Catastrophic Decline in Mollusc Diversity in Eastern Tasmania and Its Concurrence with Shellfish Fisheries

G.J. EDGAR* AND C.R. SAMSON

School of Zoology, Tasmanian Aquaculture and Fisheries Institute, University of Tasmania, Private Bag 5, GPO Box 252,
Hobart, Tasmania 7001, Australia

Abstract: *We used historical patterns of deposition of mollusc shells to infer changes to inshore benthic assemblages in the southeastern Tasmanian region over the past 120 years. We identified and counted shells in slices embedded within 1m long ^{210}Pb-dated sediment cores were collected at 13 sites in water depths of 8–16 m. Declines in mollusc species richness and shell production occurred during the past century at all sites studied, with a mean decline per 5-cm sediment slice from 21 species in 1890 to 7 species in 1990 and in shell abundance from 150 to 30 individuals over the same period. The time course of decline notably corresponded with the history of the scallop dredge fishery, presumably either because scallop dredging caused general declines in populations of mollusc species or because other factors caused a catastrophic regional decline in molluscs that included scallops. As a consequence, the fishery was forced to close. Of major concern is that losses had not previously been recognized but extended throughout the 100-km coastal span of the study. Given that fishing and other anthropogenic impacts, as well as a lack of observational data, are virtually ubiquitous for the coastal zone, major recent losses in mollusc biodiversity may be globally widespread but have gone unnoticed.*

Key Words: benthos, biodiversity loss, death assemblage, ^{210}Pb dating, historical change, macrofauna, sediment cores

Declinación Catastrófica de la Diversidad de Moluscos en Tasmania Oriental y su Concurrencia con Pesquerías de Mariscos

Resumen: *Utilizamos patrones históricos de deposición de conchas de moluscos para inferir cambios en los ensambles bénticos cercanos a la costa a lo largo de los últimos 120 años en la región sureste de Tasmania. Identificamos y contamos conchas en capas embebidas en núcleos de sedimento de 1 m de largo y fechadas con ^{210}Pb colectados en 13 sitios con profundidades de 8-16m. Durante el siglo pasado ocurrieron declinaciones en la riqueza de especies y producción de conchas de moluscos en todos los sitios muestreados, con una declinación promedio de/5-cm por capa de sedimento de 21 especies en 1890 a 7 en 1990 y en abundancia de conchas de 150 a 30 individuos en el mismo período. El curso cronológico de la declinación correspondió notablemente con la historia de la pesquería de veneras por arrastre, probablemente porque el arrastre de veneras causó declinaciones generales de las poblaciones de especies de moluscos o porque otros factores causaron una declinación regional catastrófica de moluscos incluyendo a las veneras y en consecuencia la pesquería fue forzada a cerrar. De mayor preocupación es que las pérdidas no habían sido reconocidas previamente pero se extendieron en los 100 km de costa que abarcó este estudio. Dado que la pesca y otros impactos antropogénicos y la carencia de datos de observación son virtualmente ubicuos en la zona costera, pérdidas mayores recientes en la biodiversidad de moluscos pueden estar globalmente distribuidas pero han pasado desapercibidas.*

Palabras Clave: bentos, cambio histórico, ensamble de mortalidad, fechado con ^{210}Pb, macrofauna, núcleos de sedimento, pérdida de biodiversidad

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**Current address: Tasmanian Aquaculture and Fisheries Institute, University of Tasmania, Private Bag 5, GPO Box 252, Hobart, Tasmania 7001, Australia, email g.edgar@utas.edu.au*

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Introduction

Investigation of human impacts on the marine environment increasingly represents a contentious area of research, in part because of the scale of economic consequences. Substantial investments in capital resources and social well-being may be affected by management decisions made to alleviate impacts, particularly if such decisions involve the closure of areas to development. Decisions to protect seabed environments inevitably receive close scrutiny (Malakoff 2002), and the burden of proof in such situations is often heavily weighted toward a need for clear evidence of impacts (Dayton 1998; Gerrodette et al. 2002). Such proof is rarely available because the out-of-sight nature of the marine environment has precluded knowledge of former conditions and observation of changes over time.

A consequence of the general lack of historical baseline data for marine ecosystems is that the interpretation of results from human impact studies requires considerable inference. For example, comparisons between fished and unfished areas or between heavily fished and lightly fished regions are confounded by the historical pervasiveness of fishing. The usual reason unfished reference sites have not been exploited historically is that they possess little in the way of valuable stocks and therefore are likely to be intrinsically different from fished areas. Also, impacts of fishing may not correspond linearly with fishing intensity, such that ecosystems once lightly fished become modified to a similar extent as heavily fished areas. In this situation, comparisons between lightly fished and heavily fished areas provide little indication of overall effects of fishing.

All else being equal, the assessment of human impacts through temporal comparisons for variables such as animal density, biomass, and species richness generate more powerful tests than spatial comparisons because natural variation within sites is generally substantially less than variation between sites (e.g., Edgar & Barrett 2002). Much better are before-after control-impact (BACI) comparisons with data simultaneously collected at replicated times in both affected and unaffected (reference) locations (Green 1979). Nevertheless, in the case of fishing, unless the region under study has not been fished historically—for example, an area of deep seamounts that is becoming commercially accessible with improving fishing technology (Koslow et al. 2001)—the possibility remains that long-term ecosystem change facilitated by fishing occurred before the study began (Kaiser 1998).

We circumvented the lack of baseline information by using "death assemblages" of mollusc shells in ^{210}Pb-dated sediment cores to infer historical changes in marine biodiversity. Although ^{210}Pb dating is a well-established radiometric technique for dating sediments deposited in coastal marine, estuarine, and lacustrine environments over the last 120 years (Appleby & Oldfield 1983, 1992; French et al. 1994; McMinn et al. 1997), somewhat surprisingly, we could find no published studies where this technique has been used previously to infer historical changes to mollusc communities over the past century, other than a Florida study of effects of changing salinity on benthos (Brewster-Wingard et al. 1998).

Naturally accumulated mollusc shells generally provide reliable data on local mollusc abundance, with low levels of postmortem transport to site (Kidwell 2001; Kidwell & Flessa 1996). We relate observed changes in biodiversity to the chronology of scallop-dredging activities because fishing with toothed dredges has occurred widely in the region and has been suggested anecdotally by local fishers as responsible for bed destruction and possible ecosystem damage (Young & Martin 1989).

Methods

We collected single cores from 13 sites over a coastal distance of ≈100 km between Geilston Bay, a suburb of Hobart on the Derwent estuary, and Petchys Bay in the Huon estuary (Fig. 1, Table 1). Five sites were located in the Derwent estuary, six in the D'Entrecasteaux Channel, and two in the Huon estuary. Site selection was haphazard and unrelated to levels of fishing activity. Salinity at all sites at the time of sampling was close to fully marine (34–36 mg/L).

Divers collected cores between 30 October 2000 and 21 January 2002 by rotating and hammering 2-m-long, 10-cm-diameter polyvinyl chloride (PVC) tubes into the sediment in water depths of 8–16 m. After the tubes had been pushed as far as possible into the seabed, the distance from the top of the core tube to the sediment surface inside and outside the tube was measured to provide an estimate of sediment compaction during collection. The tube was capped to maintain suction, withdrawn, and immediately sealed at the bottom. Water above the sediment was siphoned off after transportation in a vertical position to the boat, and the core was plugged with polystyrene to prevent disturbance of the surface sediment. A short 4.5-cm-diameter core was also taken before the large core was collected to ensure an undisturbed sediment-water interface, a necessary reference for the ^{210}Pb dating technique.

In the laboratory, we split the large core tubes longitudinally and removed the top half. Sediments were descriptively logged and then sampled in 1-cm slices at 5-cm intervals for ^{210}Pb analysis, 5-cm slices at 10-cm intervals for shell analysis, and 2-cm slices at 5-cm intervals for heavy metal analysis. The outer 3 to 4 mm of sediment was discarded to avoid contamination.

We undertook ^{210}Pb analyses at the Australian Nuclear Science and Technology Organization in Sydney, Australia. Alpha spectrometric measurements were made of ^{210}Po, the granddaughter of total ^{210}Pb, with the two

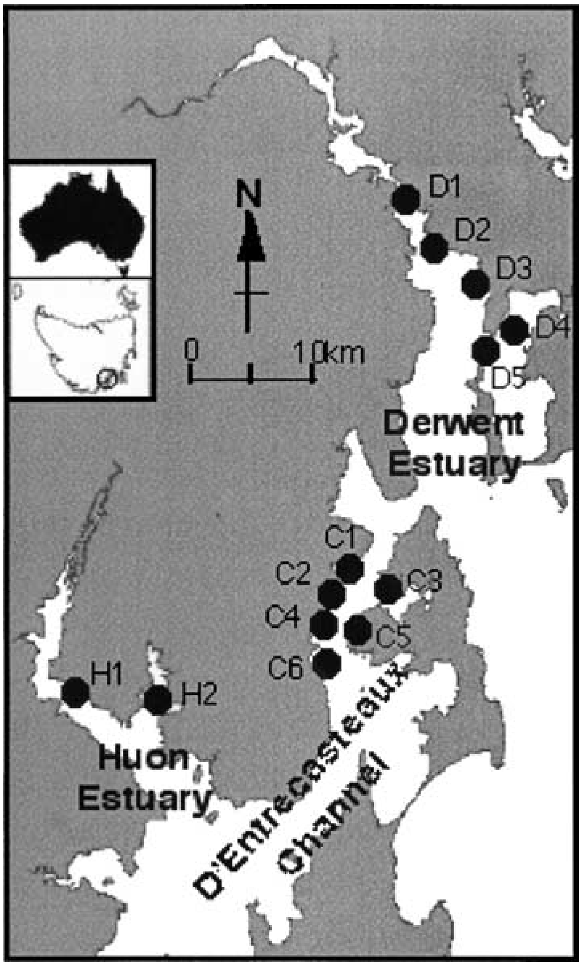

Figure 1. Map of the southeastern Tasmania region showing the location of sites sampled (black circles) in Derwent (D1-D5), D'Entrecasteaux (C1-C6), and Huon (H1-H2) subregions.

isotopes assumed to be in secular equilibrium. We estimated the supported ^{210}Pb component, formed by the in situ decay of radium in the sediments, by determining the amount of ^{226}Ra in each sample. Excess ^{210}Pb values were obtained by subtracting supported ^{210}Pb from total ^{210}Pb. Only the excess ^{210}Pb component is used to derive sedimentation rates (and thus core chronologies) because once incorporated into the sediments, it decays exponentially with time in accordance with its half-life of 22.26 years.

We used the Brugam model, a variant of the constant initial concentration (CIC) model, to derive sedimentation rates (Appleby & Oldfield 1978; Heijnis 2001). The CIC model is most appropriate when input of excess ^{210}Pb occurs predominantly via the erosion of catchment surfaces, as in this case, whereas the alternative constant rate of supply model (CRS) should be applied when input of excess ^{210}Pb to the sediments occurs predominantly by direct atmospheric input (Turner & Delorme 1996). Because the CIC model assumes a constant sedimentation rate, a plot of the log of excess ^{210}Pb versus depth should yield a straight line indicating sedimentation rate. Sedimentation rates, however, were nonlinear for many of our sites; for this reason, we applied the Brugam model, in which the profile is divided into linear segments and separate sedimentation rates are calculated for each segment (Heijnis 2001). Sedimentation rates are assumed to change instantaneously at points of inflection.

We confined our study to the past 120 years because older ^{210}Pb dates possess large confidence intervals (determined by the ^{210}Pb half-life and relatively low core-top-excess ^{210}Pb values for our region) and because there is no other reliable method for dating sediments deposited between approximately 400 and 120 years ago.

The ^{210}Pb chronologies were verified by comparison with an independent stratigraphic marker of known age—the sudden increase in heavy metal concentrations in sediments that occurred when an electrolytic zinc manufacturing plant began operating in 1917. Effluent from this zinc works was directly discharged, untreated, into the Derwent Estuary from 1917 to 1981. We measured heavy metal concentrations with inductively coupled plasma atomic emission spectrometry. Before analysis, dried bulk sediment (1–2 g) was digested for 4 hours

Table 1. Sample locations and dates, with water depth, core length, and mean fraction of silt (<63 μm) in sediments.

Location (code)	*Date collected*	*Latitude*	*Longitude*	*Water depth (m)*	*Silt fraction (%)*	*Core length (cm)*
Geilston Bay (D1)	Jan., ‘01	42°50.562′S	147°20.119′E	9.1	97.9	110
Kangaroo Bay (D2)	Jan., ‘01	42.52.840′S	147°21.508′E	14.5	73.1	121
Tranmere (D3)	Jan., ‘01	42°54.220′S	147°24.606′E	13.7	27.8	98
Ralphs Bay north (D4)	Nov., ‘00	42°56.340′S	147°26.192′E	10.5	34.1	68
Ralphs Bay entrance (D5)	Nov., ‘00	42°57.214′S	147°25.379′E	16.0	92.8	105
Oyster Cove (C1)	Oct., ‘00	43°06.649′S	147°16.584′E	14.0	38.7	127
Little Oyster Cove (C2)	Jan., ‘02	43°07.580′S	147°15.780′E	13.1	85.8	96
Shelter Cove (C3)	Oct., ‘00	43°07.145′S	147°19.623′E	8.6	67.8	130
Trial Bay (C4)	Jan., ‘02	43°08.150′S	147°15.400′E	12.3	57.9	90
Apollo Bay (C5)	Jan., ‘02	43°09.270′S	147°17.120′E	13.8	83.1	89
Birchs Bay (C6)	Jan., ‘02	43°10.413′S	147°15.154′E	13.1	60.9	80
Petchys Bay (H1)	Jan., ‘02	43°11.791′S	147°00.445′E	10.5	94.9	88
Port Cygnet (H2)	Jan., ‘02	43°12.226′S	147°04.982′E	13.3	96.5	104

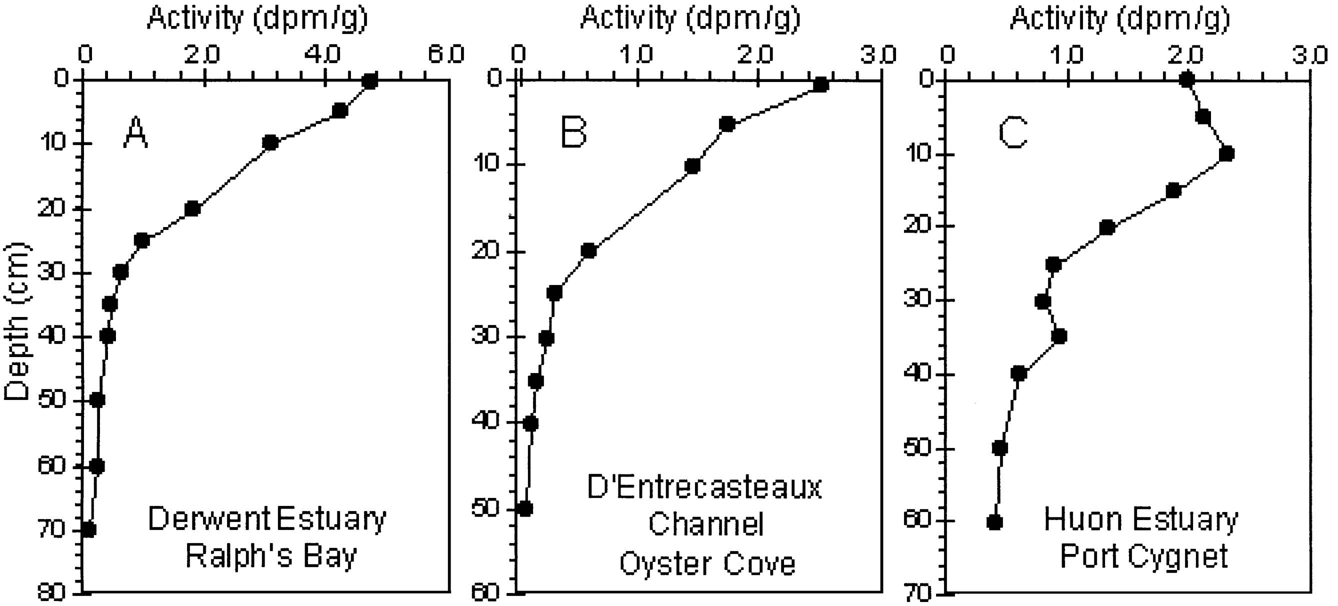

Figure 2. Typical examples of excess ^{210}Pb activity profiles with depth for a site in each of the Derwent, D'Entrecasteaux, and Huon subregions.

in a 5-mL aqua regia at 100° C and diluted to 50 mL with deionized water.

For faunal analysis, we passed sediment material through nested 1-mm and 8-mm mesh sieves. Shell fragments >1 mm were identified under a binocular microscope, and the number of unbroken or nearly intact gastropod shells and bivalve shells with intact hinges was recorded. We calculated the date assigned to the death assemblage in each 5-cm slice with the age model for the site and sediment depth at the midpoint of the slice.

We investigated relationships between mollusc death assemblages at different sites and sediment depths with nonmetric multidimensional scaling (nMDS). The data matrix showing total abundance of species at each site was square root transformed and then converted to a symmetric matrix of biotic similarity between pairs of sites with the Bray-Curtis similarity index (Clarke 1993).

To assess whether differences in species richness between samples were primarily a function of differing total abundance of shells in samples or of regional variation in the species pool, we calculated species accumulation curves with the EstimateS program (Colwell 1997). Total numbers of species in pooled samples (with associated standard deviations) were determined by randomizing 5000 times the order in which samples were added to the pool without replacement.

Results

All cores exhibited the expected decline in excess ^{210}Pb values with depth. The majority showed a consistent monotonic decline (Fig. 2a, b), although in a few cores the activity in the surface sample was depressed relative to the sample at 5.5 cm. This is apparently a common occurrence and perhaps results from either diffusion of ^{210}Pb in the top few centimeters or disequilibrium between ^{210}Pb and ^{210}Po (Eakins & Morrison 1978). Unlike cores from the Derwent Estuary and the D'Entrecasteaux Channel, the two cores from the Huon Estuary possessed excess ^{210}Pb values that were essentially constant in the top 10 cm (Fig. 2c), suggesting a 10-cm-thick mixed layer.

The initial concentrations (core-top-excess ^{210}Pb values) varied between cores within the range of 1.0–5.8 dpm/g. At one site, Shelter Cove in D'Entrecasteaux Channel, age models based on the decline of ^{210}Pb indicated very low sediment accumulation rates, such that only two dated sediment slices lay within the 120-year span of the study. This site was excluded from further analysis because of the large error associated with interpolated ages.

The independent ^{210}Pb and heavy metal dating methods typically showed good correspondence. At four of the five Derwent sites, the depth in the core at which 1917 occurs agreed for the two techniques within the 5-cm sampling resolution. At the other Derwent site, the two techniques disagreed by $\approx$10 cm, which corresponded to $\approx$15 years in that core.

The time span covered by each 5-cm slice varied little throughout the past century, averaging 7 years during the period from 1980 to 2000 and 7, 8, 6, 9, and 12 years during the preceding five 20-year periods. The number of mollusc shell species in sediment cores declined at all sites, from an overall mean of 21 species per sediment slice in 1890 to 7 in 1990. The downward trend was

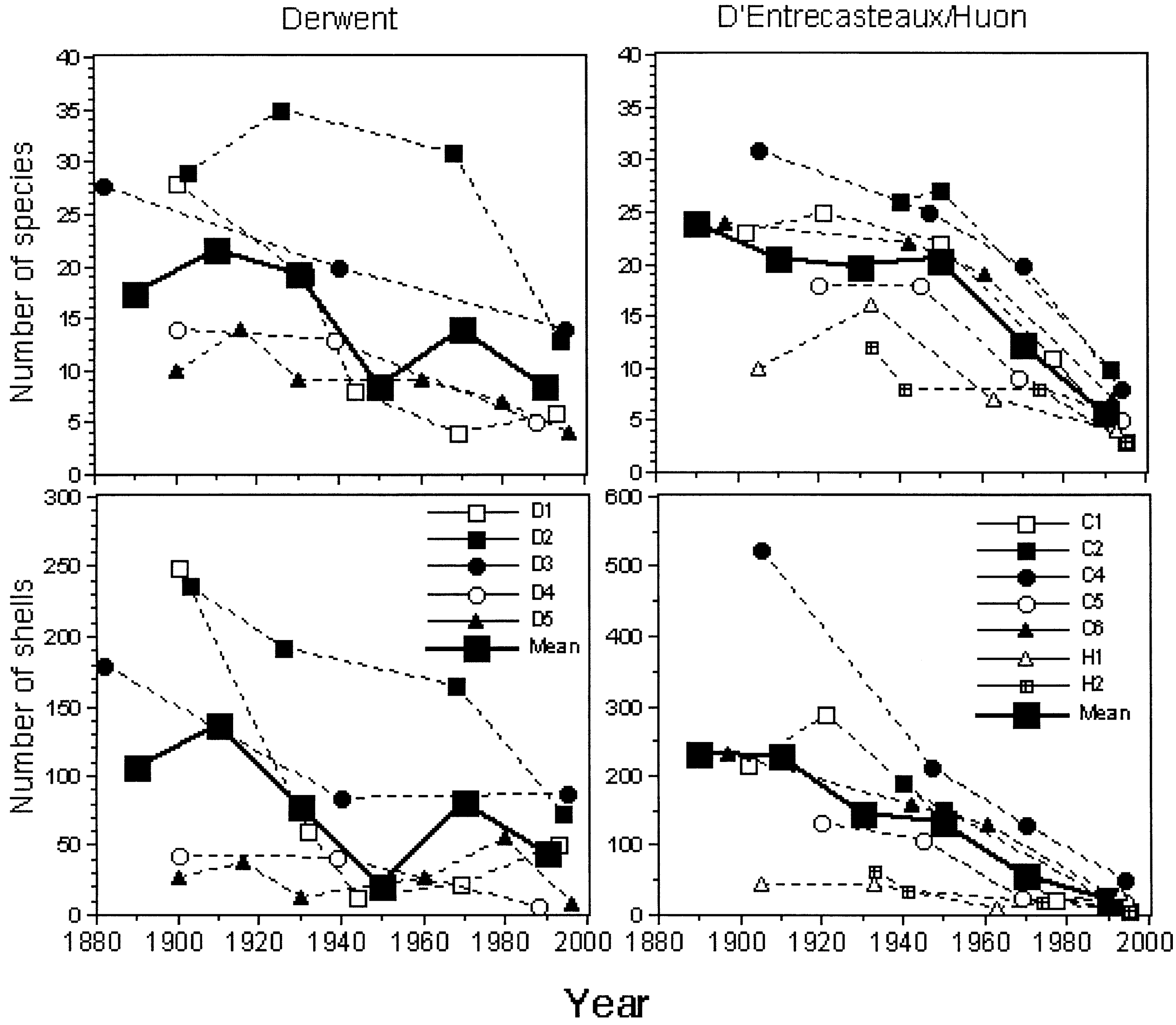

Figure 3. Changes over time in mean number of mollusc species and mean number of shells in 5-cm core slices for sites in the Derwent and the D'Entrecasteaux plus Huon subregions. Trajectories of individual sites (locations coded as in Table 1) are shown with dashed lines; the mean regional trajectory (binned in 20-year intervals) is shown with solid line.

most pronounced after 1930 in the Derwent subregion and after 1950 in the D'Entrecasteaux/Huon subregion (Fig. 3).

The decrease in mollusc species richness was partly associated with a general decline in mollusc shell production. Shell abundance declined overall from a mean of 146 shells per sediment slice in 1890 to 32 in 1990. Shell numbers in the Derwent at the start of observation (1890) were about half the numbers in the D'Entrecasteaux Channel subregion.

Scallops (*Pecten fumatus* and *Chlamys asperrimus*) and oysters (*Ostrea angasi*)—the only bivalves commercially harvested in large quantities from Tasmanian waters—exhibited population declines that were even more abrupt than for molluscs in general (Fig. 4). Scallop shells declined precipitously after 1910 in the D'Entrecasteaux subregion and from at least 1890 in the Derwent. The time course of decline for scallops was virtually identical between the two subregions, apart from a lag of 20 years. No oyster shell fragments were observed in cores after 1930 and few scallop shell fragments were seen after 1950.

The decline in shell species richness was caused, at least in part, by a regional decline in the species pool. When total number of species in pooled samples was plotted against abundance of shells, considerably more

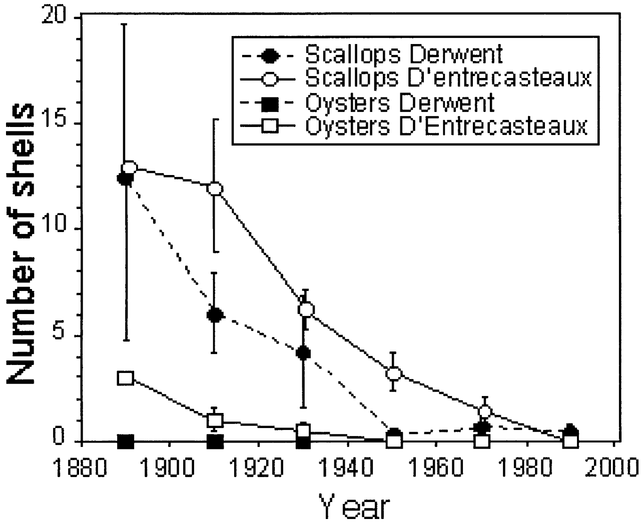

Figure 4. Mean number (±SE) of scallop and oyster shells in dated 5-cm-deep sediment slices collected at five sites in the Derwent and seven sites in the D'Entrecasteaux Channel plus Huon subregions.

species were present in samples deposited around 1930 than around 1990 (Fig. 5a). For example, approximately 44 species were present among accumulated samples of 400 shells from 1920 to 1940, compared with approximately 31 species from 1980 to 2000. This trend occurred despite the fact that the 1980–2000 samples were collected at 12 sites compared with 3.5 sites for 1920–1940 samples, and therefore the later samples encompassed a much wider range of habitat variability.

Spatial confounding associated with the number of sites sampled has been removed for the species accumulation curves shown in Fig. 5b. For these curves, a subsample of 32 shells, the average number of shells in the 1980–2000 samples, was selected at random from the collection of shells in each larger sample for use in calculations. Once this collection of equal-sized samples is examined, the number of species in pooled samples deposited in 1930 is seen to be almost double that for samples deposited in 1990.

A consistent regional change in the shell fauna is also evident in the plot of nonparametric multidimensional scaling (Fig. 6). The death assemblages at different sites consistently converge on the lower right of Fig. 6 over time. The most rapid faunal change occurred during the past 40 years for all sites, other than at Derwent sites Geilston Bay (D1) and Ralphs Bay (D5), where the shift occurred earlier.

Discussion

The usefulness of the sedimentary record for deriving species diversity estimates depends on several factors, most importantly, (1) preservation, (2) number of individuals counted, (3) time interval over which the sample accumulated, and (4) extent of vertical and lateral postmortem transport. We assumed that factors controlling preservation have remained constant over the last century. This assumption seems acceptable given that the time scale of interest is short. We considered it unlikely that the observed decline in molluscan species richness resulted from poorer preservation farther up core because damage to shells should accumulate over time rather than in the other direction (although damage may occur at an inconsequential rate once shells are buried to a few centimeters depth).

The number of species in a sample depends partly on the size of the regional species pool and the number of individuals present in the sample. Both these factors declined over time in our study. Although a similar volume of sediment was sorted for all samples, the number of individuals declined by a factor of five toward the core top, indicating a major loss of shell production over the past century. Species accumulation curves indicated that the decline in species richness was greater than accounted for by this decline in shell abundance. The death assemblage throughout the region also became more homogenous over time, with faunas in the Huon and D'Entrecasteaux/Derwent subregions, which were initially dissimilar, converging in recent decades. These results are all consistent with the hypothesis that shell production declined catastrophically and that some species were affected more heavily than others.

Interpretation of temporal changes in death assemblages nevertheless requires considerable caution because of the complex array of postmortem processes that influence shell preservation and accumulation (Kidwell & Bosence 1991; Kidwell & Flessa 1996). Regardless, mollusc death assemblages are generally considered to retain a strong signal of the original rank orders of species (Kidwell 2001), although the paucity of prior research on the topic means that potential artifacts associated with our results are inadequately known.

One potential artifact relates to sediment slices deposited over short time intervals that possess relatively few species because they miss some episodic recruitment events. Other than for slices representing the 1880–1900 period, however, where the 5-cm sediment slices accumulated over a mean period of 12 years, the time span encompassed by each slice varied little between depths, being close to the mean of 7 years recorded in the core-top sediment slice. For our data, then, this potential artifact was probably slight.

Research on the depth distribution of death assemblages at control sites—if any can be found that are outside human influence—has yet to be undertaken but is required to fully assess the importance of other factors with the potential to cause misleading interpretation of results. Such potential artifacts include mid-domain effects,

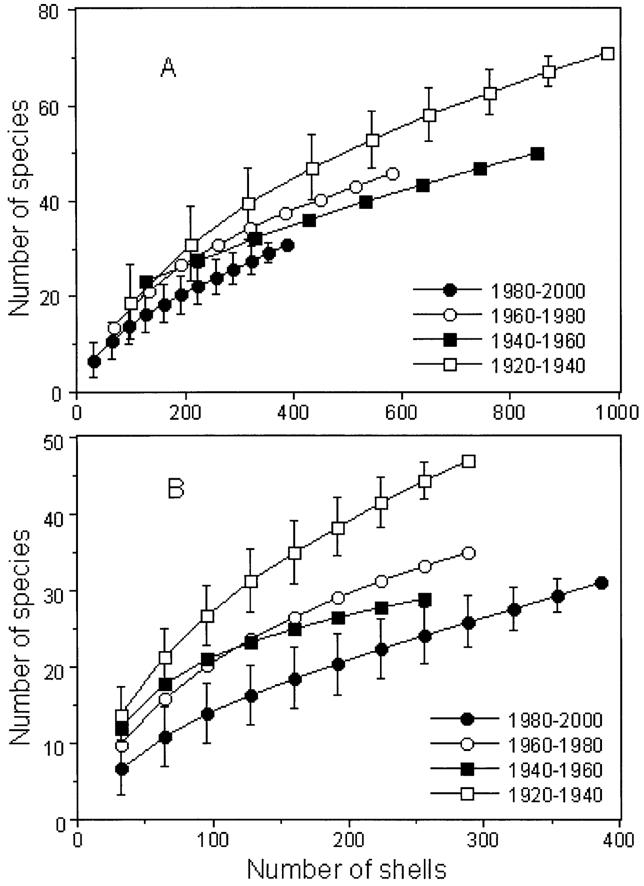

Figure 5. Total number of species (±SD) in pooled samples plotted against total number of shells for the four most recent 20-year periods based on (a) total data and (b) data standardized to a mean of 32 shells in a sample (the average number of shells in 1980–2000 samples). Error bars for intermediate time periods are not shown for reasons of graphical clarity but were comparable to 1920–1940.

sinking through fine sediment of large or dense particles, and time averaging (Kidwell & Bosence 1991).

Mid-domain effects result from two-way vertical exchange of shells in buried slices, whereas only one-way exchange of shells (i.e., with sediments below) is possible for the surface slice. Thus, some mollusc shells included in deep slices will be from both older and younger strata, whereas the core-top slice cannot include younger shells. Consequently, sediments close to the surface are more likely to include a mixture of shells from a shorter time span than deeply buried slices. This potential artifact affects the number of species in the sample rather than shell abundance and therefore does not account for the observed loss of shell production over time.

Processes of bioturbation, sediment erosion, and deposition that locally rework sediments and smear the depth profile (time averaging) can completely obscure short-term temporal patterns in some situations (Kowalewski et al. 1998). However, relatively rapid sedimentation rates (≈5 mm/year) and clear faunal changes between different strata indicate that time averaging was relatively slight for our data set.

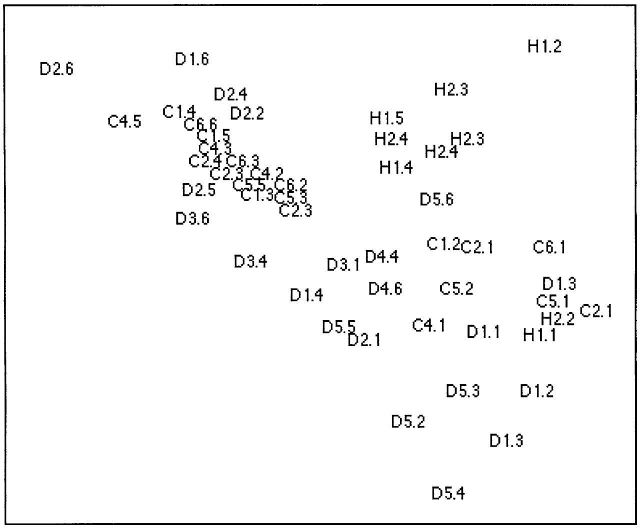

Figure 6. Results of nonparametric multidimensional scaling showing relationships between death assemblages in different dated sediment slices at different sites. Data labels are coded "site.time." Site codes are in Table 1 and time periods are 1, 1980–2000; 2, 1960–1980; 3, 1940–1960; 4, 1920–1940; 5, 1900–1920; 6, 1880–2000. Thus, D1.3 indicates Geilston Bay, 1940–1960. Samples that included <10 individual shells were excluded from analysis (stress = 0.13).

Rather than enhancing trends, time averaging in fact probably obscured a greater decline in regional mollusc production than indicated by the data. If a shell has an equal probability of moving up or down in the sediment, time averaging of sediments following major loss in shell production will cause net transport of older shell fragments from deep layers, where high abundance occurs, to the sparsely populated sediment surface. Consider an extreme hypothetical example of complete mortality of the mollusc community in 1950. Despite such an extinction, an indeterminate but possibly substantial number of older shells would have moved up core into younger surface layers (Meldahl 1990).

Despite limitations in terms of time span, resolution, and low number of samples analyzed, the latter resulting primarily from high analytical costs (≈US$3500 per core), the use of ^{210}Pb-dated sediment cores proved an effective tool for assessing recent changes to inshore mollusc communities. Our investigation, along with other recent studies (Dayton et al. 1998; Jackson 2001; Jackson et al.

2001), emphasizes the need for historical perspectives in assessing human impacts on marine ecosystems.

As with virtually every other inshore region of the globe, the marine environment off eastern Tasmanian has been subjected to an increasingly rich mixture of anthropogenic impacts over the past century (Crawford et al. 2000; Edgar & Barrett 2000). Without historical data, the specific consequences of different effects are virtually impossible to tease apart. Nevertheless, most human impacts possess distinctive historical and spatial signatures, albeit at scales much finer than the resolution of the present study. For example, effects of introduced species should correspond in time and place with the arrival of particular taxa; effects of land clearance and siltation should extend out from estuaries through time; effects of scallop dredging should correspond with the date when local fishing began; and effects of heavy metal pollution should correspond with contaminated sediments.

The decline in the southeastern Tasmanian mollusc community coincides with the history of the scallop-dredging industry. Europeans colonized southeastern Tasmania comparatively recently—in 1803 when Hobart was established as the first settlement in Tasmania and the second in Australia. Scallops (*P. fumatus* and *C. asperrimus*) were first harvested in the late nineteenth century by recreational fishers with small hand-hauled dredges working in the Derwent estuary near Hobart (Fairbridge 1953). The fishery gradually moved seaward as areas closer to Hobart were fished out. By 1925 the fishery in the Derwent had collapsed, and the fishing effort transferred to the D'Entrecasteaux Channel, where catches peaked at a total catch of 23.9 million in 1947 (Fairbridge 1953). Catches then quickly collapsed, and the region closed to scallop fishing in 1967.

These patterns of exploitation closely reflect observed changes in scallop shell abundance. Densities of dead scallop shells declined rapidly in the period prior to 1910 in the Derwent and prior to 1930 in the D'Entrecasteaux Channel (Fig. 4). The time course of decline was similar for the two subregions with a 20-year time lag.

The other large bivalve that has been commercially harvested in Tasmanian waters, the oyster *O. angasi*, declined earlier than scallops and was not observed in our dated sediment cores after 1930. This species was itself subjected to a dredge fishery, with catches in excess of 20 million oysters per year taken in the late 1880s (Anonymous 1882); however, the fishery then quickly declined to negligible levels (Webster 1908).

Collapses of native oyster fisheries to a few percent of original catches appear to be a general phenomenon worldwide, including in North America (Rothschild et al. 1994), Europe (Gross & Smyth 1946; Korringa 1946), and Australia. Millions rather than thousands of oysters may be needed for the establishment of oyster beds because of requirements for successful fertilization, spat production, and provision of settlement substrata (Gross & Smyth 1946; Korringa 1946).

The relationship between the collapse in mollusc biodiversity and dredging is correlative; for this reason, no inference on causality can be made. Presumably scallop dredging affected the benthic fauna over a wide geographic area or other factors caused a regional decline in molluscs that included scallops, and their disappearance ended the fishery.

The coincidence in timing between catastrophic observed ecosystem changes and dredging activities, coupled with the severity of dredge impacts identified in other studies (Flynn 1918, 1919; Jenkins et al. 2001; Currie & Parry 1996), warrants alarm. It is also notable that following the activities of the D'Entrecasteaux Channel scallop fishery, further serial loss of Tasmanian scallop fishing grounds occurred with little subsequent recovery. Scallop fishing in Tasmanian waters progressively moved offshore, to Great Oyster Bay and Norfolk Bay on the Tasmanian east coast around 1955 and to Bass Strait waters in 1964 (Young & Martin 1989), but in no inshore area have depleted beds recovered enough to support a major fishery in subsequent years. Further study is urgently needed to determine whether scallop dredging is causally related to long-term regional losses in biodiversity, perhaps by transformation of the geochemical properties of the sediment and water interface, by sediment resuspension, or by textural changes to the sediment including the removal of large shell fragments that affect boundary conditions and are used as settlement sites for spat (Sparks-McConkey & Watling 2001; Kaiser et al. 2002).

The other explanation for catastrophic loss of molluscs—that unknown factors have negatively affected the fauna at regional scales—warrants even greater alarm than the impact of scallop dredging. In this case, the factors remain unrecognized and are presumably continuing and uncontrolled. Siltation, introduced species, and effluent outfalls may have contributed to observed patterns, as well as natural oceanographic processes. Heavy-metal pollution may also have been a factor, although metal levels measured in sediments from locations other than the Derwent were low to moderate rather than high.

Of most concern is that a massive regional decline in mollusc biodiversity can go unnoticed. This loss of biodiversity is globally important considering that Tasmania possesses a rich molluscan fauna with >1000 species recorded (May & Macpherson 1958). Southeastern Tasmania was once regarded as a separate biogeographic province because of the large number of endemic molluscan taxa (Iredale & May 1916).

Factors contributing to the lack of prior awareness of ecosystem change in Tasmania include the absence of any long-term study of local marine benthos; the hidden nature of the subtidal environment; and the "sliding baseline syndrome," whereby perceptions of the normal state of ecosystems change gradually between generations without awareness that change has occurred (Dayton et al. 1998). Tasmania is far from unique in having these factors operating concurrently.

Among the more important questions raised by our study is whether other components of the benthic fauna, particularly polychaetes and crustaceans, have declined synchronously or perhaps increased in number to compensate for losses in molluscs (Currie & Parry 1999; Bradshaw et al. 2002). The paramount unresolved question is whether substantial but unrecognized recent declines in inshore biodiversity are common and geographically widespread. In this context, the only long-term information on benthos in southern Australia, a regional study of Port Phillip Bay, indicates a decline in mollusc species richness by half between 1969 and 1995, from 5.3 to 2.1 taxa/0.1-m^2 grab sample (Wilson et al. 1998). Long-term changes associated with scallop dredging, including declines in molluscs, have also been detected for benthic communities in the Irish and North seas (Bradshaw et al. 2002; Rumohr & Kujawski 2000). Moreover, Jackson et al. (2001) provide compelling evidence from a variety of sources that ecosystem change associated with dredging and other fishing activities is probably global. If mollusc decline in Tasmania is related to European colonization and technological development, losses may be much more pervasive.

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