

Potential impacts of mechanical cockle harvesting on shorebirds in Golden and Tasman Bays, New Zealand

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ABSTRACT

The cockle (*Austrovenus stutchburyi*) is abundant in Golden and Tasman Bays, but since mechanical harvesting began there, concern has been expressed about potential adverse impacts of harvesting on shorebirds. Information for Golden and Tasman Bays is sparse, but studies in Europe show a link between the state of shellfish stocks and oystercatcher survival. Bird numbers/densities did not need to be at 'carrying capacity' to be negatively impacted by changes in their food supplies, but only within a range where density-dependent factors were operating. In addition to the direct impacts of food loss, there could also be indirect impacts of harvesting on non-target invertebrates such as tube-dwelling polychaete worms on which shorebirds feed. In a study on the Burry Inlet, UK, using methods similar to those in Golden and Tasman Bays on muddy sand with high cockle densities, impacts on non-target species were significant, recovery rates slow, and foraging by shorebirds declined after a short-term increase immediately following the harvesting. Other adverse impacts from harvesting included damage of small cockles; reduced cockle spat numbers over the short term; mixing of anoxic layers of mud with upper layers; and chemical changes in the sediments which inhibit recolonisation. Work in Europe also showed almost complete losses of eel grass (*Zostera*) beds as a result of harvesting, suggesting that eel grass beds in Golden and Tasman Bays, which are particularly rich communities and may also be important sources of cockle spat, could be sensitive to mechanical harvesting of cockles. Research needs and recommendations for Golden and Tasman Bays are listed. A precautionary approach to management is recommended, in which potentially sensitive areas (those with high cockle densities and tube-dwelling polychaetes) are set aside from harvest until the impacts on sediment structure and sustainability of the fisheries are better understood.

Keywords: mechanical cockle harvesting, benthic disturbance, fishery effects, fishing impact, foraging ecology, habitat disturbance, oystercatcher, shorebirds, New Zealand.

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1. Introduction

‘In the early 1990s oystercatchers in the UK were seen feeding in unusual areas, such as roadside verges, probing for worms. It reflected desperate times. Hundreds of starved oystercatcher corpses were being picked up along the adjacent Wash estuary and between 1988 and 1998 their numbers plunged from 47,000 to 10,000. This was due to a collapse of the estuary’s cockle fishery probably due to weather and fishing...’

‘In recent years new cultivation methods have been introduced that involve moving young mussel stock from the subtidal areas to mussel beds in the intertidal zone. The shellfish develop well in these farmed areas, but they also make them accessible to the oystercatchers and so provide crucial winter food. But the birds have performed a reciprocal service to the fishermen, since they selectively ‘weed’ the smaller shellfish and reduce overall densities, which allows a healthy harvestable mussel crop to develop.’ (Mark Cocker, *Guardian Weekly* 20 September 2000)

Cockles, also known as littleneck clams (*Chione* or *Austrovenus stutchburyi*), have been commercially harvested using a small mechanical harvester along Pakawau Beach, Golden Bay, New Zealand since 1984. The areas and amounts of cockles being harvested in the Tasman and Golden Bay areas have increased considerably since, with further applications to expand the areas pending. It has been assumed that the scale of damage such methods cause to intertidal biota is small. However, as mechanical methods become increasingly widespread both along Golden and Tasman Bays and worldwide, concern has been expressed that these may be damaging non-target infauna (Cox 1991) and may lead to depletion of food supplies for predators such as oystercatchers (*Haematopus* spp.) and other shorebirds (Lambeck et al. 1996; Piersma & Koolhaas, unpubl. report 1997*). Mechanised fishing for cockles may potentially harm the ecosystem in two main ways: directly, by causing food shortages for birds feeding on shellfish, or indirectly, by damaging non-target species and/or important habitats like eel grass beds (*Zostera* spp.) and sediment characteristics of the tidal flats.

The objectives of this report are therefore to answer the following questions:

- What scientific research has been published on the impact of mechanical cockle harvesting on shorebirds?
- What conclusions can be drawn from this research that are relevant to current and proposed cockle harvesting in Golden and Tasman Bays?
- What further information (detail research requirements) is needed to allow informed decisions on how existing and planned cockle harvesting operations may need to be controlled to mitigate any adverse impacts on shorebirds?

* Unpublished reports are listed in Appendix 1.

1.1 HISTORY OF HARVESTING

Currently two companies, Westhaven and Tallies, hold permits to harvest cockles. Since 1992 there has been a moratorium on new commercial cockle harvesting/fishing permits (Friends of Nelson Haven and Tasman Bay 2000), but both companies have been granted increases (variations) to the amount and areas of their current permits.

Golden Bay. This area has been seen as an ideal area for cockle harvesting: 'Golden Bay offers large areas of habitat suitable for the New Zealand clam (*Austrovenus stutchburyi*) and as a result, wild populations are abundant. A large tidal range and gently sloping foreshores provide broad expanses of intertidal, sandy shore which are ideally suited to large scale mechanical harvesting of wild clam stocks, artificial enhancement procedures or aquaculture activities' (Osborne & Seager, unpubl. report 1994). Commercial mechanical harvesting in the intertidal zone was begun along the Puponga/Pakawau beaches in Golden Bay by Westhaven Shellfish Company Ltd in 1984. Landings increased from 37.8 tonnes in 1984/85 to perhaps 600 tonnes in 1998/99, and averaged 325 tonnes per annum between 1986/87 and 1997/98 (T.A. Osborne, unpubl. report 1999a).

Westhaven Shellfish Ltd has also made an application for a 100 ha cockle harvesting site within Ruataniwha Inlet, near Ferry Point, north of Collingwood in Golden Bay (D. Brown, unpubl. report 2000).

Non-commercial fishing on Pakawau Beach has not been quantified. M. Bull (unpubl. report 1984) reported the main area of non-commercial fishing to be south of the picnic area at Pakawau, with the level of non-commercial fishing in the area of the commercial fishery considered to be very low (T.A. Osborne, unpubl. report 1999a).

Tasman Bay. Mechanical harvesting in Tasman Bay commenced later, but may expand. Talley's Fisheries Ltd has held a permit since 1992 for harvesting of 35 tonnes per annum from a site of approximately 500 ha at Tapu Bay, north of Motueka (Robinson & Richardson 1998). In 1999, Talley's applied for a special permit for a biomass survey at Delaware Bay, just north of Nelson, Tasman Bay. In addition, Westhaven applied for a special permit for a biomass surveys at Moutere Inlet, east of Motueka, Tasman Bay (Friends of Nelson Haven and Tasman Bay 2000).

Areas outside the Golden and Tasman Bays may also be affected in the future.

1.2 DESCRIPTION OF HARVESTING

Harvesting is undertaken with low-pressure tyred harvesters, which remove the top 5-10 cm of sand containing the cockles and pass it over a sorting grill. Four-wheeled farm bikes with trailers are used for transport. Unlike most fishing activities, all of these operations are sited in the intertidal areas.

At present there is no legal shell-size limit in the fishery, but the market determines the minimum size in the fishery (T.A. Osborne, unpubl. report 1999b). The main size range of harvested clams is 40-50 mm shell length

(regular grade). The small grade can include clams down to 35–44 mm. A petite grade includes clams from about 32–38 mm. This grade was occasionally supplied to customers in the past, but its harvesting had been discontinued by 1998. Harvesting of smaller size classes may, however, develop in the future. An assessment to determine if there were commercial quantities of cockles inside Ruataniwha Inlet found the size distribution strongly skewed towards juvenile cockles, and concluded that the huge biomass of cockles less than 30 mm would support a total allowable catch (TAC) in excess of 174 tonnes if cockles were harvested smaller than 30 mm (T.A. Osborne, unpubl. report 1999b).

1.3 PAST STUDIES AND ASSESSMENTS OF HARVESTING IN GOLDEN AND TASMAN BAYS

To my knowledge, only one EIA (environmental impact assessment) (N. Wilson et al., unpubl. report 1988), one brief assessment of the value of Ruataniwha Inlet for birds (D. Brown, unpubl. report 2000), one study on the short-term effects of harvesting (S. Brown, unpubl. report 2001), and one brief survey of the Pakawau/Puoponga intertidal area (M. Bull, unpubl. report 1984) have been completed in Golden and Tasman Bays to assess the potential impacts of cockle harvesting on shorebirds and other fauna.

For the EIA, a small experiment was conducted to assess the potential impacts of mechanical cockle harvesting on the Puoponga/Pakawau Beach. An area of 18 m by 50 m was marked out, with one section dug over and sampled and one section left as a control on 24 July 1988, with further samples taken again on 27 and 30 July 1988 (N. Wilson et al., unpubl. report 1988). The results were inconclusive and general, but the conclusion was drawn that based on ‘our inability to find evidence to the contrary, we conclude that is no threat to bird life...imposed from this source’ (N. Wilson et al., unpubl. report 1988).

However, there are several problems with this study and its conclusions. The experiment was conducted over a very short period with only a single test plot. At least one assumption of the experiment was violated (i.e. that the area was reasonably homogeneous), and this was further aggravated by a 30-year storm and flood. Because of the high degree of variation between samples, an additional control/dug area was created and sampled. It is also not clear if the area examined had been previously fished or not; if it had been fished before, the community might have been changed prior to the experiment (see Roberts 1997; Hall 1999).

Finally, the results suggest changes in the benthic community indicative of disturbance. Wilson et al. (unpubl. report 1988) state that there might have been an increase in gastropod numbers including *Xymene plebeius*, a predatory gastropod which bores holes in bivalve shells, and in polychaete worms, which are often characteristic of disturbed areas (Dauer & Simon 1976; Roberts 1997). There were problems with sampling of the worms, as well as confounding factors resulting from a storm which exposed deeper areas where worms tend to be more common. It was recommended in Wilson et al. (unpubl. report

1988) that, to better monitor changes to the invertebrate populations, plots within the commercial area be marked out to act as reserves or controls.

In addition to the EIA, there have been two brief studies of the intertidal cockle beds in Golden Bay in association with trials of a mechanical cockle harvester. Neither study revealed any major changes in the ecology of the harvested area (M. Bull, unpubl. report 1984, S. Brown, unpubl. report 2001). However, G.A. Knox (pers. comm. 12 June 1991) in an evaluation of the first of these studies felt more detailed studies would be needed to substantiate this. He recommended that quantitative surveys of the distribution and density of the invertebrate fauna should be carried out in areas prior to harvesting and at intervals after the harvesting and adjacent control transects should also be monitored. It does not appear that any of these recommendations were implemented.

There were problems of methodology and analysis of the study by S. Brown (unpubl. report 2001) including: only a single re-survey one week after the harvest event; small sample sizes (three replicate cores for each plot/treatment); relatively small, narrow plots (3 × 10 m) (potentially allowing rapid recolonisation by surrounding benthic fauna); high natural variability (masking changes due to harvesting); and methods of analysis which would be unlikely to detect changes in abundance of individual species (A. Baxter, pers. comm. 12 July 2001). A significant reduction in species diversity was reported for one of the three samples but was discounted in the report as not being generally applicable; however, detecting a change in one-third of the sites in spite of the naturally high variability is potentially significant.

A survey of birds using the Ruataniwha Inlet was done by D. Brown (unpubl. report 2000) on 7-8 June 2000 to quantify the relative importance of the site to shorebirds, especially the South Island pied oystercatcher. He recorded 150-180 South Island pied oystercatcher using the inlet, and a total of at least 785 foraging for food in the vicinity of Collingwood/Ruataniwha Inlet. The assumption is made that since the proposed harvesting site occupies no more than 20% of the estuary its potential effect would be limited to this relative proportion of the birds utilising the inlet. However, extrapolating from high-tide roost counts can be quite inaccurate for estimating the impacts of habitat loss on birds (Evans 1995). As an example, there were plans to reclaim an area on the Medway Estuary in south-eastern England. It was argued this would have little or no effect on important populations of wintering shorebirds as the amount of area to be removed was only 1% of the area. Using high-tide counts, the only way to predict the effect on wintering waders was to assume that if 1% of the area was to be removed, 1% of the total numbers of the bird at roost sites would be affected. However, using data from low-tide counts it was possible to compare the two. The numbers of redshank (*Tringa totanus*) and dunlin (*Calidris alpina*) predicted to be impacted by high-tide counts were 31 and 298, but actual low-tide counts showed many more—200 and 520, respectively. A survey of this type cannot estimate use by arctic migrants (since peak use by these species is over summer). Nor can it accurately estimate the potential impacts since the importance of this area may be highly variable depending on the time of year, or the importance of the area proposed for harvest may be disproportionately high relative to the other nearby areas.

Numerous unpublished assessments of the cockle resource have been carried out for the permit-holding companies (Stark & Asher, unpubl. report 1991; T.A. Osborne, unpubl. reports 1992, 1998, 1999; Osborne & Seager, unpubl. report 1994; K. Grange, unpubl. report 1995; Forrest & Asher, unpubl. report 1997; Robertson & Asher, unpubl. report 1999).

2. Ecology of Golden and Tasman Bays

2.1 GENERAL BENTHOS AND SEDIMENT

The Golden Bay intertidal area of beaches and sand flats, relative to the length of coast, is one of the largest in New Zealand (Grange, unpubl. report 1995). In the southern portion, the beaches are exposed to westerly winds with a fetch equivalent to the width of the Bay. This creates more dynamic, steeper beaches with coarser, well sorted sediments which generally support few intertidal species (Grange, unpubl. report 1995). In the northern portion, where there is little fetch and wave energy is dissipated by broad, gradually sloping beaches or banks of sand and mud, organic material collects and enriches the beaches and allows development of sea grass or eel grass (*Zostera*) beds (Grange, unpubl. report 1995). The intertidal environment of Golden Bay has been reported qualitatively by Bergquist et al. (unpubl. report 1975).

Benthic invertebrates have been sampled at various sites around Golden and Tasman Bays. The most extensive study has been completed at Farewell Spit (Battley 1996). He found 70 taxa were present, mostly worms (41 taxa). There were also 20 Crustacea (including 10 amphipods, 2 isopods, 1 cunacean, 3 shrimp, 4 crabs); 4 bivalves, and 5 gastropods (Battley 1996). At Pakawau/Puponga, the intertidal sand flats supported a rich fauna of at least 32 species in total comprising 2 anemones, 1 nemertine worm, 9 gastropods, 4 bivalves, 16 polychaete worms, 6 amphipods, 5 isopods, 4 mysid crustaceans, 1 shrimp and 4 crabs (M. Bull, unpubl. report 1984; Wilson et al., unpubl. report 1988). NIWA was commissioned to do a biological survey for SeaLife Investments Ltd in Golden Bay at the Motupipi River mouth, near Pohara Beach (K. Grange, unpubl. report 1995). They recorded a total of 31 species, of which the cockle was the most widespread and abundant species. NIWA also described five communities and their associated habitats (communities being defined by the composition and abundance of the dominant species), and reported which species were endemic to each community.

Inter-annual differences in invertebrate populations can be pronounced, which has repercussions both for bird numbers and predation pressure on alternative prey. Feeding conditions vary, especially as prey and size-selection differs between species. Battley (1996) found that the polychaete worm *Travisia* showed a great increase in flesh content over summer at Farewell Spit, reaching a peak in April, but very few were present in the winter of 1994, so there had

been a collapse in population over a couple of months. Cockles did not show large changes in density through a year of study, but there was a variation in flesh content and size-structure, causing biomass to be lowest in winter. Another bivalve mollusc, *Macomona*, showed little variation in condition through the year, but in winter lived at a greater depth, making it less easily available to waders (Battley 1996).

***Zostera* beds**

The *Zostera* beds are especially rich areas in terms of numbers and variety of invertebrates and also biomass of cockles present. K. Grange (unpubl. report 1995) reported that, of the areas surveyed at Motupipi, Golden Bay, the most important community ecologically was the *Tellina/Zeacumantus* or wedge shell/turret community, which also had the highest densities of cockles recorded in the area. This community occurred among the *Zostera* beds and supported the greatest number of species, had significantly more species per sample, and contained the largest number of endemic species.

In some sites significant populations of cockles are unavailable for harvest because they are located within *Zostera* beds, which are protected under the conditions of the harvesting permits. For example, in 1999 at Pakawau/Puoponga Beach, 54% of recruited biomass was within *Zostera* patches, and so unavailable for harvest. However, there is evidence of changes in the distribution and abundance of *Zostera* throughout the year, so that some areas of clams currently protected by *Zostera* may become available to the fishery subsequently (T.A. Osborne, unpubl. report 1992, 1999). Of the 641 ha of cockle habitat within the boundaries of the Westhaven fishery, 132 ha was estimated to be covered by *Zostera* in 1998, compared with 190 ha in 1997, showing the amount of change that can happen within 12 months (T.A. Osborne, unpubl. report 1998). Several areas surveyed for potential cockle harvest have few or no *Zostera* beds (Stark & Asher 1991; Osborne & Seager, unpubl. report 1994; T.A. Osborne, unpubl. report 1999b).

2.2 COCKLES

Ecology

Cockles are filter feeders which burrow only 25 mm or so deep. *Austrovenus stutchburyi*, found only in New Zealand, is among the most numerous of all bivalves in sheltered, stable shores, and may also be found in harbour waters and along estuary channels. Cockles only flourish between low and mid-tide and in sediments containing less than 50% mud (Stephenson 1981). Smaller cockles (probably two-year-old) tend to predominate at the top of the neap flat, with larger (five- or six-year-old) cockles at the bottom (T.A. Osborne, unpubl. report 1998).

Cockles are sexually mature and reproduce at c. 18 mm shell length. Planktonic larval stages last c. 3 weeks (Forrest & Asher, unpubl. report 1997). Spat probably settle chiefly in the coarser sand at the beach top (Morton & Miller 1973). Although usually rather immobile, they can move along the surface, leaving a furrow a metre or more long (Morton & Miller 1973), but the extent of

migration of cockles across the beach and between substrate types is unknown (T.A. Osborne, unpubl. report 1998). Cockles may live up to 20 years (Owen 1992).

Similar to the New Zealand cockle (*Austrovenus stutchburyi*), the cockle in Europe (*Cerastoderma edule*) is a shallow-burrowing, suspension-feeding bivalve with relatively short siphons (Taylor 1995). The clam *Katelysia scalarina* in Tasmania is also ecologically similar to both *C. edule* and *A. stutchburyi* (Taylor 1995).

In Golden and Tasman Bays, virtually nothing is known about natural year-to-year fluctuations in biomass which may occur as a result of a complexity of pre- and post-settlement processes which affect the size of the population. Annala & Sullivan (unpubl. report 1996) note with reference to cockles in Whangarei, that sporadic recruitment of juveniles probably leads to fluctuations in biomass. Variable recruitment is a well recognised feature in most marine species having planktonic larval phases (Forrest & Asher, unpubl. report 1997). In the Wadden Sea, Netherlands, the total benthic biomass varied twofold in the course of the year (Beukema 1974). Spatfall of cockles does not occur each year. In one study area, settlement of cockles occurred in seven of eleven summers, of which only three were substantial (Zwarts et al. 1996a).

Densities and biomass

The impacts of harvesting may be related to cockle densities, with more negative impacts associated with areas of higher cockle density (see, for example, Ferns et al. 2000). At the intertidal sandflats of Motupipi, Golden Bay cockles were recorded at a maximum abundance of 450/m² (K. Grange, unpubl. report 1995). Bergquist et al. (unpubl. report 1975) also recorded cockle densities at a maximum of 460/m². At Ferry Point, in the two strata with the highest densities, cockles were found at 623 and 535 individuals/m² (Forrest & Asher, unpubl. report 1997). Biomass (all sizes) for these areas was 10.25 and 9.56 kg/m². In a later sample, mean densities for the three areas of highest densities were 974 (eel grass beds), 443, and 386 (unharvestable tidal channel) cockles/m² (Robertson & Asher, unpubl. report 1999).

Some studies have only reported the biomass of cockles, and not the numerical densities. Although density and biomass are related, some areas with many smaller cockles could have lower biomass than areas with fewer, larger cockles. In the Tapu Bay/Motueka River area there were nearly 4000 tonnes of cockles within the 306 ha area surveyed, at an average density of 12.75 tonnes/ha. Of four areas of high cockle biomass identified, the two highest had 52 and 41 tonnes/ha and the other areas had an estimated 26 and 32 tonnes/ha (Stark & Asher, unpubl. report 1991). At Ruataniwha Inlet, cockle biomass in the two richest areas surveyed was 39.2 and 35.5 tonnes/ha (or 3.92 kg/m² and 3.55 kg/m²), with an averaged biomass of 23.1 tonnes/ha (calculated from data in T.A. Osborne, unpubl. report 1999b). At Pakawau/Puponga average biomass over the entire area of 642 ha was 1.35 kg/m² (T.A. Osborne, unpubl. report 1999a).

2.3 SHOREBIRDS

General

Waders or shorebirds are important, abundant and highly visible components of coastal areas (Sagar et al. 1999), and this group of birds has been particularly affected by a wide variety of human activities such as loss of intertidal feeding habitat, recreational use of estuaries (Davidson et al. 1991), global warming, changes in sea-level, and adverse factors in other parts of their range. Waders migrating to New Zealand from the Northern Hemisphere mainly belong to the East Asian-Australasian flyway population (Watkins 1993), and at stop-over sites in Asia they are under increasing pressure from habitat loss and degradation caused by human settlement and encroachment, drainage for agriculture, pollution, and fishing and associated disturbance as well as hunting (Melville 1997; Parish et al. 1987; Scott & Poole 1989).

Along the coastlines of Golden and Tasman Bay the main species likely to be affected by cockle harvesting are South Island pied oystercatcher (*Haematopus finschi*) and bar-tailed godwit (*Limosa lapponica*), although variable oystercatcher (*Haematopus unicolor*) and banded dotterel (*Charadrius bicinctus*) may also be negatively impacted. Red knot (*Calidris canutus*) occur on Farewell Spit, would potentially be affected by harvesting, and have been showing a long-term decline in numbers. It is not known if they feed in the areas currently subject to cockle harvesting/applications.

Regional wader counts by Ornithological Society of New Zealand (OSNZ) from 1983 to 2001 along the northern coast of the South Island (from Delaware Bay to Farewell Spit, plus Whanganui Inlet) reveal the importance of the area to various shorebirds (Table 1).

Along the northern coastline of the South Island, Farewell Spit is the area of most importance to migratory waders (see for example Sagar et al. 1999) and has been set aside as a Nature Reserve. The shorebird fauna of Farewell Spit is dominated bar-tailed godwit and red knot (*Calidris canutus*), both arctic breeders, and South Island pied oystercatcher.

Various shorebirds use the areas that are being, or are proposed for, harvest. For example, at Tapu Bay/Motueka River area, Tasman Bay, Stark & Asher (unpubl. report 1991) noted that 'pied, variable and black phase oystercatchers

TABLE 1. WADER NUMBERS FOR THE NORTHERN COAST OF THE SOUTH ISLAND (R. Schuckard, pers. comm. 2001).

	AVERAGE 1983-2001	NEW ZEALAND TOTAL*	FAREWELL SPIT
Bar-tailed godwit	20 000	101 698	14 000
SI pied oystercatcher	13 500	112 675	7 000
Banded dotterel	1 300	20 000	1 000
Variable oystercatcher	285	4 000	60
Red knot	14 000	60 000	14 000

* Figures for 1983-94, from Sagar et al. (1999).

were observed feeding through out the entire area in most substrates. They gathered in large numbers...when the tide was in....' Flocks of up to 100 bar-tailed godwit, plus solitary individuals, were seen over the entire area, and banded dotterels were 'quite numerous and distributed over the area feeding.' In Golden Bay, Butler (in Wilson et al., unpubl. report 1988) recorded all four species using the area being harvested by the Westhaven Shellfish company (Table 2).

Species accounts

South Island pied oystercatcher

Oystercatchers (family Haematopodidae) are large and conspicuous waders. Two species occur in the Golden and Tasman Bay areas, variable and South Island pied oystercatcher. Until recently the South Island pied oystercatcher was considered to be a subspecies of the cosmopolitan species that occurs in Europe. However, recent authorities (e.g. Marchant & Higgins 1993) consider that it is a species (*H. finschi*) endemic to New Zealand. The population declined over the period from 1870 to 1940, but this trend reversed following legislation in 1940 which prohibited shooting of shorebirds (Sibson 1966). The population subsequently increased, with numbers rising from an estimated 49 000 in 1970-71 (Baker 1973) to an estimated 112 000 in the period 1983-1994 (Sagar et al. 1999).

South Island pied oystercatcher breed mainly inland east of the Southern Alps, from Marlborough to Southland, on shingle riverbeds and farmland (Turbott 1990; Heather & Robertson 2000). After the breeding season (usually late December), birds migrate to non-breeding grounds which are usually in coastal areas within New Zealand (Baker 1974a; Sagar & Geddes 1999). Birds have high fidelity to wintering sites within and between years (Sagar & Geddes 1999). In a study of birds in breeding in Canterbury, Sagar & Geddes (1999) found that, with only two exceptions, all birds wintered north of breeding area. Of the 23 251 South Island pied oystercatcher wintering in the South Island, on average, about half occurred in the Nelson area (Sagar et al. 1999). Numbers have increased dramatically in the North Island. At Manukau and Firth of Thames on the North Island there were fewer than 500 in 1941 (Sibson 1966), but in 1972 there were about 12 000 birds (Baker 1973; Veitch 1978). Similar to European oyster-catchers, it appears that first-year birds may have the highest mortality rates. Of five banded South Island pied oystercatcher found dead, four were under one year-old (Sagar & Geddes 1999).

TABLE 2. WADER NUMBERS REPORTED IN EIA HARVEST AREAS.

	WINTER			SUMMER
	Jun 87	Jun 88	Jul-Sep1988	Nov 1987
Bar-tailed godwit	0	64	133-135	445
SI pied oystercatcher	972	145	324-125	147
Banded dotterel	0	0	0-c. 30	2
Variable oystercatcher	9	2	c.10-13	3

Baker (1969, 1974b) made a detailed study of winter feeding of South Island pied oystercatcher. Mollusca, particularly bivalves, made up 90–95% of their diet by weight. Predominant species were cockle, pipi (*Amphidesma australe*) or tuatua (*A. subtriangulatum*), depending on location. At a beach study area near Collingwood, cockles of valve lengths from 20 to 55 mm were taken by birds, apparently in proportion to their occurrence. Pipsis were also taken, and were the dominant item farther south at Parapara. At Collingwood, some South Island pied oystercatcher fed only on the sea-anemone (*Isactinia olivacea*) which was locally abundant on the shells of pipsis. Most of the birds, however, preferred to eat pipsis and ignored the anemones. Earthworms (*Allobophora caliginosa*) and porina larvae (*Wiseana* spp.) were also taken. Battley (1996) found that oystercatchers at Farewell Spit specialised on cockles, mussels, and *Travisia olens* or other polychaete worms. *Travisia* gave the highest energy intake but appeared to be depleted over autumn one year (1994). Most observations by D. Butler at Puponga/Pakawau beaches (in Wilson et al., unpubl. report 1988) of birds feeding were in water, with substrates being sand (85%) and *Zostera* (7%).

Bar-tailed godwit

Bar-tailed godwit breed from northern Scandinavia across northern Siberia to Alaska. In New Zealand they are found on estuaries and sandy coasts. During summer they are the most abundant of the Northern Hemisphere migrants in New Zealand, with estimated populations of c. 102 000, about 30% in the South Island (Sagar et al. 1999). The three most important areas (over 10 000 counted each summer) are at Manukau and Kaipara Harbours and Farewell Spit; each site also supported > 1000 bar-tailed godwits during winter (Barter 1989; Sagar et al. 1999). Each year a small proportion (8–20%), presumably mainly youngsters, remain for the southern winter (Heather & Robertson 2000).

The numbers of bar-tailed godwit in New Zealand represent a significant proportion of the East Asian–Australasian flyway populations of these species (Sagar et al. 1999). The subspecies *Limosa lapponica baueri* breeds in northern Chukotka and western Alaska, and is the one found in south-eastern Australia and New Zealand. If only this subspecies is considered, New Zealand supports the majority of its population (Barter 1989; Sagar et al. 1999; D. Melville pers. comm.).

Bar-tailed godwit feed on tidal flats, most moving with the tideline, but birds slowly scatter as the tide falls (Heather & Robertson 2000). The only detailed study of godwit diet in New Zealand is at Farewell Spit, where the main prey item was the orbiniid worm *Travisia*. Birds also regularly caught slender worms (malidanids, probably spionids and orbiniids). Additionally, they may have fed on amphipods, and small fish or shrimps were also regularly caught (Battley 1996). D. Butler (in Wilson et al., unpubl. report 1988) observed godwit at Pakawau/Puponga feeding primarily in water over sand (73%) and *Zostera* (24%), with preferred food items being crustaceans and polychaete worms, with small molluscs also being taken. In the Northern Hemisphere, preferred food items are crustaceans, annelid worms, small molluscs and polychaete worms, in that order (Cramp & Simmons 1983). Similarly, in Australia, polychaete worms, bivalves, and crustaceans are the main foods taken (reviewed in Taylor 1995).

Variable oystercatcher

A protected, rare endemic, variable oystercatcher is found along the coasts of the main islands and offshore islands. Breeding and wintering distributions are similar, but there is some local movement to estuaries outside the breeding season. Some pairs defend territories year-round (Baker 1969). Nests are usually on a sandy beach just above the spring-tide level, but also on shingle beaches, rock platforms, and rarely on lake shores up to 30 km inland (Heather & Robertson 2000). Variable oystercatcher is listed as being of conservation concern (Cromarty & Scott 1996).

The diet of variable oystercatcher is mainly molluscs (especially bivalves), worms, and crabs (Marchant & Higgins 1993; Heather & Robertson 2000). D. Butler (in Wilson et al., unpubl. report 1988) observed variable oystercatcher feeding at Pakawau/Puoponga; of eight feeding observations, six were on *Zostera* beds and two on sand at edge of tide. Baker (1969) noted that bivalve molluscs form a major part of the winter diet, and that variable oystercatcher show a decided preference for mussels and limpets. Polychaetes were also a common food item.

Banded dotterels

Banded dotterel (*Charadrius bicinctus*) breeds only in New Zealand, around sandy coasts and on shingle riverbeds. It is widespread and moderately common. The population is estimated at c. 50 000, mainly in the South Island. They are found throughout the North and South Island and offshore islands. Inland breeding birds migrate to Australia and coastal areas of New Zealand. Birds breeding in move mainly to Farewell Spit. Birds breeding at coastal sites are mostly sedentary. They suffer from predation by introduced mammals and loss of habitat, but seem to be holding their own. The diet is primarily a variety of terrestrial and aquatic invertebrates (Marchant & Higgins 1993; Heather & Robertson 2000).

Foraging ecology

The shorebirds found in Golden and Tasman Bays that are most likely to be negatively affected by cockle harvesting are South Island pied oystercatcher, variable oystercatcher, bar-tailed godwit, and perhaps banded dotterel. Variable oystercatchers are also known to forage on cockles (Marchant & Higgins 1993). Bar-tailed godwit, South Island pied oystercatcher, and possibly variable oystercatcher could be indirectly affected by mechanical harvesting of cockles through losses of non-target species such as the polychaete worm *Travisia* and changes in the sediment and related communities. Dotterels and other species might be disturbed by harvesting activities and vehicle movements.

Peak metabolic costs for arctic migrant waders at Farewell Spit are expected to occur in late summer with pre-migratory preparation when nutrient deposition is occurring, and in July when maintenance metabolism is highest (Battley 1996). The thermostatic cost of residing at Farewell Spit is lowest over summer, with oystercatchers and other overwintering birds facing rising costs over the winter. Energy requirements for European oystercatchers increase by 50% if temperatures drop from thermoneutrality (10° C) to freezing point (Kersten &

Piersma 1985). Therefore, the minimal intake rate will increase by 5% for every degree that the average daily temperatures falls below the lower critical temperature of 10° C (Zwarts et al. 1996b).

In wet conditions, paddocks may be an important supplementary source of food for both oystercatcher species, godwits, and banded dotterels. During a survey of shorebirds on 17 August 1988, approximately half the godwits and 90% of South Island pied oystercatcher were feeding in wet paddocks when counted at high tide (Wilson et al., unpubl. report 1988).

3. Impacts of mechanical cockle harvesting

There are two main types of impacts likely from harvesting—direct, through removal of cockle biomass and thereby a direct food source of shorebirds, and indirect, through impacts on non-target species which provide food, or from disturbance to birds of the harvest activities.

Attempting to determine the impacts of fishing on shorebirds is difficult due to the large areas and long time-frames involved, in addition to the inevitable confounding factors that must be teased out of any ecological study. Therefore, there are no studies in New Zealand and only a few published studies from Europe directly on this topic, such as a report using 30 years of data on The Wash estuary in the UK (Atkinson et al. 2000) that has established links between fishing and oystercatcher numbers.

There is, however, an expanding literature on the impacts of mechanical fishing methods on non-target species. These studies are mostly experimental, at a small scale, and most vary in some aspects from Golden and Tasman Bays (e.g. sediment type, harvest regime, depth of harvest). There is, however, an extensive literature on the foraging ecology of the European oystercatcher (*Haematopus ostralegus*), a very close relative of the South Island pied oystercatcher (*Haematopus ostralegus* or *H.o. finschi*), which also feeds on cockles. Simulations to determine the impacts of habitat loss on this species are extensive and well tested.

3.1 DIRECT EFFECTS ON COCKLES AND THEIR PREDATORS

3.1.1 The Wash and other studies

The Wash is the most important British estuary in terms of the number of wintering waterfowl it supports (310 000). Over 30 years from 1968 to 1998 there were losses of 100 000 oystercatcher and knot (Atkinson et al. 2000). Stocks of cockle and mussel declined at a similar time to the wintering bird populations. Fishing quotas were set yearly at approximately 30% of current

stock. Mussel and cockle stock indices and severity of the winter were significant factors in determining winter (Oct-Mar) survival of adult oystercatcher. Major kills occurred when stocks of both shellfish species were low. The proportion of birds showing abnormal moult was a good predictor of survival over winter. Both suspended moult and small moult-gap successfully predicted that survival was lower in the three years when large numbers of birds died.

From the data, a link has been shown between the state of shellfish stocks on the Wash and oystercatcher survival. As expected in a long-lived species, sensitivity analysis indicated that even a small change in adult mortality can have a very large effect on population size and cause a stable or increasing population to undergo a major crash. The Wash is internationally important for oystercatcher and also contains economically important fisheries, and therefore the authors felt there was a need for both a healthy shellfishery and stable populations of birds.

The exact role of the fishery and its effect on birds is not yet fully understood as different fractions of the stock may be exploited by fishers and birds. There is evidence from the Burry Inlet, South Wales, that even though birds and fishers overlap in size of cockles taken, most bird predation is of a size class smaller than the fishers take (Norris et al. 1998). Competition may be mostly sequential rather than direct and the effects of fishing on birds may be through the effects of fishing mortality on the cockle spawning stock and its consequences for future spatfall. The implication is that sustainable management of fisheries will lead to healthy bird populations.

In other studies of oystercatcher mortality, both winter weather and shellfish stocks have been important in determining over-winter survival. Severe weather seems to have a larger effect in the Wadden Sea (Camphuysen et al. 1996), where winters tend to be harsher than in other north-west European study areas, e.g. the Exe estuary in south-west Britain (Durell et al. 2001). Cockle and mussel stock levels are also important in determining over-winter survival in the Wadden Sea (Camphuysen et al. 1996).

The Thames Estuary and Burry Inlet also support economically important cockle fisheries and internationally important populations of wintering knot and oystercatchers. Trends in these areas contrast with one another (Norris et al. 1998). On the Burry Inlet, numbers of oystercatcher remained stable or increased slightly from 1970 to 1986 before declining through to 1993 and then recovering slightly. On the Thames, there has been a consistent increase in numbers, despite an increase in cockle dredging, from 5000 birds at the start of the 1970s to 16 000 in 1997/98.

3.1.2 Oystercatcher foraging and fisheries

‘There can be little doubt that the European oystercatcher is the best researched wader in the world...’ (Blomert et al. 1996)

There is good information on the links between the prey species of oystercatchers such as cockle and worms, and oystercatcher population dynamics in Europe. Because European oystercatchers and South Island pied oystercatcher are closely related, and the cockles are similar in their ecology,

the ecosystems in Europe and Golden and Tasman Bays may be operating in a similar manner. Mechanical harvesting of cockles can remove not only cockles, but also associated species such as polychaete worms that are an important food source for shorebirds (see below). Since oystercatchers and humans may compete for the same food resource, it is important to know to what degree oystercatchers may switch to other food resources if, for example due to fishing, cockles or mussels numbers are reduced. Zwarts et al. (1996b) found that, especially in winter, there are only limited possibilities of taking alternative prey along the NW European coast because of changes in availability (e.g. worm species burrowing more deeply and are less) or profitability (low intake rates on grasslands).

Importance of cockles as food for oystercatchers

Cockles can be a very important food for oystercatchers. In north-west Europe, using data from 57 articles (plus student reports, theses and unpublished data), Zwarts et al. (1996b) found that mussels and cockles were universally important foods for oystercatchers, especially in winter. Along the Frisian coast, Dutch Wadden Sea, more than half of average annual consumption consisted of cockle flesh (Zwarts et al. 1996d). Oystercatchers were able to remove between 10% and 80% of their prey in winter, but in most studies it was 20% to 40% (Zwarts et al. 1996d).

In New Zealand, Baker (1969) found that molluscs, particularly bivalves, made up 90-95% of the diet by weight, with cockles being the predominant prey in some locations (e.g. Avon-Heathcote estuary). Owen & Sell (1985) note: 'Baker calculated that, at the Avon-Heathcote estuary, near Christchurch, the mean daily cockle consumption in winter by 4000 South Island pied oystercatchers was about 1 472 000 cockles, with a mean yearly food intake per oystercatcher of 190 179 cockles, and by 4000 oystercatchers of 438 876 600 cockles (A.J. Baker, unpubl. data). Battley (1996) also observed extensive and probably specialised cockle foraging by South Island pied oystercatchers at Farewell Spit.

Profitability as energy intake and species eaten

The fishery in the Burry Inlet removes less than 25% of the available stock of cockles, but even this caused a decrease in the numbers of oystercatchers feeding in the Inlet in some years (Norris et al. 1998).

Oystercatchers and other shorebirds tend to prey on a fraction of the total biomass of potential prey within a tidal flat or other feeding area. This is because not all of the potential prey is harvestable by them. The response of shorebirds to decreases in cockle number depends, at least in part, on what proportion of the decrease occurred in harvestable cockles. Zwarts & Wanink (1993) developed a model that defines the harvestable fraction of any prey species as a function of its profitability and availability. Profitability is defined as energy intake per unit handling time. Availability is a function of three factors—detectability, accessibility and ingestibility (Hulsman et al. 1996).

European oystercatchers refuse small prey due to low profitability (Zwarts et al. 1996a). Similarly, birds may also select only the most profitable prey, which will be living at a shallow depth, have slightly opened valves, and/or have thin shells. European oystercatchers are more selective when their intake rates are

high; as rates rise they successively drop least profitable prey from their diet. For instance, European oystercatchers take prey from the upper 70 mm of the substrate when intake rates are low, but only from the upper 30 mm when intake rates are high (Zwarts et al. 1996b).

Areas and sizes of cockles providing the best food

Zwarts (1996d) found from a ten-year study on the Dutch Wadden Sea a reasonably good correlation, $r = 0.74$, between bird density and total biomass (which is actually surprising, since total biomass is an inaccurate measure of the food supply, due to the highly variable fraction that is unharvestable). There was an almost perfect correlation between bird densities and biomass of prey that is both profitable as well as available, $r = 0.95$ (Zwarts et al. 1996d).

Intake rates of European oystercatcher feeding on cockles showed a clear positive correlation with size of cockles present (Ens et al. 1996a). The size of bivalves selected depends to some degree on what is available and the average intake rate during feeding (Zwarts et al. 1996a). Oystercatchers do not take cockles less than 10 mm, when older cockles, 20–40 mm long, are available (Hulscher 1982; Zwarts et al. 1996b). They do not necessarily choose the largest cockles either, because they can be difficult and sometimes even dangerous to handle. Both Sutherland (1982) and Triplet (1989 in Zwarts et al. 1996b) found that larger cockles were refused more often than small ones.

There is a general consensus that, along the coastal gradient, the best feeding areas for oystercatchers feeding on cockles are located downshore. Ens et al. (1996a) noted, ‘...it is easy to see why this should be so. Bivalves are suspension feeders, so the time they can feed increases downshore. As a consequence these bivalves generally grow better and have a higher condition the further they are down the shore.’ Feeding conditions for oystercatcher are therefore better downshore, even when prey densities are lower.

Ability to switch foods

Oystercatchers often specialise on certain food types. Those breeding on the saltmarsh of Schiermonnikoog, Netherlands, relied on two staple foods during the breeding season: the bivalve *Macoma balthica* and the ragworm *Nereis diversicolor*. Both prey are highly profitable, yet individual birds tended to specialise on one or the other for prolonged periods. Males often specialised on *Macoma*, while females often specialised on *Nereis* (Ens et al. 1996b). Individuals observed during summer at Schiermonnikoog usually kept to the same prey species for several months. Thus, most individuals can be assigned as specialising on either *Nereis* or *Macoma* (Bunskoeke et al. 1996).

Specialisation includes physical adaptations to the bill, making prey switching difficult and potentially unprofitable. Battley (1996), during his study at Farewell Spit, only once saw a worm-feeder taking a cockle at Farewell Spit, suggesting short-term prey alternation is unlikely, or at least unprofitable. Since specialisation often occurs by age or gender class, losses in food resources may impact different sexes/ages differently, which may in turn impact recruitment and/or mortality and therefore the population. In the Exe Estuary, Britain, the birds in the best condition were: adults, mussel hammerers; immatures, mussel stabbers; and juveniles, worm/clam feeders (Durell et al. 2001).

Nevertheless, switching and supplementation does occur. Oystercatchers breeding on the Isle of Schiermonnikoog in the Dutch Wadden Sea switched from a diet dominated by the bivalve *Macoma balthica* in late spring to a diet dominated by the ragworms in early summer (Bunskoeke et al. 1996). Many juveniles on the Exe Estuary switched from mussels to other prey in autumn, and most juveniles in September fed on ragworms (Durell et al. 1996).

If oystercatchers switch prey there may be a cost in terms of foraging efficiency. American oystercatchers (*Haematopus palliatus*) responded to declines in the density of oysters (*Crassostrea virginica*) on a commercial bed examined in 1979 and 1995 by not only increasing the number of species of prey eaten, but also increasing their search times (Tuckwell & Nol 1997). Juvenile birds generally have lower intake rates than adults feeding in the same place (Wunderle 1991). On the Exe Estuary, juvenile oystercatchers feeding on mussels fed at under half the rate of adults feeding in the same place and same time, but those feeding on worms were as efficient as adults (Durell et al. 1996). The adult oystercatchers were cockle specialists that had changed to worm feeding at a time of cockle shortage. Their bill tips were adapted to cockle feeding, and so may have been less effective as the pointed juvenile bill tips for catching worms.

Alternative prey may not always be available, depending on the season. For example, the ragworm is a short-lived, fast-growing species that European oystercatchers sometimes specialise on. They live in burrows that provide protection. Burrows are shallowest during summer, but even then only small ragworms are always within reach of the oystercatchers' bill wherever the worm is lying in its burrow. Therefore, it is almost certain that the availability of ragworms to oystercatchers depends mainly on their surface activity. The worms have two types of feeding methods: filter-feeding, which allows them to stay in the relative safety of their burrows; and surface feeding, when they emerge partly out of their burrows to graze the surrounding surfacing, making them much more vulnerable to predation. This type of surface feeding is more common in the middle of the emersion period, and surface activity is probably highest in summer, making them less available during winter when activity decreases and burrows are deeper (Bunskoeke et al. 1996).

Inland fields are heavily exploited in spring and summer, but only used as a supplemental food resource by most oystercatchers in winter (Heppleston 1971; Dann & Loene 1981; Goss-Custard & Durell 1984). In 22 studies of the intake rate of oystercatchers feeding on earthworms and leatherjackets (Tipulidae) all arrived at rather low intake rates. This may explain why grassland is only used as a supplemental food resource at high water in winter (Zwarts et al. 1996c). Often there is a higher proportion of young birds on fields than older birds. This is related to both age and diet. Birds eating food that is less accessible in winter (e.g. ragworms) feed in fields more often than birds feeding principally on mussels, winkles (*Littorina* sp.) or cockles. Amongst mussel-feeders, adults are seen in fields less (18%) than immatures (56%) (Goss-Custard & Durell 1983).

Decreases in cockle numbers may increase exposure of oystercatchers to parasites. The infective stages of cestodes and trematodes are probably widespread in oystercatcher marine prey (Goater 1988) and it has been shown

that, when taking clam, *H. ostralegus* is able to distinguish and avoid prey that are heavily infected by parasites (Hulscher 1982). In New Zealand, cockles are the intermediate host of the trematode *Curtuteria australis*. The incidence of metacercariae in cockles is 64%, but the incidence in oystercatcher is 50% (Allison 1979) is low in comparison. This suggests that *H. finschi* may be partially successful in rejecting parasitised bivalves. Reduced cockle numbers might result in oystercatchers having less 'choice' and therefore having to take more infected cockles, although this remains to be tested.

Critical season

In many temperate-zone birds, food shortage is an important cause of death outside the breeding season (Lack 1954; Newton 1980). It is the reason for concern when winter feeding areas are threatened by human activities that take place on the foreshore (Lambeck et al. 1996). Food shortage is an important factor that prevents some oystercatchers from surviving the non-breeding season, or from arriving on the breeding grounds in good condition (Hulscher et al. 1996).

Severe winter weather can result in the death of many oystercatchers. This has been reported in Britain, the German and Dutch Wadden Sea, along the Belgian coast and in France (reviewed in Goss-Custard et al. 1996a). Body weights on such occasions are some 40% below the typical weight of live birds. For oystercatchers on the Dutch Wadden Sea, the risk of dying in January or February was nearly four times as large as during the rest of the year (Zwarts et al. 1996c).

Severe weather has two adverse effects on the ability of oystercatchers to satisfy their energy requirements. First, it considerably increases the amount of energy required to maintain body temperature. Secondly, low temperatures and strong winds may make it more difficult to collect food. Visual cues to detect prey may become scarcer as prey become less active and, perhaps, move deeper into the substrate (Hulscher 1996). The rate at which oystercatchers caught earthworms (Lumbricidae) in fields decreased from over 7 per five minutes to less than 1 per five minutes as soil temperatures fell from 5°C to 0°C (Goss-Custard et al. 1987). Similarly on mudflats, ragworms move deeper into the sediment during the first cold spell of winter (Esselink & Zwarts 1989). However, such changes do not always occur and sometimes there are no detectable effects of low temperatures on intake rates of oystercatchers feeding on cockles, mussels, ragworms or clams (reviewed in Goss-Custard et al. 1996a). In addition, during winter, daylength shortens, and an increasing proportion of low water occurs in darkness when feeding rates may be lower for at least some prey species. At the very time when energy requirements of oystercatchers increase, ambient temperatures fall and storms become more frequent, and so the opportunities to satisfy them decrease (Goss-Custard et al. 1996a).

The main response of birds to these deteriorating conditions appears to be to increase the time they spend feeding. They feed more at the higher shore-levels on the receding and advancing tides. They feed for a greater proportion of time over the low water period and they may feed more at night than at other times of year. In some areas they feed increasingly in fields at high tide, mainly on earthworms, especially when it has rained (Goss-Custard et al. 1996a).

Alternative prey to large mussels and cockles

Beukema (1993) describes what oystercatchers did when there were no large mussels and cockles in the western part of the Dutch Wadden Sea. They did not leave the area, and the disappearance rate of alternative prey was exceptionally high, suggesting that birds started to take prey they usually ignored. Although the winter of 1990/91 was mild, mortality was relatively high (Camphuysen et al. 1996). Apparently, the consumption rate was too low to meet energy requirements, as some of the birds starved to death (Zwarts et al. 1996b).

Oystercatchers tend to be quite faithful to their wintering sites. Studies of marked individuals confirm that oystercatchers on mussel beds, in the Exe Estuary, Britain were extremely faithful to their wintering grounds. They occupied well-defined home ranges to which they returned year after year (Goss-Custard et al. 1982). Others roam more widely, although even these birds usually have only a few favoured feeding areas. In New Zealand, Sagar & Geddes (1999) found South Island pied oystercatchers also tend to be faithful to their wintering sites.

Reduced food availability might also increase susceptibility to parasites and disease. Camphuysen (2000) reported over 20 000 eiders (*Somateria mollissima*) were found dead between November 1999 and May 2000 on the Wadden Sea. Food shortages (primarily cockles and mussels) were thought to have caused starving eiders to become more prone to parasites and infections (2000). Interspecific competition for scarce prey and starvation can lead to chronic stress (high levels of corticosterone) which in turn leads to immunosuppression.

Differing effects of decreases in food supplies

Some birds are more at risk of dying in severe weather than others (reviewed in Goss-Custard et al. 1996a). First, young birds are often particularly at risk, as confirmed by several studies. Secondly, birds older than one year that had not finished the moult of primary flight feathers by the time severe weather arrived were at risk. Thirdly, so were birds with anatomical abnormalities of the bill or foot. How well birds feed earlier in the winter is an important factor. Subdominant individuals (young birds, and those with deformities) occur in higher proportions on the lower-quality habitats. Their mean weights are lower, probably due to poorer feeding conditions and the inexperience of the younger birds. In one instance, 40% of the birds in a roost with a high proportion of young birds and individuals with deformities died during severe weather compared with only 2% in a roost adjacent to the widest shore (high-quality habitat) (Swenon 1984).

There are a variety of reasons why young birds and birds in poor condition are particularly vulnerable in all winters. Evidence suggests that a number of sometimes inter-related factors might affect feeding rates. Apart from age, these include sex, diet, feeding method, dominance rank, foraging skill and, probably, injuries to feet and mandibles (Goss-Custard et al. 1996a and references therein). There are some overall associations that are consistent with this hypothesis. Juveniles have generally low rates of intake and dominance, use fields most at high tide, have the lowest average body weights, have the highest

intensity of parasites, and are also most at risk of dying in severe weather (Goss-Custard et al. 1996a).

On the Exe Estuary, there is a male bias in the oystercatcher population. It only occurs in adults, which implies a sex difference in the choice of wintering area in the first year. It is also likely to occur in older birds, as there are differences in diet, and so in choice of feeding area, between sexes (Durell et al. 1996). There are two explanations: one is that sexes overwinter in different areas; the other is that there is a bias in the population as a whole. In the latter case, if males and females are fledged in equal numbers, such a bias implies differential mortality rates.

There is some evidence from the Exe to suggest that females are subject to higher rates of overwinter mortality. Birds that feed on worms and clams are more likely to supplement feed in fields. This not only means they are probably having difficulty in maintaining intake rates at low water, but are more susceptible to harsh weather because when fields freeze birds are unable to feed. As more females than males feed on worms and clams, they could suffer higher winter mortality rates. Indeed, of 14 corpses collected on the Exe after cold weather, 11 were female and all but one had pointed bills, indicating that they were feeding on worms rather than bivalves (Durell et al. 1996). Alone, emigration and immigration rates in young birds seemed insufficient to explain the change in sex ratios on the Exe. Modelling showed that even without a sex difference in emigration and immigration rates, female mortality would only have to be 5% higher to result in 67% of the adult population being male (Durell et al. 1996).

Baker (1975) reported a male bias in a South Island pied oystercatcher population and considered higher female mortality to be the most likely explanation. However, mortality rates of inland-breeding South Island pied oystercatcher are not significantly different between males and females (P.M. Sagar, pers. comm.). In either case, there are important implications if wintering habitat is affected. Any loss of habitat or increase in overwinter mortality could affect one gender more than the other.

Competition between fisheries and oystercatchers

Oystercatchers and fishers tend to concentrate on the same areas (Taylor 1995; Goss-Custard et al. 1996b). Much depends on the minimum size of cockle that is allowed to be fished and on local conditions. For example, in the Somme Estuary, France, the minimum legal size is 30 mm. Local oystercatchers mostly eat smaller cockles. However, if cockle fishing includes small sizes (16–18 mm) birds cannot escape effects of fishing. Larger minimum sizes are more favourable to oystercatchers. If sizes overlap, there can be a genuine conflict of interest between birds and the fishery (Lambeck et al. 1996).

In the Netherlands, the usual minimum harvest size of cockles is 20–22 mm, and a conflict does exist. Intensive fishing in 1987 in the Oosterschelde made it difficult for oystercatchers to meet their needs, and average weights were 30 g lower than normal. In areas with normal food supplies, weights were normal. However, mortalities were not unusually high, as the winter was mild and wet and birds were able to supplement their intake by increased inland feeding. However, the difficulties the birds were experiencing were clearly demon-

strated during three frosty days in December, when unusually large numbers of birds emigrated; which they would normally do only after much longer spells of really severe weather (Lambeck 1991; Hulscher et al. 1996). In winter of 1990/91, cockle stocks were even lower than in 1987/88, due to a combination of earlier low spatfall and commercial fishing and the local winter population of oystercatchers decreased from 18 000 to 6000 (Lambeck et al. 1989). Many oystercatchers apparently moved into other areas, but despite this redistribution, over 1200 starved oystercatchers (2% of the midwinter population) were found dead during a cold spell that lasted two weeks. It is in periods of severe weather that oystercatchers are most likely to pay the penalty for not having been able to feed at adequate rates (Lambeck et al. 1996).

When habitat/food loss affects oystercatcher numbers

There is an unfortunate tendency for some authors to imply that habitat loss will only affect population size if the carrying capacity of an area has already been reached. This is a dangerous misunderstanding (Goss-Custard & West 1995). When predicting the effect of habitat loss, the key issue is whether density-dependent processes are already operating, or will do so after the habitat is lost (Goss-Custard & West 1995). Reducing the quantity or quality of food in a locality will affect the numbers of birds remaining if competition for food is already sufficiently intense for the rates of emigration or mortality to be density-dependent. Reducing the food supply—thus intensifying competition between birds—will reduce local numbers (Goss-Custard 1977; Goss-Custard et al. 1995a). A simulation using information from decades of studies demonstrated how oystercatchers feeding on mussels would be affected by changes in food supply if the numbers were between 1000 and 8000 settling in the Exe estuary in September (Goss-Custard & West 1995).

Winter habitat loss

Simulations by Goss-Custard et al. (1996c) demonstrated that human activities that change density-dependent rates, even by quite small amounts, could greatly affect population size, especially when the summer density-dependent effect is weak. Habitat loss in winter would cause density to increase, at least temporarily, so that whether or not winter mortality is density-dependent was critically important. Up to a certain point, the strength of winter density-dependent function had a large influence on the effect that habitat loss has on population size. Most simulations showed that, in comparison to the proportion lost, removing habitat of average quality led only to subproportional reductions in population size. Two circumstances led to supraproportional reductions:

1. Where two subpopulations with different reproductive rates used and lost the same winter habitat, the poorer-breeding population decreased at a disproportionately high rate.
2. When the best quality, rather than average quality, habitat was removed.

Another general conclusion of the simulations by Goss-Custard et al. (1996c) was that a given percentage reduction in winter habitat of average quality did not ever result in an even greater percentage reduction in population size. Reductions even approaching proportionality to the area lost only occurred when the winter density-dependent function was steep. This important conclusion was robust across the probable range of values of many of the other

parameters and clearly implies that, at the European scale, the oystercatcher population is unlikely to be greatly reduced in the early phases of the removal of typical winter habitat. This conclusion was, however, very sensitive to the condition that the lost habitat was of average quality. When habitat of above average quality was removed first, the reduction in population size became markedly supraproportional. Habitat loss in certain areas will, therefore, have a much greater effect than it will in others.

The alternative food supply also had an important influence on winter survival, and thus the size of the whole population on the estuary (Goss-Custard et al. 1996d).

Role of summer breeding habitat and potential recruits

Simulations were also run on loss of summer breeding habitat. In general, the results showed that, in a migratory species such as the oystercatcher, the effect of habitat loss in one season depended not only on density-dependence factors in that season, but also, to varying degrees, on the density-dependent factors operating in the other season (Goss-Custard et al. 1996c).

The model was run to explore the effects of increasing summer habitat by a factor of up to two. Without a winter density-dependent effect, habitat gains resulted in a proportional increase in breeding numbers and population size. However, when winter density-dependent losses were introduced the increase in numbers was subproportional. Whether increasing summer habitat makes a difference to population size, therefore, depends a great deal on the strength of the density-dependent effect in the coastal areas in winter (Goss-Custard et al. 1996a).

3.1.3 The bigger picture/metapopulations

Simulations with the oystercatcher population model illustrate how the year-round population size is affected by processes occurring on both the breeding and wintering grounds. In most circumstances, anything that affects the reproductive and mortality rates, whether their action is independent or dependent of density, will affect population size. Similarly, it is not useful to ask whether a population is limited on its breeding or wintering grounds, as often has been done; to do so is rather like asking whether the area of a rectangle depends on its width or breadth. The simulations also showed that the effects of habitat loss at one time of year on population size could depend a great deal on the shape of the density-dependent function (Goss-Custard et al. 1996a).

For reproductive rates to be density-dependent either the proportion of adults that breed or the fledgling production per breeding pair, or both, must decrease as the total population increases. The sizes of both subpopulations (coastal and inland) simulated were very sensitive to the levels of all the major density-dependent rates: the production rate of fledglings, the survival of young birds during their first and second winters, and adult mortality rate (Goss-Custard et al. 1996a).

The effect of habitat loss on local numbers depends in part on the effect that the loss has on the supply of potential recruits, and thus on the size of the metapopulation itself. If only a small fraction of the metapopulation spends the

winter in the affected area, the number of potential recruits to that area may be little affected. However, if the feeding grounds of a large number of oystercatchers were affected, the size of the metapopulation itself would be reduced because of the generally increased winter mortality rate (Goss-Custard et al. 1996d).

Density dependence at Golden and Tasman Bays

South Island pied oystercatcher numbers around Farewell Spit appear to be independent of national counts (Battley 1996), although this needs to be verified with more recent data (D. Melville, pers. com.). The New Zealand population of South Island pied oystercatchers has increased greatly over the past 50 years, and evidence suggests that competition at southern sites is driving the northward migration of oystercatchers (Battley 1996). In an analysis of wader populations and habitat in the UK, the explanatory variable that recurred most frequently in the analyses was longitude (Rehfishch et al. 1995). Birds may tend to winter as close to their breeding grounds as possible to minimise energy spent on migrating. These pieces of evidence, taken together, would tend to support the notion that South Island pied oystercatcher wintering in Golden and Tasman Bays are at numbers where density dependence is occurring.

In contrast to South Island pied oystercatcher, there are positive relationships between counts for godwits, indicating that their numbers on the Farewell Spit are determined to a large degree by numbers arriving in New Zealand (Battley 1996). Nevertheless, as modelling by Goss-Custard et al. (1996d) has demonstrated, density dependence can occur across a wide range of densities among birds.

Other literature reviews and ecosystems

In a detailed literature review, Taylor (1995) concluded that, of the bird species found in the areas of proposed clam/cockle harvesting in Tasmania, pied oystercatchers (*H. longirostris*) would be the species mostly likely to be affected. He based his conclusions on the similarities between the species of oystercatcher and clam/cockle found in Tasmania and those in Europe, where the majority of research has been done on the relationships between cockles and oystercatcher populations.

Unintended effects of harvesting on cockles

Besides removing cockles, harvesting may damage cockles or change their abundance in an area, although this appears to be mostly in the shorter term. Experiments were conducted in Europe on the cockle (*Cerastoderma edule*) to quantify effects of physical impacts on shell damage. The minimum free falling distance in air required to damage cockle shells was approximately 0.3–0.5 m (Coffen-Smout 1998). Additional field experiments were conducted to study effects of simulated harvesting on reburrowing behaviour, displacement by tides, and recolonisation of cleared patches. Simulation of machine-induced physical shocks caused delays to the normal cockle reburrowing response. Small (< 20 mm) cockles were less affected than those of a size to be retained in the catch (> 20 mm), and more of all sizes reburrowed if deposited in pools rather than on wet sand. None reburrowed into drained sand. Tagged and

marked cockles failing to reburrow were transported up the shore with the flood tide, some being found 200 m away. Many of these reburrowed at new positions. The majority of tagged cockles that reburrowed before the first flood tide subsequently remained where they had re-established themselves. Those moved to new positions stayed where they had been carried to (Coffen-Smout & Rees 1999).

However, because of injury, increased susceptibility to scavengers, and displaced sediment, many undersized animals may be unable to resettle. In the Thames Estuary, 25% of undersized, usually juvenile cockles, were killed by dredging (Franklin & Pickett 1978 in Lambeck et al. 1996), while estimates of mortality in the Dutch Wadden Sea varied from 10 to 50% (Anon., unpubl. report 1987 in Lambeck et al. 1996). Afterwards, differences in densities equalised in about six months.

The effects on cockle populations in the Burry Inlet of mechanical harvesting using a tractor dredger were investigated with an experimental trial conducted on 29 October 1992. Appreciable losses of spat and one-year-olds from the dredged plots were observed even though they were not taken in the catch. Counts of damaged individuals remaining on the plots on the day after dredging were generally low for all age groups. Spatfall success in 1993 was depressed by 11% on dredged plots compared to that on control plots in the low density area. However, it was concluded that delayed effects of the dredging on cockle stocks were negligible (Cotter et al. 1997).

3.2 INDIRECT EFFECTS ON NON-TARGET SPECIES

Mechanical cockle harvesting disturbs both the seabed and the organisms within it. This in turn may affect shorebirds through impacts on the non-target species that they feed on. This could occur in a number of ways including direct mortality of the invertebrates, changes in recruitment or settlement of spat, or changes in the sediments or structure of the sediments and the associated communities. Important issues include the recovery rates of these communities after disturbance, the effects of large- versus small-scale disturbances, the intensity of fishing, and susceptibility of different sediment types and communities to disturbance.

Two types of studies have examined effects of mobile fishing gear: experimental studies in which the sea bottom is disturbed and the post-disturbance biota is compared with an undisturbed nearby area; and observational studies in which fished areas are compared with areas that are either off-limits or unfished areas (Watling & Norse 1998). There are shortcomings with both, and more 'research by management' designs that incorporate the best of both (i.e. realistic scale and regime of disturbance plus good controls) may be needed to resolve some of the outstanding questions.

3.2.1 Meta-analysis of fisheries impacts

Studies into the impacts of fishing tend to be highly specific with respect to fishing gear, disturbance regime, habitat and environment. As a result the generality of fishing effects with respect to gear and habitat types is poorly

understood. To address this deficiency, Collie et al. (2000) did a meta-analysis of published fishing impact studies. A total of 57 different manipulations or observations of the effects of fishing disturbance on benthic fauna and communities was extracted from 39 publications. The studies were classified with respect to a range of variable that might affect the degree of impact. The classifications were: gear type, regime, minimum scale, region (broad geographical area of study), depth, and habitat. The gear type 'inter-tidal raking' was used for tractor and cockle dredging. Regime was how often the same area was disturbed. Habitat was classified into sand, mud, muddy sand, biogenic (e.g. seagrass meadows), and gravel. Fifteen studies were similar to mechanical harvesting in Golden and Tasman Bays with respect to the gear used and depth of area fished (Table 3).

The analysis showed that inter-tidal dredging was among the types of fishing that had the greatest initial effect on benthic biota. The fauna in less consolidated coarse sediments was less adversely affected than the fauna in more stable habitats such as mud. Recovery rates were fastest in less physically stable habitats, which are generally inhabited by more opportunistic species. However, areas that are fished in excess of three times per year are likely to be maintained in a permanently altered state.

3.2.2 Sandy substrates and impact assessment

Looking broadly at all types of mechanical fishing including trawling and dredging, often the literature has found limited impacts in sandy sediments (Watling & Norse 1998; Collie et al. 2000). There is probably a variety of reasons for this, including: the difficulty of detecting impacts, flawed methodologies, real differences between various types of substrates and within

TABLE 3. SUMMARY OF FISHING IMPACT STUDIES REVIEWED BY COLLIE ET AL. (2000).

Inter-tidal dredging and raking gear. Region codes are: ENA, Eastern North America; NE, Northern Europe; SA, South Africa.

GEAR	HABITAT	REGION	SCALE (m)	DEPTH (m)	REGIME	RECOVERY PERIOD (days)	REFERENCE
raking	mud	ENA	1	0	2		Collie et al. 2000
raking	sand	NE	20	0	1	400	Cotter et al. 1997
raking	sand	NE	45	0	1	56	Hall & Harding 1997
hydraulic dredging	muddy sand	ENA	150	3	1	300	Ismail 1985
dredging	biogenic	ENA	35	1	2	730	Peterson et al. 1987
dredging	mud	ENA	1	0	4		Brown & Wilson 1997
dredging	mud	NE	2	0	1	210	Kaiser 1996
dredging	muddy sand	NE	1.5	0	1	365	Kaiser et al. 1998
dredging	muddy sand	SA	3	0	1	606	Wynberg & Branch 1994
dredging	sand	ENA	35	1	2	730	Peterson et al. 1987
dredging	sand	NE	45	0	1	56	Hall & Harding 1997
dredging	sand	NE	5	0	1	180	Cryer et al. 1987
dredging	sand	NE	1	0	1	140	McLusky et al. 1983
dredging	sand	NE	7	0	1	140	Heiligenberg 1987
dredging	sand	NE	1	0	1	180	Heiligenberg 1987
dredging	sand	NE	1	0	1	140	Heiligenberg 1987
dredging	sand	NE	50	7	1	40	Hall et al. 1990

substrate types (i.e. muddy sand versus clean sand), previous levels of disturbance, differences in the type of disturbance, and finally the natural disturbance regime that is present already.

Hall & Harding (1997), for example, reported recovery of the faunal community within 56 days in disturbed plots in a study of the effects of dredging for cockles on non-target species in an intertidal area in Scotland. Both tractor dredging and suction dredging were tested. The tractor dredging had less impact than suction dredging, and recovery occurred when the natural seasonal decline was occurring. They concluded that 'recovery is rapid and overall effects on populations is probably low.' There were, however, problems with the Hall & Harding (1997) study. The experimental plots were very small (0.02 ha to 0.5 ha) and, as they themselves point out, 'the effects for small plots are unlikely to persist if sediments and fauna are moving'. Relocating the smallest of Hall & Harding's areas (15 × 15 m) using a global positioning system capable of accuracy to only ± 15 m must have also been difficult.

An additional and more serious flaw with the Hall & Harding (1997) study, as well as many other studies, is that they took place in an area that had long been exposed to fishing and was thus 'ecologically primed' to such disturbances (see Roberts 1997; Hall 1999). There is a lack of good, undisturbed control sites available for these types of studies, and often the mistake may be made of assuming the community of an area is what was originally there, when in fact it is not. Carlton (1998) commented upon the lack of historical knowledge of marine environments and how, without such a framework, our sense of history often defaults to viewing the step on which we are standing as the second step of the staircase, no matter how far down the staircase we have gone. This phenomenon has been called the 'shifting baseline syndrome' (Pauly 1995; Sheppard 1995).

Most studies (42 of 56) were conducted in either North America or northern Europe. Lack of undisturbed control sites has frustrated recent research in both locations. In North America, Engel & Kvitek (1998) lamented the nearly universal absence of true unfished control sites which, '...severely limits our ability to determine appropriate levels of harvest pressure for maintaining sustainable fisheries and marine biodiversity...' Likewise, studies conducted in the North Sea have also been hampered by the lack of comparable control areas protected from fishing over the last century. For example, Kaiser (1998) noted that the communities they observed might have been the products of continuous fishing disturbance, making it difficult to infer the ecological implications of the results because the environment was potentially predisturbed. The implications of this widespread lack of undisturbed sites are that many studies have probably underestimated the impacts and changes that fishing has had on the benthic communities.

It is also probable that coarse, wave-disturbed sandy sites are more adapted to disturbance than some other sites. Watling & Norse (1998) felt that it was hardly surprising many studies had detected few impacts in soft sediments given that the sites were primarily sand, and tended to have either strong currents or were swept by storm waves, resulting in infaunal communities dominated by species adapted to frequent physical disturbance. Even so, each community studied showed the loss of some species, usually the larger-bodied

species living buried in the sand. Collie et al. (2000) also suggested that sandy, disturbed substrates may be more adapted to disturbance than other substrate types. They also make clear that the Hall & Harding's (1997) conclusion that 'under most circumstances, populations of non-target fauna are likely to be robust to the disturbance' caused by cockle dredging needs qualification.

Finally, soft substrates may be harder to study. The effects of trawling, digging, and dredging on substrate characteristics and animal communities may be harder to compare with, for example, rock substrates, and the effects may be more 'thinly spread' over larger spatial scales (Hall et al. 1990; Hall & Harding 1997; Piersma & Koolhaas, unpubl. report 1997).

3.2.3 Impacts of intertidal mechanical cockle harvesting

A more recent study in the Burry Inlet (Ferns et al. 2000) was conducted using a tractor-towed cockle harvester, similar to that used in New Zealand. Two intertidal sandy habitats were sampled: muddy sand with high cockle densities (1850/m²) and clean sand with low densities (120/m²). One important difference between the two harvesting systems is that the tractors used in Wales used a horizontal-mounted rotating drum with holes in the screens of the drum to sort the sediment from the cockles, rather than the conveyor belt type of system used in Golden Bay. The Burry Inlet is an important wintering area for oystercatchers and knots and has experienced low levels of manual cockle harvesting since mediaeval times. More recently, mechanical harvesting methods have been introduced, and a proposal to use tractor-towed machines within the Burry Inlet provided the impetus for a study to determine the potential effect of such harvesting upon non-target invertebrates and their dependent bird populations.

Harvesting resulted in a loss of a significant proportion of the most common invertebrates from both areas. Populations of some invertebrates (*Pygospio elegans* and *Hydrobia ulvae* (Gastropoda) remained significantly depleted in the area of muddy sand more than 100 days after harvesting. Areas in clean sand with relatively few cockles recovered more quickly than those in muddy sand.

The poorly sorted fine muddy sand had more tube-dwelling and sedentary species than the clean sand which had more mobile species. A dark, anoxic mud layer was brought to the surface by the action of the harvester. (In Golden Bay, this mixing of dark, anoxic mud layer also occurred when a storm disturbed the experimental plot reported in Wilson et al. (unpubl. report 1988), but was not observed to occur during harvesting of cockles in Golden Bay during a site visit by the author, 10 Jan 2001). In the Burry Inlet plots, harvesting damaged a great many annelids and thin-shelled molluscs, including some of the smaller cockles. These were deposited on the surface of the sediment and afforded an immediate source of food for predators such as gulls and waders. Reasonably intact invertebrates buried themselves in the sediment within a few minutes, leaving only moribund ones on the surface. There was thus a relatively short-lived food supply available. By 86 days after harvesting, species richness was significantly lower in both areas than at the outset. Annelids declined by 74% on muddy sand and 32% on clean sand. Mollusc populations declined by 55% and 45%, and crustaceans declined by 56% and 81%, respectively. Invertebrate densities remained depleted in muddy sand for longer than in clean sand, with some

species (*Hydrobia ulvae* and *Pygospio elegans*) having failed to recover 174 days after harvesting.

The ability of invertebrates to recolonise depleted areas is variable. The movement of adults of most species, or their passive transport, was apparently sufficient to allow recovery of the modest invertebrate population in clean sand, but was inadequate to allow recovery of the richer community in muddy sand. The recovery rate of invertebrates in clean sand was similar to that reported by Hall & Hardy (1997), even though an important difference between the two studies is that this study was carried out during the winter months and Hall & Hardy's was carried out during the summer.

Small invertebrates were subjected to a more powerful grinding action from the larger number of cockles in the harvester drum in clean sand. For example, the damage to cockles small enough to escape harvesting was much greater in clean sand than muddy sand. The slow recovery of the adult populations in clean sand may also have been a consequence of the physical disruption caused by the harvester to the complex layered structure of the sediment and the communities it supported. Evidence from the study also supported the notion that adverse physical and chemical conditioning of the sediment was a factor inhibiting settlement.

This study demonstrated that tractor dredging for cockles can cause depletion of populations of non-target invertebrates for several months, and so reduce bird feeding activity, especially in more productive areas of intertidal sediment. Bird feeding activity increased in the first few days following harvesting, with gulls and waders taking advantage of invertebrates made available by harvesting. However, harvesting also resulted in a subsequent reduction of feeding opportunities that extended over a long period. Significant reductions in bird activity on the harvested sector became apparent 21 and 45 days after harvesting. For oystercatchers, feeding activity in the harvested plot in muddy sand remained significantly reduced for more than 50 days after harvest. Its effects would also impinge on birds in unharvested areas as a consequence of movements of individuals away from harvested zones. It was concluded that tractor dredging for cockles in high-density areas causes a sufficiently large mortality of non-target invertebrates that harvesters should be excluded from areas of conservation importance for intertidal communities such as invertebrates, fish and birds.

3.2.4 Sediments, habitat and waders

Sediment characteristics are very important to wader distribution. Inter-related variables such as sediment particle size, degree of organic content, and salinity influence invertebrate and, therefore, shorebird distributions (Yates et al. 1993; Rehfish et al. 1995). Benthic organisms can substantially alter the sediments in many ways, including adhesion, grain-size distribution, water content and formation of structures. Interactions between animals and sediments work in both directions. In unconsolidated sediments, physical disturbances from natural (storms) and unnatural causes (dredging) can greatly affect the benthic fauna. Physical disturbances that take place on a large scale, even if they occur only once, thus have cascading effects throughout the benthic community (reviewed in Piersma et al. 2001).

In a study of the Wadden Sea at realistic scales (1500 ha of a total of 5000 ha of intertidal flats) it appears that losses of bivalves within the system changed the sediments, causing them to become coarser and less productive (Piersma & Koolhaas, unpubl. report 1997; Piersma et al. 2001). In the Griend area of the Wadden Sea stocks of *Cerastoderma*, *Macoma* and *Mytilus* had not recovered from suction-dredging during the period 1989–98 to the levels of 1988. Long-term losses of bivalves were most pronounced in the area dredged for cockles. Declines of bivalve stocks were caused by low rates of settlement in fished areas until 1996, eight years after dredging (Piersma et al. 2001). Numbers of red knot decreased in the affected areas. The benthic assemblages seem to have shifted from biomass-rich bivalve-dominated communities to biomass-poor communities where short-lived but powerfully colonising species such as small polychaete worms dominate (Piersma et al. 2001).

The hypothesis is that mechanical removal of the large filter-feeding bivalves in the Wadden Sea initiated sedimentary changes that led to the disappearance of other filter-feeders such as *Macoma*. At this point these filter feeders can no longer produce the faecal pellets that play such an important role in the build-up of fine-grained sediments, which in turn attract settling bivalve larvae. The strength of such a negative feedback loop can be increased by storms that churn up the upper layer of sediments (Piersma et al. 2001).

Piersma et al. (2001) proposed that cockle dredging in the Wadden Sea, perhaps in combination with the destruction of nearly all the intertidal mussel beds, temporarily transformed relatively unstressed midshore communities into stressed benthic communities typical of mobile sands low in silt and organic matter. That cockle dredging leads to significant long-term reduction in settlement and stocks of target species contradicts the conclusion of many short-term environmental impact studies and reviews commissioned by the shellfishing industry (see Piersma et al. 2001).

Similar dynamics have also been observed in New Zealand. After extensive experimentation on a New Zealand sandflat, Thrush et al. (1996) concluded that large scale disturbances that destroy organisms with a role in maintaining habitat stability, such as mussel and cockle beds, are likely to result in very slow recovery dynamics, particularly in wave-disturbed soft-sediment habitats.

In research on the effects of shellfisheries on bird populations it is usually taken for granted that commercial fisheries and birds only compete for a food resource, and apart from the disappearance of this resource, system properties remain the same (review by Lambeck et al. 1996). This may be justified in the few cases where the fishery practice is of low intensity and extent. But studies like those by Piersma et al. (2001) indicate that mechanical shellfishing impacting extensive intertidal areas may not be sustainable. One-off mechanical harvests not only depleted stocks, they compromised the environmental requirements of normally regenerating shellfish populations.

Like clearcutting of forests, use of mobile fishing gear does not eliminate biological activity. Rather, it converts ecosystems dominated by disturbance-intolerant equilibrial species to ones dominated by disturbance-tolerant opportunistic species (Watling & Norse 1998). Such anthropogenic change—which foresters call ‘type conversion’—has also occurred in an intensively trawled off-shore area in the Irish Sea, of which Lindeboom & de Groot (1998)

say 'the present species-poor and low biomass fauna may represent an artificial man-made community adapted to the regular fishing disturbance experienced at this site.' They concluded that, 'if trawling intensity remains high, these communities may never recover.'

Summarising extensive and detailed information from the Wadden Sea, Reise (1982) concluded that bivalves and some other groups were showing long-term decreases, while the smaller polychaete species with short lifespans were doing well. In spite of their fragility, which makes them susceptible to mechanical disturbances of soft sediments including bait-digging, small polychaetes are particularly good colonisers of defaunated areas of intertidal sediment. Small polychaetes are characteristic of stressed and unpredictable habitats and can take rapid advantage of environmental disturbances leading to faunal depletions such as oxygen depletion in hot summer periods, severe cold in harsh winter periods and heavy local pollution (see Piersma et al. 2001).

3.2.5 Scale of disturbances

An important ecological issue is to understand how patterns and processes vary with scale. Thrush et al. (1996) tested this by removing the invertebrates from three different plot sizes (0.203 m², 0.81 m², and 3.24 m²). Samples were then collected to assess recovery over a 9-month period. The sandflat used for the experiment was prone to disturbance by wind-driven waves. Most common species revealed significant relationships between density and disturbance plot size. Scale-dependent recovery was also demonstrated by differences in species assemblage structure over the course of the experiment. Relative rates of colonisation varied by approximately 50% between large and small experimental plots. The results revealed an unusually slow rate of faunal recovery.

In the study by Thrush et al. (1996), both increasing numbers of colonists and density changes in ambient sediments made an important contribution to recovery. Sediment instability was higher in all experimental plots than in the ambient sediments, due to the initial removal of a dense spionid polychaete tube mat characteristically found at the study site. Sediment instability also increased with increasing plot size. While the results demonstrate the need for caution in scaling-up from small-scale studies, they do indicate that larger-scale disturbances that destroy organisms with a role in maintaining habitat stability are likely to result in very slow recovery dynamics, particularly in wave-disturbed soft-sediment habitats.

Several mechanisms can be invoked to explain slow recolonisation of even small patches. First, colonisation is affected by patch type: type I patches—those surrounded by undisturbed communities—are colonised both from the perimeter and by dispersed propagules (Connell & Keough 1985; Sousa 1985), whereas type II patches—undisturbed spots surrounded by vast disturbed areas—are the source of colonisers, especially over short distances. Key components of benthic ecosystems, including amphipods, isopods, and other small crustaceans, do not have planktonic larvae but have direct development and characteristically short-distance dispersal across the seabed. In addition, in temperate waters at least, production of propagules is seasonal, so disturbed patches may sit for some time before recolonisation can occur. Secondly,

disturbance alters the seabed physically and chemically. In Maine, sediment food quality decreased after scallop dredging muddy sand sediments (unpubl. data, cited in Watling & Norse 1998). Several groups of invertebrates did not recolonise disturbed patches until food quality had recovered. Thirdly, there are likely to be non-linear changes in recolonisation, depending on the aggregation of individual disturbances and resulting fragmentation of the landscape (Hall 1994).

3.2.6 Bait digging

Bait digging is a common activity in many areas of the world and, although not currently common in New Zealand, the impacts are similar to cockle harvesting. Bait digging can also offer insights into the potential effects of cockle harvesting if such harvesting removes non-target species, such as marine worms. In the Bay of Fundy, Canada, baitworm harvesting had a significant negative effect on semipalmated sandpiper (*Calidris pusilla*) foraging behaviour and on the density and age of structure of their principal prey, the amphipod *Corophium volutator*. These impacts occurred after only one season of digging. Foraging efficiency decreased by 68.5% in dug sediment, which corresponded to observed reductions in prey density. The decreased foraging efficiency may also have been related to reductions in prey availability due to the obstruction of visual and tactile cues caused by the turning and loosening of the surface sediment (Shepherd & Boates 1999).

Harvesting of lugworms (*Arenicola marina*) from the Wadden Sea over 4 years (1978-82) resulted in almost doubling of the annual mortality rate and led to a gradual and substantial decline of the stock. Simultaneously, total biomass declined further due to the almost complete extinction of large gaper clams (*Mya arenaria*) that initially constituted half the total biomass. Of the other, mostly short-lived, species only *Heteromastus filiformis* showed a clear reduction through the dredging period. Recovery of the biomass took several years (Beukema 1995).

Bait digging can also have negative impacts on cockles. In popular bait digging areas, cockles left on the surface or buried under more than 10 cm of sediment often die, which can result in a considerable reduction in the cockle population (Lambeck et al. 1996). In the Dutch Delta area recent regulations limit bait digging to specially assigned tidal flats. A similar management policy was pursued in Lindisfarne, England (in Lambeck et al. 1996).

4. Applicability of European studies to New Zealand

The general conclusions of the European studies probably hold true in New Zealand. The European oystercatcher and South Island pied oystercatcher are very closely related, and both are migratory. The edible cockle (*Cerastoderma edule*) in Europe is, like the New Zealand cockle, a suspension-feeding bivalve

that lives just below the surface. The study at Farewell Spit by Battley (1996) indicates that foraging specialisation by South Island pied oystercatchers, similar to that seen in Europe, also occurs. The winters on the Wadden Sea are probably harsher than those at Golden and Tasman Bays, so some of the details of overwinter survival and responses of birds may differ. The Wash has milder winters than the Wadden Sea, and may therefore be more similar to Golden and Tasman Bays; however, the winters at the Wash are probably still colder than those at Golden and Tasman Bays (D. Melville, pers. comm.).

The intensity and scale of harvesting, plus the nature of the sediments being disturbed, is probably a very important variable which needs to be considered carefully, but more information is needed to determine differences and similarities between Golden and Tasman Bays and European areas.

The study in the Burry Inlet (Ferns et al. 2000) was probably the most analogous in most ways to the situation in Golden and Tasman Bays. There was, however, a slightly different system used to separate out sediments from cockles from that used by harvesters in Golden Bay. The local conveyer-belt type method may cause less damage to non-target species and small cockles because there would presumably be less grinding than in the rolling drum system used in Wales. Nevertheless, the high number of birds scavenging around harvesters suggests that there is a flush of prey available, presumably dead and exposed benthic invertebrates, even if they are not easily detected by casual observation.

Of the studies reviewed by Collie et al. (2000), two were conducted in New Zealand. One was otter trawling in biogenic habitat at 10 m depth (Bradstock & Gordon 1983) and the other was scallop dredging in sand at 24 m depth (Thrush et al. 1995). There were indications that impacts may vary by geographical regions, but this was not conclusive due to the small number of studies done outside North America and northern Europe. In addition, there are the confounding factors of varying habitat types, gear, regimes, scale and depth of study, plus the added problems of methodologies (e.g. previously fished areas, lack of adequate controls). The experiment by Thrush et al. (1996), which demonstrated strong evidence for size-dependent recolonisation, was conducted in New Zealand.

5. Further information needed

Little local information is available for Golden and Tasman Bays. In the areas that have been subject to fishing disturbance, no similar unfished control areas within and adjacent to the fisheries have been maintained. Below is a list of information gaps along with rationales for why the information is felt to be important. Following that are sections on why control areas are essential, some approaches to future research that could be employed based on experience from Europe, and specific methods that might be useful for further understanding the potential impacts of cockle harvesting on the intertidal ecosystems of the bays.

5.1 SPECIFIC INFORMATION GAPS

Sediment characteristics and potential impacts of harvesting on them

Changes in sediment characteristics due to fishing have been demonstrated in Europe, along with a probable conversion of the benthic invertebrate fauna from more stable productive communities to more disturbance tolerant species. These conversions have been brought about by the loss of structure and adhesion of sediments provided by cockles and tube-dwelling worms. There is an urgent need to better understand the sediment characteristics of Golden and Tasman Bays—specifically, to identify those areas that are most prone to disturbance, where fishing disturbance may be unsustainable and could lead to long-term or irreversible changes in the sediments and communities.

Yates et al. (1993) concluded that, because of the strong influences of sediment on the densities of most prey species and therefore wader numbers, the sediment particle distribution could be used to predict shorebird densities directly. It was not necessary to first predict the densities of the benthic invertebrates on which they feed. A survey of sediments across the Bays could allow identification of the main wader feeding areas, and, in future, remote sensing data could be used to track changes in sediment characteristics.

Long-term impacts on sediment benthic communities

Contrary to the findings of the two brief New Zealand studies on potential impacts of harvesting, significant impacts on key non-target species are likely to be occurring, but should be verified. An assessment of the long-term impacts on the sediment benthic communities should be undertaken. Quantitative surveys of the distribution and density of the invertebrate fauna should be carried out in areas before and after harvesting; adjacent control transects should also be monitored.

Eel grass bed dynamics

Eel grass beds are rich, dynamic habitats which support several endemic species of invertebrates as well as high densities of cockles. The latter may help ensure there is enough spat to aid recovery of harvested areas. However, little is known about how much these areas change in size or distribution or the potential affects of harvesting around the edges of them. The effects of black swan grazing on *Zostera* might also warrant investigation, as it might lead to changes in distribution and hence impact on cockle numbers and distribution.

Prey and diet of shorebirds

Feeding observations on the prey and diet of shorebirds at Golden and Tasman Bays needs to be collected, as these data are lacking (see Battley 1996). Widespread benthic surveys and mapping would indicate how representative Battley's sites were. Long-term sustainability of food resources is dependent on reproductive stocks and harvesting rates, and feedback links between the two. Given likely variation in factors, sustained sampling efforts may be required to gain robust interpretable data (Battley 1996). For sites that are under the

Ramsar Convention on Wetlands (Ramsar Bureau 2001), a site monitoring programme needs to be established with long-term sampling protocols to allow for assessment of potential 'changes in the ecological character of the site'.

Local shorebird ecology

There is much less information available about bar-tailed godwit, variable oystercatcher, banded dotterel, and red knot than South Island pied oystercatcher. Data on recruitment and mortality factors, local foraging ecology, and movements of all these species are essential if impacts are to be predicted or detected.

Rates of recovery of non-target fauna after harvesting

The rates of recovery of non-target fauna after disturbance are unknown. These will probably vary depending on the scale of harvest (size of areas being disturbed), time of year, harvest regime (how frequently the area is disturbed), the nature of the sediment types, and their exposure to wind and waves. The obvious, but trivial question, for research/impact assessment concerns the disturbance at the time of harvest. The more important question concerns long-term changes to benthos and resultant impacts on bird use/populations. A related, but separate, issue concerns the potential changes in use by birds of specific areas (i.e. there may be changes in distributions without losses to the total population).

Disturbance effects

Little is known about the potential effects, or scale, of disturbance from the presence of people and machinery on the shorebirds of Golden and Tasman Bays. Hill et al. (1997) describe levels of disturbance experienced by birds in the UK, review the studies between 1970 and 1997, and suggest improvements for the way in which results are applied. They also suggest approaches for assessing the impacts of disturbance on birds.

Relationship between shellfish, birds and fishers

There is a need to understand the relationships between shellfish, oystercatcher and fishers, and so determine whether competition is direct, whether exploitation patterns differ, and the consequences (see Atkinson et al. 2000). Specific New Zealand information, combined with information from elsewhere, may give a good indication of these relationships.

Critical seasons for shorebirds in the bays

In the Wadden Sea, mortality for oystercatchers is highest in winter. It is unknown if this true of South Island pied oystercatcher in Golden and Tasman Bays, where winters are milder. More specific weather information for European estuaries and Golden and Tasman Bays would be useful for determining if the responses of oystercatchers in Golden and Tasman Bays are likely to be similar to those on the Wadden Sea or the Wash. It would also be useful to know more about the energy needs of other shorebirds at various times of the year and the availability of prey during the most critical times of year for these other species, for example, the burrowing depths of benthic

invertebrates in summer and winter, and earthworm behaviour in pastures in relation to soil temperature and water content.

5.2 IMPORTANCE OF CONTROLS/UNFISHED AREAS

It is essential to preserve unfished control areas, representative of the fished area (these areas may also act as refugia, and so assist in recovery/sustainability), if the potential impacts of harvesting are to be assessed. The lack of unfished areas has hampered research worldwide. For example, Engel & Kvitek (1998) commented on the constraints currently imposed on fisheries research by the almost universal absence of true unfished control sites which ‘... severely limit our ability to determine appropriate levels of harvest pressure for maintaining sustainable fisheries and marine biodiversity.’ They felt that valid research in these areas would require marine reserves in which fishing effort and methods can be manipulated in collaborative studies involving fishers, researchers, and resource agencies. Studies conducted in the North Sea have similarly been hampered by the lack of comparable control areas that have been protected from fishing over the last century. The communities observed at present may be the products of continuous fishing disturbance (Groot & Lindeboom 1994; Dayton et al. 1995), so it is difficult to infer the ecological implications of the results of short-term experiments conducted in an environment that is potentially pre-disturbed (Kaiser 1998).

In fact, there is a near-universal absence of areas that can be used for true no-harvest controls, or sites where fishing effort can be systematically manipulated. One possible approach is that recommended in a paper by Engel & Kvitek (1998): addressing questions about fishing impacts through large-scale, long-term, manipulative studies in marine reserves, working with all stakeholders—resource agencies, fishery representatives, fishers, elected officials, and research institutions—to determine the size, location, and duration of an experiment and the conditions under which it would be terminated. Questions pertaining to biodiversity, habitat heterogeneity, recolonisation, and recovery rate following different levels of trawling intensity and frequency could then be easily and definitively addressed. This approach has already been applied successfully in the Gulf of Castellammare, north-west Sicily.

5.3 RECOMMENDATIONS FOR A GENERAL APPROACH TO FUTURE RESEARCH

For research results to be widely accepted the approach used by Fern et al. (2000) and Ens et al. (2000) is recommended. This involves including representatives of various interested parties in the design of the research. This is because if, for example, only the fishing industry was involved in the design of experiments, conservation groups might not accept the results as valid, and *visa versa*.

Both experimental and observational studies have limitations. Experimental manipulations of fishing disturbance at large scales are time consuming and expensive, and few studies have attempted to incorporate factors such as habitat type, scale of disturbance or disturbance regime into their design. Thus, the results of any single study tend to be highly specific. Another problem with experimental studies is their artificially small scale, which may mask the impacts of disturbance. Also, small plots may experience different rates of recovery from areas disturbed at a much larger scale. Most fisheries occur at a decidedly larger scale than experimental plot sizes. On the other hand, observational studies, especially with fisheries, have the fault of often lacking undisturbed control sites. There is a need for large-scale, realistic experiments. Collie et al. (2000) advocate the use of 'press and relaxation experiments' or switching on and off experiments. There are three principal advantages to this approach. First, results are clearly interpretable in terms of 'real-world' fishing disturbance. Secondly, the spatial scale of protected areas can probably be relatively small (and hence replicated to fulfill the requirements for sound experimental design). Thirdly, experiments are conducted in the very habitats about which questions of recovery are actually being posed.

It may also be helpful to assess the views surrounding research into marine issues, possibly in the form of a more detailed literature review or at a broader national scale. In New England, it was found that, unlike terrestrial wildlife, most commonly known marine species are usually considered food. Most people rarely venture under the sea, and they tend to see marine species only in the seafood case at the supermarket (Brailovskaya 1998). Furthermore, a public opinion poll found that 61% of respondents did not view scientific knowledge as relevant to their concerns about the ocean, or even as a valid criterion for government decision making about the marine environment (SeaWeb, unpubl. report 1996). Adding to the public relations problems that many wild marine species have is the image of the fisherman as one of contemporary society's last rugged individualists. In New England this image repeatedly has resulted in the industry's opinion on the condition of groundfish populations being viewed as more credible than scientific data and has created a fisherman-versus-scientist stereotype in both regional and national media (Brailovskaya 1998). Understanding and taking such views into account could assist with policy development and implementation.

5.4 SPECIFIC METHODS

Some specific methods recommended in Europe could also be adopted in New Zealand.

In the Wash, regular monitoring of oystercatchers by catching birds actively moulting and in late winter has been recommended. The number of birds showing abnormal moult can then be used as an indicator of over-winter survival. It has also been recommended that ways to place this information into the fisheries management process should be investigated (Atkinson et al. 2000).

Ornithological Society of New Zealand wader counts are conducted at high-tide roost sites. There is very little information available regarding the distribution of

waders at low tide. WeBS (Wetland Bird Survey) could be adopted around Golden and Tasman Bays to better understand areas used by birds for foraging at low tides and to predict the impacts of harvesting or other activities. WeBS uses counts made at spring-tide roosts where birds can be counted quickly and accurately (Evans 1995). Low-tide counts have several uses: to identify the important low-tide feeding areas; to identify changes in distribution which may be related to food supply; and to obtain a clearer picture of the use made of a system by shorebirds throughout the tidal cycle. Similarly, extensive counts before and after a land claim in the Wadden Sea allowed the authors to check on the predictions made to assess impacts on coastal bird populations of habitat loss. The predictions were remarkably accurate. Knowledge of the species' habitat requirements and the experience gained with earlier land claims had been sufficient to produce a reliable prediction. One reason for the precision was the simplicity of the data available. They were qualitative or semi-quantitative and they were restricted to the affected area itself (Hötker 1995). However, counts will at best show local effects. For assessing the impact of habitat losses on large scales, data on the population biology of the species has to be considered (Goss-Custard et al. 1995b).

Modelling has been used to explore the cockle fishery on the Burry Inlet, UK, and suggest how the current management regime might be affecting overwintering oystercatchers (Stillman et al. 2001). In the Stillman et al. (2001) simulation, model birds responded to shellfishing in the same way as real birds. They increased the time spent feeding at low tide, and higher proportions fed in fields and upshore areas at other times. When shellfishing removed the larger prey, birds ate more of the smaller prey. The results suggest that, at the levels of fishing in the Burry Inlet (hand raking only, minimum take size of 22 mm, and high levels of shellfish left relative to oystercatcher numbers), shellfishing was not causing increased oystercatcher mortality. The modelling demonstrated, however, that if management practices were changed, such as increasing fishing effort, reducing the minimum size of the shellfish collected or increasing the daily quota, oystercatcher mortality and population size could be greatly affected. Additionally, the impact of fishing might often be small within a single year, but subsequent fishing might have a greater effect if the birds or shellfish, or both, do not recover by the following year.

These models also suggest that effective monitoring in Golden and Tasman Bays could include determining the size of shellfish being taken by oystercatchers, the percentage of the local population feeding in paddocks, and percentage of time spent feeding. The Stillman et al. (2001) models predict that smaller shellfish will be taken, more time will be spent feeding, and a higher proportion of the population will feed in paddocks as shellfish supplies are reduced. If birds are being banded, losses in mean mass and overwinter mortality might also be possible to determine.

6. Conclusions and recommendations

‘The task of preserving ecosystem integrity is challenging. Even when not influenced by human activities, ecosystems show a high degree of variability... This...often makes it extremely difficult to separate the relative effects of natural and anthropogenic perturbations. However, absence of (scientific) evidence should not be interpreted that environmental impacts are absent.... There are costs in assuming an effect of human activity on ecosystem integrity when there actually is none, but the consequence of assuming no effect when there really is one is often far greater.... Waiting for a scientific consensus may delay any decision about conserving the environment, with possibly irreversible consequences.’ (De Leo & Levin 1997)

Decreases in cockles are likely to be having a direct impact on the local populations of South Island pied oystercatcher, and possibly variable oystercatcher and godwit, through removal of cockles as a food supply. However, the impacts on the larger populations are unknown. The South Island pied oystercatcher population nationally has been increasing in recent years and is not considered threatened. However, it should be noted that, although the Farewell Spit population has increased, there has been a reduction elsewhere in Golden and Tasman Bays of the past two decades (R. Schuckard, pers. comm. 2001). Loss of food may result in an advantage to inland breeders over river breeders if inland breeders are more productive. Small changes in adult survival can lead to declines in stable or increasing populations. The cumulative impacts of cockle harvesting, which is growing in scale and intensity, has never been assessed in New Zealand. Modelling would be the best way to explore these potential impacts and also help to identify future research needs and priorities. Under some circumstances there may be mutual benefits for oystercatchers and the industry due to thinning of very dense cockle beds.

Too little is known about variable oystercatcher and godwit ecology to determine if loss of cockles is having an impact on this species and, if so, what the magnitude might be. The numbers of variable oystercatcher are much lower than those of South Island pied oystercatcher and are more difficult to determine (Sagar et al. 1999). However, there is some evidence that in some areas their numbers are increasing (P. Sagar, pers. comm.). Considering the extent of coastal modification and disturbance around New Zealand, however, it is likely that their numbers have decreased in some areas and their future is not secure in more disturbed or populated areas.

Bar-tailed godwit forage on polychaete worms, which are likely to be affected by harvesting. Their numbers at Farewell Spit appear to be determined by the numbers reaching New Zealand (Battley 1996). It is very likely that harvesting would affect their food supplies and, therefore, possibly local numbers (i.e. distributions). More studies are needed to determine what the probable scale of this impact might be on the larger populations.

The main impacts to banded dotterel would probably be from disturbance and perhaps crushing of benthos and sediments by vehicles in the intertidal zones leading to changes in the invertebrate communities, but no work has been done on this area. In spite of pressure and losses from predation and other forms of disturbance, banded dotterel appear to be holding their own in New Zealand (Heather & Robertson 2000).

Overseas research indicates that indirect impacts through changes in the benthos community, and possibly the sediment, are very likely to be occurring in Golden and Tasman Bays although local past studies have not detected any such impacts. These local studies may not have detected significant impacts from harvesting due to the limited time and scale on which they were conducted, methodological weaknesses, and the nature of sandy communities, where changes can be difficult to detect. Other studies, in both New Zealand and Europe, have demonstrated impacts in situations similar to those in the Bays.

Several factors are probably important in determining the magnitude of negative impacts; these include the scale and intensity of harvesting, the size class of cockles taken, and the characteristics of the sediment and benthos. Scale is very important to recovery and recolonisation processes (Thrush et al. 1996). They found surprisingly slow recovery, particularly in the larger patches. Because the sandflat in the experiment was prone to disturbance by wind-driven waves, sediments were unstable after the experiment removed a dense mat of polychaete tubes, hampering recolonisation. This suggests that larger disturbances that destroy organisms that maintain habitat stability are likely to recover very slowly, particularly in wave-disturbed soft bottoms. Tractor dredging for cockles, similar to what occurs in Golden and Tasman Bays can cause significant depletion of non-target invertebrates for several months and consequently can reduce bird feeding activity, especially in more productive areas of intertidal sediment (Ferns et al. 2000).

Higher densities of cockles may be associated with more stable, finer, and more sorted sediments (Thrush et al. 1996). Burrow-forming polychaete worms found in these communities may also help stabilise sediments. These areas are probably more productive, but also more prone to disturbance. Disturbance by harvesting may, through a series of interactions between the environment and benthic communities, lead to the sediments becoming coarser, slow recovery rates (Collie et al. 2000), and even drastically altered benthic communities (Roberts 1997; Hall 1999). Wave-disturbed, cleaner sands with lower cockle densities are probably better adapted to disturbances, and so are less likely to be negatively affected by harvesting activities (Watling & Norse 1998; Collie et al. 2000; Ferns et al. 2000).

To the author's knowledge there has been no attempt to determine the cumulative impacts of cockle harvesting on shorebirds in Golden and Tasman Bays. There is a lot of useful, but unpublished and generally unavailable data on these bays. It would be useful for assessing possible cumulative impacts if a summary of the information on the benthic communities and ecosystems of these bays, plus the extent and intensity of harvesting, was prepared using these unpublished reports and EIAs.

Eel grass beds have especially rich benthic communities and may appear and disappear from areas. Very little is known about the impacts that harvesting at the edges of these beds has on their establishment or loss. Such beds are potentially very important sources of cockle spat as well as rich communities in their own right.

There is a lack of undisturbed control areas. However, to detect changes in the benthos and sediment, it is imperative that undisturbed areas are interspersed with harvested areas. These areas must be the same sort of sediments as the harvested areas.

There is a need to know if there is a high overlap between areas used by both harvester and birds. Sediment characteristics of Golden and Tasman Bays and the effects of fishing on these sediments needs to be better understood; including how consolidated the sediments are, and the stability provided by cockles and tube worm communities. There is much less information available on bar-tailed godwit, variable oystercatcher, red knot, and banded dotterel than on South Island pied oystercatcher. Additional data on recruitment and mortality factors, local foraging ecology, low tide distribution, and movements of these species is essential if impacts are to be predicted or detected.

Finally, future research efforts, especially those involving experiments or harvesting, should include consultation with representatives from the various interest groups in the design to help ensure that the results are accepted as valid by all concerned.

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Appendix 1.

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