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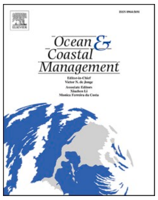

From ‘clean and green’ to ‘brown and down’: A synthesis of historical changes to biodiversity and marine ecosystems in the Marlborough Sounds, New Zealand

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ABSTRACT

Ecosystem-based management (EBM) is a potential antidote to the alleviation of multiple stressors in highly-valued and contested marine environments. An understanding of the magnitude and drivers of past ecosystem changes can inform the development of realistic ecological and social outcomes for different places. These goals should aim to increase the ecological health and resilience of coastal ecosystems and their connected land- and sea-scapes by minimising anthropogenic disturbances. To address knowledge gaps, we present a marine historical synthesis of the Marlborough Sounds in New Zealand's South Island. These rias are strongly coupled to the surrounding land and inland river catchments. We took an integrated approach by examining effects of land use change on coastal ecosystems, along with case studies of the effects of exploitation on foundational marine species. We found that ecosystems have gone through a series of transformations since Māori settlement ca. 700 years ago, with localised extirpations of marine megafauna, overharvesting of exploited species, and disruption to ecological functioning through ongoing clearfelling of terrestrial and marine biogenic communities since European settlement in the 1800s. There has been a decline from great abundance of marine life to relative scarcity, which is currently evident to local people in increased effort and reduced allowable catches of fish and shellfish. Recovery of biodiversity in the short-term within the Marlborough Sounds is uncertain, given ongoing multiple and interacting stressors from unsustainable land-use and over-exploitation of marine life. Lifting baselines are possible but will require significant changes to land and marine management to restore ecological health and enhance resilience in the face of climate change. Increased marine protection, regeneration of biodiverse biogenic habitats, spatial fishing measures to increase predators of sea urchins, stricter regulation of plantation forestry and a replanting prohibition in critical erosion source areas, are all needed within an EBM framework. Large experimental areas are proposed to develop, test and integrate different management techniques, and to facilitate community understanding, participation, and support for the transition to EBM.

1. Introduction

The field of marine historical ecology has emerged as an important contributor to the development of ecosystem-based management (EBM) (Engelhard et al., 2016; MacDiarmid et al., 2016a). This is because EBM is placed-based, and therefore bespoke to the human communities and ecosystems within a geographic region (McLeod and Leslie, 2009). The success of formulating EBM requires an inclusive and collaborative approach underpinned by a sound knowledge base (McLeod and Leslie, 2009; Hewitt et al., 2018; Davies et al., 2018).

A historical perspective is also necessary to counter shifting baselines (*sensu* Pauly, 1995). An understanding of the nature and extent of shifted baselines is helpful in engaging with communities to co-develop EBM, and to inform the development of ecologically realistic goals to ‘lift baselines’ (Jackson et al., 2001; Roman et al., 2015; Engelhard et al., 2016; McAfee et al., 2019), and thus improve ecosystem health and adaptive resilience (Benson and Craig, 2017). Local people's affinity for a place may also help them to better understand environmental changes, as they can recognise shifted baselines within their personal frame of experience (Fazey et al., 2006). As musician Bob Marley so elegantly put

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it: "*If you know your history, then you would know where you coming from*" (Marley and Williams, 1978).

The sharing of this information within the wider community can contextualise the current baseline and assist with management activities (Leach, 2006; Lotze and Worm, 2009). These activities can include: identifying shared values; co-developing management goals and objectives; setting feasible restoration outcomes; identifying critical habitats for protection to improve ecological health and resilience; contributing to adaptive spatial management and marine governance; and helping to foster a social consensus to better manage the myriad causes of stressors to address adverse effects on biodiversity and ecological function (McLeod and Leslie, 2009; Foley et al., 2010; Thrush and Dayton, 2010; Thrush et al., 2016; Davies et al., 2018; McAfee et al., 2019).

Here, we provide a synthesis of historical changes to the Marlborough Sounds region of New Zealand's South Island. As Hewitt and Thrush (2019) point out, New Zealand is a useful place for such studies as the effects of cumulative and multiple stressors are increasingly affecting coastal environments. It is also one of the most recent archipelagos to be settled (Jacomb et al., 2014), and there is a plethora of authoritative information that can be synthesised to identify the effects of human colonisation on previously pristine ecosystems (Lotze and Worm, 2009).

Despite its self-promoted reputation as 'clean and green', given the recent environmental reporting on the state of its marine environment (NZ Government, 2019), particularly from the effects of excessive terrigenous sedimentation and extensive physical disturbance to biogenic habitats, New Zealand can also now be aptly characterised as 'brown and down'. How these stressors, and others such as over-exploitation of marine resources, are affecting coastal ecosystems at a regional scale in New Zealand is generally not well-understood (but see Pinkerton et al., 2015; MacDiarmid et al. 2016b). In part, this is because marine management is fragmented, as different institutions operate at different scales under different legislation (e.g., Urlich et al., 2018a); and underpinning information such as sediment source attribution, coupled biogeochemical and hydrodynamic modelling, and seabed habitat type, extent, distribution and condition is sparse, patchy, dated, or proprietary. There are also few marine monitoring programmes integrated at different scales to detect the approach of tipping points (Hewitt and Thrush, 2019). What is known is that regional studies of historical fishing effects have shown a consistent theme of regime shifts in marine food webs arising from different forms and intensities of exploitation (Smith, 2013; Pinkerton et al., 2015; Booth, 2017; MacDiarmid et al., 2018). However, this knowledge is not being translated into management that fosters the resilience of food webs to adverse effects at different scales, which is needed as the effects of climate change unfold (cf. Benson and Craig, 2017).

In this study, we investigate how coastal food webs have potentially been disrupted in the Marlborough Sounds since human settlement. First, we examine evidence for how coastal marine and terrestrial ecosystems have been utilised over archaeological and historical time to determine the nature, scale, and magnitude of ecosystem change. This should enable detection of the circumstances and timing of any acceleration of human impacts (Lotze and Worm, 2009). Second, to explore the consequences of alteration to ecological processes on the food web structure, we examine how human activities have affected keystone (Paine, 1969) and ecosystem engineer (Jones et al., 1994) species: the southern right whale/tohorā (*Eubalaena australis*), pilchard/mohimohi (*Sardinops sagax*), green-lipped mussel/kūtai (*Perna canaliculus*), and giant kelp/rimurimu (*Macrocytsis pyrifera*).

Keystone species have pivotal roles transferring energy in the coastal food web, either by transporting macronutrients across large distances to enhance primary production (Roman et al., 2014), or by channelling primary production into significant biomass (Paul et al., 2001). Marine ecosystem engineers play a foundational role in habitat formation, and provide essential ecological functions for biodiversity (Jones et al., 1994; Coleman and Williams, 2002). There are acknowledged difficulties in separating out the effects of anthropogenic impacts from natural dynamics and climate-driven variation in attempting to reconstruct ecosystem processes after the loss of important marine species, and thereby to set benchmarks for restoration (Dayton et al., 1998; Jackson et al., 2001; Lotze and Worm, 2009). Thus, we consider the additional dimension of historical anthropogenic land use changes on ecological functioning, given the tight coupling of land to sea in the Marlborough Sounds rias.

1.1. *We suggest this type of synthesis will provide*

- important context for contemporary management;
- critical input into the future development of EBM for the region;
- bolster the impetus for agencies to work together more effectively to manage cumulative effects across different scales;
- insights that inform the co-development of feasible restoration goals;
- provide a baseline for future empirical studies.

2. Materials and methods

An extensive literature review to produce a synthesis of available marine historical information was undertaken (Table 1). This comprised a mix of published archaeological, palaeoecological, indigenous, social

Table 1
Information sources and search terms for literature used for this study. The 'Marlborough Sounds' was selected as the primary search term for the international academic databases as it is a distinctive and well-known geographical area, commonly used in the title or as a key word in journal articles across disciplines.

Sources	Search terms	Authoritative gateway	Relevance gateway
Marlborough District Council (MDC)'s bibliography of natural and human history studies (1569 items as at 2010). MDC, Fisheries NZ, and Environmental Protection Authority websites for recent peer-reviewed technical reports (25); National Library of New Zealand's Te Puna catalogue & Papers Past database	Marlborough Sounds; Pelorus Sound; Queen Charlotte Sound; Port Underwood	Academic histories with verifiable references; technical reviews of empirical studies; academic theses and peer-reviewed articles in ecology; paleoecology; archaeology; soil science; coastal processes, fisheries and marine science; authoritative observations (e. g., historical newspaper accounts; voyage diaries)	Contemporary, historical, archaeological and palaeoecological evidence for the effects of human harvesting on marine life and changes in species abundance, range and diversity over time; and/or the effects of coastal and catchment land-use activities on ecological and physical processes.
CAB Abstracts 1910–2020 (22 items); Web of Science 1980–2020 (143 items); Google Scholar (70 pages)	Marlborough Sounds	Peer-reviewed articles in categories: Marine Freshwater Biology; Fisheries; Ecology; Aquaculture; Zoology; Ornithology; Biodiversity Conservation; Geology; Environmental Sciences	As above

(historical treatises, newspaper records), institutional records (e.g., fishing stock trends), geophysical and ecological data and modelling (including grey literature) (Bolster, 2006; Lotze and Worm, 2009; MacDiarmid et al., 2016a; Maxwell and MacDiarmid, 2016; Beller et al., 2017; Thurstan et al., 2017*a*). Such a synthesis can potentially reveal previous regime shifts in comparison with contemporary interaction networks, and help identify whether subtidal and intertidal ecosystems are in hysteresis or near tipping points (Thrush et al., 2014; Seekell, 2016; Beller et al., 2017). Detection of changes in habitat type, extent and distribution, along with temporal and spatial shifts in species composition and abundance, can provide insights into the modifying effects of cumulative and multiple stressors over time (MacDiarmid et al., 2016b).

Two 'gateway' tests were devised to select from a plethora of literature on the Marlborough Sounds dating back to the late 18th Century. The first was an 'authoritative gateway', which excluded self-published family histories, even though a number had illustrative anecdotes of past fisheries abundance that would augment and inform any future studies of local ecological knowledge. For literature that met the test for inclusion, additional referenced articles were identified following more in-depth reading. The second test was a 'relevance gateway', which included information centred on ecological processes at a range of scales, extensive land-use patterns, or that documented intensive harvesting and/or widespread changes in populations and distributions of foundational species. This excluded numerous site-based assessments of benthic conditions underneath marine farms, taxonomic studies, or studies of terrestrial island habitats or species inhabiting those within the Marlborough Sounds.

Reliable historical records were identified for large biomass of *E. australis*, *S. sagax*, and *P. canaliculus* under intensive harvesting; and for distribution surveys of *M. pyrifera* dating back to the 1940s. Other species emblematic of the Marlborough Sounds were considered as case studies, such as Hector's dolphin (*Cephalorhynchus hectori*), hāpuku (*Polyprion oxygeneios*), blue cod (*Parapercis colias*), and scallop (*Pecten novaezelandiae*). There was relatively less historical information for these species; and, in the case of *P. colias* and *P. novaezelandiae*, deserving of fuller treatment elsewhere due to the complex and contested management of ongoing over-harvesting, and widespread concern from customary and recreational fishers at population declines over recent decades. Several of these species are discussed incidentally in relation to sustained anthropogenic pressures.

2.1. *Study area - soils and climate*

The Marlborough Sounds (hereinafter 'Sounds') are situated in the north-eastern top of the South Island (Fig. 1). The Sounds were formed by tectonic tilting and eustatic sea-level changes during interglacial periods throughout the Late Quaternary, which led to the partial submergence of the two main unglaciated river valley systems: Queen Charlotte Sound/Tōtaranui ('Queen Charlotte') and Pelorus Sound/Te Hoiere ('Pelorus') (Nicol, 2011; Hume et al., 2016). Together, these systems comprise ~73,000 ha (ha) of sheltered waters and convoluted coastline (Hume et al., 2016); and, along with Port Underwood (2347 ha) to the south-east, are the main focal area of this study (Fig. 1). D'Urville Island/Rangitoto ki te Tonga to the northwest of Pelorus is also discussed where germane.

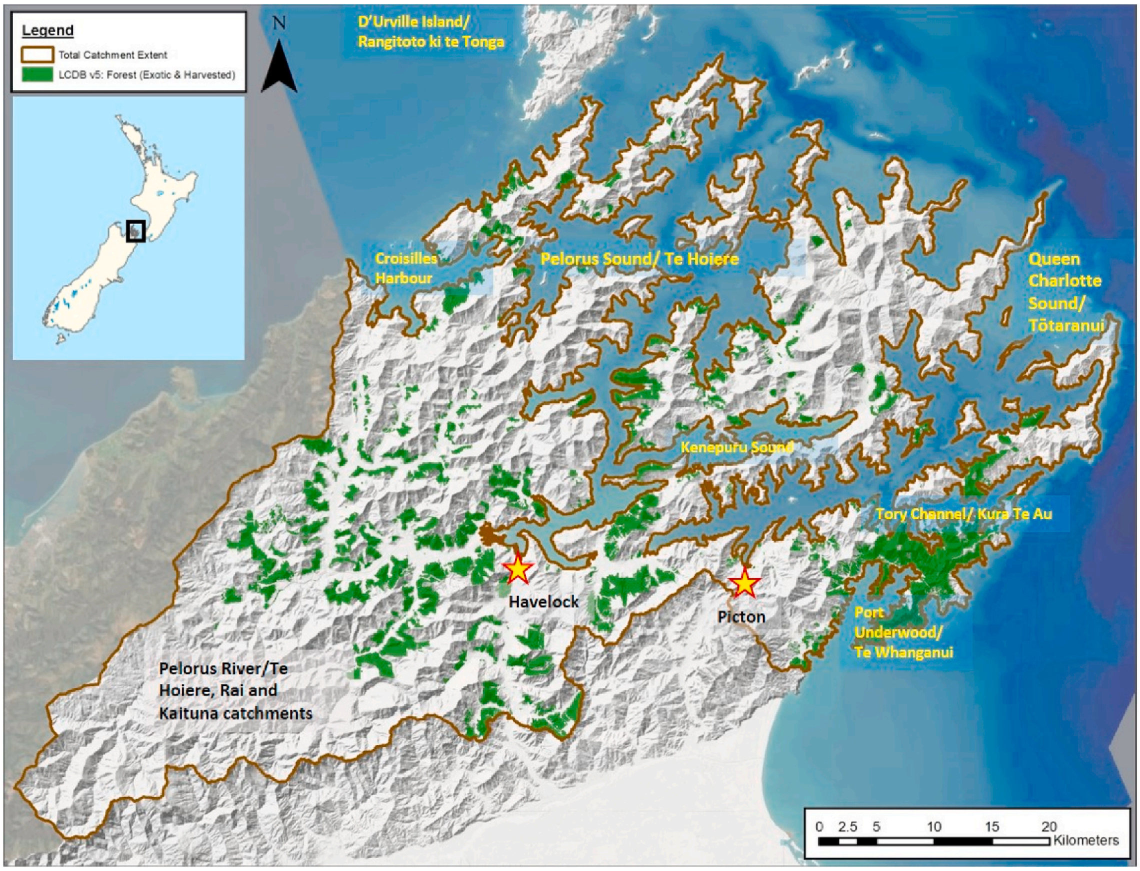

Fig. 1. The Marlborough Sounds and catchment area of the main rivers flowing into Pelorus Sound/Te Hoiere: the Pelorus River/Te Hoiere, Rai and Kaituna Rivers (approx. 1046 km^2). There is no major river discharging into Queen Charlotte Sound/Tōtaranui. The areas in green are introduced *Pinus radiata* monoculture plantations as at 2018/19 (ca. 31,688 ha) which are typically located on steep hillsides and harvested by extensive clearfelling. LCDB = Land Cover Database version 5, which is a multi-temporal, thematic classification based on multispectral satellite data. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

Highly erodible soils with clay content of up to 60%, steep topography, landforms directly coupled to the sea, and historical and contemporary widespread vegetation removal and associated land disturbance, have all resulted in excessive terrigenous sediment deposition in intertidal and subtidal areas since the mid-1800s (Lauder, 1987; Urlich, 2015; Handley, 2015; Handley et al., 2017). Parent rocks are composed of Mesozoic siliceous greywackes and schists, with bands of serpentinitic greywacke, basaltic and ultramafic rocks in the western Sounds (Laffan and Daly, 1985). Soils differ in the strength of weathering over an altitudinal gradient, with weakly weathered and thin soils generally above 200 m above sea level, and strongly weathered, deeper soils below 200 m (Laffan and Daly, 1985). Soils are relatively high in kaolinite and vermiculite clay (Molloy, 1998), which flocculate on contact with seawater (O'Loughlin, 1979) and settle out on the benthos in sheltered bays (Hadfield, 2015).

Seabed sediment accumulation rates in the last 150 years are 5–20 times above pre-European settlement levels (prior to the 1860s) at different sites within the inner Pelorus Sound (Handley et al., 2017). Relatively frequent terrigenous sediment plumes occur after moderate to heavy rainfall (Fig. 2). Flood flows over 2000 cumecs in the Pelorus River system have occurred periodically over the last 20 years: in 1998, 2008, 2010, 2012 and 2016 (MDC on-line data). Annual average rainfall varies in a north-south gradient, from 1000 to 1200 mm in the northern parts of D'Urville Island and Arapaoa Island (outer NE Queen Charlotte), to up to 2600 mm at the head of the Pelorus River catchment (Tait, 2017) (Fig. 3).

2.2. Marine habitats

The steep slopes of the Sounds terminate in a narrow fringe of rocky shoreline, and subtidal bedrock and cobble reefs (Davidson et al., 2011). These extend a short distance underwater before grading into shell-hash and sandier sediments, which in turn give way to extensive depositional areas of silts and clays, particularly within the inner Sounds (Davidson et al., 2011; Neil et al., 2018). Multi-beam echosounder sonar mapping of benthic terrain over 433 km^2 in Queen Charlotte identified the

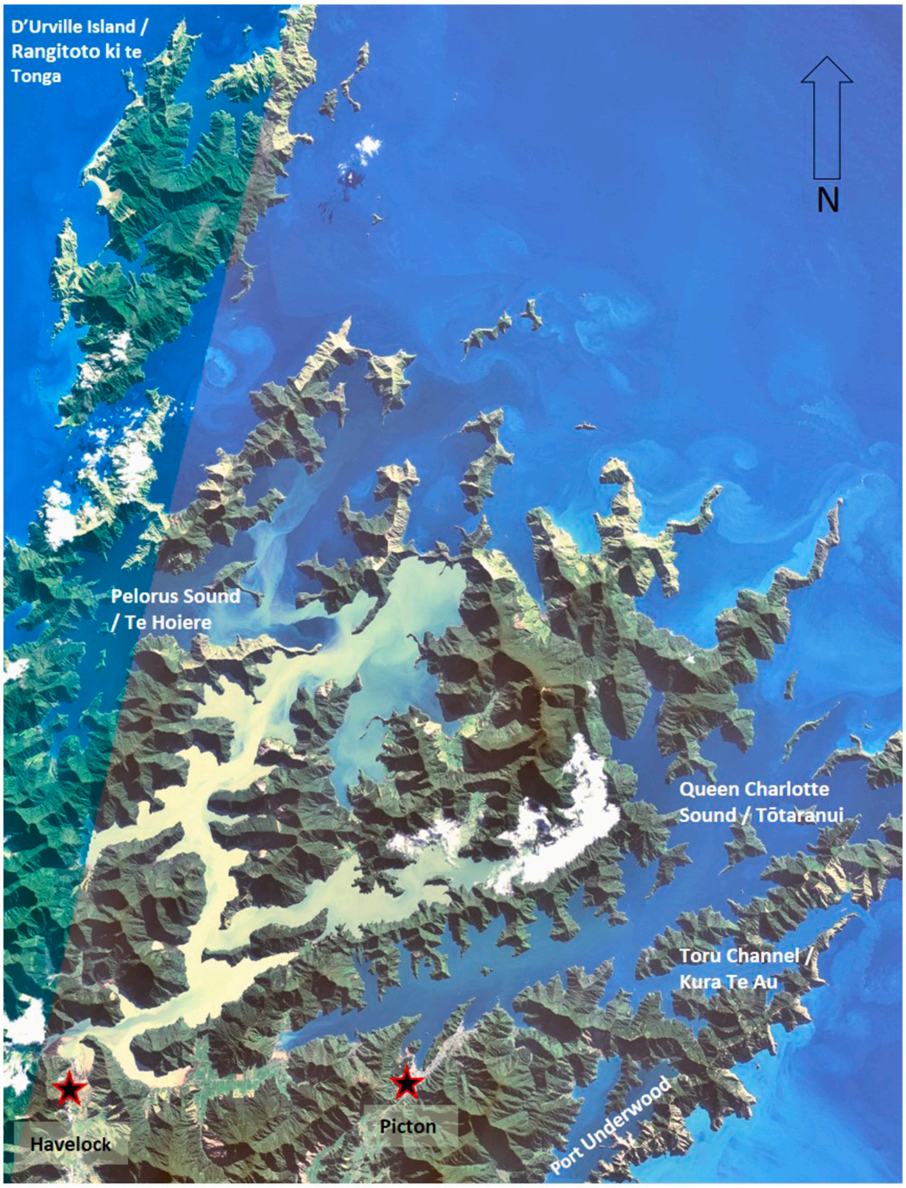

Fig. 2. European Space Agency Sentinel 2 satellite image of Pelorus Sound (discoloured yellow-brown) and Queen Charlotte Sound from July 8, 2018 after an estimated 1 in 3.1 year annual return interval rainfall event, (Marlborough District Council unpublished data). (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

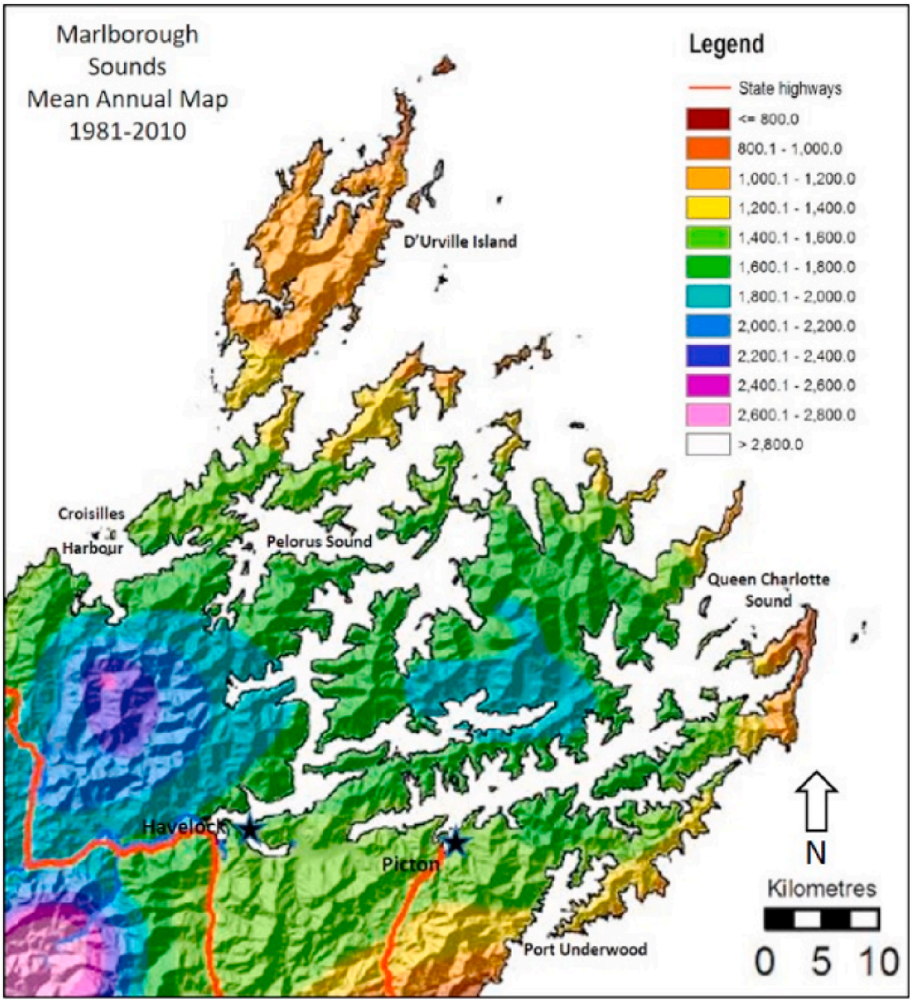

Fig. 3. Interpolation of mean annual rainfall for the Marlborough Sounds and catchments for the period 1981–2010. Redrawn from Tait (2017) with permission.

distribution and type of soft sediments, areas of coarse sands and gravels in high current areas, and rocky reefs in the outer sounds formed by sunken ridges and pinnacles (Neil et al., 2018).

Intertidal areas are typically <20 ha at the upper tidal reaches of inlets and bays, where streams and small rivers deposit eroded catchment soils. The largest estuaries are the Havelock, Mahakipaoa and Kaiuma estuary complex in the inner Pelorus encompassing ~1025 ha. Intertidal habitats in these estuaries are dominated by soft mud from fine silt/clay deposited by the Pelorus River, with seagrass (*Zostera muelleri*) now scarce and periodically covered by fine sediment (Stevens and Robertson, 2014, 2017, Skilton and Thompson, 2017).

There is a diverse array of subtidal biogenic habitat-forming species, such as giant kelp forests (*M. pyrifera*), horse mussel beds (*Atrina zelandica*), tubeworm towers (*Galeolaria hystrix*), rhodolith or maerl beds, and bryozoan, hydroid, and sponge communities (Davidson et al., 2011). Biogenic habitats are much reduced in extent and distribution from historic times (Handley, 2015, 2016) and continue to be exposed to damage (Davidson and Richards, 2015). These habitats survive mostly in high-current areas, where anchoring is difficult, bottom-trawling and dredge operations are limited, and the bottom-shear stress of tidal currents is of sufficient velocity to re-suspend fine terrigenous sediment. Biogenic habitats survive elsewhere due to the self-protecting nature of topography, or by chance.

3. Results

3.1. Māori habitation pre-1770

Māori colonisation and occupation of Marlborough has been dated to the early 14th Century CE (Jacomb et al., 2014). Before the arrival of Europeans in 1770, Māori harvested a range of marine life in the Marlborough region, including marine mammals and seabirds (Collins et al., 2014; Seersholm et al., 2018). The New Zealand fur seal (*Arctocephalus fosteri*), New Zealand sea lion (*Phocartos hookeri*), southern elephant seal (*Mirounga leonia*), and the Waitaha penguin (*Megadyptes waitaha*) were hunted extensively, leading to the decline of the fur seal population (Salis et al., 2016), and local extirpation of the sea lion, elephant seal and Waitaha penguin populations (Collins et al., 2014; Seersholm et al., 2018). The timing of the extirpations is suggested to be in the 15th century (Seersholm et al., 2018), although as late as 1770 James Cook recorded an elephant seal near Motuara Island in outer Queen Charlotte (Beaglehole, 1955) and a sea lion in 1773 near Arapaoa Island (Beaglehole, 1961). Whales and dolphins were also periodically harvested or scavenged, but probably not at levels that caused significant impacts on the viability of populations (Seersholm et al., 2018).

Māori archaeological sites in coastal locations are numerous throughout the Sounds, reflecting the long history of occupation. Wadsworth (2015) noted 343 recorded sites in Queen Charlotte including middens, ovens, gardens and pā (fortifications). Subsistence harvesting by Māori, and localised cultivation and timber use, were unlikely to have resulted in widespread damage to benthic ecosystems (see Leach, 2006 for review). Palaeological analysis of sediment accumulation rates and molluscan death assemblages from seabed sediment cores in Kenepuru Sound (inner Pelorus Sound), that span the period before Māori settlement to contemporary time, showed no detectable changes before widespread European settlement and associated economic land-use activities in the late 1800s (Handley et al., 2017). These methods did not however include palynological and diatom composition analyses, which were undertaken at nearby Tasman and Golden Bays ca. 140 km west, where benthic productivity was affected by Māori use of fire, localised coastal land clearance for dwellings, and horticultural cultivation (Handley et al., 2020). Sedimentation rates during the Māori settlement period were elevated four-fold over pre-human conditions from the analysis of a seabed sediment core, and there was a shift from benthic to pelagic productivity for diatoms. An upsurge in pollen deposition from seral plant communities and an increase in silt content reflected land use activities at sufficient scale to affect benthic processes.

The predominantly intact forest and wetland systems during Māori settlement (e.g., Wakefield in Ward, 1840:31–34) buffered sediment run-off from localised soil disturbance caused by fire, cultivation, and defensive earthworks associated with small settlements in the heads of bays and headlands (Walls and Laffan, 1986). These disturbances were also likely to have had short-term effects, as vegetation regenerates relatively quickly under the high rainfall of the Sounds, particularly in small areas (Walls and Laffan, 1986). Elsewhere in the drier eastern and southern parts of the South Island, forest landscapes were transformed into shrub and grassland-dominated mosaics by fires set by Māori (McWethy et al., 2010; Perry et al., 2014). It is unlikely that Māori cleared extensive areas of forest in the Sounds (Perry et al., 2014), as when Europeans arrived in the Sounds from the mid-1800s, they encountered extensive lowland forest areas dominated by large podocarps (e.g., Fig. 4). These forests covered thousands of hectares of the Pelorus, Rai and Kaituna catchments and were milled progressively from 1864 to 1915 (Paton, 1982).

Periodic large natural disturbances such as storms, earthquakes and tsunamis in the top of the South Island over the last ~800 years (e.g., Goff and McFadgen, 2001; Clark et al., 2015) also did not leave any detectable evidence of compositional change of molluscan death assemblages within seabed cores from the inner Pelorus (Handley et al.,

Fig. 4. Large tōtara (*Podocarpus totara*), the "Patriarch" girth 37 feet (3.6 m dbh), Carluke, Rai Valley. Courtesy Marlborough Museum (Akersten photograph. 2000.182.0016).

2017).

Pelorus Sound was characterised as potentially oligotrophic in its pre-human unmodified state (Handley et al., 2017). These authors found that molluscan death assemblages were dominated by bioturbating deposit feeders prior to European settlement in the 19th century. However, some molluscan shell types are poorly preserved in low sediment environments, due to taphonomic loss from dissolution at, or just below, the sediment-water interface, which is more acidic than the overlying water column (Davies et al., 1989). Deposit feeders may be more likely to be preserved due to burial from periodic sedimentation events, than filter feeders in clastic sediments, and high sedimentation rates may increase likelihood of preservation of shells in the death assemblage (Kidwell and Bosence, 1991). This difference can lead to an underestimation of the prehistorical importance of biogenic habitat-forming epifaunal shellfish beds.

3.2. *1770s–1860s European exploration and exploitation*

When the HMS Endeavour arrived in Queen Charlotte in 1770, the crew had little difficulty in harvesting an abundant variety of fish and shellfish (Beaglehole, 1955, 1961, 1962). "*Having the Saine with us we made a few hauls and caught 300 pounds weight of different sorts of fish which were equally distributd to the Ships Compney*" (Cook, January 16, 1770 in Beaglehole, 1955:235). On 26 January (1955:240) "… *we generally haul* [the seine] *mornings & evenings, and seldom fail of getting fish sufficient to serve all hands*.

Parkinson (1770:114 cited in Beaglehole, 1962:453) recorded a diversity of species: "… *such as cuttle-fish, large breams* [*Pagrus auratus* snapper] *small grey breams* [*Nemadactylus macropterus*, terakihi]*, small and large baracootas* [*Thyrsites atun*, barracouta]*, flying gurnards* [*Chelidonichthys kumu*]*, horse-mackerel* [*Trachurus novaezelandiae*]*, dog-fish, soles, dabs, mullets* [probably grey mullet *Mugil cephalus* according to Beaglehole]*, drums, scorpenas or rock-fish* [*Helicolenus percoides*, Jock Stewart]*, cole-fish* [*P. colias*, blue cod]*, the beautiful fish called chimera* [*Callorhinchus milii*, elephant fish] … and *muscles* [green-lipped and blue (*Mytilus* sp.) mussels]*, and sorts of shell-fish in great plenty*."

Banks observed Māori subsistence fishing that employed a low physical impact method, which was captured in a painting by Parkinson (Fig. 5): "*we saw a Man in a small canoe fishing … he took up his netts & shewd us his machine, which was a circular net about 7 or 8 feet in diameter, extended by 2 hoops; the top of this was open and to the bottom was tied Sea Ears* [pāua/abalone *Haliotis iris*] *&c, as bait; this he let down upon the ground and when he thought that fish enough were asembled over it, he lifted it up by very gentle and even motion, so that the fish were hardly sensible of being lifted till they were almost out of the water. By this simple method he had caught abundance of fish and I believe it is the general way of Fishing all over this coast, as many such netts have been seen at almost every place we have been in. In this Bay indeed fish were so plenty that it is hardly possible not to catch abundance whatever way is made use of.*" (Beaglehole, 1962, 456–457).

Fish and shellfish were commonly harvested at all sizes within discrete areas (Leach, 2006). In his review of fishing in pre-European New Zealand, Leach concluded that the method of harvesting did not deplete stocks of fish and shellfish at a regional scale, rather biomass and mean size increased for some species, such as *P. colias*, over archaeological time. Leach termed this a 'Slash and Burn and Fallow Method' of harvesting, and surmised that targeting young fish also placed selection pressure on juveniles for fast growth to reproduction. However, Aarts et al. (2019) suggested that high seal populations also alleviate density-dependent competition between remaining fish, which allows for increased fish growth. How fish population size-structures in the Sounds responded to the decimation of seal populations and Māori spatial harvesting patterns before widespread European settlement is unknown.

The low human population density and subsistence harvesting methods meant that pressures on fish and shellfish stocks were spatially and temporally localised and ephemeral. Furneaux observed in 1773 that there were unoccupied huts in every cove, and that the Māori inhabitants foraged in different areas at different times (Beaglehole, 1961:738). Cook estimated in 1770 that there were 3–400 inhabitants dispersed along the shores, who lived on fish and fern roots and did not cultivate the land (Beaglehole, 1955:247). In 1774, Cook traversed Tory Channel for the first time, and noted it was more densely populated with two large settlements in bays (Beaglehole, 1961:575–576). There are

Fig. 5. *New Zealanders fishing – some wearing Potae-taua (mourning hats) Queen Charlotte Sound* by Sydney Parkinson in 1770. (c) The British Library Board, Add. 23,920 f.44.

only 25 recorded Māori horticulture archaeological sites within Queen Charlotte, of which the majority are in side bays of Tory Channel (Jackson, 2014; Wadsworth, 2015). Kūmara (*Ipomoea batatas*) cultivation was unreliable in the region at the time of Cook's visit (Leach, 2006), which occurred in the Little Ice Age that affected southern New Zealand between ca. 1400–1850 (Shulmeister et al., 2004). The dependence on fish and shellfish during that period led to malnutrition from a diet deficient in carbohydrates and fats, as witnessed by the crew of the *Discovery* in 1777 (Leach, 2006). The relative scarcity of key diet constituents, following the decline of local pinniped populations, along with the difficulties in cultivating carbohydrates, may help explain the sparse population of Queen Charlotte at Cook's arrival (Leach, 2006). Potatoes (*Solanum tuberosum*) introduced by Cook had become an abundant constituent of Māori diet when two Russian exploration vessels arrived in 1820, in addition to kūmara, fern (*Pteridium esculentum*) and cabbage tree (*Cordyline australis*) roots, as well as fish, shellfish, Polynesian rat (*Rattus exulans*), dogs, and humans (Mitchell and Mitchell, 2004).

The abundance of fish life had not diminished by the mid-19th century in Queen Charlotte, Pelorus, and Port Underwood given observations of both Wakefield and Dieffenbach in 1839: "*The sea* [at Ships Cove] *teems with fish, of which we caught enough with hooks and lines for the whole ship before we dropped anchor. These consisted of hake, colefish, spotted dog-fish, gurnet, flounders, and joe-fish, all of which are eatable* (Wakefield in Ward, 1840:11).

Dieffenbach recorded the absence of fur seals (*A. forsteri*) in Port Underwood in 1839, "*and but very few to the southward, where they were formerly in abundance*" (Ward, 1840:28). He notes that the decline took place over the preceding decade from over-harvesting by European and North American sealers.

3.3. 1860s–2010s Extraction and transformation

The mid-1800s also marked the beginning of a radical transformation of the landscape. European settler numbers increased dramatically from 1860 to 1880 (Waitangi Tribunal, 2008). The effects on the environment from economic exploitation manifested quickly (McIntosh, 1940; Kelly, 1976; Handley et al., 2017). The pursuit of timber and gold from 1864, the widespread conversion of native forest and wetlands for farmland, and the loss of estuarine habitat for reclamation and port development all had transformative effects on the marine ecology of the Sounds (Handley, 2015, 2016). Gold mining occurred in different parts of the Pelorus catchment during the late 1800s and intensive sluicing likely contributed significant sediment loads into waterways and the eventual deposition onto intertidal and shallow subtidal areas (Handley, 2015). The removal of indigenous forest cover on the steep hills of the Pelorus catchment and in the Sounds also resulted in widespread erosion after heavy rain (McIntosh, 1940; Bowie, 1963; Lauder, 1987). McIntosh (1940:277) describes the unforeseen consequences of extensive clearance of old-growth indigenous forest in these areas: "*The history of one* [catchment] *is typical of all. In the Rai, for example, the sawmilling was followed by the grassing down of the bush burn and the introduction of sheep. The heavy rain leached out the fertility and the process of erosion denuded the steep hill faces of soil*".

By 1871, many bays in the Sounds had clearings with pasture for sheep, and following the exhaustion of the easily accessible timber in the late 1800s, pastoral farming reached its peak between the 1910s and 1930s (Bowie, 1963). Repeated burning of regenerating scrub and poor soil fertility contributed to frequent erosion after heavy rainfall on hill slope pastures, and as farming began to decline in the 1930s many farmers planted exotic trees or left the land to revert to native vegetation (Bowie, 1963; Laffan and Daly, 1985).

Hill country pastoral farming resurged after WWII when aerial superphosphate application became cost-effective with peak wool prices (Bowie, 1963), but when fertiliser subsidies were phased out in the 1980s widespread reversion into indigenous forest occurred or extensive *Pinus radiata* plantations were established (Handley et al., 2017). The first commercial-scale *P. radiata* plantings occurred in Queen Charlotte in the 1930s and the Pelorus River catchment in the late 1940s, with plantations becoming more widespread in the 1960s and 1970s incentivised by the government loans (Urlich and Handley, in press). Clearfelling of plantation forests over large areas began in the 1970s and increased over time as forests planted from the 1960s progressively matured (at ~30 years of age). Sediment accumulation rates are expected to remain elevated with ongoing *P. radiata* harvesting on steep slopes, as plantations now cover ca. 31,688 ha (Fig. 1). Fig. 6 shows examples of current forestry management practices that are being questioned (Urlich, 2015, 2020).

The consequences are visibly manifest in the Havelock estuary at the mouth of the Pelorus and Kaituna Rivers, which increased in soft mud habitat by 34 ha between 2001 and 2014 (Stevens and Robertson, 2014). This contrasts to the outer Pelorus, and in most Queen Charlotte estuaries (Stevens, 2018), where catchment land-cover is dominated by indigenous forest, and the extent of soft mud is less than 10% of monitored habitats (Fig. 7). Adverse effects of excessive sediment deposition occur on estuarine biogeochemistry and interaction networks (e.g., Thrush et al., 2004; Thrush et al., 2014). In coastal waters of the Sounds, increased turbidity from terrigenous sediment has been recognised since the late 1970s, with consequent effects on primary productivity and smothering of subtidal and intertidal habitats (Bargh, 1977; O'Loughlin, 1979; Johnston et al., 1981; Davidson and Richards, 2015).

The effects of activities within the coastal waters of the Sounds since European settlement have also been transformative. Frequent disturbance of the benthos by dredging, bottom-trawling, mooring chains, and anchors has resulted in extensive areas of biogenic habitat being damaged, modified, or destroyed over the last century (Davidson and Richards, 2015; Handley, 2015, 2016; Morrisey et al., 2018). Dredging for oysters (*Tiostrea chilensis*) in Oyster Bay, Tory Channel occurred from at least 1863 (Handley, 2016), and by 1894 "*indiscriminate dredging and careless picking is rapidly becoming a delicacy of the past*" (Pelorus Guardian and Miners Advocate, 1894). Trawling in the Cook Strait region began at least as early as 1904 (Hawke's Bay Herald, 1904; Evening Post, 1908). Concern about the effects on fisheries in Queen Charlotte were voiced in the 1930s, along with a call to ban trawling inside the sheltered waters of the Sounds due to blame for "*destroying* [fish] *breeding grounds*" (Twyford, 1939 cited in Handley, 2016). In 1974, the Ministry of Agriculture and Fisheries wrote to the Marlborough Sounds Maritime Park Board concerned about the decline in *P. colias* abundance due to the destruction and lack of recovery of biogenic habitats from physical disturbance, and overfishing of remaining populations (Pätete, 1997; Ministry of Fisheries, 2000), both of which have continued to occur (Davidson et al., 2014). Beentjes and Carbines (2012) reported from a 2010 survey that *P. colias* preferentially inhabited areas of topographic complexity, including biogenic habitats vulnerable to dredging; and they noted *P. colias* abundance decreased by 60% between 1995/96 and 2007, which indicated localised depletion. Daily permitted takes for recreational fishers progressively dropped five times from 12 fish in 1986, to 2 fish in 2012 over a minimum size of 28 cm in length. This is despite new environmental sustainability provisions being enacted within the 1996 Fisheries Act (Mace et al., 2014).

The first calls to address over-exploitation of fisheries by European settlers came from the Māori iwi (tribe) Ngāti Koata in the 1880s (Pätete, 1997; Waitangi Tribunal, 2008). Petitions to Government in 1888 and 1903 for better management of deep water and coastal fisheries around D'Urville Island, including establishing reserves under traditional customs, were unsuccessful. Pätete (1997) also documented iwi concerns expressed in 1938 about inshore set-netting depleting small fish, and new technologies which had resulted in decline of hāpuku (*P. oxygeneios*, grouper) in deeper waters. These concerns remained to be addressed by the Government, 70 years later (Waitangi Tribunal, 2008). Handley (2015, 2016) also summarised dramatic declines over the course of European settlement in populations of large sharks,

Fig. 6. Illustrative examples of soil erosion causing fine sediment deposition resulting from harvesting activities associated with widespread *Pinus radiata* plantations. *Left*– log-scoured runnels by inadequate elevation from a cable hauler cable on a ridgetop, and harvesting to the waterline resulting in sediment delivery to coastal waters. *Right* – log debris and soil deposited into coastal waters following recent clearfelling, after logging close to the water's edge (Photos: S.C.U).

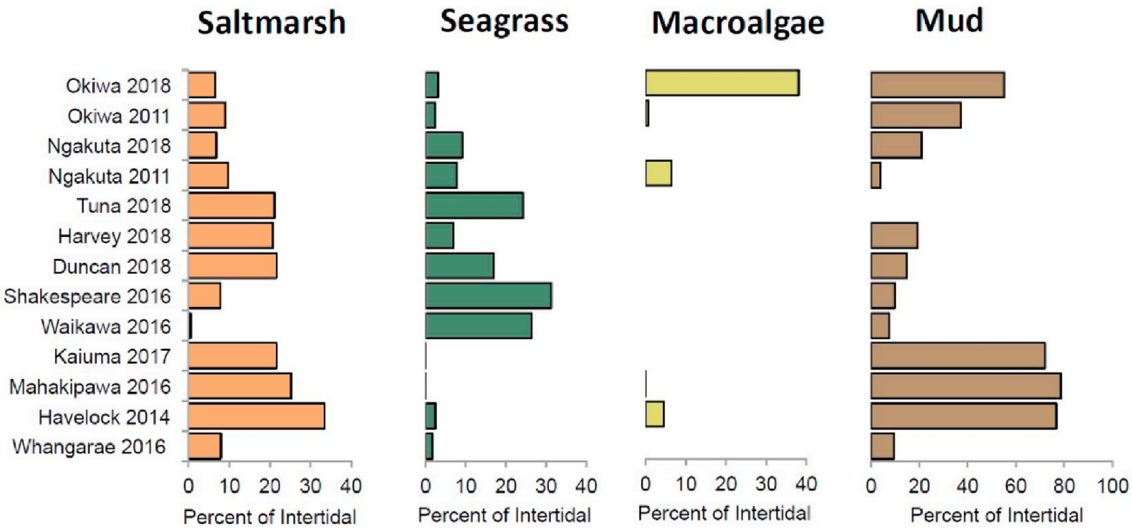

Fig. 7. Percentage of estuary with saltmarsh (dominants *Apodasima similis* and *Juncus kraussii*), seagrass (*Zostera muelleri*), nuisance macroalgae (*Enteromorpha* sp.), and soft mud habitat for 11 shallow intertidal dominated estuaries (residence time < 3days) in the Marlborough Sounds, New Zealand. Soft mud proportion in each estuary assessed according to the National Estuarine Monitoring Protocol (Robertson et al., 2002). Okiwa, Ngatuka, Shakespeare and Waikawa estuaries are located in Queen Charlotte Sound; Tuna, Duncan and Harvey are in outer Pelorus Sound; Kaiuma, Havelock and Mahakipawa estuaries are in the inner Pelorus Sound; Whangarae is in Croisilles Harbour. Open source data kindly compiled and supplied by Leigh Stevens, Salt Ecology Ltd.

P. novaezelandiae, *P. canaliculus*, *P. augratus*, *P. oxygeneios*, *J. edwardsii*, *A. zelandica*, *H. iris*, and *T. chilensis*.

We now turn to examine anthropogenic effects on the four selected foundational species.

3.4. Southern right whales/Tohorā

A steep decline in the *E. australis* population was noted by Dieffenbach in 1839 over the decade since whaling began (Ward, 1840). McIntosh (1940:24) noted that when John Guard's whaling vessel, the *Waterloo*, first visited Port Underwood in 1828 that the harbour: "*literally 'swarmed with whales'. The crew were alarmed one morning by the vessel bumping, as they thought, on a rock. But on looking overside they were startled by the sight of a whale endeavouring to rub itself against the ship*". The whale picked the wrong ship.

Dieffenbach observed that *E. australis* spent approximately six months in the Sounds: "*From May to the beginning of October the whales visit the bays and bring forth their young. They arrive from the N.W., and go to the S.E., following the tide along the shores in search of smooth water. They are often seen rubbing off against the beach and rocks the numerous barnacles and other parasitical insects with which they are covered. The mother, called the cow, is always with her offspring, whilst the male, called the bull, is rarely seen, and seldom caught,- a circumstance which must act very unfavourably on the number of these animals. The same result arises from the constant destruction of the calves, which are always secure prey to the whaler. The months of May, June, and July are regarded as the best months in Cloudy Bay* [Port Underwood from Dieffenbach's topographic description, see also McIntosh 1940]*, the three other months for Tory Channel. The cause of this may be, that they go then as far up in the inlets of the sea as they can to bring forth their young*" (Ward, 1840:97–98).

By the late 1830s, competition for *E. australis* was fierce in Port Underwood, with six shore whaling stations and a peak of 18 whaling vessels in 1836 sending out up to 70 boats to pursue each whale (McNab, 1913 in Pickett, 2002). When Dieffenbach made his observation of decline in 1839, whales were being intercepted out at sea before they could reach the port's sheltered waters (McNab, 1913 in Pickett, 2002). In a comprehensive review of whaling records, Carroll et al. (2014) calculated 82% of the total New Zealand shore-based catches occurred between 1830 and 1849, in the range between 4581 and 6728 whales. Data for whaling ships were also aggregated, making it difficult to determine the numbers killed in Port Underwood. Carroll et al. (2014) cited one ship's log, which tallied the catches of three vessels that between them caught 61 whales there in 1836. After 1900, *E. australis* were seldom seen and whalers operating from Tory Channel turned their attention primarily to the relatively numerous humpback whales (*Megaptera novaeangliae*) (Dawbin, 1956). In 1927, the last *E. australis* whales killed were by Tory Channel whalers (Prickett, 2002).

The decimation of the semi-resident *E. australis* could have had effects on primary productivity in parts of the Sounds. Roman et al. (2016) found that northern right whales (*E. glacialis*) likely played a substantial role in recycling nutrients in coastal ecosystems through the release of N, P, and Fe in faecal plumes, and probably also in urine and placentas. Whales have also been described as marine ecosystem engineers as they transfer nutrients from deep to surface waters; and cause localised ocean mixing (see Roman et al., 2014 for review). In calving grounds, whales release N in the form of urea as they are usually fasting or lactating. Roman et al. (2014) term this the "great whale conveyor belt" of nutrients transferred from productive feeding grounds to nutrient-limited oligotrophic waters. In Port Underwood and Tory Channel, the uptake by phytoplankton of whale-derived nutrients may have been light-limited in the colder winter months, but could have contributed to the annual spring blooms before whales migrated back to their feeding waters in October. Whether this was a significant contribution in the context of upwelling from adjoining Cook Strait waters (Gillespie et al., 2011; Hadfield et al., 2014), and fluctuating productivity between El Niño and La Niña years (Zeldis et al., 2008), is not known. What is known is that *E. australis* populations within New Zealand waters have not yet recovered to pre-whaling levels (Carroll et al., 2014).

3.5. Pilchards/mohimohi

The Sounds have a diverse range of reef and pelagic fish species (Taylor and Dempster, 2016). One of the most important was the planktivorous *S. sagax*, which entered the local vernacular as the 'Picton herring' or 'Picton bloater' from the late 1800s (Brehaut, 2017). In a review of *S. sagax* biology and ecology, Paul et al. (2001) noted its pivotal position in transferring energy in the coastal food web by converting and channelling primary production into significant biomass for helping to sustain kingfish (*Seriola lalandi*), kahawai (*Arripis trutta*), *P. auratus*, *P. colias*, as well as barracouta, sharks, dolphins, and seabirds (Phillipps, 1929). *S. sagax* can directly use nutrients and energy captured during diatom blooms, and so can rapidly increase in numbers when these blooms occur. Although studies of pilchard stocks and fisheries elsewhere revealed considerable short and long-term fluctuations in biomass size, linked to changes in climatic and oceanographic conditions, population declines may be due to overfishing a naturally shrinking stock (Paul et al., 2001).

From the 1860s, large *S. sagax* populations were recorded and harvested in Queen Charlotte (The Press, 1864; Arthur, 1883). Shoals moved into shallow bays in winter, and into deeper waters in the spring (Marlborough Express, 1881; Arthur, 1883). Average hauls of 1.5–2 tonnes up to 10 tonnes kept four smoke houses going in the winter months (Arthur, 1883). Large shoals stranded on Picton beach in 1865 and 1903 (Kelly, 1976) and in 1909 (Fig. 8), a phenomenon that could have been related to epizootic disease (Paul et al., 2001). Phillipps (1929:343) recorded suggestions from local fisherman that "*the amount of decayed seaweed and black floating ooze*" in Queen Charlotte may have been linked to a decline in shoals in the early 20th Century. By the 1940s, the fishery was large enough for the commencement of commercial seining to supply a cannery in Picton (Baker, 1972; Kelly, 1976). Catches quickly declined from 274 tonnes in 1942 to 11 tonnes in 1949, contributing to the closure of the factory along with the loss of the seine (Baker, 1972). A commercial quota of 150 tonnes currently exists for the Challenger fisheries reporting area, which includes the Sounds, Cook Strait, Tasman Bay, and the South Island's west coast (Fisheries NZ, 2019a). Annual catches over the last six years were below 25 tonnes from 2014 to 2017, peaking at 232 tonnes in 2018 and 58 tonnes in 2019 with the vast bulk landed in late spring and summer. Young and Thomson (1927:319) noted that "*enormous shoals*" moved along the coast of the South Island between November and April. Large shoals within the Sounds are now rarely seen (Brehaut, 2017). The lack of biomass recovery suggests a tipping point has been passed either within and/or outside the Sounds. The cause is unknown, but could be related to all or a combination of overfishing reducing spawning biomass,

Fig. 8. Mass stranding of Picton herring *Sardinops sagax* in Picton, Queen Charlotte Sound/Tōtaranui, New Zealand August 1909. Courtesy: Picton Historical Society.

disease, climatic oscillations affecting upwelling, increased sea-surface temperatures, seabed disturbance, and/or sedimentation. The consequences to the food web are also speculative, but it is plausible that it has been significant for species in higher trophic levels.

3.6. Green-lipped mussels/Kūtai

A tipping point has also passed for the recovery of extensive subtidal *P. canaliculus* beds and intertidal reefs that occurred throughout Pelorus Sound up until the early 1970s (Handley, 2015). Commercial dredging of subtidal beds commenced in the early 1960s, along with harvesting by hand gathering on intertidal reefs, resulting in the collapse of the biogenic habitats (Stead, 1971; Flaws, 1975). About 350 ha of subtidal beds remained in 1969 (Stead, 1971). No surficial evidence for these beds was found when sediment cores were taken in 2015 from several of the subtidal locations mapped in 1969 (Handley et al., 2017).

Mussel beds provide a range of ecosystem services, including the formation of three-dimensional structures that provide attachment surfaces for invertebrates and seaweeds, and feeding sites and refuges for invertebrates and fish (Hewitt et al., 2005; McLeod et al., 2013). Nutrient cycling, enhancement of productivity, sediment stabilisation and sequestration, and modification of neighbouring macrofauna communities are also key ecological functions of large shellfish assemblages (Norkko et al., 2001; Hewitt et al., 2002; Gutiérrez et al., 2003).

A culturing industry developed from the 1970s using suspended ropes anchored onto surface buoys to replace the exhausted benthic stock (Dawber, 2004). By 2014, there were 565 *P. canaliculus* farms producing 50–60,000 tonnes annually, and occupying approximately 3000 ha of the Sounds (Clough and Corong, 2015). There are system-wide ecological effects related to habitat and food web alteration and change in biogeochemical processes from these farms, but it is difficult to compare ecological equivalency with the extinct benthic mussel reefs due to the lack of underpinning data (Stenton-Dozey and Broekhuizen, 2019). Live mussels frequently drop onto the benthos, but survival is generally low, likely due to predation by the 11-arm starfish *Coscinasterias muricata* (Inglis and Gust, 2003), and/or burial by fine sediment (Handley, 2015). *C. muricata* numbers are elevated under mussel farms over non-farmed areas (Inglis and Gust, 2003), which may partly be due to a loss of predators such as rock lobster (*Jasus edwardsii*) (Wing and Jack, 2014). Several *P. canaliculus* beds have been located in the outer Pelorus beneath spat collection ropes (Davidson in Handley, 2015).

Failure of the beds to re-establish more widely may be related to the loss of suitable primary and/or secondary settlement habitat for larval mussels (Handley and Brown, 2012; Handley, 2015). Bull (1976) reported that scallops (*P. novaezelandiae*), which also attach themselves as larvae using byssal threads, were found in areas of Pelorus Sound attached to brown alga (*Cystophora retroflexa*), red algae attached to *A. zelandica*, and drifting *Z. muelleri* debris; however, sand, mud, and broken shell in the area was not colonised. The loss of species that once provided settlement surfaces for *P. canaliculus* and *P. novaezelandiae* has likely been driven by siltation, concomitant loss of water clarity for photosynthesis in deeper water, and bottom-contact fishing methods (Handley and Brown, 2012). For these reasons, Handley and Brown (2012) proposed for neighbouring Tasman Bay that recruitment failure due to habitat change appears to be the most likely reason for lack of mussel bed recovery. A non-mutually exclusive hypothesis advanced by Handley et al. (2017) is that changes in land-use from the 1980s, and consequent reduction in nutrient run-off that coincided with declining abundance of filter-feeders in the recent death assemblage, may be implicated in recruitment failure.

3.7. Macrocystis pyrifera - giant kelp/rimurimu

Large brown seaweeds are important ecological engineers in temperate reef ecosystems (Foster and Schiel, 1985; Dayton et al., 1998; Coleman and Williams, 2002). Kelp also provide organic matter, which helps to drive productivity and support fish biomass in sheltered temperate coastal systems (Wing and Jack, 2014; Udy et al., 2019a). Warming coastal waters, and increased herbivory from sea urchins after loss of top predators, have caused localised and regional declines of kelp (Estes et al., 1989; Dayton et al., 1998; Wernberg et al., 2016).

M. pyrifera is one of several brown seaweeds in the Sounds, and is valued by commercial, recreational and customary fishers for its close association with, and as a food source for, the abalone, *H. iris*. Allowable annual commercial and non-commercial catches of *H. iris* have reduced in the Sounds and in the adjoining fisheries reporting area (Fisheries NZ, 2019b), in response to the decline in harvestable populations (Fu, 2016).

A contraction of the range occupied by *M. pyrifera* from inner Queen Charlotte since 1965 was suggested from local ecological knowledge as a contributing cause of *H. iris* decline, along with overharvesting (Handley, 2016). This memory may be confused with the sympatric *Ecklonia radiata* or *Carpophyllum flexuosum* however, as a survey in the 1940s located *M. pyrifera* in the outer Sounds and Tory Channel only (Rapson et al., 1942). Hay (1990) also did not observe *M. pyrifera* within inner Queen Charlotte between 1984 and 1988, but noted a retraction in range in outer Queen Charlotte from 1942. A systematic acoustic survey of kelp distributions by Neil et al. (2018) showed broad concordance with the locations of mixed tall and low statured macroalgae (species unidentified) with previous surveys, but with notable differences in the eastern and western parts of outer Queen Charlotte (Fig. 9). A subsequent seafloor imaging survey has identified extensive urchin/kina (*Evechinus chloroticus*) barrens in some of those areas (Simon Thrush, pers comm). The survival of *M. pyrifera* may reflect sites with more energetic and turbid waters, as Shears and Babcock (2007) found that *E. chloroticus* were more abundant at sheltered sites with high water clarity in outer Queen Charlotte. Two isolated small patches of low and mixed low/tall statured macroalgae, possibly *C. flexuosum*, were recently located around headlands in inner Queen Charlotte (Neil et al., 2018).

Notable size differences in *P. colias* and *J. edwardsii*, predators of *E. chloroticus*, were identified in the sole no-take marine reserve in Queen Charlotte compared to control sites after 15 years of annual monitoring (Davidson et al., 2014). Both species were significantly larger inside the reserve and were in higher densities, although not for *P. colias* in macroalgal habitats. Mean size of *H. iris* was also significantly larger inside the reserve, but abundance was likely influenced by poaching. No differences in *E. chloroticus* abundance were detected between the reserve and controls, although individuals were significantly larger within the reserve, which was attributed to predation of smaller size classes (Davidson et al., 2014).

Several non-mutually exclusive explanations have been posited for a putative *M. pyrifera* range retraction in Queen Charlotte. These included: suspended sediment levels smothering kelp and inhibiting light levels (Baker in Handley, 2016), warming sea surface temperatures (Hay, 1990), and a trophic cascade from increasing *E. chloroticus* numbers as a result of overfishing of large *P. colias* and *J. edwardsii* capable of limiting grazing urchin populations (Shears and Babcock, 2007). Sediment seems unlikely for the main stem of Tory Channel and the outer Sound, as bottom-shear stresses and current speeds are generally above the resuspension threshold for terrigenous sediment (Hadfield, 2015). Moreover, kelp of mixed stature survives in the shallower, more quiescent side bays of Tory Channel (Neil et al., 2018), where fine sediment deposition, generated by land disturbance from extensive clearfelling of *P. radiata*, has periodically occurred since the 1990s (Urlich, 2015). *Evechinus chloroticus* may be inhibited in those side bays, as their larval settlement and recruitment has been shown to be inhibited by low levels of sediments in wave-sheltered reefs (Walker, 2007). It seems plausible that there are complex interactions occurring between multiple stressors that act on *M. pyrifera* (e.g., Mabin et al., 2019) and *E. chloroticus* populations. In the inner Sound, and in more quiescent, shallower side bays of the outer Sound, overfishing of *E. chloroticus* predators, and high

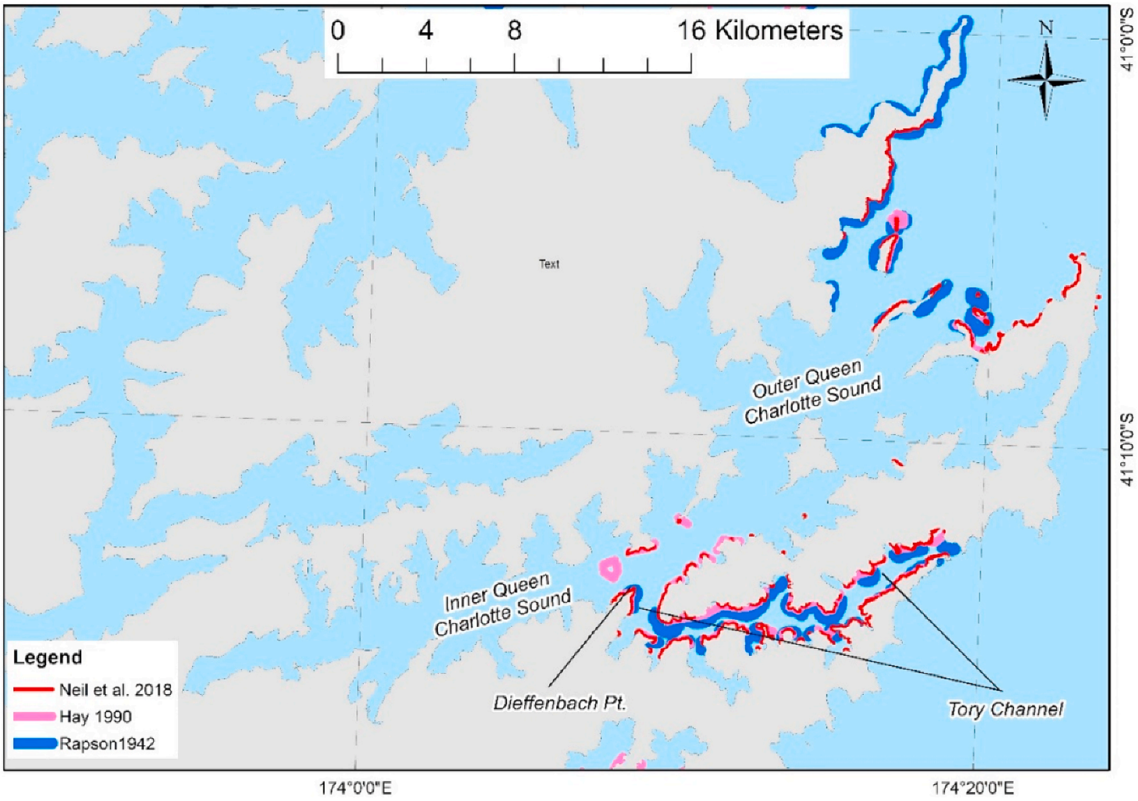

Fig. 9. Maps of conspicuous kelp distributions from surveys done in 1942 (Rapson et al., 1942), 1984–88 (Hay, 1990) and 2018 (Neil et al., 2018). Note: the 2018 survey did not include the large neighbouring bay to the northwest of outer Queen Charlotte (partially covered by the scale bar).

summer water temperatures are potentially more important limiting factors for *M. pyrifera* and other large macroalgae. Whereas, in outer Queen Charlotte waters, localised overfishing of *E. chloroticus* predators may help explain the patchy changes in *M. pyrifera* distribution (Shears and Babcock, 2007). Temperature data from 2011 to 2018 shows that Tory Channel is 2–3 °C cooler and outer Queen Charlotte 1-2 °C cooler in summer, and less stratified than side bays and within the inner Sound (Broekhuizen and Plew, 2018).

Supporting the hypothesis of a macroalgal decline, Udy et al. (2019b) used stable isotopes to argue that organic matter derived from macroalgae is significantly less in Queen Charlotte than in Fiordland ca. 1000 km southwest. These authors compared the contribution of macroalgal-derived organic matter with pelagic sources in supporting exploited and non-exploited fish species common to both areas. Backed by unpublished data of δ13C of collagen from fish bones, Udy et al. (2019b) hypothesise that anthropogenic impacts have contributed to up to 40% less organic matter to support omnivorous fish species in Queen Charlotte than in Fiordland waters. In a companion paper, Udy et al. (2019a) examined the movement of organic matter derived from macroalgae through different trophic levels of fish between the same regions. In Fiordland, a greater proportion of cumulative fish biomass at different trophic levels was supported by macroalgal subsidies; in contrast, phytoplankton from pelagic sources subsidised higher trophic levels in Queen Charlotte. Whether this may change as the volumes of organic matter discharge from additive feed to cage-farmed chinook salmon (*Oncorhynchus tshawytscha*) increase is a subject for future research.

4. Discussion

4.1. Five significant transformative periods

The arrival of European and North American sealers and whalers in the 1820s marked the beginning of a major change to the marine ecology of the Sounds. Decimation of southern right whale (*E. australis*) populations within a decade, and the extirpation of the remaining fur seal (*A. fosteri*) stronghold in Port Underwood marked the second significant transformation of the food web (Table 2). The first was the extirpation by Māori of Waitaha penguin and pinniped species several hundred years prior. The third transformation occurred from 1864 to the early 1900s characterised by the overexploitation of the pilchard (*S. sagax*), and pressure on other fisheries as reflected in a petition to Parliament by Māori in the 1880s, and newspaper records of localised depletion of oysters in 1894. On land, the widespread clearance of old-growth indigenous forests for timber, and the sluicing and soil disturbance of the gold rush, resulted in significantly elevated quantities of fine sediment deposited onto intertidal and subtidal habitats. Seabed disturbance characterised the fourth transformation, which became more widespread with the arrival of the first trawler around 1904. By the late 1930s, alarm over trawling damage was being linked to the decline in recreational fish species abundance. Extensive destruction of subtidal *P. canaliculus* beds occurred in the 1960s from the effects of commercial dredging. Soil erosion under pastoral farming on steep hill country continued to contribute terrigenous sediment into estuaries, side bays and inlets. The fifth transformation began in the mid-1980s, which was a time of economic and social upheaval in New Zealand. On land, following the cessation of subsidies for fertiliser, pastoralism on steep hill country in the Sounds and its inflowing catchments gave way to regeneration of indigenous vegetation, or the widespread establishment and extensive harvesting of *P. radiata* plantations. There are now ca. 31,688 ha of different-aged *P. radiata* stands, which are clearfelled at maturity (ca. every 30 years), with continuous harvesting in different patches across the landscape exposing soils to intensive erosion in the period between harvesting and reestablishment of stabilising root networks from new plantings. In coastal waters, a surge in development of suspended rope *P. canaliculus* aquaculture replaced the depleted

Table 2
Five major transformative periods of ecological change to ecosystems in the Marlborough Sounds, New Zealand, identified from a literature synthesis.

Period	Drivers of change in the marine environment	Effects and food web consequences
1300–1828	(i) Severe decline of fur seals, localised extirpation of sea lions, elephant seals, Waitaha penguin. (ii) Slash and burn and fallow method of seafood harvesting, causing localised and temporal depletion of marine life.	(i) Release of fish from predation, likely increased biomass of middle trophic groups. (ii) Patchy localised depletion of shellfish and fish stocks.
1828–1864	Decimation of southern right whale populations and localised extirpation of fur seals.	Reduced nutrient transfer from whale feeding areas. Reduced fish predation.
1864–1904	(i) Overexploitation of the pilchard 'Picton Bloater'. (ii) Depletion of dredge oyster beds in Oyster Bay, Tory Channel from overexploitation. (iii) Extensive deforestation of native forests for timber and intensive mining for gold. (iv) Repeated burning to retard secondary forest regeneration and induce pasture on steeplands. Severe periodic wildfires from escaped burns.	(i) Loss of biomass available to middle/upper trophic groups in winter months. (ii) Loss of biogenic oyster bed habitat. (iii - iv) Significant elevated sediment deposition onto subtidal and intertidal areas. Alteration of nutrient and carbon cycling from change in interaction networks. Decline in species richness and community compositional change. Stress on biogenic habitats and reduction of blue carbon (seagrass).
1904–1986	(i) Bottom trawling commenced and by 1939 blamed for destroying fish breeding grounds. (ii) Dredging resulted in destruction of subtidal green-lipped mussel beds. (iii) Introduction of motorised boats in 1910s increased fishing pressure, as well as efficiency of whaling. Switch from southern right whale harvesting to humpbacks and other species before cessation of whaling in 1964. (iv) Foreign fishing vessel commercial catch peaked prior to 1986 Quota Management System introduction. (v) Pastoralism at maximal extent but repeated soil erosion on steeplands, associated with underlying erodability and repeated burning to retard forest regeneration and maintain pasture. Widespread aerial application of fertilisers on pasture from 1950s largely replaced burning in succeeding decades. (vi) First plantations of exotic *P. radiata* planted on steeplands from 1930s, increased 1960s–1980s. Harvesting commenced 1970s	(i-ii) Loss of biogenic habitat and failure to recover. Alterations in benthic-pelagic coupling, increased contribution of pelagic productivity to food web. Decline in fish and shellfish abundance. (iii-iv) Loss of significant biomass from overfishing of pilchards, affecting food webs. Change in fish distributions as long-lived species such as hāpuku (grouper) became rarer and restricted to deeper waters. (v-vi) Significant elevated sediment deposition onto subtidal and intertidal areas (see row above for effects). (v) Nutrient runoff from repeated fertiliser applications may have increased primary productivity and affected benthic community response.
1986-present	(i) Warming sea surface temperatures (ii) Fishing restrictions in response to concern about declining blue cod fishery. First marine reserve gazetted at Long Island/Kokomohua. (iii) Overexploitation of scallop beds from dredging caused loss of beds in inner Pelorus (Mahau), and loss of horse mussel beds in outer Sounds. (iv) Government subsidies for fertiliser to maximise pastoralism phased out in 1985. Pastoral areas either abandoned, or planted in *P. radiata* plantations and extensive clearfell harvesting commences from late 1990s/early 2000s. (v) Increased development and expansion of suspended rope culture industry of green-lipped mussels to replace exhausted benthic stocks. (vi) Development and ongoing expansion of introduced chinook salmon cage farming using additive feed in both sounds. (vii) High-speed catamarans introduced on the inter-island ferry route along Tory Channel and inner Queen Charlotte, operated from 1994 to 2005.	(i) Stress on giant kelp and pāua habitat. Possible changes to wider system processes. Sea-level rise. (ii) Population declines of many fish species. Widespread urchin barrens as urchin-predator populations declined. (iii) Continued loss of biogenic habitat and failure to recover, including critical fish habitat (shellfish, macroalgae). (iv) Widespread regeneration of plant cover on hillsides abandoned for farming, stabilising some hillslopes. Inner Pelorus increased in soft mud habitat. Decline in kelp and seagrass. (v-vi) Localised over-enrichment underneath marine farms from ongoing deposition. Alteration of benthic processes from prevention of dredging/trawling underneath farms. Far-field water column effects imprecisely known. (vii) Coastal erosion and altered sediment transport. Cast mortality of intertidal and shallow subtidal organisms along ferry route.

dredged fishery, and cage farming of *O. tshawytscha* using feed pellets expanded. Serial overfishing of *P. colias*, *P. auratus*, *P. novaezelandiae*, and *J. edwardsii* resulted in localised depletion and progressive reduction in daily bag limits and/or temporary moratoriums. Overfishing of *E. chloroticus* predators has likely contributed to change in kelp bed distributions including *M. pyrifera* at a local scale. Sustained excessive terrigenous sediment inputs, ongoing physical disturbance to the seabed, and continued overfishing, combined with invasive species, increasing sea surface temperatures, and ocean acidification are likely to exacerbate cumulative effects on key components of the Sounds marine ecosystem, possibly foreshadowing a sixth transformation.

4.2. Food web effects

Consequential to these transformations, primarily caused by overfishing, seabed disturbance and terrigenous sedimentation, has been a shift from the relative importance of benthic productivity to a greater proportion of pelagic primary productivity (Udy et al., 2019b). This is consistent with studies elsewhere where trophic cascades have occurred, disrupting the flow of organic matter following the release of urchins from top-down control by predation (Estes et al., 1989; Jackson et al., 2001). Any contraction or increasing patchiness of macroalgae distribution in the Sounds may also place a limitation on the amount of fish that can be supported (Udy et al., 2019b). Sustained fishing pressure targeting the urchin predators, *P. colias* and *J. edwardsii*, could be holding large areas in an alternate stable state dominated by *E. chloroticus* where sea conditions limit fishing or provide unfavourable habitat (Shears and Babcock, 2007; Karatayev and Baskett, 2019). The loss of *S. sagax* biomass is in effect a triple whammy, as the availability of pelagic productivity via the massive historic shoals has significantly reduced since the late 1800s. Given the reliance on pelagic subsidies of organic matter for fish at higher trophic levels (Udy et al., 2019a), light inhibition from increased turbidity may also have been an ongoing additional factor affecting primary productivity, particularly within inner Pelorus Sound (Fig. 2).

The loss of *P. canaliculus* and other large shellfish beds, and the widespread failure of regeneration, has also likely to have affected benthic pelagic-coupling, disrupted nutrient cycling and carbon burial, and altered infaunal communities (e.g., Norkko et al., 2001). The destruction of habitat has also negatively affected the abundance of the highly valued fish *P. auratus* in the Pelorus (Handley, 2015). Cultured mussel farms by contrast were found to support small demersal fish characteristic of rocky reefs (Morrisey et al., 2006). This was not always the case, as gelignite was used to protect mussel spat from *P. auratus* and other fish predators when the industry was in its developmental stage in the 1970s (Dawber, 2004), and dynamite was known to be used as a fishing tool by commercial fishers (Handley, 2015). A broadly similar pattern of significant biomass loss from overexploitation of major fish groups was also identified in the Hauraki Gulf, in northern New Zealand (Pinkerton et al., 2015). Using multiple lines of evidence, these authors developed sophisticated food web models of organic matter flow representing all the major biota, from bacteria to whales, to estimate the trophic importance of each group at different time periods over the last 800 years. Significant alterations of the food web occurred over time, with cetaceans and pinnipeds decimated or locally extirpated, during a similar historical period to the Marlborough Sounds, with concomitant reduction in trophic importance. These authors found that many middle trophic groups changed in biomass, such as small and large pelagic fishes, macrobenthos, squid, macrozooplankton, and gelatinous zooplankton, which are generally important prey items of the middle and upper level predators. These groups, along with benthic epifauna, macroalgae, and phytoplankton did not change much in their modelled rank trophic importance, and this may have buffered any effects of changes in higher trophic levels on the lower food-web. Pinkerton et al. (2015) noted that fur seals have reappeared in the Hauraki Gulf, as in the Sounds, some half a millennium after their removal; and highlight the potential for changes in trophic importance and consequential system-level effects.

There has also been profound loss of biogenic habitats in the Hauraki Gulf from seabed disturbance, and removal of extensive subtidal *P. canaliculus* beds by dredging in the 20th century (Pinkerton et al., 2015). Recent attempts to restore beds in the Gulf by placing adult *P. canaliculus* on the benthos, had had mixed success due to predation by *C. muricata* (Wilcox and Jeffs, 2019). Restoration trials are also planned for Pelorus, but may require a multi-functional approach with contemporaneous placement of habitat for *J. edwardsii* to attempt to control *C. muricata* attracted to benthic mussels (Inglis and Gust, 2003).

4.3. Managing stressors to protect foundational species

The decline and loss of foundational species has had significant effects on ecological functioning as well as the integrity and extent of biogenic habitats. The top anthropogenic stressors in the Sounds, in addition to ocean acidification and warming sea surface temperatures, are terrigenous sedimentation, dredging, and bottom-trawling, consistent with a national assessment of the direct threats to New Zealand's aquatic ecosystems and biodiversity (MacDiarmid et al., 2012). Bottom trawling is not permitted in Queen Charlotte and the inner Pelorus and, but is near ubiquitous in the rest of Marlborough's entire 725,000 ha coastal marine area (Halley, 2018). Prior to the moratorium in 2017, recreational dredging for *P. novaezelandiae* had limited spatial restrictions and was controlled by a seasonal closure, whereas commercial dredging was confined to outer Queen Charlotte and outer Pelorus outside of the seasonal closure (Halley, 2018). Controls on the recreational fisheries include daily bag limits, minimum legal sizes, restrictions on certain fishing methods, and seasonal and spatial closures (Halley, 2018). Despite these constraints on fishing activity, the decline of valued commercial, recreational and customary species has not been stemmed.

Marine protection is also inadequate within the Sounds with only one fully protected reserve under the 1971 Marine Reserves Act comprising 625 ha or $<$0.001%. The management of biogenic habitats outside of this reserve has become the subject of a legal battle between the local authority and central government, as to whether the 1991 Resource Management Act or the 1996 Fisheries Act takes primacy in the protection of biodiversity within the Territorial Sea. The local authority implemented a ban in 2016 on dredging and bottom-trawling at identified biogenic habitats assessed as having significant biodiversity values. This was contested by the government who argued that this was reserved under fisheries legislation. However, a recent judicial ruling determined that local authorities are compelled by the resource management law to protect biodiversity, which goes beyond the ambit of fisheries law with its narrower focus on protecting fisheries resources (Court of Appeal, 2019). The consequences of this are yet to be played out, but it is likely that there will be more fine-scale zoning of areas in Marlborough's coastal waters where limited seabed disturbance could be permitted, along with better integration between agencies acting under different legislation.

The local authority also has responsibility for managing sources of terrigenous sediment, particularly from *P. radiata* clearfelling, which has generally not been effective despite compelling research of the environmental effects since the late 1970s (Urlich, 2015, 2020). The National Environment Standard for Plantation Forestry (NZ Government, 2017) enables the local authority to put more stringent rules in place to protect coastal water and benthic habitats in the Sounds to comply with the New Zealand Coastal Policy Statement (2010). The retirement of erosion-prone land by instituting replanting controls on steep faces and in gullies is a matter of urgency, given the increasingly muddy state of estuaries (Urlich, 2015). The local authority failed to do this in its recent proposed coastal plan in 2020 (Urlich, 2020).

4.4. Ecosystem-based management (EBM)

The need for improved catchment management, along with more effective integration of marine management responsibilities and marine spatial planning, point towards EBM. We suggest the aim of EBM in the Sounds would be to restore ecological functions and processes to enable biodiversity to be maintained (McLeod and Leslie, 2009; Gladstone-Gallagher et al., 2019). By 'maintain', we use the definition of Urlich et al. (2018b): "*Take action to preserve or retain natural species diversity (including foundational species) from loss and keep the functioning of ecological complexes effective and unimpaired from deterioration.*"

To do this, the objective should be to minimise or eliminate disturbances that disrupt ecological functioning (McLeod and Leslie, 2009; Benson and Craig, 2017) with the goal of improving ecological health and ecosystem resilience in the face of the dynamic uncertainties and consequences of climate change (Lotze et al., 2006; Benson and Craig, 2017). Benson and Craig (2017) argue that this should include a strong societal dimension, which in the New Zealand context includes a partnership with Māori iwi under the Treaty of Waitangi and a collaborative approach involving the wider community (Hewitt et al., 2018). We suggest that such an inclusive method will be a means of facilitating the restoration of the mauri (life-force) of the Sounds. Māori hold traditional knowledge known as Mātauranga Māori in transplanting shellfish and other marine management techniques (Waitangi Tribunal, 2008). The outcome of applying these knowledge streams would be to enhance the wise use of marine resources by safeguarding and restoring ecological resilience and integrity, which allows ecosystems to thrive for themselves and for current and future generations.

Marine protection is a core component of EBM in New Zealand (Thrush and Dayton, 2010; Wing and Jack, 2014). This is to protect what high quality habitat that remains, which includes spawning and nursery habitat, so as to lead to the recovery of more diverse trophic linkages including omnivorous species and top predators (Lotze et al., 2006; Wing and Jack, 2014). Calls for additional marine protection in Marlborough have been largely unheeded since the 1880s, such as the most recent request from local youth direct to the Prime Minister at her invitation (Angeloni, 2019; Urlich et al., 2019). Several community-based initiatives for additional marine protected areas since 2000 have also failed to gain any institutional mandate or support.

The recent Hauraki Gulf (ca. 700 km north) marine spatial planning process offers a potential guide for the Marlborough Sounds (Sea Change, 2017). It seeks to institute marine protection, improve fisheries management, undertake restoration efforts, and contribute to regional economic development. However, despite significant resources and the involvement of Māori, community groups, ENGOs, local authorities and government agencies, the non-statutory plan has yet to be implemented three years after it was produced. This is perhaps not surprising given the systemic institutional failure to manage New Zealand's marine environment effectively, as evidenced by the ecosystem health and biodiversity crisis reported in the Our Marine Environment 2016, 2019 reports (Urlich et al., 2018a).

The urgent need for transformative change to conserve and restore nature is now recognised as requiring innovative, adaptive, inclusive, informed, and integrative management (Diaz et al., 2019). To help change the narrative back to 'clean and green' from 'brown and down' in the Marlborough Sounds, we suggest EBM needs to be configured to achieve the following:

- Manage seabed disturbance to protect remaining biogenic habitat and encourage habitat regeneration and restoration, and enhance connectivity between existing habitats.
- Undertake a participatory process to co-develop and institute a network of different types of protected marine areas, including no-take to provide biodiversity sanctuaries.
- Rebuild populations of pilchards (*S. sagax*).
- Manage exploitation of fisheries within the Sounds such that top predators are more numerous and widespread to control populations of the urchin (*E. chloroticus*).
- Reduce sedimentation by stabilising land to reduce frequent and extensive disturbance through retirement of steep erosion-prone faces and gullies from plantation forestry.
- Co-create and implement just-transition schemes to buy out forestry cutting rights and transition commercial fishers to less environmentally damaging fishing methods.
- Co-develop a research strategy to inform management and meet community aspirations.
- Include a Māori approach to ecosystem-based management through recognition of traditional knowledge (Mātauranga Māori) and indigenous stewardship (Kaitiakitanga).

These approaches are consistent with the levers of transformative change identified by Diaz et al. (2019). We also support the call of Jackson et al. (2001) for bold experiments on the scale of ecosystems to test integrated management methods to achieve multiple goals (see also Leslie and McLeod, 2007). We suggest a staged transition to EBM could be achieved by large experimental areas in the Sounds, which would become accepted as "our marine labs" in the public consciousness, and so build knowledge and support for eventual full implementation of EBM. The lessons learnt along the way will, we anticipate, facilitate the co-development and eventual realisation of a vision for a more resilient social and ecological system, which is essential under climate change (Benson and Craig, 2017; Diaz et al., 2019). The historical synthesis presented here is aimed at informing that process and stimulating discussion about the future. Studies such as these go beyond individual over-exploited marine populations by taking an ecosystem approach (Lotze and Worm, 2009). This study extends the seminal work of Leach (2006) by examining post-European effects of fishing within a region. It builds on a comprehensive multidisciplinary synthesis of historical changes nationally (MacDiarmid et al., 2016b), by the inclusion of the detailed effects of historical and contemporary regional land use on the coastal marine area. The integration of changing land-use effects enables an ecosystem-based management process to more fully consider how to manage multiple, cumulative and interacting stressors in an ecologically holistic way to improve the resilience of ecosystems.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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