia, I. (2022). Impact of urbanization and landscape changes on the vegetation of coastal dunes along the Gulf of Mexico, *Écoscience, 29:2*, 103-116, DOI: 10.1080/11956860.2021.1934299.

Schlacher, T.A., Schoeman, D.S., Dugan, J., Lastra, M., Jones, A., Scapini, F., McLachlan, A. (2008). Sandy beach ecosystems: key features, sampling issues, management challenges and climate change impacts. *Marine Ecology, 29*, SUPPL. 1, 70-90.

Seabloom, E., Ruggiero, P., Hacker, S., Zarnetske, P., & Mull, J. (2013). Invasive grasses, climate change, and exposure to storm-wave overtopping in coastal dune ecosystems. *Global Change Biology, 19*(3), 824. https://doi.org/10.1111/gcb.12078.

Shadbolt, A. (2014). *Site significance statement, Christchurch Coastal Strip, SES/LP/6. In, Christchurch (N.Z.)*. City Council. (2017). Christchurch District Plan. [Appendix 9.1.6.1

Schedule of Sites of Ecological Significance, site SES/LP/6]. Christchurch, New Zealand: The Council, 2017. Retrieved October 28th, 2022, from https://districtplan.ccc.govt.nz/pages/plan/book.aspx?exhibit=DistrictPlan

Shepherd, M. J. and Hesp, P. A., (2003). *N.Z. coastal barriers and dunes*. Ch. 8, In Nichol, N, Goff., J and Rouse, H.L. (Eds.), The New Zealand Coast, Te Tai O Aotearoa. Dunmore Press, Wellington, New Zealand.

Short, A. D., & Hesp, P. A. (1982). Wave, beach and dune interactions in southeastern Australia. *Marine Geology, 48*(3–4), 259–284. https://doi.org/10.1016/0025-3227(82)90100-1.

Silliman, B. R., Schrack, E., He, Q., Cope, R., Santoni, A., van der Heide, T., . . . van de Koppel, J. (2015). Facilitation shifts paradigms and can amplify coastal restoration efforts. *Proceedings of the National Academy of Sciences of the United States of America, 112*(46), 14295.

Skidmore, A. K., Pettorelli, N., Coops, N. C., Geller, G. N., Hansen, M., Lucas, R., . . . Wegmann, M. (2015). Environmental science: Agree on biodiversity metrics to track from space. *Nature, 523*(7561), 403-404. https://doi.org/10.1038/523403a

Sloss, C. R., Shepherd, M. & Hesp, P. (2012) Coastal Dunes: Geomorphology. Nature Education Knowledge 3(10): 2Smyth, T., Wilson, R., Rooney, P., Yates, K. (2022). Extent, accuracy and repeatability of bare sand and vegetation cover in dunes mapped from aerial imagery is highly variable. *Aeolian Research. 56*. 100799. 10.1016/j.aeolia.2022.100799.

Stallins, J.A. (2005). Stability domains in barrier island dune systems, *Ecological Complexity, Volume 2*, Issue 4, 2005.

Stats New Zealand. (2015). *Environmental Indicators, Te Taiao Aotearoa, Active Dune Extent.* Retrieved October 28th, 2022, from https://www.stats.govt.New Zealand/indicators/active-sand-dune-extent.

Stats NZ. (2015). *Active sand dune extent 1950s–2008.* New Zealand's Environment Reporting Series: The Ministry for the Environment. Retrieved from: https://www.stats.govt.nz/indicators/active-sand-dune-extent/

Stats NZ. (2017). *Age and income in 2013 by TALB2013*. Retrieved from https://datafinder.stats.govt.nz/layer/64334-age-and-income-in-2013-by-talb2013/

Stats NZ. (2019). 2018 *Census usually resident population and age groups by Statistical Area 2*. Retrieved from https://datafinder.stats.govt.nz/layer/103902-2018-census-usually-resident-population-and-age-groups-by-statistical-area-2/.

Stockdon, H.F., Sallenger, A.H. Jr, Holman, R.A; Howd, P.A. (2007). A simple model for the spatially-variable coastal response to hurricanes. *Marine Geology, 238* (2007) 1–20.

Sykes, M. T., Wilson, J. B. (1991). Vegetation of a Coastal Sand Dune System in Southern New Zealand. *Journal of Vegetation Science, 2*(4), 531–538. https://doi.org/10.2307/3236035.

Tāwharanui Open Sanctuary. (n.d). *What we do [revegetation].* Retrieved October 28th, 2022, from https://www.tossi.org.nz/?page_id=73

Therneau T., Atkinson, B. (2022). *_rpart: Recursive Partitioning and Regression Trees_.* R package version 4.1.19, https://CRAN.R-project.org/package=rpart.

Tittensor, D. P., Walpole, M., Hill, S. L., Boyce, D. G., Britten, G. L., Burgess, N. D., Butchart, S. H., Leadley, P. W., Regan, E. C., Alkemade, R., Baumung, R., Bellard, C., Bouwman, L., Bowles-Newark, N. J., Chenery, A. M., Cheung, W. W., Christensen, V., Cooper, H. D., Crowther, A. R., Dixon, M. J., … Ye, Y. (2014). A mid-term analysis of progress toward international biodiversity targets. *Science* (New York, N.Y.), *346*(6206), 241–244. https://doi.org/10.1126/science.1257484

Todd, D., MacDonald, K., Debski, D. (2022). *Kapiti Coast District Council Coastal Hazard Technical Assessment Technical Report. Kāpiti Coast Coastal Hazards Susceptibility and Vulnerability Assessment Volume 2: Results* (draft). Report no. IS355300-NC-RPT0004. Jacobs New Zealand Limited.

Tordoni, E., Bacaro, G., Weigelt, P., Cameletti, M., Janssen, J. A. M., Acosta, A. T. R., … Kreft, H. (2021). Disentangling native and alien plant diversity in coastal sand dune ecosystems worldwide. *Journal of Vegetation Science, 32*(1). [Dataset for New Zealand component, supplied by Hannah Buckley, collected 2010]. https://doi.org/10.1111/jvs.12961

Tsoar, H. (2005). Sand dunes mobility and stability in relation to climate. In *Physica A: Statistical Mechanics and its Applications (Vol. 357*, pp. 50–56). ttps://doi.org/10.1016/j.physa.2005.05.067

Turner, M. G. (1989). Landscape Ecology: The Effect of Pattern on Process. *Annual Review of Ecology and Systematics, 20*, 171–197. http://www.jstor.org/stable/2097089.

Turner, M. G., Gardner, R. H., & Golley, F. B. (2015). *Landscape ecology in theory and practice: Pattern and process / Monica G. Turner, Robert H. Gardner* (Second edition.). Springer.

United Nations. (1992). *Convention on Biological Diversity*. United Nations. Accessed online 05 November 2022 from: https://www.cbd.int/convention/text/

United Nations. (2021). *System of Environmental-Economic Accounting-Ecosystem Accounting* (SEEA EA). White cover publication, pre-edited text subject to official editing. Retrieved October 28th, 2022, from https://seea.un.org/ecosystem-accounting

United Nations Environment Programme. (2022a). CBD/COP/DEC/15/5 *Monitoring framework for the Kunming-Montreal Global Biodiversity Framework*. Retrieved 21 November, 2023 from https://www.post-2020indicators.org/resources.

United Nations Environment Programme. (2022b). CBD/COP/DEC/15/4 *Kunming-Montreal Global Biodiversity Framework*. Retrieved 21 November, 2023 from https://www.post-2020indicators.org/resources.

United Nations Environment Programme -World Conservation Monitoring Centre. (2024). *Indicators for the Kunming – Montreal Global Biodiversity Framework.* United Nations Environment Program - World Comservation Monitoring Centre. Retrieved October 5th, 2024, from https://www.gbf-indicators.org/

United Nations Statistics Division. (2021). *Global Assessment of Environmental-Economic Accounting and Supporting Statistics* 2021. United Nations Statistical Commission.

United Nations Statistics Division (2023). *Global Assessment of Environmental-Economic Accounting and Supporting Statistics 2023*. United Nations Statistical Commission.

United States Department of Agriculture. (2006). Plant Fact Sheet, *American Beachgrass*. Retrieved from https://plants.usda.gov/DocumentLibrary/factsheet/pdf/fs_ambr.pdf

United States Geological Society. (2024a). *Coastal Vegetation Maps to Improve Forecasting of Hurricane Impacts and Coastal Resilience*. https://www.usgs.gov/software/a-tool-rapid-repeat-high-resolution-coastal-vegetation-maps-improve-forecasting-hurricane

United States Geological Society. (2024b). *Lidar-derived Beach Morphology (Dune Crest, Dune Toe, and Shoreline) for U.S. Sandy Coastlines.* https://www.usgs.gov/apps/coastalsciencenavigator/productdetails.html?id=61

Ustin, S.L. and Gamon, J.A. (2010). Remote sensing of plant functional types. *New Phytologist, 186*: 795-816. https://doi.org/10.1111/j.1469-8137.2010.03284.x

Uuemaa, E., Antrop, M., Roosaare, J., Marja, R., Mander, U. (2009). Landscape Metrics and Indices: An Overview of Their Use in Landscape Research. *Living Reviews in Landscape Research*, 3, (2009), 1.

Uys, R., Crisp, P. (2019). *Duneland Health State of the Environment monitoring programme: Annual data report,* 2017/18 and 2018/19. Greater Wellington Regional Council, Publication No. GW/ESCI-T-19/126.

Van De Maarel, E. (Ed.). (2005). *Vegetation ecology*. Blackwell Pub.

Van der Maarel, E. (Ed.). (1993a). *Ecosystems of the World*, *2A*: *Dry coastal ecosystems: Polar Regions and Europe*. Elsevier, Amsterdam.

Van der Maarel, E. (Ed.). (1993b). *Ecosystems of the World, 2B*: *Dry Coastal Ecosystems: Africa, America, Asia and Oceania*. Elsevier, Amsterdam.

Van der Maarel, E. (1997). Coastal Dunes: Pattern and process, zonation and succession. In E. van der Maarel (Ed). *Dry coastal ecosystems: General aspects. Volume C*. Elsevier, Amsterdam.

Van der Maarel, E., Van der Maarel-Versluys, M. (1997). Biogeographic Features of Dry Coastal Ecosystems. In E. Van der Maarel, (Ed). *Dry coastal ecosystems: General aspects Volume C* (pp. 1-10). Elsevier, Amsterdam.

Verhoeven, D., Buckley, H.L., Curran, T.J. (2014). Functional traits of common New Zealand foredune species at New Brighton, Canterbury, New Zealand *Journal of Botany, 52*:4, 460-466, DOI: 10.1080/0028825X.2014.939663.

Violle, C., Garnier, E., Lecoeur, J. (2009). Competition, traits and resource depletion in plant communities. *Oecologia* **160**, 747–755. https://doi.org/10.1007/s00442-009-1333-x.

Waikato Regional Council. (2011). *Our Fragile Dunes. A Guide to Waikato Dunes*. Waikato Regional Council. Hamilton, New Zealand.

Waiotahi Stream and Dune Care. (n.d). *Waiotahi Stream and Dune Care Ōtaki*. Retrieved October 28th, 2022, from http://www.waitohudunecare.org/

Walker, I. J., Eamer, J. B. R., & Darke, I. B. (2013). Assessing significant geomorphic changes and effectiveness of dynamic restoration in a coastal dune ecosystem. *Geomorphology, 199,* 192–204. https://doi.org/10.1016/j.geomorph.2013.04.023

Walker, S., L., Zinnert, J. (2022). Whole Plant Traits of Coastal Dune Vegetation and Implications for Interactions with Dune Dynamics. *Ecosphere 13*(5): e4065. https://doi.org/10.1002/ecs2.4065.

Wardle, P. (1991). *Vegetation of New Zealand*. Cambridge University Press. Cambridge, New York, Port Chester, Melbourne, Sydney, 1991. ISBN 0-521-25873-1.

Watt, A. S. (1947). Pattern and Process in the Plant Community. *Journal of Ecology, 35*(1/2), 1–22. https://doi.org/10.2307/2256497.

West, C. J. (n.d). *Coal River Beach Weeds* (B279). New Zealand Plant Conservation Network. https://www.nzpcn.org.nz/publications/plant-lists/plant-lists-by-region/coal-river-beach-weeds-b279/

Wickham H., Averick M., Bryan J., Chang W., McGowan L.D., François R., Grolemund G., Hayes A., Henry L., Hester J., Kuhn M., Pedersen T.L., Miller E., Bache S.M., Müller K., Ooms J., Robinson D., Seidel D.P., Spinu V., Takahashi K., Vaughan D., Wilke C., Woo K., Yutani H. (2019). Welcome to the tidyverse. *Journal of Open-Source Software*, *4*(43), 1686. doi:10.21105/joss.01686.

Wickham, H. (2016). *ggplot2: Elegant Graphics for Data Analysis*. Springer-Verlag New York.

Wiedemann, A.M. (1998). Coastal foredune development, Oregon, USA. *Journal of Coastal Research. 26*. 45-51.

Wiedemann, A.M., Pickart, A.J. (2008). Temperate Zone Coastal Dunes. In: Martínez, M.L., Psuty, N.P. (eds) Coastal Dunes. *Ecological Studies, vol 171*. Springer, Berlin, Heidelberg. https://doi.org/10.1007/978-3-540-74002-5_4

Wildlands Ltd. (2008). *Bay of Plenty region sand dune vegetation mapping and condition assessment methods for Tauranga Ecological District*. Prepared for Environment Bay of Plenty, P.O Box 364, Whakatane. Retrieved October 28th, 2022, from https://cdn.boprc.govt.nz/media/32395/EnvReport-201002-SandDuneVegetationMapping.pdf

Williams, P.A., Wiser, S., Clarkson, B., Stanley, M. C. (2007). New Zealand's historically rare terrestrial ecosystems set in a physical and physiognomic framework. *New Zealand Journal of Ecology 31*(2), 119–128. https://newzealandecology.org/nzje/2829.

Wolff, F., Kolari, T., Villoslada, M., Tahvanainen, T., Korpelainen, P., Zamboni, P., Kumpula, T. (2023). RGB vs. Multispectral imagery: Mapping aapa mire plant communities with UAVs. *Ecological Indicators*. doi: 10.1016/j.ecolind.2023.110140.

Wolff, F., Kolari, T., Villoslada, M., Tahvanainen, T., Korpelainen, P., Zamboni, P., Kumpula, T. (2023). RGB vs. Multispectral imagery: Mapping aapa mire plant communities with UAVs. *Ecological Indicators*. doi: 10.1016/j.ecolind.2023.110140.

Ye, S., Pontius, R.G., Rakshit, R. (2018). A review of accuracy assessment for object-based image analysis: From per-pixel to per-polygon approaches, *ISPRS Journal of Photogrammetry and Remote Sensing, Volume 141*, 2018, Pages 137-147.

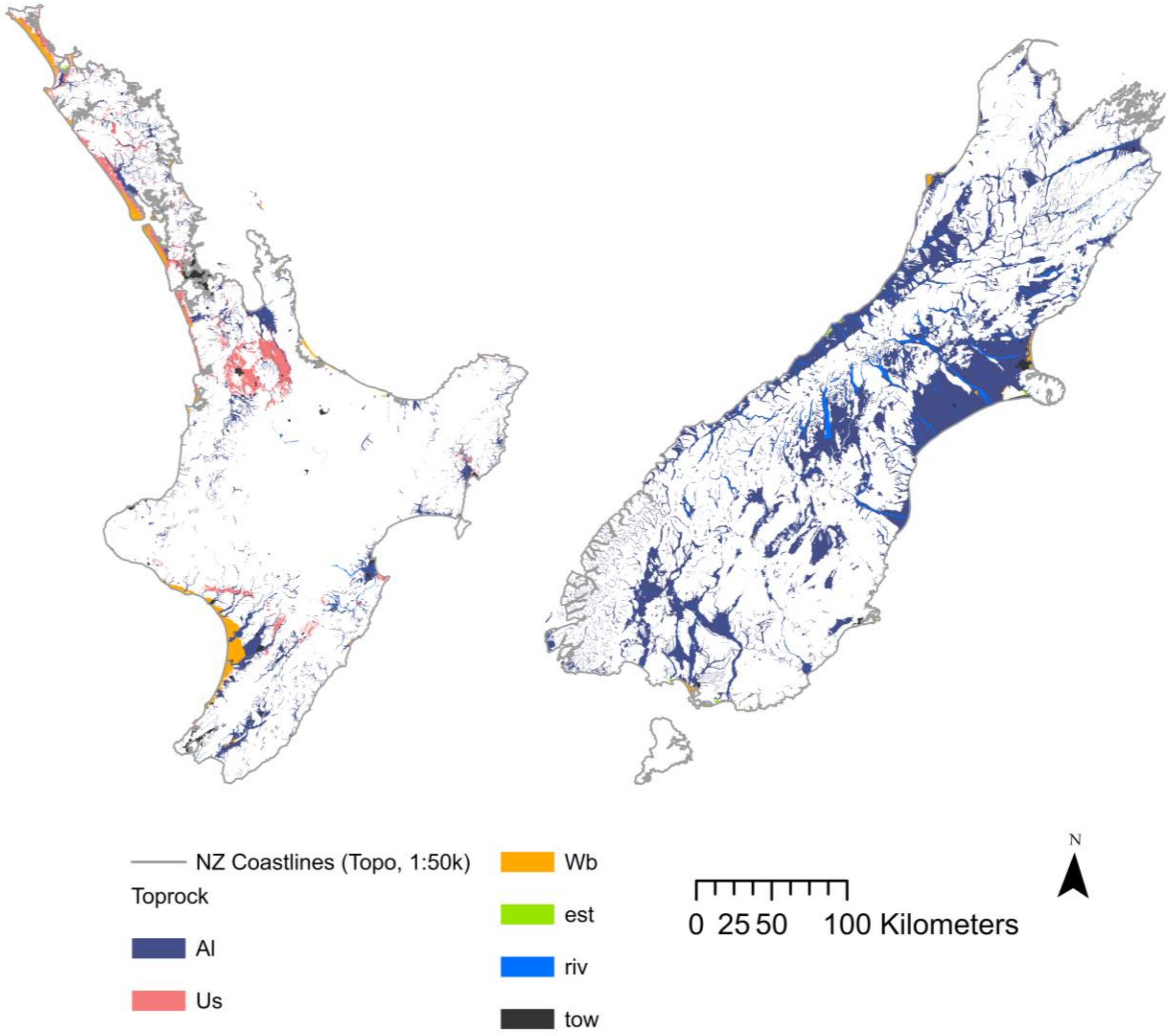
Zarnetske, P. L., Hacker, S. D., Seabloom, E. W., Ruggiero, P., Killian, J. R., Maddux, T. B., & Cox, D. (2012). Biophysical feedback mediates effects of invasive grasses on coastal dune shape. *Ecology, 93*(6), 1439–1450. http://www.jstor.org/stable/23213773

Zarnetske, P., Hacker, S., Ruggiero, P., & Seabloom, E. (2015). Coastal foredune evolution: The relative influence of vegetation and sand supply in the US Pacific Northwest. *Journal of the Royal Society Interface, 12*(106). https://doi.org/10.1098/rsif.2015.0017.

Zhao, Q., Yu, L., Du, Z., Peng, D., Hao, P., Zhang, Y., Gong, P. (2022). An Overview of the Applications of Earth Observation Satellite Data: Impacts and Future Trends. *Remote Sensing, 14*(8), 1863-1880. https://doi.org/10.3390/rs14081863.

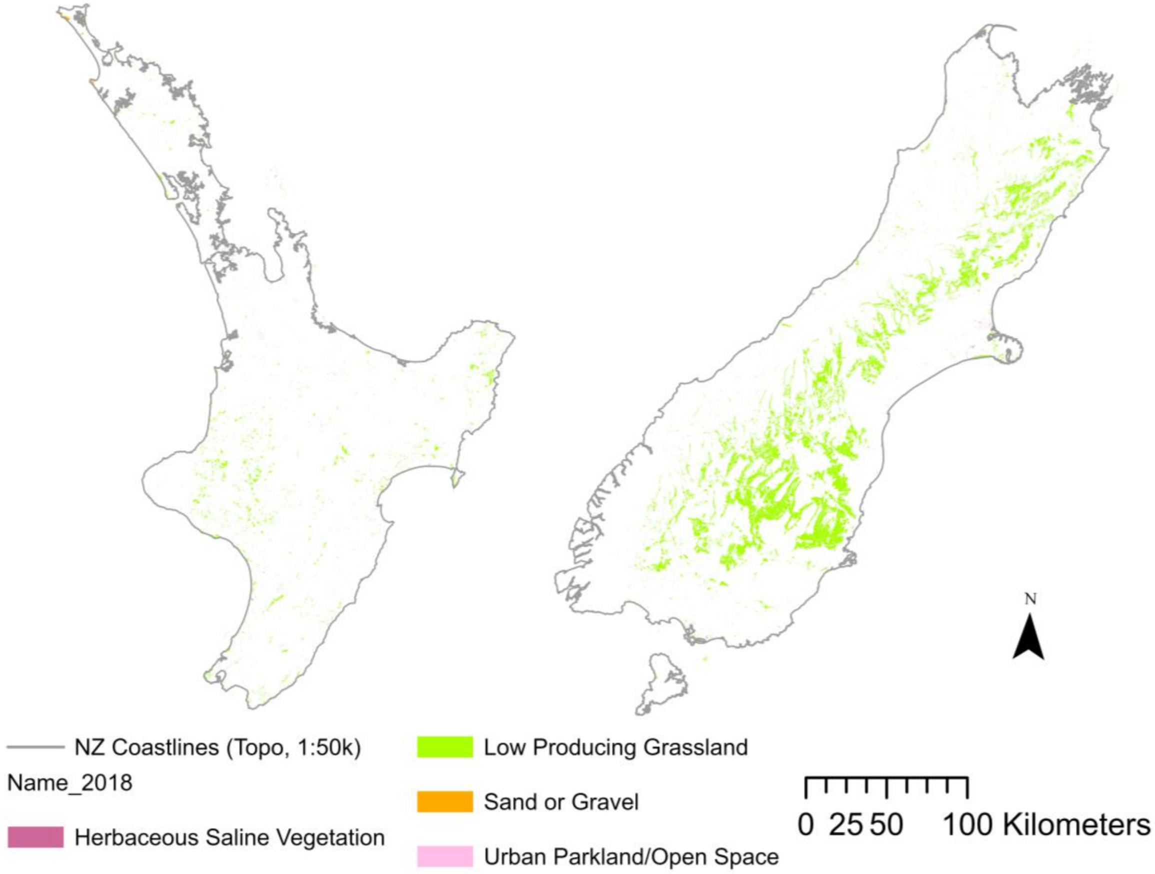
Appendices

Chapter 2 Appendices



Chapter 2. Appendix 1. Selected LRI surface lithology types that could potentially form active dunes.

The lithology types are sourced from the New Zealand Land Resource Inventory, third edition (Newsome, 1987; Newsome et al. 2008). Types include: 'Wb' = Wind blown sand, 'Us' = Unconsolidated sand, 'est' = Estuary, 'riv' = River, 'tow' = Town, 'Al' = Aluvium. The arrangement of the three main constituent islands of the Aotearoa New Zealand archipelago given here is to show all islands on the same page. The correct geographic placement is the North Island, Te-Ika-a-Māui (left hand side) above the South Island, Te-Waipounamu and Stewart Island, Rakiura (right hand side).



Chapter 2. Appendix 2. Selected land cover types that could potentially form active dunes.

New Zealand Land Cover Database v.5.0 (Landcare Research / Manaaki Whenua, 2020)

Chapter 2. Appendix 3. List of sources from which plant community composition data was compiled

Data for biotic indicators

Site	Data source
Cable Bay	Shaft, L. (2022). Coastcare co-ordinator, Northland Regional Council -Te Kaunihera ā rohe o Te Taitokerau (personal correspondence – email, 31/05/2022).
Coal River	Johnson, P. (1992). The Sand Dune and Beach Inventory of News Zealand II. South Island and Stewart Island. Pg 61 (Coal River).
	West, C. J. (n.d). Coal River Beach Weeds (B279). New Zealand Plant Conservation Network. Retrieved October 28th, 2022 from https://www.nzpcn.org.nz/publications/plant-lists/plant-lists-by-region/coal-river-beach-weeds-b279/
Himatangi Beach	Rapson, G.L., Smith, A., Murphy, A.L. (2016). Sand-dune vegetation of the Foxtangi Region, Manawatu Coast, New Zealand. Report to the Department of Conservation by the Ecology Group, Institute of Agriculture and Environment, Massey University, Palmerston North, New Zealand. Retrieved October 28th, 2022, from https://www.massey.ac.nz/massey/expertise/profile.cfm?stref=361030
	Milne, R., Sawyer, J. (2002). Coastal foredune vegetation in Wellington Conservancy, current status and future management. Published by Department of Conservation, Wellington Conservancy. Retrieved October 28th, 2022 from https://www.doc.govt.nz/about-us/science-publications/conservation-publications/native-plants/coastal-foredune-vegetation-in-wellington-conservancy/
	Ogle, C.C., (2008). Vegetation and threatened and adventive plants of the Wanganui sand country. Paper presented to the Dune Restoration Trust, March, 2008. Coastal Restoration Trust. Retrieved October 28th, 2022 from https://ref.coastalrestorationtrust.org.nz/site/assets/files/5987/weeds_and_indigenous_biodiversity_of_the_fed_northern_half_2008.pdf
	Manawatu District Council (2019). Coastal Reserves Management Plan. Manawatu District Council. Retrieved October 28th, 2022 from mdc.govt.nz/documents/plans/reserve-management-plans
Kaitorete Spit	Tordoni, E., Bacaro, G., Weigelt, P., Cameletti, M., Janssen, J. A. M., Acosta, A. T. R., … Kreft, H. (2021). Disentangling native and alien plant diversity in coastal sand dune ecosystems worldwide. Journal of Vegetation Science, 32(1). [Dataset for New Zealand component, supplied by Hannah Buckley, collected 2010]. https://doi.org/10.1111/jvs.12961
	Hoosen, S. (2017). Site Significance Statement, Kaitorete Spit. SES/E/2. In, Christchurch (N.Z.). City Council. (2017). Christchurch District Plan. [Appendix 9.1.6.1 Schedule of Sites of Ecological Significance, site SES/E/2]. Christchurch, New Zealand: The Council, 2017. Retrieved October 28th, 2022 from https://districtplan.ccc.govt.nz/pages/plan/book.aspx?exhibit=DistrictPlan
	Hetherington, J., & Bastow Wilson, J. (2014). Spatial associations between invasive tree lupin and populations of two katipo spiders at Kaitorete Spit, New Zealand. New Zealand Journal of Ecology, 38(2), 279–287.

Chapter 2, Appendix 3 (cont.)

Site	**Data source**
	Case, B. S., Buckley, H. L., Fake, M., Bryan, S., & Bilkey, J. (2019). Assessing the use of UAV-collected data for characterising the distributions and frequencies of sand dune vegetation cover types at Kaitorete Spit, Canterbury. DOC Research & Development Series 359, 25. Retrieved October 28th, 2022 from https://www.doc.govt.nz/globalassets/documents/science-and-technical/doc-research-and-development-series/drds359entire.pdf
	Hutchison, M., Patrick, B. (2020). Canterbury Botanical Society report on Kaitorete Spit October field trip. Trilepidea, Newsletter of the New Zealand Plant Conservation Network. No.134, Jan 2015. Retrieved October 28th, 2022 from https://www.researchgate.net/publication/345753083_Canterbury_Botanical_Society_report_on_Kaitorete_Spit_October_field_trip
	New Zealand National Vegetation Survey Databank (2012). Birdlings Flat Gravel Beach Survey 2012 – Recce inventory (National Vegetation Survey). https://www.nvs.landcareresearch.co.nz.
Mangangu Stream Beach	Partridge, T. R. (1992). The sand dune and beach vegetation inventory of New Zealand: 1. North Island. DSIR Land Resources scientific report (Vol. 15).
	Ryan, C. (2022). [Site visit]
Muriwai Beach	Partridge, T. R. (1992). The sand dune and beach vegetation inventory of New Zealand: 1. North Island. DSIR Land Resources scientific report (Vol. 15).
	Ryan, C. (2022). [Site visit]
New Brighton	Tordoni, E., Bacaro, G., Weigelt, P., Cameletti, M., Janssen, J. A. M., Acosta, A. T. R., … Kreft, H. (2021). Disentangling native and alien plant diversity in coastal sand dune ecosystems worldwide. Journal of Vegetation Science, 32(1). [Dataset for New Zealand component, supplied by Hannah Buckley, collected 2010]. https://doi.org/10.1111/jvs.12961
	Shadbolt, A. (2014). Site significance statement, Christchurch Coastal Strip, SES/LP/6. In, Christchurch (N.Z.). City Council. (2017). Christchurch District Plan. [Appendix 9.1.6.1 Schedule of Sites of Ecological Significance, site SES/LP/6]. Christchurch, New Zealand: The Council, 2017. Retrieved October 28th, 2022 from https://districtplan.ccc.govt.nz/pages/plan/book.aspx?exhibit=DistrictPlan
	Ryan, C. (2022). [Site visit]
Ōhope Beach	Wildlands (2008). Bay of Plenty region sand dune vegetation mapping and condition assessment methods for Tauranga Ecological District. Prepared for Environment Bay of Plenty, P.O Box 364, Whakatane. Retrieved October 28th, 2022 from https://cdn.boprc.govt.nz/media/32395/EnvReport-201002-SandDuneVegetationMapping.pdf
	Bay of Plenty Regional Council (2017). [Duneland Survey data sheets for 2017 and 2008, supplied by Bay of Plenty Regional Council].
	Coastcare Bay of Plenty (n.d). Dune Management in Ohope. Coastcare Bay of Plenty Retrieved October 28th, 2022 from https://cdn.boprc.govt.nz/media/30282/CoastCare-090527-RestorationOhope.pdf
	Ryan, C. (2022). [Site visit]

Chapter 2, Appendix 3 (cont.)

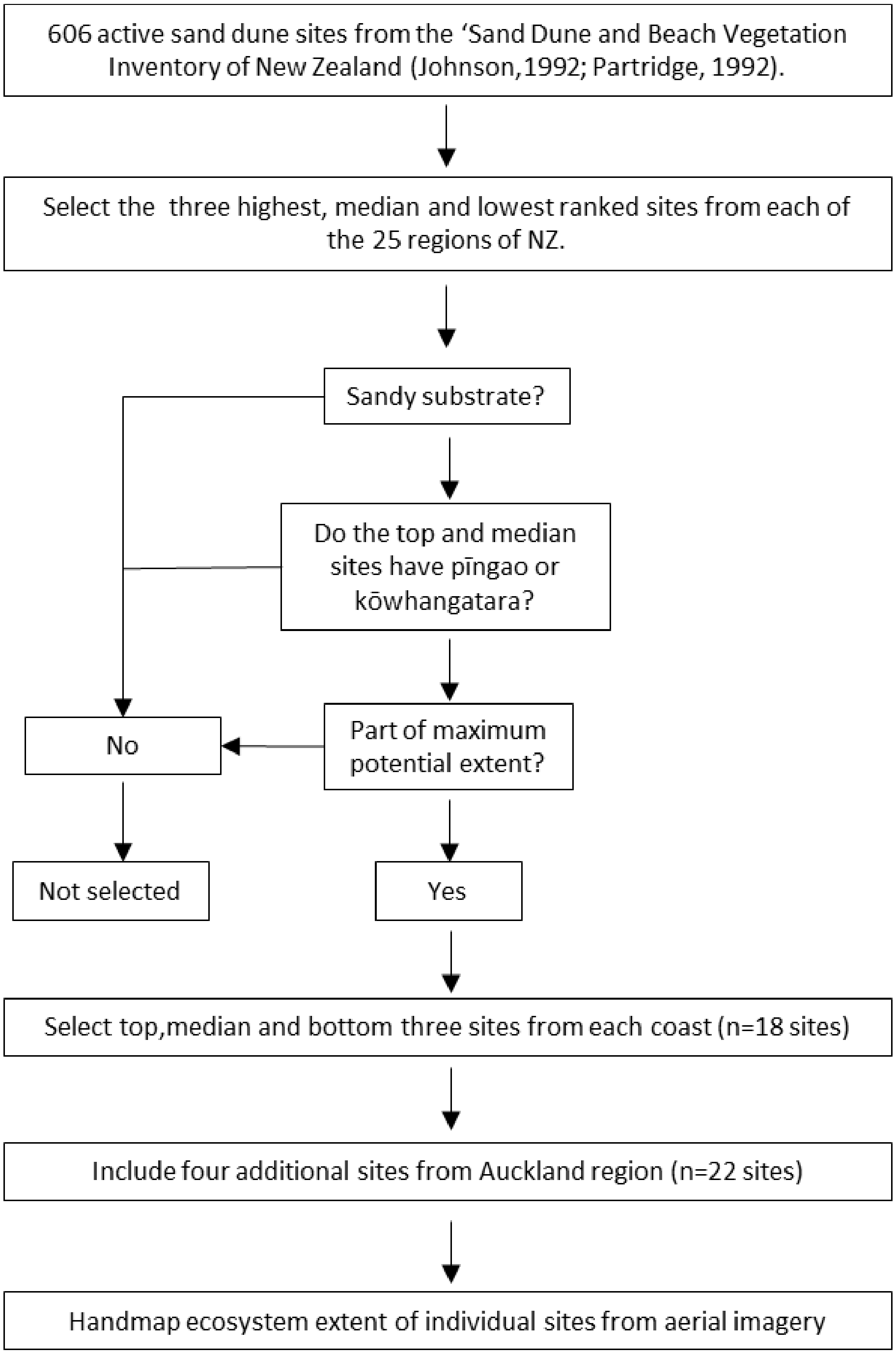
Site	**Data source**
Ōrewa Beach	Bishop, C. (2022) [Site visit]
Ōtaki Beach	Tordoni, E., Bacaro, G., Weigelt, P., Cameletti, M., Janssen, J. A. M., Acosta, A. T. R., … Kreft, H. (2021). Disentangling native and alien plant diversity in coastal sand dune ecosystems worldwide. Journal of Vegetation Science, 32(1). [Dataset for New Zealand component, supplied by Hannah Buckley, collected 2010]. https://doi.org/10.1111/jvs.12961
	Milne, R., Sawyer, J. (2002). Coastal foredune vegetation in Wellington Conservancy, current status and future management. Published by Department of Conservation, Wellington Conservancy. Retrieved October 28th, 2022 from https://www.doc.govt.nz/about-us/science-publications/conservation-publications/native-plants/coastal-foredune-vegetation-in-wellington-conservancy/
	Waiotahi Stream and Dune Care (n.d). Waiotahi Stream and Dune Care Ōtaki. Retrieved October 28th, 2022 from http://www.waitohudunecare.org/
Ōtama Beach	Auckland Botanical Society (2005). Field Trip: Coromandel Peninsula, Auckland Anniversary Weekend 28/01/05 to 1/02/05. Auckland Botanical Society. Retrieved October 28th, 2022 from https://bts.nzpcn.org.nz/articles/field-trip-coromandel-peninsula-auckland-anniversary-weekend-28-01-05-to-1-02-05/
	Havel, D. (2017). Site assessment, Otama Beach Dunes.
	Megan Graeme (ecologist, personal communications – phone call, 03/03/2022).
	Otama Reserves Group (n.d). History of Otama. Retrieved October 28th, 2022 from https://www.otamareservesgroup.co.nz/history-of-otama/
	Brown (2011). Coromandel Peninsula Landscape Assessment, draft. [see Landscape Unit 38, Otama Beach]. Prepared for the Thames Coromandel District Council. Retrieved October 28th, 2022 from https://www.tcdc.govt.nz/Global/1_Your%20Council/Council%20Projects/Current%20Projects/District%20Plan%20Review/Introduction.pdf
Pakiri Beach	Goldwater, N. (2013). The botanical treasures of Middle Pakiri Beach. Auckland Botanical Society, 2013. Retrieved October 28th, 2022 from https://bts.nzpcn.org.nz/articles/the-botanical-treasures-of-middle-pakiri-beach-farm-pakiri/
	Shaft, L. (2022). Coastcare co-ordinator, Northland Regional Council -Te Kaunihera ā rohe o Te Taitokerau (personal correspondence – email, 6 July, 2022).

Chapter 2, Appendix 3 (cont.)

Site	**Data source**
Paraparaumu Beach	Bergin, D., 2005. Establishment of spinifex planting trial Paraparaumu Beach, Kapiti Coast. CSIRO and SCION, Ensis, Rotorua, New Zealand. Retrieved October 28th, 2022 from https://takutaikapiti.nz/wp-content/uploads/2020/11/Establishment-of-spinifex-planting-trial_Paraparaumu-Beach_Bergin_Ensis_2005.pdf
	New Zealand National Survey Databank (1991). Foxton Protected Natural Area Programme / National Vegetation Survey 1989 – 91. [Data sheets supplied by Landcare Research / Manaaki Whenua]. https://www.nvs.landcareresearch.co.nz)
	Kāpiti Coast District Council (n.d). Caring for Sand Dunes in Kāpiti. Retrieved October 28th, 2022 from https://waikawabeach.org.nz/wp-content/uploads/2019/02/1092-kap-dune-posterbrochure-reading-ff2-for-web.pdf
Ponaki Beach (also known as Ngakengo Beach)	Lux, J., Holland, W., Rate, S., Beadel, S. (2009). Natural areas of Te Paki Ecological District: reconnaissance survey report for the Protected Natural Areas Programme, New Zealand Department of Conservation. [Ngakengo Beach, PNAP Survey no. N02/062]. Retrieved October 28th, 2022 from https://www.doc.govt.nz/about-us/science-publications/conservation-publications/land-and-freshwater/land/northland-conservancy-ecological-districts-survey-reports/natural-areas-of-te-paki-ecological-district/
Rarawa Beach	Jane, G., Donaghy, G. (2008). Rarawa Beach dunes (RARW). [plant species list]. New Zealand Plant Conservation Network. Retrieved October 28th, 2022 from https://www.nzpcn.org.nz/publications/plant-lists/plant-lists-by-region/rarawa-beach-dunes-rarw/
	Northland Age (2021). Ngataki school kids restore Rarawa Beach dunes with *F. spiralis*. NZ Herald. Retrieved October 28th, 2022 from https://www.nzherald.co.nz/northland-age/news/ngataki-school-kids-restore-rarawa-beach-dunes-with-pingao/PWM7D4GMKKUM7ZS6GI77YL5JTY/
Scott Point to Waikoropupunoa Stream	Northland Regional Council (2014). Northland Regional Landscape Assessment. [Unit name – Twilight Beach / Kahokawa Beach Dunefields, Wetlands and Bush]. Retrieved October 28th, 2022 from https://www.nrc.govt.nz/media/kiwe3v4y/twilightbeachkahokawabeachdunefieldandwetlands.pdf
	Lux, J., Holland, W., Rate, S., Beadel, S. (2009). Natural areas of Te Paki Ecological District: reconnaissance survey report for the Protected Natural Areas Programme, New Zealand Department of Conservation. Retrieved October 28th, 2022 from https://www.doc.govt.nz/about-us/science-publications/conservation-publications/land-and-freshwater/land/northland-conservancy-ecological-districts-survey-reports/natural-areas-of-te-paki-ecological-district/
	Partridge, T. R. (1992). The sand dune and beach vegetation inventory of New Zealand: 1. North Island. DSIR Land Resources scientific report (Vol. 15).

Chapter 2, Appendix 3 (cont.)

Site	**Data source**
Smoky Beach	Johnson, P. (1992). The Sand Dune and Beach Inventory of News Zealand II. South Island and Stewart Island. Pg 61, Coal River.
	Rance, B. (personal communication, email, 25/03/2022)
	Department of Conservation (2012). Stewart Island / Rakiura Conservation Management Strategy and Rakiura National Park Management Plan (pp. 1–316). New Zealand Department of Conservation. Retrieved October 28th, 2022 from https://www.doc.govt.nz/about-us/our-policies-and-plans/statutory-plans/statutory-plan-publications/conservation-management-strategies/stewart-island-rakiura/
	Hilton, H., Konlechner, T. (2021). The Rakiura Dune Restoration Programme (1999-2021). Lessons Learned from 21 Years of Operations, Monitoring & Research. School of Geography, University of Otago, PO Box 56, Dunedin, New Zealand.
Spirits Bay	Lux, J., Holland, W., Rate, S., Beadel, S. (2009). Natural areas of Te Paki Ecological District: reconnaissance survey report for the Protected Natural Areas Programme, New Zealand Department of Conservation. [Site name Kapowairua, Survey no. N02/027]. Retrieved October 28th, 2022 from https://www.doc.govt.nz/about-us/science-publications/conservation-publications/land-and-freshwater/land/northland-conservancy-ecological-districts-survey-reports/natural-areas-of-te-paki-ecological-district/
St Kilda and Clair Beaches	Johnson, P. (1992). The Sand Dune and Beach Inventory of News Zealand II. South Island and Stewart Island.
Tāwharanui	Ryan, C. (2022). [Site visit] reviewed by Bishop, C.
	Tāwharanui Open Sanctuary (n.d). What we do [revegetation]. Retrieved October 28th, 2022 from https://www.tossi.org.nz/?page_id=73
Waikouaiti Beach	Johnson, P. (1992). The Sand Dune and Beach Inventory of News Zealand II. South Island and Stewart Island.
Whatipu Beach	Auckland Botanical Society (2013). A Visit to the Whatipu Sands, West Auckland. Auckland Botanical Society. Retrieved October 28th, 2022 from https://bts.nzpcn.org.nz/articles/a-visit-to-the-whatipu-sands-west-auckland/
	Cameron, E.K. 1989: Vegetation of the Whatipu Sands, north Manukau Heads. Auckland Botanical Society Journal 44: 3–10. Retrieved October 28th, 2022 from https://bts.nzpcn.org.nz/articles/vegetation-of-the-whatipu-sands-north-manukau-heads/
	Pegman, A.P. McK., Rapson, G.L. 2005: Plant succession and dune dynamics on actively prograding dunes, Whatipu Beach, northern New Zealand. New Zealand Journal of Botany 43: 223–24



Chapter 2. Appendix 4. Workflow to select a nationally representative sub-sample of sand dune sites

Chapter 3 Appendices

Chapter 3. Appendix 1. Aerial imagery datasets used in this research:

All imagery listed was sourced from Land Information New Zealand (LINZ) Data Service under a Creative Commons open licence (CC BY 4.0), and retrieved from https://data.linz.govt.nz/data/category/aerial-photos/

Imagery used	Sites
Northland 0.4m Rural Aerial Photos (2014-2016)	Cable Bay Ponaki Beach Rarawa Beach Muriwai Beach Scott Point to Waikoropupunoa Stream Spirits Bay
Southland 0.75m Rural Aerial Photos (2005-2011)	Coal River Smoky Beach
Waikato 0.3m Rural Aerial Photos (2016-2019)	Ōtama Beach
Auckland 0.5m Rural Aerial Photos (2010-2012)	Tāwharanui Pakiri Beach
Canterbury 0.3m Rural Aerial Photos (2015-2016)	New Brighton Kaitorete Spit
Manawatū - Whanganui 0.3m Rural Aerial Photos (2015-2016)	Himatangi Beach
Bay of Plenty 0.1m Urban Aerial Photos (2018-2019)	Ōhope Beach
Auckland 0.075m Urban Aerial Photos (2017)	Ōrewa Beach
Dunedin 0.1m Urban Aerial Photos (2018-2019)	St. Kilda and St. Clair Beaches Waikouaiti Beach
Auckland 0.075m Urban Aerial Photos (2017)	Whatipu Beach
Wellington 0.3m Rural Aerial Photos (2021)	Ōtaki Beach Paraparaumu Beach

Chapter 3. Appendix 2. Normalised Green-Red Vegetation Index (NGRDI) Equation

NGRDI was calculated as:

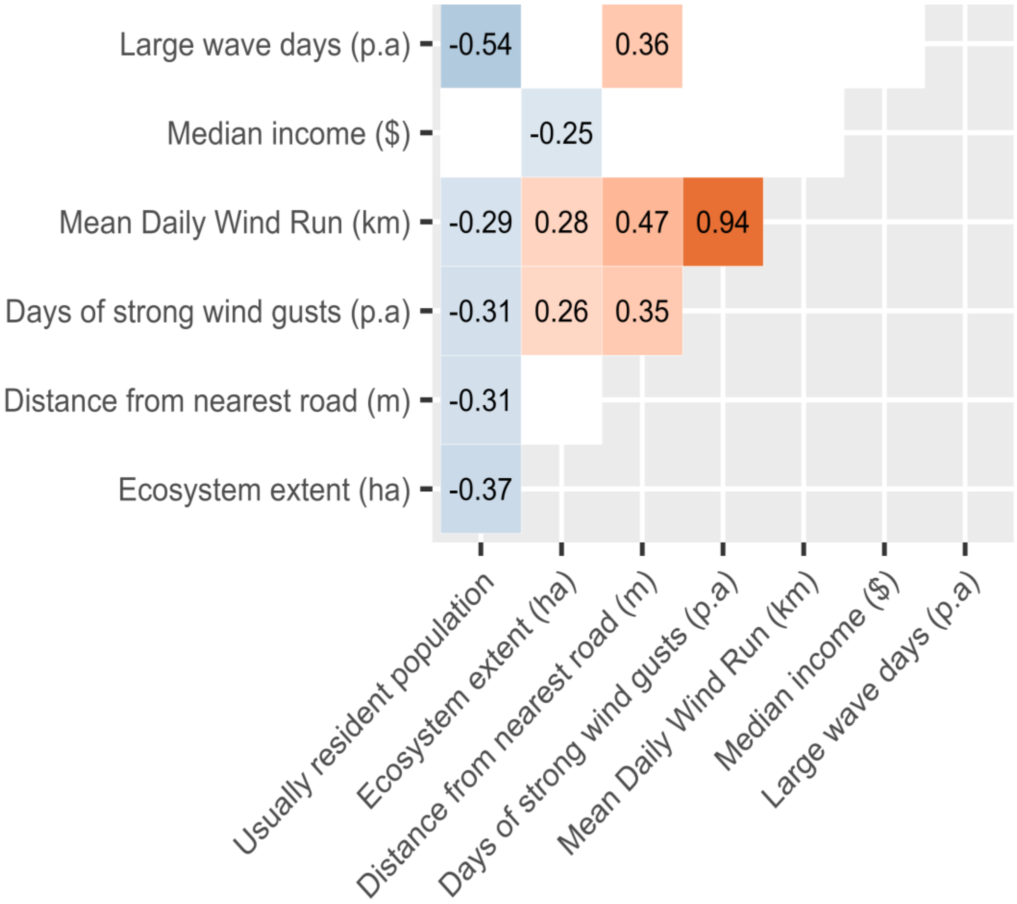
(GREEN-RED) / (GREEN+RED) (Hunt et al., 2005), where GREEN was the part of the electromagnetic spectrum with wavelengths between 495 - 570 nm, and RED was between 620 – 750 nm (Hunt et al., 2005).

Chapter 3. Appendix 3. Environmental variables used in the CART Classification Tree analysis

Variable name	**Dataset and author**	**Dataset description**
Adjacent land cover type	Land Cover Database version 5.0, Mainland, New Zealand. Manaaki Whenua Landcare Research (2019).	The most common, adjacent land cover type in terms of area (ha), from the Land Cover Database within a 400 m radius from the centre of sample plots. The seaward side of the coastline (i.e. the sea) is excluded.
Coast	Ryan et al. (2023).	The coast where the active dune ecosystems in this research occur.
Distance from nearest road (m)	NZ Roads (Addressing). Land Information NZ.	Distance (m) to the closest road to a sample plot (all types of roads).
Ecosystem extent (ha)	Ryan et al. (2023).	The extent of active dune ecosystems, digitised from the same aerial imagery used for classification in this research.
Days of large waves (p.a)	Number of days waves exceeded 4 m for at least 12 hours per year in coastal regions, over the years 2008 - 2015. Gorman (2016), for the Ministry for the Environment, New Zealand.	Coastal extreme wave indices are derived for eighteen coastal regions around Aotearoa. Four-metre-tall waves are considered extreme in the northernmost parts of Aotearoa but are more common in the south. In all cases, they are the number of days when large waves occur over 12 hours or more (p.a).
Days of strong wind gusts (p.a)	Number of days of wind gusts ≥ 24 knots (per annum). CliFlo, New Zealand's National Climate Database. NIWA (2023).	Days of wind gusts ≥ 24 knots across all months for the year or years that aerial imagery used in this research was captured. Data was taken from the nearest climate station to research beaches in operation over the research period ($\bar{x}$ = 29.4 km, min = 6.4, max = 77.3, SD =20.5).
Mean daily wind run (km)	CliFlo, New Zealand's National Climate Database. NIWA (2023).	The mean distance covered by the wind in a day (km), over a month, across all months for the year, for years that aerial imagery used in this research was captured. Data was taken from the nearest climate station to research beaches in operation over the research period ($\bar{x}$ = 29.4 km, min = 6.4, max = 77.3, SD =20.5).

Chapter 3, Appendix 3 (cont.)

Variable name	**Dataset and author**	**Dataset description**
Median personal income ($)	Age and income in 2013 by Territorial Authorities and Local Boards (Statistical Area 2). Stats NZ (2017).	Median personal income for residents. Due to high non-response rates, median personal income is used because Stats NZ states, "total household income is considered: Poor: fit for use – but use with caution due to some significant data quality issues." Note median personal income is income from all sources. An alternative would be the "Household Economic Survey", but this is only a sample from 5000 homes and is less well suited to geography-based analyses. Data was collected for census mesh blocks within 1km of plots.
Total rainfall (mm)	CliFlo, New Zealand's National Climate Database. NIWA (2023).	Mean total rainfall (mm) across all months for the year or years, that the aerial imagery used in this research was captured. Data was taken from the nearest climate station to research beaches in operation over the research period ($\bar{x}$ = 29.4 km, min = 6.4, max = 77.3, SD =20.5).
Usually resident population	Census counts, 2013, New Zealand (Statistical Area 2). Stats NZ (2019).	The population usually living in an area on census day, 2013. Data was collected for census mesh blocks (Statistical Area 2) within 1 km of plots. Statistical Area 2 meshblocks reflect communities that interact together socially and economically. The population of a meshblock varies depending on the authority (district or regional councils) and whether the area is urban or rural. The geographical size of vary depending on the community sampled.



Chapter 3. Appendix 4. Pearson's pairwise correlation coefficients among the explanatory environmental variables used in the CART Classification Tree analysis.

The CART analysis predicted dominant plot vegetation type for 135 plots on 21 beaches across Aotearoa New Zealand. Statistically non-significant relationships are represented by white squares. Statistically significant relationships are represented by squares shaded either blue (negative correlations) or red (positive correlations). The matrix shows that most variables were moderately or not significantly correlated, except for mean daily wind run and days of strong wind gusts, which were highly correlated.

Chapter 3: Appendix 5. Distances between plots on beaches and climates stations (rain and wind data), and the associated extreme coastal wave index region (modelled extreme wave data).

The average distance between climate stations and beaches was 25.2 km, and the average number of beaches to one climate station was 1.6, with a maximum of three. The average number of sites per modelled extreme wave index region was two, with a maximum of four. A total of fifteen sites shared extreme wave index data. Distances are given as a straight line between the climate station and the centre plot on associated beaches.

Beach	Distance to climate station (km)	Climate station no.	Extreme coastal wave index region
St Kilda and St Clair Beaches	0.7	1002	Chalmers
Tāwharanui	10.5	1002	Colville
New Brighton	11.3	1002	Conway
Pakiri Beach	11.6	1340	Colville
Ōrewa Beach	12.6	1340	Colville
Spirits Bay	13.5	1400	Kaipara
Scott Point to Waikoropupunoa Stream	15.0	1520	Kaipara
Ōtama Beach	15.6	1673	Colville
Ōtaki Beach	16.6	1962	Stephens
Ōhope Beach	18.5	1962	Plenty
Cable Bay	19.5	3275	Brett
Himatangi Beach	27.6	3275	Stephens
Whatipu Beach	27.9	3275	Raglan
Ponaki Beach	30.1	4903	Brett
Waikouaiti Beach	33.2	4960	Chalmers
Coal River	33.7	5909	Puysegur
Paraparaumu Beach	37.1	9533	Stephens
Rarawa Beach	43.9	15752	Brett
Kaitorete Spit	47.0	15752	Rangitata
Muriwai Beach	51.0	18183	Kaipara
Smoky Beach	70.0	18183	Foveaux

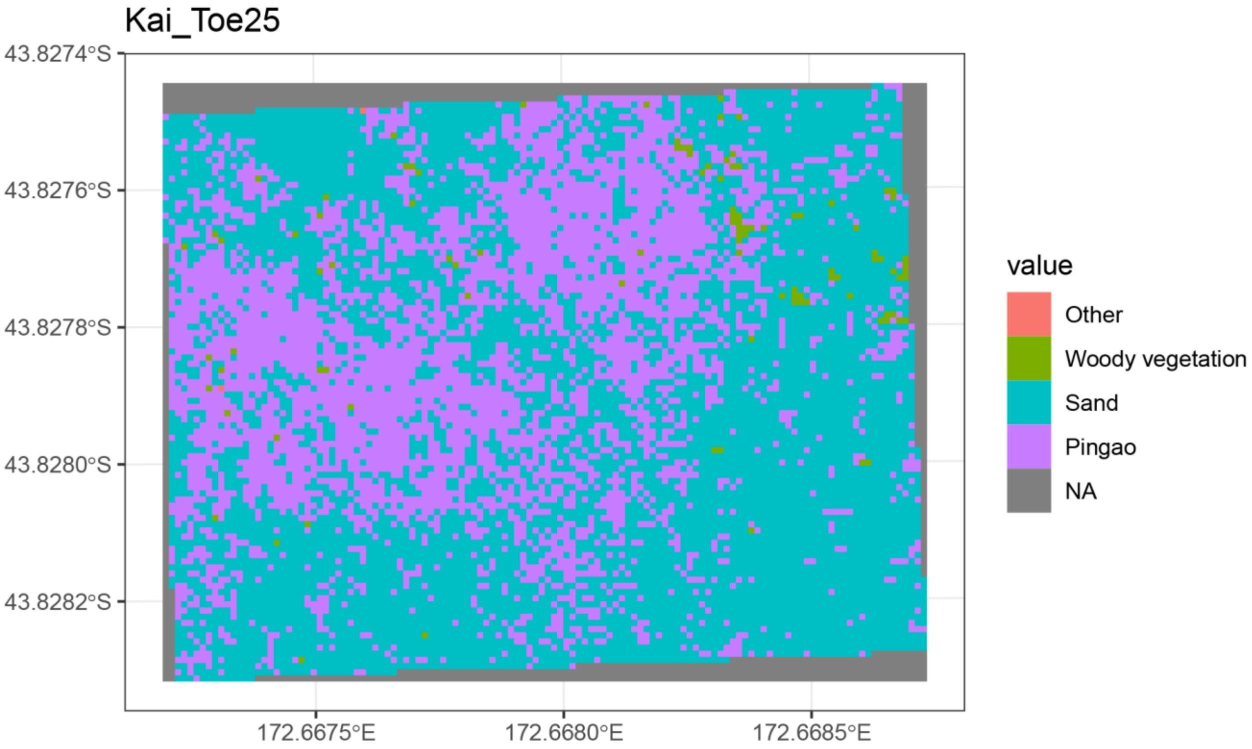
Chapter 3. Appendix 6. Examples of species or objects that could potentially occur in cover classes discriminated in image classification.

Cover class	**Description**	**Aggregate cover class**
F. spiralis	*Ficinia spiralis* (A.Rich.) Muasya et de Lange	*F. spiralis*
S. sericeus	*Spinifex sericeus* R.Br.	*S. sericeus*
A. arenaria	*Ammophila arenaria* (L.) Link.	*A. arenaria*
Mixed native and exotic trees	Examples of native species that could occur are pōhutukawa (*Metrosideros excelsa* Sol. ex Gaertn.) and karo (*Pittosporum crassifolium* Banks et Sol. ex A.Cunn). Exotic species that could occur are Monterey pine (*Pinus radiata* D. Don) and cluster pine (*Pinus pinaster* Aiton), either wilding or in plantation, including cutover stands.	Woody vegetation
Mixed native and exotic shrubs	Examples of native shrubs that could occur are sand coprosma (*Coprosma acerosa* A.Cunn.), sand daphne (*Pimelia villosa* Sol. ex Sm.), pōhuehue (*Meuhlenbeckia complexa* A.Cunn.) tauhinu (*Ozothamnus leptophyllus* G.Forst.), karamū (*Coprosma robusta* Raoul), taupata (*Coprosma repens* A. Rich), kānuka (*Kunzea robusta* de Lange et Toelken), māhoe (*Melicytus ramiflorus* J.R.Forst. *et G.Forst. subsp. Ramiflorus*). Exotic woody shrub species could be tree lupin (*Lupinus arboreus* Sims), gorse (*Ulex europaeus* L.), barberry (*Berberis glaucocarpa* Stapf), boxthorn (*Lycium ferocissimum* Miers) and wattle (*Acacia mearnsii* De Wild).	Woody vegetation
Exotic trees and shrubs	Examples are given in the tree and shrub cover classes above.	Woody vegetation
Shadow from trees and shrubs	Shadow from trees and shrubs when clearly falling within an area of woody vegetation.	Woody vegetation
Exotic pasture grasses	Examples could include buffalo grass (*Stenotaphrum secundatum* Walter), kikuyu grass (*Cenchrus clandestinus* (Hochst. ex Chiov.) Morrone), and Bermuda grass (*Cynodon dactylon* (L.) Pers.)	Exotic pasture grasses

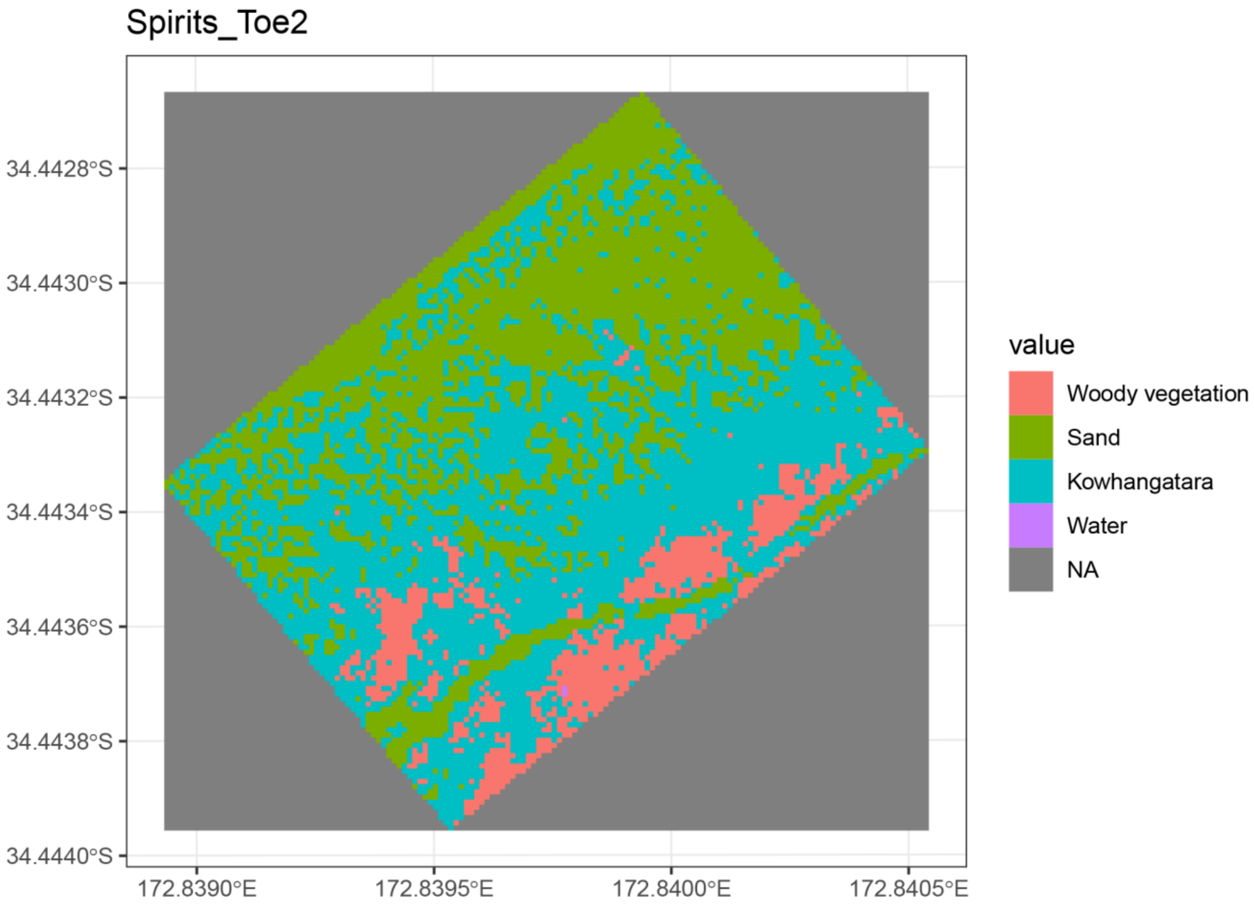
Cover class	**Description**	**Aggregate cover class**
Exotic herbaceous species	Examples could include sea rocket (Cakile edentula (Bigelow) Hook. var. edentula) and South African ice plant (*Carpobrotus edulis* (L.) N.E.Brown).	Other herbaceous species
Mixed native and exotic herbaceous species	Examples could include oioio (*Apodasmia similis* (Edgar) Briggs et L.A.S. Johnson); wiwi (*Ficinia nodosa* (Rottb.) Goetgh., Muasya et D.A.Simpson), bracken (*Pteridium esculentum* (G. Forst.) Cockayne), harakeke (*Phormium tenax* J.R.Forst. et G.Forst.) and toetoe (*Austroderia fulvida* (Buchanan) N.P.Barker et H.P.Linder). Exotic species could include Pampas grass (*Cortaderia selloana* (Schult. et Schult.f.) Asch. et Graebn.).	Other herbaceous species
Shadow	Shadows from dunes, vegetation, or impermeable structures or any other feature (except for trees and shrubs, included above).	Other
Buildings	Houses, sheds or other similar structures.	Other
Bare earth	Patches of exposed non-sand sediment.	Other
Other impermeable	Small impermeable features such as cars or signs.	Other
Dry sand	Areas of dry sand without tidal or estuarine water.	Sand
Water over sand	Sand areas with shallow tidal or estuarine water present.	Sand
Wet sand	Sand areas still wet and darker in colour from tidal water.	Sand
Gravel	Weathered, rounded rock fragments, larger than coarse sand and smaller than pebbles.	Sand
Driftwood	Trees and branches washed up on the beach, generally around the high tide mark.	Sand
Rocks	Stones, rocks, boulders and bedrock.	Sand
Water	Estuaries, rivers or ponded water (this class is deeper than ‘Water over sand’)	Water

Chapter 3. Appendix 7. Selected examples of the classified output

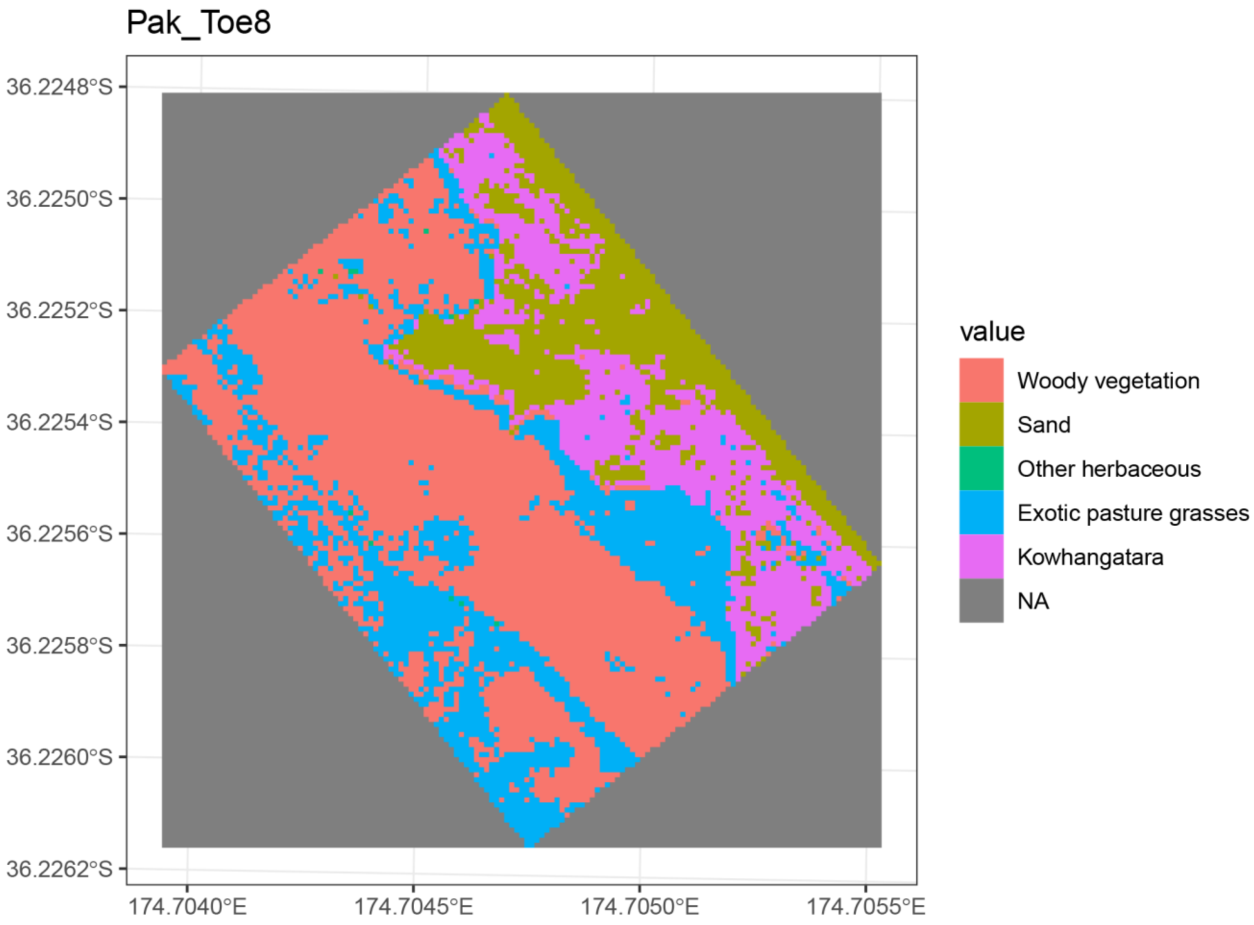
Kaitorete Spit



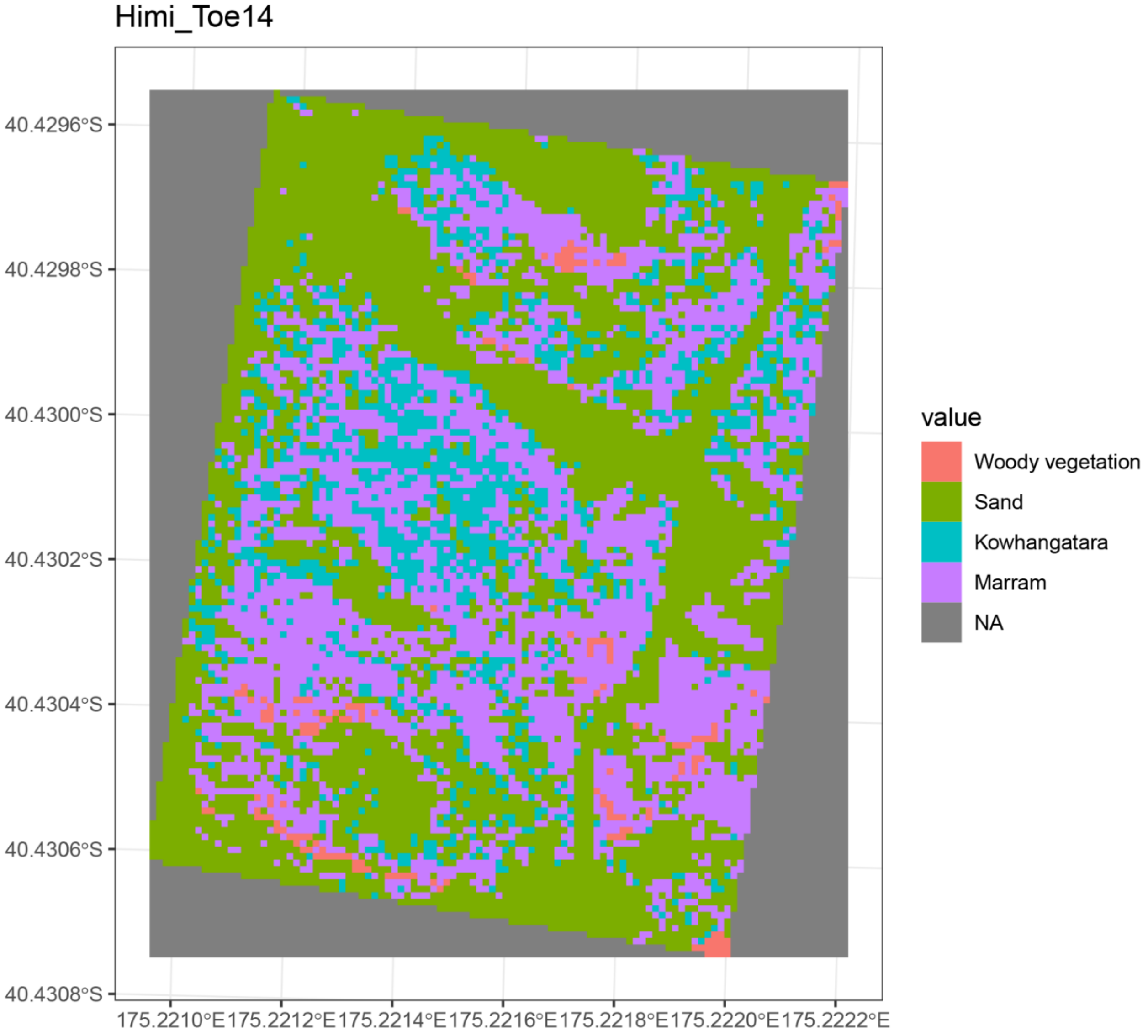
Spirits Bay


Spirits_Toe2
34.4428°S
34.4430°S
34.4432°S
34.4434°S
34.4436°S
34.4438°S
34.4440°S
172.8390°E
172.8395°E
172.8400°E
172.8405°E
value
Woody vegetation
Sand
Kowhangatara
Water
NA

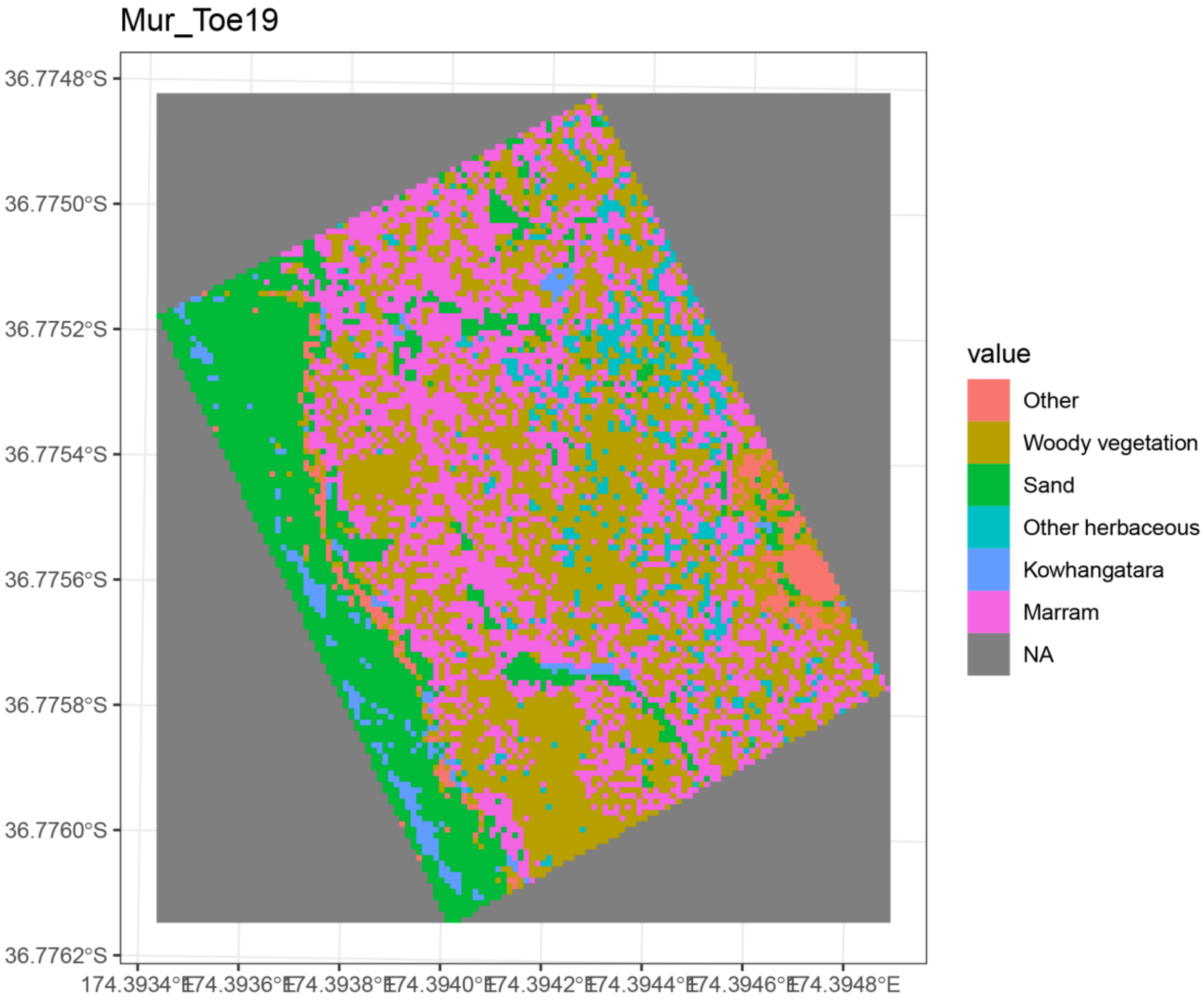
Pakiri Beach



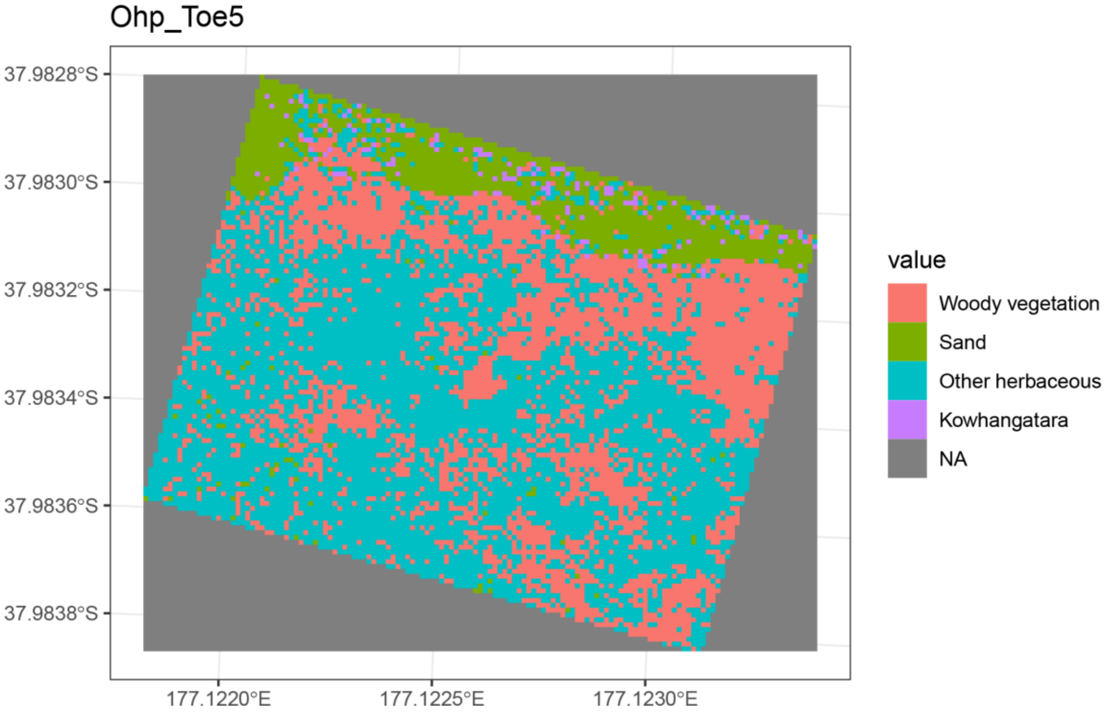
Himatangi Beach



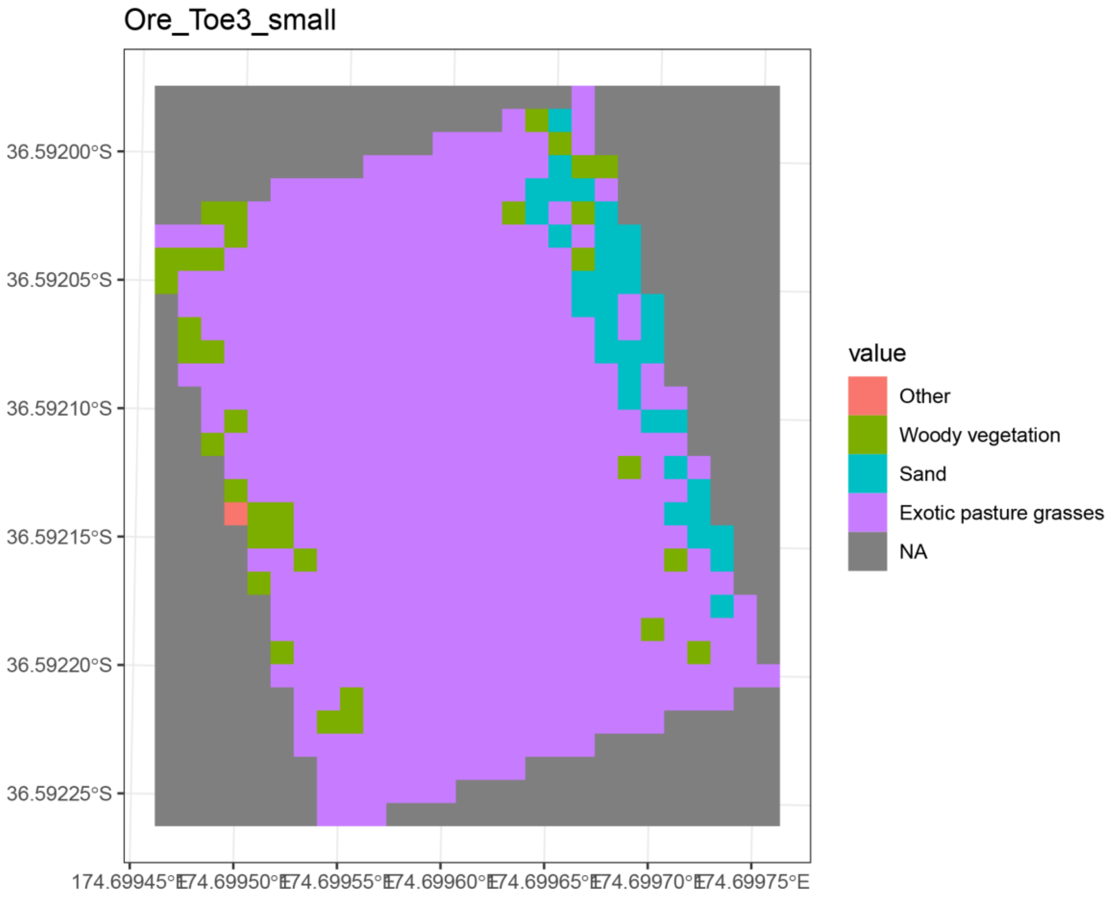
Muriwai Beach



Ōhope Beach


Ohp_Toe5
37.9828°S
37.9830°S
37.9832°S
37.9834°S
37.9836°S
37.9838°S
177.1220°E
177.1225°E
177.1230°E
value
Woody vegetation
Sand
Other herbaceous
Kowhangatara
NA

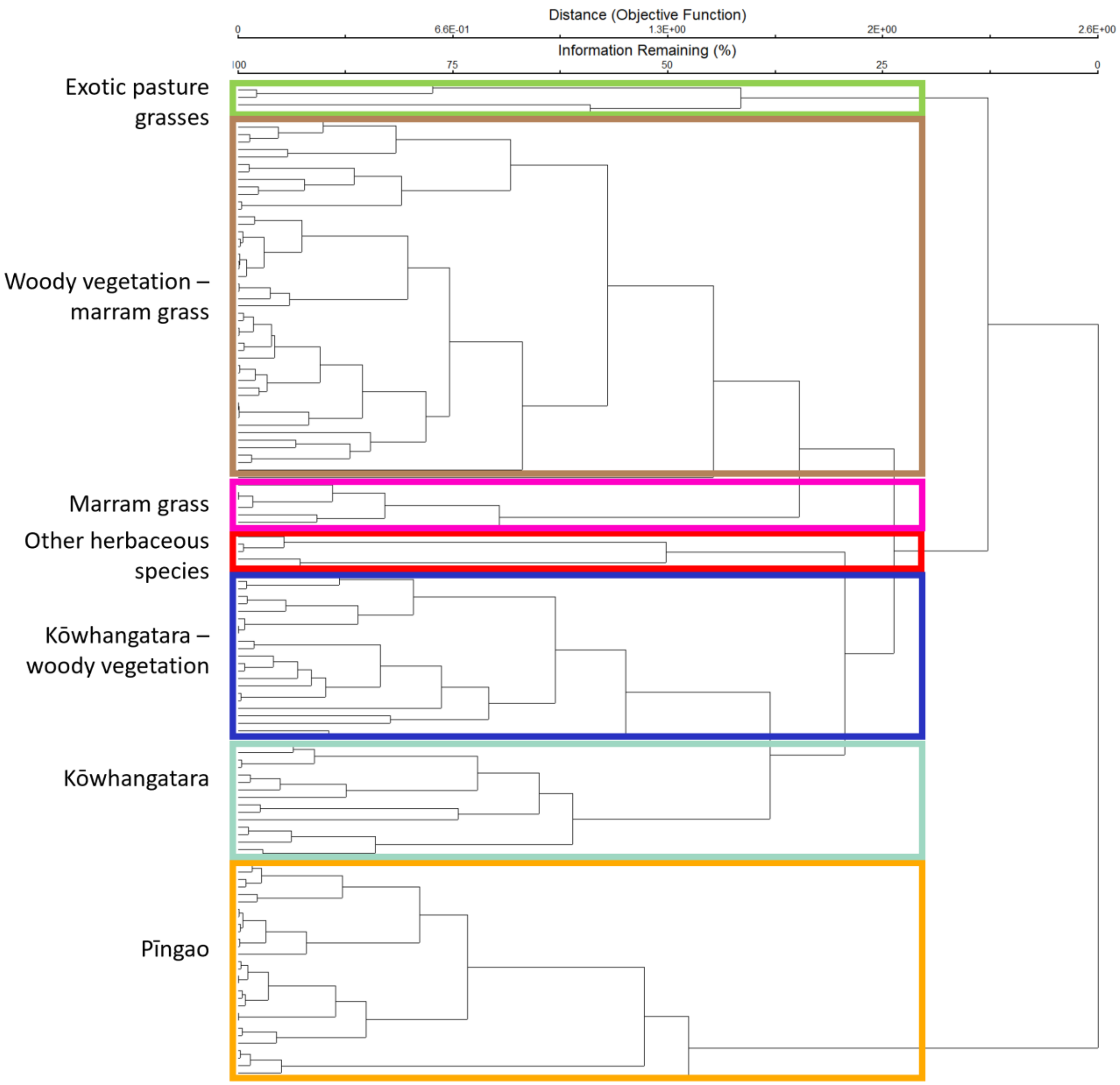
Ōrewa Beach


Ore_Toe3_small
36.59200°S
36.59205°S
36.59210°S
36.59215°S
36.59220°S
36.59225°S
174.69945°E
174.69950°E
174.69955°E
174.69960°E
174.69965°E
174.69970°E
174.69975°E
value
Other
Woody vegetation
Sand
Exotic pasture grasses
NA

Class	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	**Total**	**UA**
1	957	4	1	39	0	0	0	12	16	0	17	70	14	0	9	0	0	9	0	0	0	2	1150	0.83
2	64	125	5	4	0	0	0	0	0	0	0	3	44	0	0	0	0	0	0	0	0	2	247	0.51
3	0	0	40	0	3	0	0	0	0	0	0	1	5	0	0	0	0	0	0	0	0	0	49	0.82
4	34	11	0	716	10	0	0	13	17	2	5	17	44	0	0	0	0	9	0	0	0	14	892	0.80
5	5	0	2	109	421	0	0	22	0	1	0	2	22	0	4	0	0	0	0	0	0	9	597	0.71
6	0	0	0	27	0	11	9	3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	50	0.22
7	38	0	0	19	2	0	19	16	0	0	0	5	0	0	0	0	0	0	0	0	0	1	100	0.19
8	17	0	0	20	1	0	0	341	0	0	0	2	37	0	0	0	0	0	0	0	0	1	419	0.81
9	25	0	0	11	1	0	0	3	149	2	0	5	2	0	0	0	0	2	0	0	0	0	200	0.75
10	4	0	0	8	0	0	0	0	0	31	0	0	0	0	0	0	0	7	0	0	0	0	50	0.62
11	38	10	0	27	0	0	0	0	0	0	163	4	0	3	0	0	0	0	0	0	2	3	250	0.65
12	96	15	1	46	1	0	0	19	7	2	2	462	24	2	0	0	0	9	0	0	0	2	688	0.67
13	10	0	2	33	0	0	0	9	0	0	0	9	334	0	0	0	0	2	0	0	0	0	399	0.84
14	10	0	0	48	0	0	0	0	0	0	3	10	7	22	0	0	0	0	0	0	0	0	100	0.22
15	25	0	0	1	0	0	0	2	0	0	7	0	0	0	6	0	0	0	8	0	3	1	53	0.11
16	0	0	0	6	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	7	0.00
17	0	0	0	0	0	0	0	1	0	0	0	0	5	0	2	0	9	0	0	0	0	3	20	0.45
18	55	0	0	25	0	0	0	0	11	4	0	2	11	0	0	0	0	78	0	0	0	0	186	0.42
19	49	0	0	1	0	0	0	0	0	0	0	0	4	0	0	0	0	0	18	0	2	0	74	0.24
20	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0.00
21	3	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	46	0	50	0.92
22	36	10	0	59	6	0	0	1	1	0	1	2	1	3	0	0	0	0	0	0	0	175	295	0.59
Total	1466	175	51	1200	446	11	28	444	201	42	198	594	554	30	21	0	9	116	26	0	53	213	5878	0.00
PA	0.65	0.71	0.78	0.60	0.94	1.00	0.68	0.77	0.74	0.74	0.82	0.78	0.60	0.73	0.29	0.00	1.00	0.67	0.69	0.00	0.87	0.82	0.00	0.70
Kappa																								0.66

Chapter 3. Appendix 8. Confusion matrix for all cover classes before aggregation into the nine cover classes across all beaches.

The cover classes assigned to each validation segment through the classification process were compared to those assigned visually against reference sources. Diagonal shaded cells represent the number of correctly classified segments for each cover class, and the off-diagonal cells indicate the misclassifications between different cover classes. Overall accuracy is the number of correctly classified segments divided by the total number of segments in the sample. Kappa is another misclassification measure that compares overall accuracy to a random classification. Class codes in the left-hand column are: 1) Sand; 2) Wet sand; 3) Mixed native and exotic shrubs; 4) Mixed native and exotic shrubs 2; 5) Mixed native and exotic trees; 6) Water; 7) Water over sand; 8) Exotic pasture grasses; 9) Other herbaceous species; 10) Exotic trees and shrubs; 11) *F. spiralis*; 12) *S. sericeus*; 13) *A. arenaria*; 14) Other; 15) Rocks; 16) Buildings; 17) Shadow from trees and shrubs; 18) Other herbaceous species 2; 19) Driftwood; 20) Bare earth. The overall accuracy of the classification for the non-aggregated classes was 0.70, and the Kappa score was 0.66.



Chapter 3. Appendix 9. Summarised Hierarchical Agglomerative Cluster analysis dendrogram

The dendrogram shows the percent cover values for cover classes, using Bray-Curtis pairwise distances among plots and the Average Group Linkage method. The dendrogram is scaled using Wishart's (1969) objective function and was pruned where the tree retained the most ecological meaning, with around 50 – 75% of information remaining.

Chapter 4 Appendices

Chapter 4. Appendix 1. Environmental variable datasets

Variable name	Dataset and author	Dataset description
Adjacent land cover type	Land Cover Database version 5.0, Mainland, New Zealand. Manaaki Whenua Landcare Research (2019).	The most common, adjacent land cover type in terms of area (ha), from the Land Cover Database within a 400 m radius from the centre of sample plots. The seaward side of the coastline (i.e. the sea) is excluded.
Coast	Ryan et al. (2023).	The coast where the active dune ecosystems in this research occur.
Distance from nearest road (m)	NZ Roads (Addressing). Land Information NZ.	Distance (m) to the closest road to a sample plot (all types of roads).
Ecosystem extent (ha)	Ryan et al. (2023).	The extent of active dune ecosystems, digitised from the same aerial imagery used for classification in this research.
Days of large waves (p.a)	Number of days waves exceeded 4 m for at least 12 hours per year in coastal regions, over the years 2008 - 2015. Gorman (2016), for the Ministry for the Environment, New Zealand.	Coastal extreme wave indices are derived for eighteen coastal regions around Aotearoa. Four-metre-tall waves are considered extreme in the northernmost parts of Aotearoa but are more common in the south. In all cases, they are the number of days when large waves occur over 12 hours or more (p.a).
Days of strong wind gusts (p.a)	Number of days of wind gusts ≥ 24 knots (per annum). CliFlo, New Zealand's National Climate Database. NIWA (2023).	Days of wind gusts ≥ 24 knots across all months for the year or years that aerial imagery used in this research was captured. Data was taken from the nearest climate station to research beaches in operation over the research period ($\bar{x}$ = 29.4 km, min = 6.4, max = 77.3, SD =20.5).

Chapter 4, Appendix 1 (cont.)

Variable name	**Dataset and author**	**Dataset description**
Median personal income ($)	Age and income in 2013 by Territorial Authorities and Local Boards (Statistical Area 2). Stats NZ (2017).	Median personal income for residents. Due to high non-response rates, median personal income is used because Stats NZ states, "total household income is considered: Poor: fit for use – but use with caution due to some significant data quality issues." Note median personal income is income from all sources. An alternative would be the "Household Economic Survey", but this is only a sample from 5000 homes and is less well suited to geography-based analyses. Data was collected for census mesh blocks within 1km of plots.
Total rainfall (mm)	CliFlo, New Zealand's National Climate Database. NIWA (2023).	Mean total rainfall (mm) across all months for the year or years, that the aerial imagery used in this research was captured. Data was taken from the nearest climate station to research beaches in operation over the research period ($\bar{x}$ = 29.4 km, min = 6.4, max = 77.3, SD =20.5).
Usually resident population	Census counts, 2013, New Zealand (Statistical Area 2). Stats NZ (2019).	The population usually living in an area on census day, 2013. Data was collected for census mesh blocks (Statistical Area 2) within 1 km of plots. Statistical Area 2 meshblocks reflect communities that interact together socially and economically. The population of a meshblock varies depending on the authority (district or regional councils) and whether the area is urban or rural. The geographical size of vary depending on the community sampled.

Chapter 4. Appendix 2. Correlation matrix for the third principal component for the PCA of landscape metrics for the sand – vegetation binary raster