PERSPECTIVE / PERSPECTIVE

Precaution in the harvest of Methuselah's clams — the difficulty of getting timely feedback from slow-paced dynamics[1]

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Abstract: Geoduck (*Panopea abrupta*) stocks are perceived as stable and their fisheries as sustainable, but this may reflect a mismatch between slow-paced dynamics (maximum recorded age 168 years) and short-term perception. Management is based on biological reference points, whose appropriateness as a means to ensure sustainability is limited by a sedentary lifestyle and long-term trends in productivity. Analysis of age frequency distributions for 1979–1983, postharvest recovery rates measured in Washington in tracts pulse-fished during the 1980s and 1990s, and age frequency distributions compiled in British Columbia during the 1990s consistently suggest that recruitment declined for decades (long before the onset of the fishery), reaching a minimum around 1975, and rebounded afterwards. In such scenario, reliance on biological-reference-point-based harvest rules without timely feedback could accelerate population declines, eventually driving an apparently sustainable fishery to collapse. The merits of approaches that rely on monitoring and feedback using data-driven decision rules are stressed. Transition from a biological-reference-point-based strategy to one based on monitoring and feedback will demand a shift in research focus to the design of practical monitoring programs and the evaluation of management procedures by means of simulations. For geoducks and other long-lived organisms, monitoring should integrate data informative at different temporal scales.

Résumé : Les stocks de panopes du Pacifique (*Panopea abrupta*) sont perçus comme stables et leur pêche comme durable, mais il peut s'agir d'une discordance entre leur dynamique très lente (âge maximal enregistré de 168 ans) et notre perception à court terme. La gestion se base sur les points de référence biologiques dont la pertinence pour le maintien de la durabilité de la pêche est limitée à cause du mode de vie sédentaire de l'animal et des tendances à long terme de sa productivité. Des analyses des répartitions de fréquence des âges en 1979–1983, des taux de récupération après la récolte mesurés dans des zones soumises à la pêche périodique durant les années 1980 et 1990 dans l'état de Washington et des analyses des répartitions de fréquence des âges compilés en Colombie-Britannique durant les années 1990 indiquent toutes de façon concordante que le recrutement a décliné pendant des décennies (bien avant le début de la pêche commerciale) pour atteindre un minimum vers 1975 et remonter ensuite. Dans un tel contexte, l'utilisation de règlements de récolte basés sur les points de référence biologiques sans rétroaction peut accélérer le déclin de la population et éventuellement causer l'effondrement d'une pêche en apparence durable. Nous mettons l'accent sur les approches qui se basent sur le suivi démographique et la rétroaction et qui utilisent des règles de décision basées sur les données. Le remplacement d'une stratégie basée sur les points de référence biologiques par une autre fondée sur le suivi et la rétroaction exigera une modification des orientations de recherche vers l'élaboration de plans de suivi démographique et l'évaluation des procédés de gestion à l'aide de simulations. Dans le cas des panopes et des autres organismes à vie très longue, le suivi doit intégrer les informations à différentes échelles temporelles.

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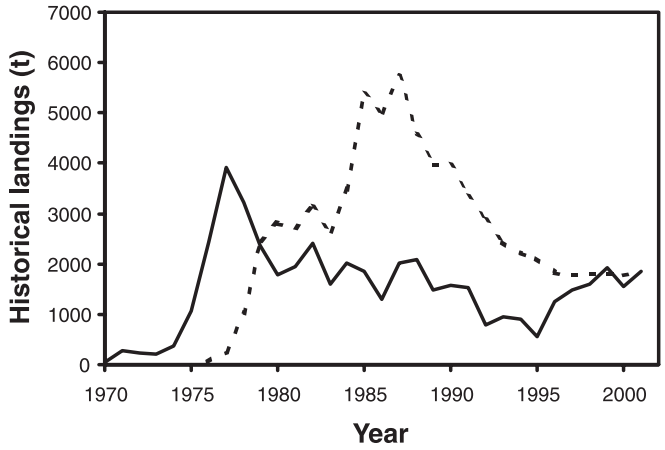
Introduction

Marine invertebrates of commercial value constitute a heterogeneous assemblage, confronting scientists and managers with a diverse array of peculiar problems (Orensanz and Jamieson 1998; Perry et al. 1999). Sedentary (not necessarily immobile) species with broadcast spawning, external fertilization, and pelagic larvae comprise a significant and well-defined subset of the assemblage that includes, among others, commercial bivalves. The assessment and management of these sedentary stocks present very specific difficulties mostly associated with the spatial structure of populations at different scales. Geoducks (*Panopea abrupta*) have an additional idiosyncrasy: they are extremely long-lived (Shaul and Goodwin 1982). Once a geoduck settles and buries itself into the substrate, it stays in place for up to 168 years (Bureau et al. 2002).

Harvested in subtidal grounds by commercial divers, geoducks support one of the most profitable fisheries in the northeastern Pacific from southeast Alaska to the Puget Sound Basin, Washington. Landings from the Washington and British Columbia fisheries (Fig. 1) constitute nearly all of the world's supply of this highly valued product. Interestingly, management systems differ radically between the two major fisheries: in Washington, geoducks on selected seabed tracts are auctioned yearly, while in British Columbia, there is limited entry and individual vessel quotas. In both cases, it is claimed that sustainability is ensured by very low harvest rates, on the order of 1%–3% of the harvestable biomass per year. The population dynamics of long-lived animals is necessarily slow paced, with the vagaries of recruitment and mortality being buffered by the coexistence of many year classes. At the time scale of the observation process, which is typically on an annual cycle, slow change in population structure conveys an impression of stability. When the annual survival of unharvested animals is on the order of 97%, stocks harvested at rates commensurate with natural mortality may secure a harvest for years into the future, even in the absence of recruitment. Given the observational errors typical in stock assessments, slow changes in abundance may not be readily detectable in survey data. This appearance of stability is further enhanced by hyperstability in the catch per unit of effort through the depletion process owing to shifts in the spatial allocation of effort (Hilborn and Walters 1992).

The sustainability of geoduck fisheries could be more apparent than real. Scenarios that could render the fishery unsustainable in the long run, even if harvest rates were low, include long-term declines in productivity, errors in the estimation of abundance, and (or) errors in the estimation of parameters commonly used to prescribe harvest rates. What would constitute a reasonable approach to manage these fisheries with precaution? In recent years, there has been a drive towards generic harvest control rules based on biological reference points (BRPs) (Restrepo et al. 1998). Mangel et al. (2002), in his synopsis of a symposium on "Targets, thresholds, and the burden of proof" (Coleman et al. 2002), listed species of limited mobility and long-term changes in productivity among situations that may render BRPs meaningless. Limited mobility is an intrinsic problem in the case of bivalves, and its implications have been discussed elsewhere (Orensanz and Jamieson 1998). The present study addresses the issue of slow-paced dynamics. Publicly available data (primarily age frequency distributions (AFDs)) are scrutinized to explore trends and patterns in recruitment. The merits of the assessment-based approach are discussed in light of the findings and alternative approaches that rely more on monitoring and feedback, using decision rules driven by data rather than mediated by models and assessments, are presented.

Fig. 1. Historical landings (tonnes (t)) in Washington (solid line) and British Columbia (broken line).



What kind of data would provide appropriate short-term feedback of value to management? AFDs reveal long-term trends in recruitment and abundance but they are not informative in the short term. Given the low rate of change of biomass, and considering uncertainty about the latter, surveys or ad hoc updates are unlikely to provide for timely feedback. One possibility would be a network of monitoring sites with variable histories of exploitation. As an example, results of an experiment conducted in Washington involving the monitoring of postharvest recovery at 15 sites are analyzed and discussed. This experiment received little attention in the past. The consistency between those results and results from analysis of AFDs is explored. It is argued that a narrow focus on BRPs leads management to overlook some significant phenomena evident in these data. The value of monitoring and feedback is emphasized and, in the particular case of long-lived organisms, the need identified to integrate pieces of information that are informative at different time scales.

Geoduck life history and its implications

Geoducks are extraordinary animals

Geoducks are extraordinary because of their size, lifestyle, growth pattern, and longevity. They are the largest infaunal bivalve in the world (individual weight up to 4+ kg) and live buried in the sediment down to a depth of approximately 1 m (Andersen 1971). Geoducks are dioecious broadcast spawners with pelagic larvae. The spawning season in Puget Sound runs between spring and early summer, peaking in June (Goodwin 1976); a single female may release millions of eggs during a spawning season. The pelagic larval stage lasts 25 days under hatchery conditions at 17 °C (Beattie

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and Goodwin 1993) but might extend longer in nature (up to 40–50 days). After settlement, postlarvae move at the sediment surface for several weeks (King 1986; Cole 1991) feeding on surface deposits until metamorphosis. Siphons then develop and suspension feeding begins while the juveniles bury themselves into the sediment. A technique to read internal rings in the hinge of the shells was first developed by Shaul and Goodwin (1982) and soon after utilized in several studies (Table 1). Geoducks have been convincingly shown to be extremely long-lived (Shaul and Goodwin 1982; Noakes and Campbell 1992; Strom 2002) with a recorded maximum age of 168 years (Bureau et al. 2002). Unlike trees and many deepwater fishes, geoducks grow fast during their first 7–12 years, and afterwards, their size remains virtually unchanged. On average, geoducks of 15 and 100 years of age do not differ significantly in weight (Breen and Shields 1983; Harbo et al. 1983; Bureau et al. 2002). Growth parameters for the von Bertalanffy model have been estimated in several studies (Noakes and Campbell 1992; Hoffmann et al. 2000; Bureau et al. 2002). Growth rate varies substantially along environmental gradients and between geographic regions (Goodwin and Pease 1991; Hoffmann et al. 2000; Bureau et al. 2002).

Given such longevity, natural mortality rates should be very low. Predation risk is very high during early benthic life (Goodwin and Shaul 1984) but decreases quickly after the age of 1 year (Sloan and Robinson 1984). Predation on adult geoducks has been generally considered rare under normal circumstances (Andersen 1971; Fyfe 1984). In recent years, however, predation by sea otters (*Enhydra lutris*) appears to have increased along the west coast of Vancouver Island, where the ranges of the two species overlap (Underwater Harvesters Association 2001). Published estimates of the coefficient of natural mortality (M) are in the range 0.014–0.054·year^{-1}.

The geoduck's lifestyle constrains monitoring, commercial harvesting techniques, and management options

The technique used to harvest geoducks is unique: commercial hookah-divers locate them by the tips of their siphons, which extend ("show") a few centimetres above the substrate, and then use hand-held water jets (stingers) to liquefy the sediment and extract the clams one at a time (for a description, see Washington Department of Fish and Wildlife (WDFW) and Washington Department of Natural Resources (WDNR) 1985). Orensanz et al. (2000), using data from Campbell et al. (1998), estimated a handling time of 0.3 min per geoduck, within the range of 0.25–0.5 min per geoduck indicated by WDFW and WDNR (1985). Once an adult geoduck is removed from the sediment, it cannot rebury itself; left on the bottom, it dies within a short period of time.

The vulnerability of geoducks to harvest gear depends on several factors: (*i*) the fraction of geoduck necks showing ("show factor") varies with sea conditions and is at a minimum during the winter (Goodwin 1977; Fyfe 1984), (*ii*) geoducks become available to commercial divers at ages between 4 and 12 years (Harbo et al. 1983), although for practical purposes, it can be assumed that they are fully recruited by an age of 8–10 years (Harbo et al. 1983;

Table 1. Published age frequency distributions of live geoduck samples from Washington and British Columbia with an indication of maximum recorded ages.

Site	Year(s)	Type of sample	Sample size	Maximum age (years)	Study
Puget Sound and Hood Canal (Washington)	1979–1982	11 unharvested sites, commercial gear	1 736	131	Goodwin and Shaul 1984
		3 unharvested sites, Venturi dredge	421		
Clayoquot Sound, west coast of Vancouver Island (British Columbia)	1980	Unharvested site, commercial gear	260	101	Fyfe 1984
Both coasts of Vancouver Island, five sites	1981	Harvested and unharvested sites, depletion of plots with commercial gear	2 276	126	Breen and Shields 1983
British Columbia (mostly west coast of Vancouver Island), eight landing sites	1981–1982	Commercial catch	1 396	146	Harbo et al. 1983
Hammond Bay near Nanaimo (British Columbia)	1982–1983	Commercial gear	365	107	Sloan and Robinson 1984
Ladysmith Harbour near Nanaimo	Nov.–Dec., 1980	Previously unharvested site, presumably commercial gear	403	102	Noakes and Campbell 1992
Near Gabriola Island (British Columbia)	1990	Presumably commercial gear	394	105	Noakes 1992
34 locations spread all over British Columbia waters	1993–2000	Suvey samples, sites with diverse harvesting history, commercial gear	14 210	168	Bureau et al. 2002

Note: Noakes (1992) also reported the age frequency distribution of samples from Bamfield (his fig. 3A, n = 993) and Ladysmith (his fig. 3B, n = 847). It is not clear if the sample from Ladysmith used in the study by Noakes and Campbell (1992) (n = 403) is a subset of that reported by Noakes (1992).

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Bradbury et al. 1998; Bradbury and Taggart 2000), as show factor and body size are not correlated for adults, (*iii*) harvest success is lower in shelly or compacted sediments, and (*iv*) visibility, often decreased by divers themselves because of sediment disturbance, can have an impact on harvest success. More important in determining the selectivity of the harvest are the factors that control the spatial allocation of effort (Campbell et al. 1998). Geographic gradients of density, size, and meat color are well documented and known to fishers, and so market preferences translate into patterns of spatial effort allocation. The proximity to port was an important factor during the early years of the fishery, particularly in British Columbia, leading to a gradual expansion from south to north (Campbell et al. 1998). As well, there are areas closed to harvest because of contaminated water, parks, etc., and deepwater areas that are beyond the operational reach of divers.

Because divers cannot determine the size of a geoduck before pulling it from the sediment, size limits are not a viable management measure. In Washington, meat color and price are correlated (the lighter the better) but color (as size) cannot be assessed until a geoduck is removed from the sediment. This introduces the possibility of high-grading, which occurs largely underwater as geoducks of poor quality are discarded in situ, creating serious problems for the assessment of the stock and the management of the fishery.

The dynamics of geoduck populations are poorly understood

Stocks are structured as metapopulations, collections of subpopulations of sedentary adults connected with each other by means of the dispersal of planktonic larvae. Relatively small segments of populations (not necessarily discrete, typically with an area of 20–50 ha) are defined as "tracts" in Washington or "beds" in British Columbia. In Washington, Sizemore et al. (1998) listed 267 tracts covering approximately 11 940 ha in total (average tract area is about 45 ha). In British Columbia, the fishing grounds are partitioned into geoduck management areas, of which currently there are a total of 243. Each geoduck management area comprises a variable number of beds. The estimated total bed area is close to 25 200 ha (Hand et al. 1998, their table 12), with an approximate mean bed area of 22 ha. No serious effort has been made to group beds into metapopulations, although some stocks (e.g., Hood Canal or southern Puget Sound) may correspond to metapopulation-level units. Any segment of such benthic metapopulations is open to recruitment by larvae originating at other beds. Availability of new settlers depends on local conditions prevalent in other regions as well as larger-scale environmental conditions that affect spawning and advection and survival of larvae. There is virtually no connection between the reproductive contribution of a given site and recruitment to that site. Pre- and post-dispersal phenomena are decoupled, with the result that key processes in the dynamics may be completely blurred if analyzed at the wrong spatial scale.

As is the case with most benthic sedentary organisms, density dependence in geoduck metapopulations is likely to be significant only during benthic stages. Density-dependent interactions have small operational scales, as individuals are influenced only by their neighbors (Orensanz et al. 1998). The effects of localized (small-scale) density dependence on the dynamics of the metapopulation are mediated by larval dispersal. Thus, it is convenient to distinguish between pre- and post-dispersal density-dependent mechanisms (Botsford and Hobbs 1995) and whether these are compensatory or depensatory. In the two latter, survival and (or) per capita reproductive contribution respectively decrease or increase as density increases. Several types of processes have been invoked for benthic sedentary invertebrates with broadcast spawning, external fertilization, and pelagic larvae (Fig. 2). Predispersal density-dependent processes may affect reproductive output either directly through direct depensatory effects (e.g., low adult density results in poor fertilization success, the "Allee effect") or indirectly through compensatory effects (e.g., low adult density results in faster individual growth and higher reproductive output). These processes may be important depending on the concentration profile of the populations, which summarizes the density conditions experienced by individuals (second-order properties of the spatial process; Orensanz et al. 1998). Postdispersal processes may affect the rate of recruitment through the effect of resident density on larval settlement and survival of prerecruits. In the case of geoduck populations, there is no clear evidence of density dependence either pre- or post-dispersal. Although density-dependent growth has occasionally been invoked (e.g., WDFW and WDNR 1985), several inquiries (Breen and Shields 1983; Fyfe 1984; Goodwin and Pease 1991) have been inconclusive. Depensatory effects on fertilization rate, although likely given geoduck's sedentary habits, also remain undocumented. As for postdispersal density dependence, no evidence exists that high density of resident geoducks inhibits settlement rate, growth, and (or) survival of postlarvae and juveniles. Also, experiments conducted to evaluate the hypothesis that larval settlement is facilitated in the vicinity of adult siphons (whether due to chemical attraction or hydrodynamics-mediated mechanisms) were inconclusive (Fyfe 1984; Goodwin and Shaul 1984).

The dynamic response of geoduck beds to fishing remains virtually unknown. There is only scant information available on postharvest recruitment, all pertaining to postdispersal dynamics (Table 2). A few purposeful experiments were conducted by Goodwin and Shaul (1984), but results were inconclusive because of problems with design, logistics, and low natural recruitment. Interestingly, whatever information is available has been interpreted in support of two radically different hypotheses. Hypothesis 1 ("negative" effects): "The decreased number of juveniles in the fished beds is probably due to an adverse effect of fishing on recruitment. The low number of juveniles 1–3 years age group was probably due to lower post-harvest recruitment, suggesting that commercial geoduck fishing has an adverse effect on recruitment" (Goodwin and Shaul 1984, p. 14). Hypothesis 2 ("positive" effects): "There have been recent reports from commercial fishers of high proportions of juveniles in some beds that have been heavily fished in the past. This is substantiated by some aged biological samples taken during surveys" (Hand et al. 1998, p. 17). Hypothesis 1 is empirically substantiated by the recurrent observation of juveniles dislodged from the sediment during harvest operations (Goodwin and Shaul 1984). The significance of this effect has not been measured; juvenile geoducks are able to rebury themselves (Goodwin

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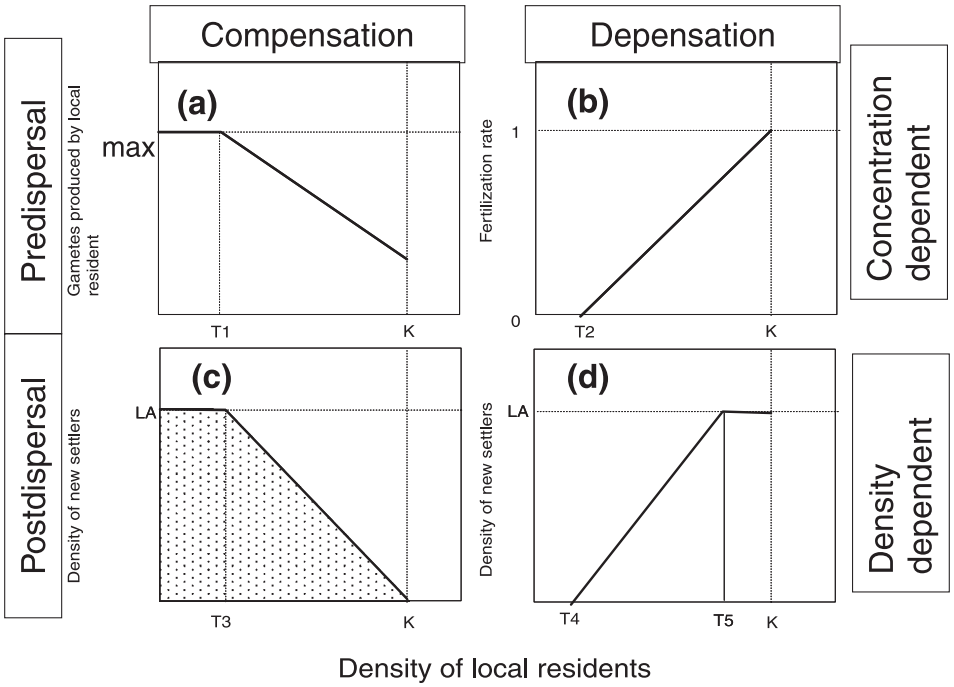
Fig. 2. Basic forms of density dependence documented or hypothesized for sedentary benthic invertebrates with broadcast spawning and pelagic larvae. Notice that in Fig. 2*c*, the solid line represents an "envelope" indicating the maximum density of new settlers expected for a given density of residents. In all others, the solid line corresponds to the expected value (mean) of the response variable, conditioned on the density of resident adults. *K*, local carrying capacity; *T*1–*T*5, density threshold levels; LA, index of larval availability; max, maximum of an index of relative gamete production by an individual.



and Pease 1989). Other ecological side effects of geoduck fishing are generally regarded as minor (Goodwin 1978; Breen and Shields 1983). Notice that environmental deterioration, incidental juvenile mortality, and postdispersal depensation could be confounded in experimental results. Hypothesis 2 is intuitively appealing. Adults are found aggregated in dense beds and have tremendous reproductive potential: millions of larvae are produced per capita during each reproductive season. Local (small-scale) abundance in such populations could be regulated through compensatory postdispersal density dependence; a virtually unlimited supply of prospective settlers would be regularly available to fill in gaps created by mortality of residents in crowded beds. Using closed-population stock–recruitment relationships as a metaphor, these are situations in which steepness is high: few breeders can quickly saturate a population decimated by mortality (e.g., by fishing) as long as aggregations exist to ensure successful fertilization. The implications for management are obvious and far-reaching. Recovery of harvested patches could be slow under Hypothesis 1, with a consequent reduction in productivity. Under Hypothesis 2, and considering that geoducks virtually stop growing after the age of 10 years, sustainable harvest rate could be possibly higher than dictated by rules of thumb based on natural mortality.

Postdispersal dynamics aside, fishing could pervasively increase predispersal depensation, even if this effect were practically undetectable. In sedentary broadcast spawners, the reproductive contribution of the stock may depend heavily on dense patches, but it is precisely there that fishers selectively allocate effort. As a bed or tract is depleted, there is, besides the obvious change in overall density, a less apparent effect on the spatial pattern of distribution (i.e., concentration or distribution of organisms relative to each other; Orensanz et al. 1998), with consequences on predispersal density-dependent processes.

Two contrasting management systems

The management systems in place in British Columbia and Washington are radically different, and so are their implications for the dynamics of the harvested stocks under different hypotheses about density dependence, in spite of harvest rate being very low in both cases. Summing up the contrasts, Freeman (1984) wrote that "British Columbia manages the geoduck like a fish, Washington like a tree. Both methods are controversial."

In British Columbia, the fishery is managed through a combination of limited entry and individual quotas (Muse 1998; Heizer 2000). The annual quota is set at 1% of virgin biomass (B_0), the long-term goal being to harvest 50% of B_0 over a 50-year horizon. Only a license was required during the earlier years. The number of licenses grew fast to a maximum of 101 in 1979, when a moratorium was introduced and licenses were made nontransferable. Between 1980 and 1981, the moratorium evolved into a formal limited entry program, with the number of licenses reduced to 55. Licenses, held by vessel owners, were made transferable between vessels but not between persons. Starting in 1989, the total quota has been split into equal parts among license owners, as requested by the Underwater Harvesters Association, the goal being "even distribution of the catch". Provin-

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Table 2. Hypotheses proposed to explain the local effects of the harvest on postharvest trends in recruitment rate and supporting evidence.

Hypothesis	Recruitment trends expected under hypothesis	Existing evidence
No effect	Same recruitment trends in harvested and unharvested plots	Inconsistent with the repeated direct observation of juveniles being dislodged from the sediment by harvesting operations; inconsistent with observation of higher density of prerecruits in unharvested plots than in harvested ones of the same tracts (Goodwin and Shaul (1984), their experiment 3)
Postdispersal overcompensation	Recruitment rate high immediately after harvesting and then gradually decreasing	No experimental evidence of postdispersal compensation; if observed, declining recruitment rate after the harvest could be confounded with global declining trends
Postdispersal depensation	Negative effect of removal of resident adults, settlement–recruitment improves as the adult resident population builds up	Consistent with positive correlation between number of adults left by the harvest and recovery rate in a recovery experiment (Orensanz et al. 2000); suggested by recurrently observed aggregation of juveniles in the vicinity of adults; enhanced survival consistent with experimental results on survival of planted juveniles (Goodwin and Shaul 1984, their experiment 4)
Incidental mortality	Recruitment drops after the harvest because of the incidental mortality of juveniles and then recovers as postharvest settlers start reaching harvestable size	Indicated by the repeated direct observation of juveniles being dislodged from the sediment by the harvest; presumably exacerbated by the tendency of juveniles to occur in the vicinity of adults (e.g., Goodwin and Shaul 1984, their experiment 5); for that reason, consistent with positive correlation between number of adults left by the harvest and recovery rate in the recovery experiment
Environmental impact	Gradual increase in recruitment rate after "healing" of the substrate from disturbance caused by the harvest	Inconsistent with lack of correlation between number of clams harvested (an index of disturbance) and recovery rate in a recovery experiment (Orensanz et al. 2000); inconsistent with repeated demonstration of moderate ecological effects of harvesting; experimental results inconclusive (Goodwin and Shaul 1984, their experiment 6)

Note: Only hypotheses involving local-scale processes are listed. Recruits include preharvest resident juveniles and postharvest new settlers. Owing to incidental juvenile mortality, recruitment rate will tend to increase over time following the harvest, even if settlement rate is constant and not affected.

cial waters are divided into three regions (north, central, and south) and each region into three subregions. Under a "three-year rotation time" introduced in 1989, one out of three subregions within each region is harvested every 3 years. The quota for that year is three times the calculated annual quota. The rationales for the 3-year rotation are purely logistic: easy monitoring, reduced number of landing ports, and concentration of the assessment effort. A so-called "amortization program" (in place since 1995) was motivated by the desire to compensate for high quotas in some areas that result from various factors (estimation errors among them). Annual quotas are adjusted (down) to evenly spread the remainder of $0.5B_0$ over time through the end of the 50-year horizon. Given the geographic extent of subregions, fishers have considerable freedom in the spatial allocation of effort. The management system now in place has been credited with reducing effort while increasing rent through improved product quality and access to new markets. In recent years, the fishery grew to become the major invertebrate fishery in the province; an all-time record of CDN$ 42.5 million was reached in 1995 (Hand et al. 1998).

A unique auction-based system is in place in Washington (Sizemore 2000; WDNR 2000*a*). Tracts are leased to private individuals or companies through a public auction. The State contracts to the highest responsible bidder and generally collects above 50% of the landed value; the average annual revenue to the State during the last decade has been approximately US$ 6.2 million. Before 1998, the nominal harvest rate in Washington was 2% of B_0. More recently, a constant harvest rate strategy based on $F_{40\%}$ was adopted (Bradbury et al. 1998; Bradbury and Tagart 2000); the current target harvest rate is set at 2.7% of the estimated current biomass. The allowable catch is calculated under the $F_{40\%}$ policy and then a group of tracts estimated to support an exploitable biomass in excess of that catch is selected and leased through the auctions. A preharvest survey is conducted in each tract and WDNR auctions close to approximately 80% of the estimated biomass; leasees typically do not harvest the

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entire amount because of economic and logistic constrains (e.g., catch per unit of effort dropping to a nonprofitable level). According to a "recovery proviso", a tract cannot be opened to the harvest until a survey indicates that it has recovered to preharvest conditions (WDNR 2000*a*). In principle, at least, this proviso should result in de facto rotation and spatial tracking of recovery rate, as fast-recovery tracts would be revisited more often than slow-recovery ones.

The contrast between the two strategies was early highlighted by Fyfe (1984): "In Washington State, with a smaller area to manage, the harvest concentrates within well-defined boundaries", while "in British Columbia the fishery is spread over the whole coast. The relative merits of an intensive harvest in discrete patches vs. a moderate harvest over large areas are presently unknown." Twenty years later these "relative merits", which relate to prevalent types of density dependence, remain unknown and unexplored.

Beyond low harvest rates and the spatially explicit elements of harvest strategies, perhaps the most significant component of the actual (albeit implicit) management system is a contingency: the existence of de facto reproductive refugia. The most obvious are beds located at depths beyond the reach of commercial divers and regions where density is below profitable thresholds. The known depth range of the species extends from the low intertidal zone to 110+ m (Jamison et al. 1984). At least in some locations, density increases gradually with depth down to at least the operational limit of surveys (25 m; Campbell et al. 1996) and commercial diving. Many tracts in Washington cannot be harvested, and they are not considered in quota calculations. Areas can be designated as noncommercial because of pollution, low density, small extension, substrate, land-use conflicts, navigation, depth (too deep or too shallow), proximity of shore, conflicts with conservation priorities (e.g., eelgrass beds or proximity of bald eagle nests), etc. The extent, distribution, and potential reproductive contribution of deep beds are unknown in the two fisheries. An approximate calculation by WDNR (2000*a*, 2000*b*) suggests that only one fourth of the total Puget Sound stock is considered as commercial biomass (the basis for allowable catch calculations).

Four important pieces of information

Much of the published literature on geoducks reports the results of surveys of abundance and estimation of growth and natural mortality parameters. Here, we focus on four pieces of information that are key to understanding the dynamics of harvested stocks but either have received little attention in the past or have become available only recently. AFDs of dead animals (Goodwin and Shaul 1984) and results from a network of monitored sites (Goodwin 1996; Bradbury et al. 1998) were never analyzed before; data summaries were published only in technical reports of little circulation. AFDs of live animals collected in 1979–1983 were published but only used to estimate natural mortality (Breen and Shields 1983; Goodwin and Shaul 1984; Sloan and Robinson 1984). AFDs from samples collected in 1993–2000 from British Columbia became available very recently (Bureau et al. 2002). Exploratory analyses and supporting information are contained in two recent technical reports. Orensanz et al. (2000) scrutinized publicly available information about the ecology and dynamics of geoduck populations in Washington and British Columbia before 2000. Bureau et al. (2002) presented results on age and growth in British Columbia populations, the first study of this type to be published in 20 years, and the largest ever (14 210 geoducks aged).

Natural mortality estimated from AFDs of dead individuals (Washington)

Most published estimates of natural mortality rate are based on catch-curve analysis of AFDs of animals collected alive between 1979 and 1982 (Table 1). In all cases, a constant coefficient of natural mortality (M) was estimated by fitting the exponential model to a range of ages selected ad hoc (Table 3). All of the AFDs used showed a remarkable scarcity of individuals younger than 20–50 years, a pattern that likely reflects low recruitment before the 1980s, as discussed later. Under such long-term trends in recruitment, estimates of M based on catch-curve analysis are likely to be strongly biased. The mortality rate of adult geoducks can also be assessed directly by tracking the fate of individuals whose location has been mapped in experimental plots (Fyfe 1984; Bradbury et al. 2000). Bradbury et al. (2000) reported results of an experiment (n = 1128 tagged individuals) in Hood Canal, Washington. Their estimated value of M (0.016·year^{-1}, 95% confidence interval 0.007–0.025·year^{-1}) is close to the lower end of catch-curve estimates. A third method, proposed by Orensanz et al. (2000), was utilized to estimate M. Goodwin and Shaul (1984, their fig. 6) collected dead shells from 1-m^2 quadrats at three locations with a Venturi dredge, which is reputed to be non-size-selective. Only paired shells of dead animals found in the upright position within the sediment were retained to prevent the possible bias introduced by shell transport. Goodwin and Shaul (1984) claimed that "geoduck shells remain in the substrate many years after the death of the clam", which is generally the case for thick-shelled infaunal bivalves. Thus

$$N_i = \sum_{a=i}^{a} n_a$$

where i and a index age, A is maximum age, n_a is the number of shells of age a in the sample, and N_i is the number of shells of individuals that died at age i or older (i.e., that lived at least to age i) should provide a schedule of average survival more reasonable than the AFD of live animals. Situations that might bias the pattern are nonstationarity in the survival schedule or an overwhelming contribution to the dead-shell pool by one or a few year classes that have still not vanished completely from the living population. The latter is unlikely in the case of geoducks because of high longevity and low recruitment rate. Simulation exercises confirmed that this method is robust to long-term trends in recruitment, our main concern. There are two main possible sources of bias. First, a bias could occur if the rate of decay of empty buried shells varied with the age of the individual at the time of death and, second, age specificity in the rates of mortality because of sources that result in empty shells left at the sediment surface after death. Goodwin and Shaul

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Table 3. Published estimates of the exponential coefficient of natural mortality (M), all based on catch-curve analysis.

Study	Low cutoff	High cutoff	Age interval for grouping the data (years)	Range of estimated M values (year^{-1})	Observations
Breen and Shields 1983	10	None	2?	0.014–0.019	The sample from one location, deemed not interpretable, was excluded from the analysis
Sloan and Robinson 1984	0	None	5	0.035	
	21			0.054	
Noakes 1992	20–50	None	5	0.034–0.038	Based on data published by Breen and Shields (1983), Harbo et al. (1983), and Noakes and Campbell (1992)
Bradbury et al. 1998; Bradbury and Tagart 2000	28	98	1	0.0226	Based on data published by Goodwin and Shaul (1984)
Orensanz et al. 2000	20	90	1	0.036	

(1984) argued that most adult shells remain within the sediment after death. This is consistent with direct observation of M by Fyfe (1984). While stressed individuals may be preyed by starfish, their shells ending up at the sediment surface, this seems to be a comparatively rare phenomenon. Removal of whole animals (shells included) by predators has been well documented for juvenile (small) geoducks, whose burrows are comparatively short, but seems to be rare in adults. A low cutoff point of 5–10 years should be utilized to minimize these potential biases. The cumulative frequency distribution of the AFD of the dead-shell sample (n = 459) departs from linearity (Fig. 3); the latter would be expected under the exponential model. Mortality seems to increase with age, which is to be expected under most biologically reasonable models and noticeable in populations of long-lived organisms. Age-dependent survival can be modeled with the Weibull function

$$N(t) = N_0\, e^{-\alpha t^{\beta}}$$

where α and β are parameters and t is the age of a cohort. The exponential model is a particular case of the Weibull model for $\beta = 1$. The Weibull model was fitted to the observed cumulative distributions (Fig. 3, left). Estimated parameters are $\hat{\alpha} = 0.00174 \cdot \text{year}^{-1}$ and $\hat{\beta} = 1.64$. The Weibull model provides a very good fit over the entire age range. The exponential model gives a reasonable approximation to average survival for the age range 20–90 years (Fig. 3, right). The estimated value of M ($0.036 \cdot \text{year}^{-1}$) is within the narrow range estimated by Noakes (1992) (Table 3) and also the median of all published estimates of M.

Trends in recruitment extracted from AFDs compiled between 1979 and 1983 (Washington and British Columbia)

In the absence of long-term survey data, the history of the stocks was reconstructed based on AFDs compiled in six studies conducted in Washington and British Columbia, mostly between 1979 and 1983 (n = 7251 geoducks aged) (Table 1; Fig. 4). Examination of these data showed a consistent pattern: age classes younger than (at least) 30 years appeared to be underrepresented. The pattern was the same in samples from harvested and unharvested sites in British Columbia (data from Breen and Shields 1983). Three hypotheses have been implicitly or explicitly advanced to explain this intriguing pattern: (1) gear selectivity, usually invoked in catch-curve analysis (Breen and Shields 1983; Noakes 1992), (2) highly variable and patchy recruitment: gaps in recruitment could simply reflect the vagaries of recruitment at single locations (Breen and Shields 1983; Goodwin and Shaul 1984), and (3) a declining trend in recruitment (Sloan and Robinson 1984; Bradbury et al. 1998; Bradbury and Tagart 2000).

Hypothesis 1 is not tenable given that age selectivity is unlikely for geoducks older than 4–10 years (Bradbury and Tagart 2000). Hypothesis 2 is inconsistent with the geographic coherence of the pattern. To scrutinize Hypothesis 3, a time series of relative year-class strength was computed by back-calculating relative abundance at age 5+ years for all of the age groups represented in each sample. In these backwards projections, the parameters estimated for the Weibull

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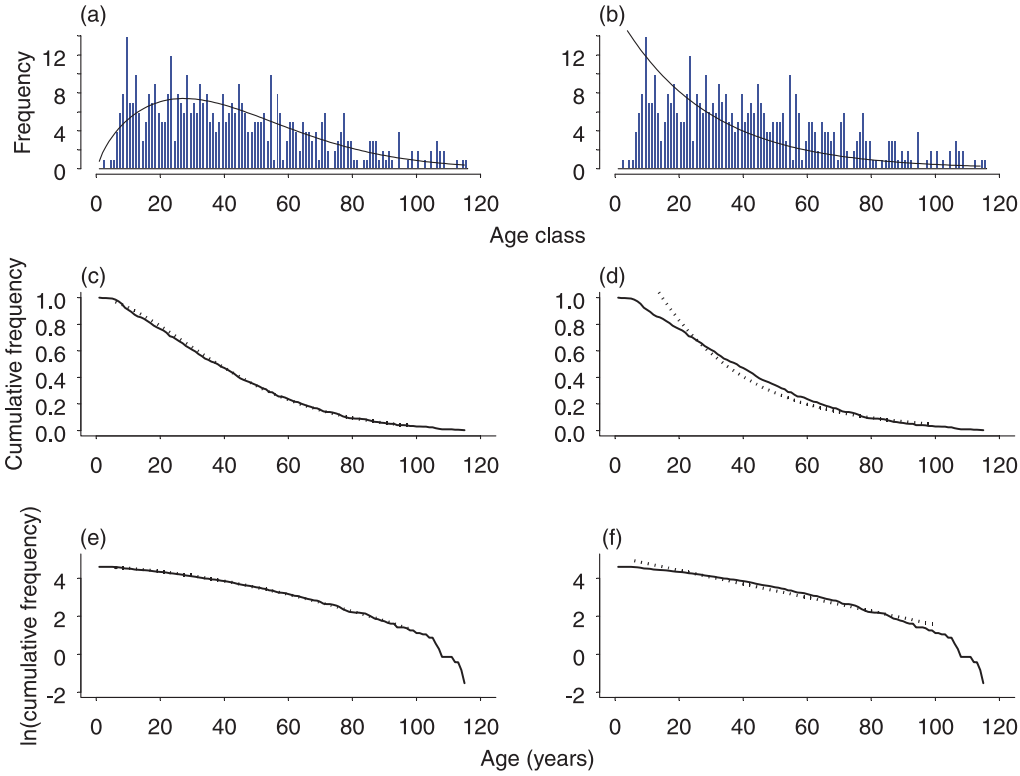
Fig. 3. Natural mortality estimated from samples of dead shell: (*a*, *c*, and *e*) Weibull distribution; (*b*, *d*, and *f*) exponential function; (*a* and *b*) age frequency distributions and model fit; (*c* and *d*) fit of the cumulative age frequency distribution by the two models; (*e* and *f*) log scale.



distribution were used. Time series of year-class strength back-calculated for the largest samples from Washington (Goodwin and Shaul 1984) and British Columbia (Breen and Shields 1983) show intriguingly coincidental patterns (Fig. 5). The rate of decline in year-class strength is comparable across these two large geographic domains, spanning five degrees of latitude (55–47°N) of intricate coastscapes. Most of the AFDs were obtained around 1980 and are thus indicative of a declining trend in annual recruitment rate during at least six decades (1920–1975) and at a large geographical scale. Consistently with Hypothesis 3, a generalized perception of very low and spatially variable recruitment rate (Goodwin 1976) discouraged direct observations of geoduck prerecruit abundance during the late 1970s.

Postharvest recovery in experimental plots (Washington)

This study was started during the early 1980s by Lynn Goodwin (WDFW). Fifteen tracts (Fig. 6*a*), due to be fished, were selected and sampled before the harvest, mostly between 1979 and 1985. Each site was revisited after the harvest (1.45–9.1 years after the preharvest survey) and again several years later (7–20 years after the first postharvest survey). Within each tract, a variable number of fixed stations (between 8 and 36) were established, each consisting of an 83.6-m^2 stripe. The total number of stations that were visited on the three occasions (all tracts pooled) was 324 (see Goodwin (1996) and Orensanz et al. (2000, their table 8) for complete information). Adult geoducks were surveyed visually (siphon counts) on each occasion.The experiment provided valuable information about fishing-related changes in abundance and the spatial pattern of abundance. At the time of the first postharvest survey, density (D_{post}) was 5%–56% of the preharvest level (D_{pre}). After a few years of recovery, density observed during the second postharvest visit (D_{rec}) had increased in all the beds (Fig. 6*b*), attaining between 13% and 117% of the preharvest level. Close examination reveals the existence of two clusters of trajectories: a fast-recovery cluster (seven tracts), where average annual recovery rate ranged between 5.1 and 11.2 geoducks per plot, and a slow-recovery cluster (nine tracts), where average annual recovery rate was in the range of 1.0–2.6 geoducks per plot (Fig. 6*b*). Evidence of highly variable recovery rate in WDFW's experimental plots prompts the question of whether there is some identifiable environmental correlate. Goodwin (1990, p. 29) offered a clue: "in general, recruitment is higher in tracts where water currents are of medium velocity". Post hoc speculation about geographic patterns in the dynamics of recovery provides a basis to propose hypotheses about geographical gradients in recovery and recruitment rates. Recovery rate is a result of the balance between recruitment and mortality. Orensanz et al. (2000) analyzed the recovery process under the assumption that natural mortality is invariant across space, which would be justified if (as we believe is the case) geographic variation in recovery largely reflects variation in recruitment. Four models of recovery were considered, all dealing exclusively with postdispersal processes: constant recruitment rate, constant recovery rate (postulated by Goodwin (1996) and Bradbury et al. (1998)), pure compensation, and a sequence of depensatory and compensatory phases. In the latter three models, abundance was assumed to return to D_{pre}, where recruitment and natural mortality would balance each other. Such an assumption was

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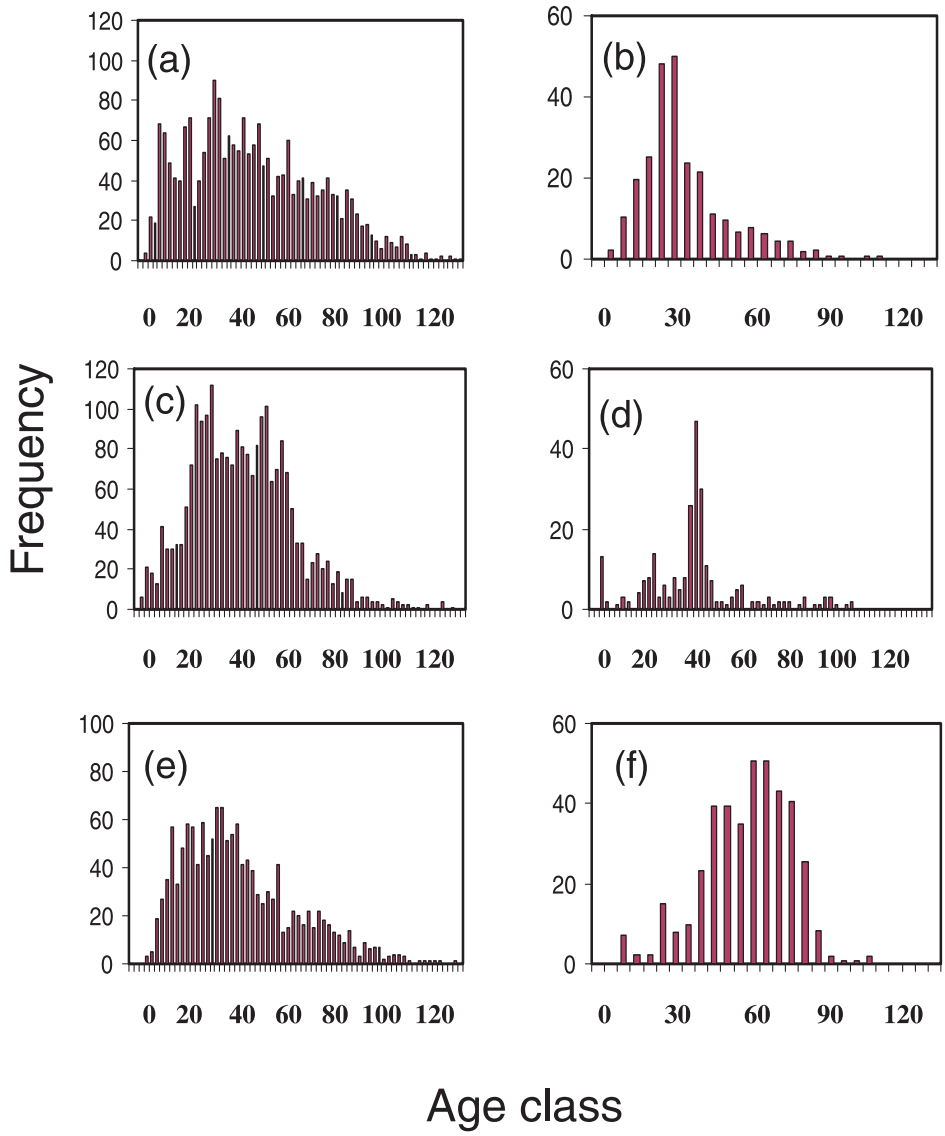
Fig. 4. Age frequency distributions published for Washington and British Columbia. The geographical level of aggregation is the highest used by the original authors. Interval widths are same as in original publications: (*a*) Goodwin and Shaul (1984); (*b*) Sloan and Robinson (1984); (*c*) Breen and Shields (1983); (*d*) Fyfe (1984); (*e*) Harbo et al. (1983); (*f*) Noakes and Campbell (1992).



not made in the constant recruitment model, which was therefore the most appropriate one to evaluate the rate of recruitment during the experiment relative to the preharvest condition. Under constant recruitment, the recovery trajectory is nonlinear and is described by

$$D_t = D_{\text{post}}\mathrm{e}^{-Mt} + R\sum_{i=1}^{t}\mathrm{e}^{-M(i-1)}$$

where R is the constant recruitment rate. There is no post-dispersal density dependence, i.e., no inhibition or facilitation of recruitment by the resident population. Abundance increases asymptotically towards equilibrium, at which deaths are balanced by recruitment. A value of R was estimated for each site by forcing the trajectory to pass through D_{post} and D_{rec} and using a constant coefficient of natural mortality ($M = 0.035\cdot\text{year}^{-1}$). The exponential model of survival, as discussed earlier, is a reasonable approximation for the range of ages expected to be represented in the population. Density in the slow-recovery cluster converges to D_{pre}, whereas in the fast-recovery cluster, it seems to converge towards a much higher level. Lower M, e.g., the value of M utilized by WDFW ($0.0226\cdot\text{year}^{-1}$), results in even higher equilibrium density (D_{eq}). AFD data suggest decades of declining recruitment through the 1970s. Had this continued into the 1980s and 1990s, the recovery rates documented by WDFW would be inconsistent. Orensanz et al. (2000) scaled back-calculated trends of year-class strength to match D_{pre} levels observed in the recovery plots. Estimated annual recruitment rate was in the range of 0.013–0.049 recruits·m^{-2} in the slow-recovery cluster of plots and 0.071–0.150 recruits·m^{-2} in the fast-recovery cluster. On average, the recruitment rate estimated for the slow-recovery cluster is within the bounds expected from AFDs for the 1970s, whereas the rates estimated for the fast-recovery plots is comparable with recruitment levels back-calculated for the 1930s and

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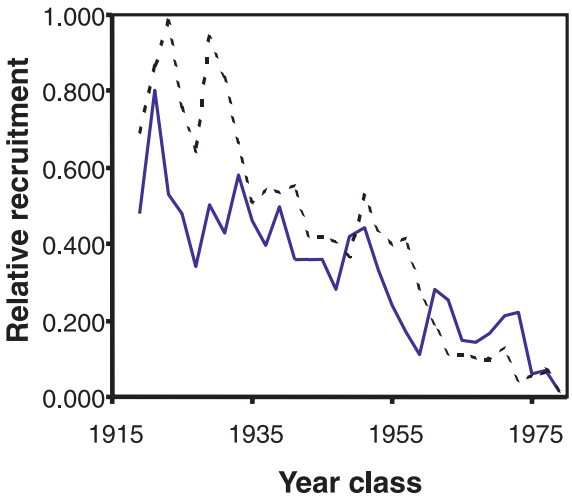
Fig. 5. Trends in relative recruitment (1915–1975) back-calculated from age frequency distributions from British Columbia (broken line) (data from Breen and Shields 1983) and Washington (solid line) (data from Goodwin and Shaul 1984) compared. The Weibull model of natural mortality was used in back-calculations.



the 1940s. This suggests that the low levels of recruitment calculated from AFDs for pre-1980 decades persisted during the 1970s in some regions, while recruitment may have rebounded to predecline levels in others. In no site did it decline further.

Concentration profiles were constructed to compare the spatial pattern of distribution of abundance between pre- and post-harvest surveys and between postharvest and post-recovery surveys (Fig. 7). The preharvest concentration profile corresponds to Type C of Orensanz et al. (1998) (equivalent to Type III of Prince and Hilborn 1998), which is typical of benthic invertebrate populations observed at scales on the order of one to several square metres and associated with hyperdepletion (i.e., abundance dropping faster than indicated by catch rate) in the fishing process. The postharvest profile corresponds to Type A (Type I of Prince and Hilborn 1998), which is typical of strongly thinned populations at the scale of observation. The fishery removed a substantial fraction of the geoducks, and those that survived the harvest were very "diluted" (Figs. 7*a*–7*d*). After a few years of recovery, the profile began converging back to Type C; as abundance increased, the absolute number of isolated individuals decreased (Figs. 7*c*–7*f*). The three cumulative profiles are compared in Fig. 8. The percentage of geoducks above the commercial threshold (as perceived at the spatial scale defined by quadrat size, which is not necessarily the perception scale of searching commercial divers) was 95% in the preharvest condition and then dropped to 65% after the harvest and rebounded to approximately 80% after some years of recovery.

Trends in recruitment extracted from AFDs compiled during the 1990s (British Columbia)

An extensive ageing project was conducted in British Columbia based on biological samples collected from 34 sites throughout the coast between 1993 and 2000 (Bureau et al. 2002). Those data were used to back-calculate year-class strength (cohort biomass at age 5 years) for each of the sites utilizing (*i*) the AFDs compiled by Bureau et al. (2002), (*ii*) estimates of B_0 (extracted from Department of Fisheries and Oceans reports), (*iii*) annual catch data for each bed (made available by the Department of Fisheries and Oceans), and (*iv*) a constant natural mortality coefficient of $M = 0.02 \cdot \text{year}^{-1}$. The latter is close to the experimental estimate of Bradbury et al. (2000) ($0.016 \cdot \text{year}^{-1}$) and to the value used by the WDFW ($0.0226 \cdot \text{year}^{-1}$). Simulation exercises showed that back-calculation of recruitment index series using the exponential model (with $M = 0.02 \cdot \text{year}^{-1}$) yields results that are very similar to those obtained with the parameters of the Weibull distribution, discussed earlier. To combine AFDs with catch data and estimates of B_0 (both reported in metric tons), biomass was assumed to be linearly related to numbers, which is justified by the fact that geoducks virtually stop growing in weight over the range of ages of interest (Bureau et al. 2002). Recruitment indices by site were pooled into six broad geographical areas to smooth out mesoscale spatial variation (Fig. 9). Because the samples were collected from different sites in different years, only samples collected in 1996 or later were used; only indices for 1996 and earlier year classes were included in the analyses (results are shown in Fig. 9).

Bureau et al. (2002) found indications that recruitment rebounded after 1980 in most regions, most notably in the south (west coast of Vancouver Island and Georgia Basin). The density of geoducks younger than 10 years was considered "significant" in comparison with estimated B_0. Good recruitment events appeared to have occurred during the 1980s and 1990s in beds with a wide range of harvest histories. The 1988 year class was particularly strong throughout the coast, but localized pulses of other year classes were also observed in all regions. That perception was confirmed by the back-calculated year-class strength analyses, which revealed some significant features (Fig. 9): year-class strength was at a global minimum around 1975 (indicated in Fig. 9 by a vertical broken line) and there was a post-1975 rebound in recruitment across all British Columbia coastal zones. The rebound occurred coherently in zones with very different harvest histories, reaching historical maximum levels along the west coast of Vancouver Island and the Georgia Basin (both heavily fished in recent decades). In the north coast and Queen Charlotte zones, which have been lightly fished, year-class strength shows a decades-long decline before 1975, a pattern comparable with that observed during the early 1980s in the south coast (Fig. 5). A comparable decline is not apparent in the data from Georgia Basin, the zone most heavily fished during the last two decades. The relatively strong rebound in this zone may indicate that effective harvest rate was higher than calculated, resulting in an underestimation of the relative strength of pre-1975 year classes.

An emerging large-scale pattern

Altogether, these four pieces of information are strongly suggestive of long-term trends in recruitment, coherent at a very large geographical scale. AFDs collected in 1979–1983 indicate that recruitment declined for decades, reaching a minimum around 1975. Pre- and post-harvest density data in several tracts in Puget Sound suggest a rebound in recruitment during the 1980s and early 1990s. The pattern that

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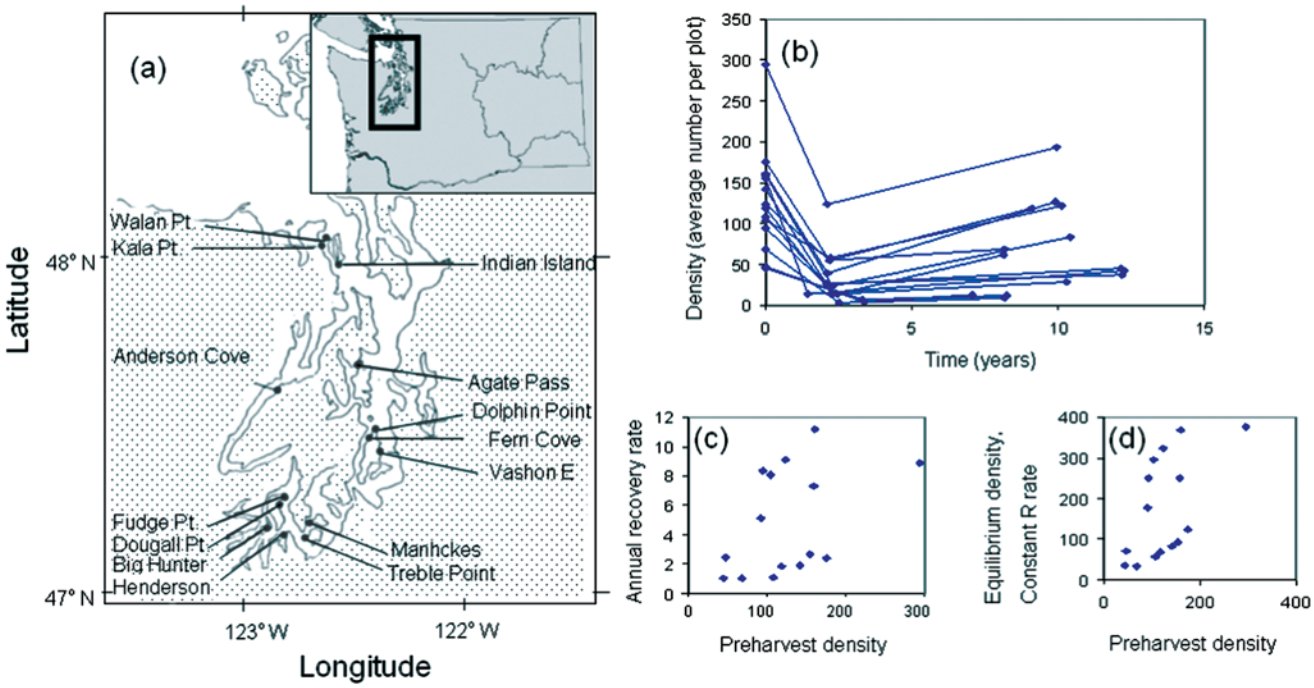
Fig. 6. (*a*) Puget Sound (inset map shows the location on the coast of Washington), sites of Washington Department of Fish and Wildlife's recovery experiment. Labeled on the right, fast-recovery cluster; labeled on the left, slow-recovery cluster. (*b*) Recovery in harvested plots; the three points correspond to mean preharvest, postharvest, and postrecovery densities. (*c*) Relationship between preharvest density and annual recovery rate. (*d*) Calculated equilibrium density reached under the constant recruitment hypothesis compared with preharvest density.



emerged from the AFDs compiled in British Columbia during the 1990s is consistent with that reconstruction of events. Preliminary results from Washington (J. Valero, unpublished data) indicate that strong recruitment pulses also occurred in the Puget Sound Basin during the last two decades, consistent with the geographic coherence documented for the pre-1980 decades.

BRPs are not appropriate as the sole basis for the sustainable harvest of geoduck stocks

Over recent years, there has been a drive in the United States and Canada towards generic harvest control rules based on target and limit BRPs proposed as default harvest guidelines that could perform adequately for most fisheries and circumstances (Restrepo et al. 1998). Generic harvest rules specify the desired harvest rate or catch as a function of the actual stock size, the case of the $F_{40\%}$ policy currently in place in the Washington geoduck fishery. Some BRPs are ad hoc rather than generic, as is the case for the $0.01B_0$ annual harvest rate implemented in the British Columbia fishery. For these harvest rules to work, the estimates of stock size must be reliable and BRPs must be well determined and meaningful in terms of the stock dynamics. Both conditions are difficult to meet in reality. Hilborn (2002) discussed some concerns with regard to the use of BRPs, among them the effect of uncertainty about virgin or actual biomass. This is a serious problem in geoduck stock assessment, as there is no expedient way of surveying geoducks over large areas. Current choices are either annual surveys of actual biomass in small areas to construct a composite map combining data collected over many years (Washington) or very coarse updates of estimated virgin biomass over broad areas (British Columbia). Details are provided in Appendix A. Uncertainty is high in the Washington case because estimated abundance is always outdated for large areas and in the British Columbia case because estimates are based on quantities (like bed area) that are even difficult to define. In British Columbia, for example, estimated B_0 went up during the early years of the fishery as new beds were discovered (Fig. 10) and then down as a result of a combination of lower estimates of density from survey data, arbitrary downscaling of area estimates, and bed boundary revisions. A later rebound of estimated B_0 was due to a combination of abandoning area downscaling, an increase in regional density estimates from surveys, a small increase in mean weight estimates, and some new beds found. Summing up, in neither of the two systems does this limited surveying capability provide adequate feedback about population trends, as required for the implementation of BRP-based harvest rules. Limited mobility and long-term changes in productivity, such as those found in geoduck stocks, bring to question the applicability of BRPs (Mangel et al. 2002; Parma 2002). Consider first the issue of spatial structure. In metapopulations of species of limited mobility, as discussed earlier, the decoupling of pre- and post-dispersal processes involved in stock–recruitment dynamics and the open nature of subpopulations preclude the use of simple stock–recruitment relationships. Yet, generic reference points, such as the $F_{40\%}$ utilized in Washington, have been proposed on the basis of their performance

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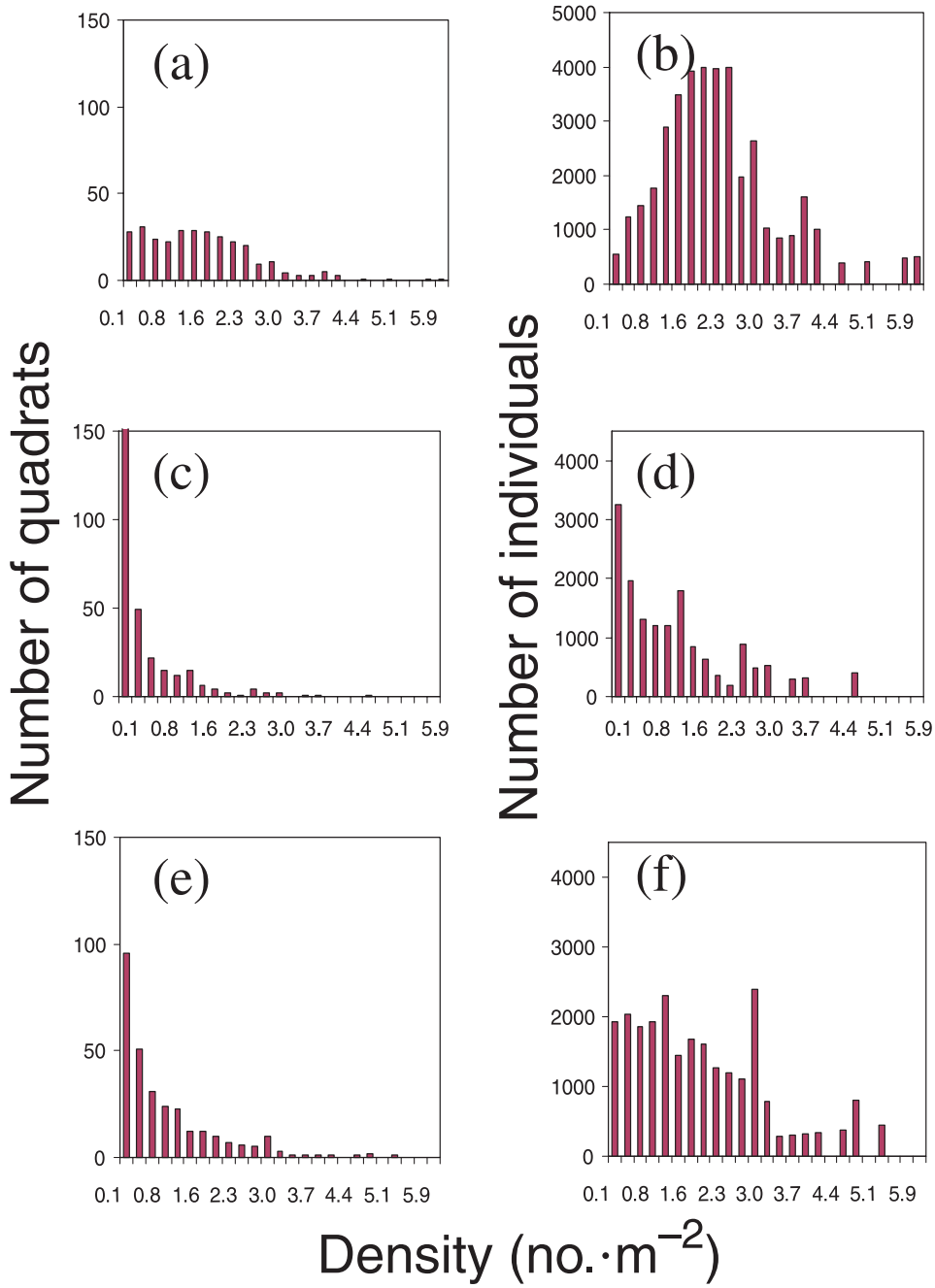
Fig. 7. Recovery sites (individual plots pooled): (*a*, *c*, and *e*) density frequency distributions; (*b*, *d*, and *f*) concentration profiles; (*a* and *b*) preharvest; (*c* and *d*) postharvest; (*e* and *f*) postrecovery.



evaluated using a range of stock–recruitment relationships under the closed-population assumption (Orensanz and Jamieson 1998). Significant elements of the renewal process, including connectivity and different forms of density dependence specified at the appropriate operating scale (Fig. 2), have never been properly incorporated into the evaluation of the strategies implemented in either fishery. The implications are not trivial. Recovery of harvested beds or tracts can be very different under the two management systems depending on the nature of density dependence. If concentration dependence of fertilization rate were significant, recruitment could depend on the reproductive contribution of high-density hotspots, precisely the ones that are selectively harvested by unconstrained fishers. This is illustrated by changes in concentration profiles produced by fishing in Washington's recovery experiment, described earlier (Figs. 7 and 8). A combination of concentration-dependent fertilization rate and fishers' freedom to allocate effort across large geographical areas would be a matter of concern in the British Columbia system. On the other side, if compensation were important, there would be a forgone productivity in a Washington-like system.

A second problem highlighted by Mangel et al. (2002) refers to long-term trends in productivity. Given the geoduck's extreme longevity, alleged stability of populations and sustainability of the fisheries that they support could possi-

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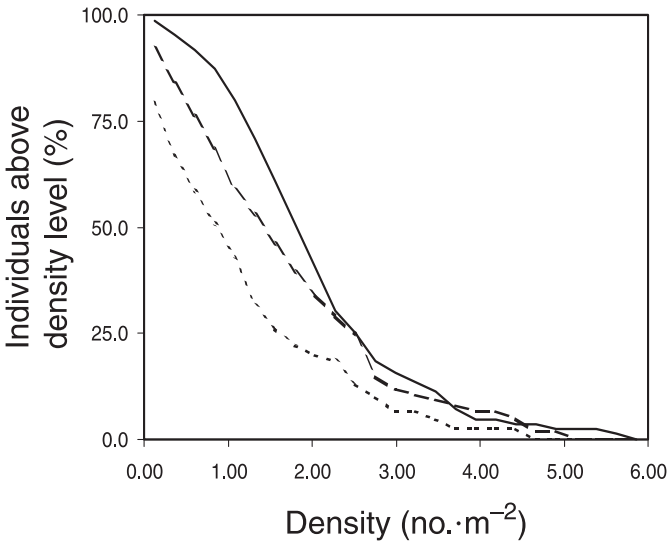
Fig. 8. Recovery sites (individual plots pooled): cumulative concentration profiles (percentage of individuals concentrated above density level): preharvest (solid line), postharvest (dotted line), and postrecovery (broken line) samples.



bly be more apparent than real. Geoduck abundance changes slowly, the impact of recruitment variability being buffered by the large number of year classes present in a population. Thus, eventual trends in abundance could be perceived only through the examination of long time series of data. Short-term variation in abundance is likely to go undetected because of large observation errors. Complicating things even further, and as is the case for other benthic fisheries, serial spatial allocation of fishing effort could result in hyperstable trends in catch per unit of effort underestimating the real rate of decline of total abundance. In fact, the large data set on catch and effort available in British Columbia has not been used in stock assessment for that reason. If the six-decade declining trend in recruitment detected in the AFDs from around 1980 had continued into the 1980s and 1990s, the application of BRP-based harvest rules without timely feedback could have accelerated the decline, eventually driving stocks to commercial extinction and an apparently sustainable fishery to a collapse. Nonstationary productivity also aggravates uncertainty about virgin biomass. Because geoduck fisheries started operating during the 1970s, biomass back-calculated to that period was used as an estimate of B_0. Back-calculation of relative abundance of geoducks in Washington based on AFDs suggests, however, that biomass in the 1970s was about 50% of the biomass during the 1920s and still declining, even in the absence of a fishery (Orensanz et al. 2000). As pointed out by Mangel et al. (2002) (after Hilborn 2002), virgin biomass is based on a theoretical construct that depends on existing environmental conditions.

The geoduck fishery started in 1970 in Washington and in 1976 in British Columbia and by 1980 was well developed in both jurisdictions. Since then, it evolved to become the most lucrative invertebrate fishery in the Pacific Northwest. Yet, evidence of a decades-long decline in recruitment ending around the mid-1970s and the rebound during the subsequent two decades went virtually unnoticed, this in spite of clear signs of the decline being present in AFDs collected between 1979 and 1983 and of stronger more recent recruitment in the results from Washington monitoring sites. A narrow focus on BRPs and BRP-based harvest rules had no use for information that did not contribute to the estimation of parameters required by standard models (B_0, M, K, L_∞). Once growth and mortality parameters became available, ageing efforts were interrupted for close to 20 years in both jurisdictions; Washington monitoring sites survived through the individual efforts of a few committed scientists. Distraction of scientists from important problems by narrowly focusing their attention on BRPs has been diagnosed as a general problem with this approach (Hilborn 2002).

An alternative approach, with emphasis on monitoring and feedback

Is there an alternative to the so-called "assessment-based approach" and reliance on BRPs? One worth considering emphasizes monitoring and feedback using decision rules driven by data rather than mediated by assessment models. In a "management-procedure approach", harvest regulations are driven by a predetermined feedback control rule, usually a simple algorithm for adjusting catches in response to monitoring indices. The possible outcomes of alternative management procedures are evaluated through simulation of the dynamics of the system (stock, fishery, and monitoring) with an operating model, tailored to specific situations (e.g., Butterworth and Punt 1999; Parma 2002). The operating model can be as complex and realistic as desired, but decision rules usually depend on simple calculations. The management-procedure approach appears as a reasonable option for the design of stock-specific harvesting strategies, which in the case of geoduck could include different sorts of spatially explicit regulations. While spatial components are present in both management systems (de facto reproductive reserves, even distribution of the harvest in British Columbia, de facto rotation in Washington), their merits, significance, and implications have never been evaluated. This management-procedure approach is currently being explored by some of us (J. Valero, A.M. Parma, and R. Hilborn); results will be reported elsewhere.

The question, now, is what kind of monitoring would be most appropriate in the case of geoduck fisheries. Monitoring may prove difficult in a situation where abundance changes at a very slow pace because of low rates of mortality (natural and fishing) and recruitment and complicated by the fact that actual abundance cannot be accurately surveyed every year over the entire region of interest. AFDs are informative about stock dynamics at a time scale far larger than that required for timely management response. Is there any practical way of complementing AFDs with information meaningful at a shorter time scale, more commensurate with the scale of management decisions? Given that short-term changes in total abundance are buffered by the coexistence of many year classes, and thus difficult to detect, a more proximate index of short-term recruitment success would be more suitable. A network of monitoring sites (plots) with variable histories of exploitation is a possibility, well exemplified by the experimental study of postharvest recovery described earlier. Such a network has several merits: (*i*) it is

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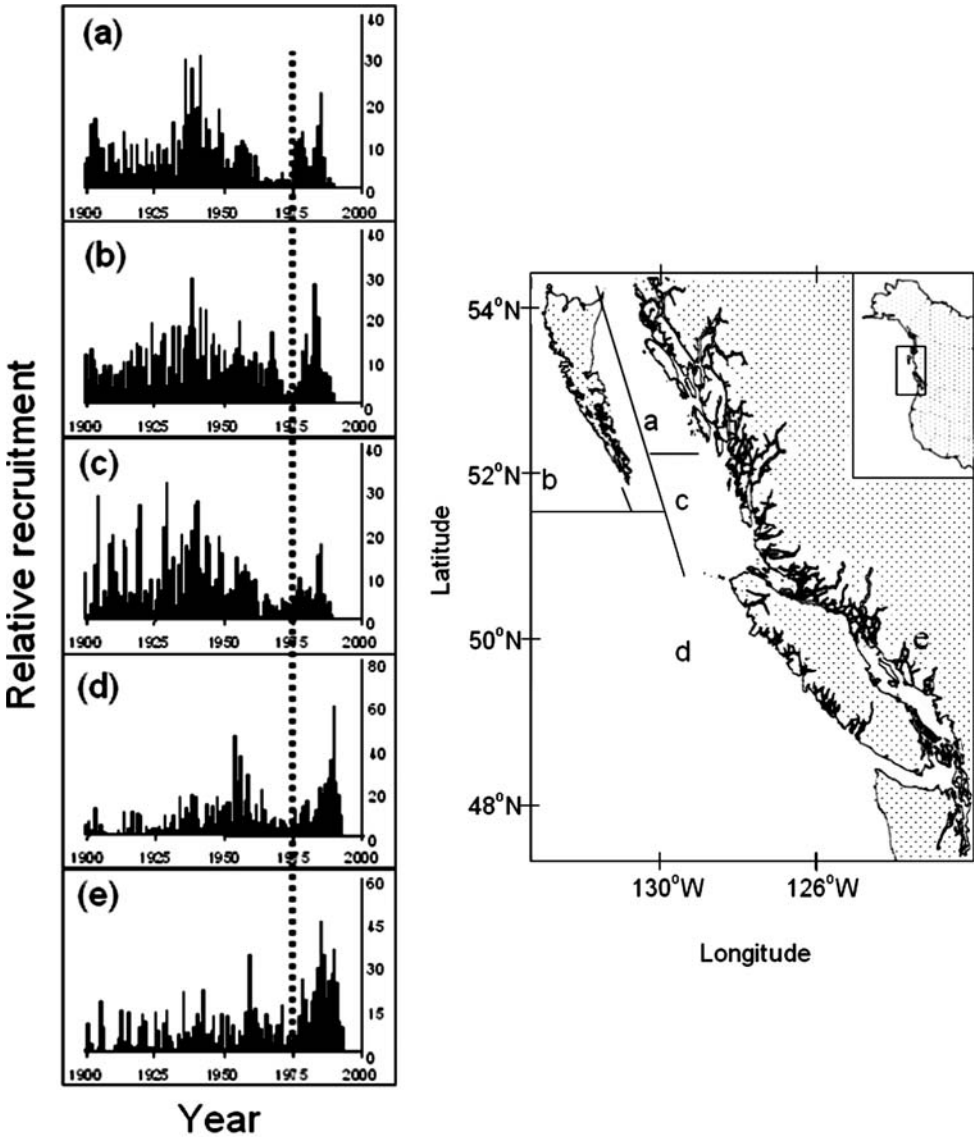
Fig. 9. Time series of relative recruitment backcalculated for five regions in British Columbia from age frequency distributions in Bureau et al. (2002): (*a*) north coast; (*b*) Queen Charlotte Islands; (*c*) central coast; (*d*) west coast of Vancouver Island; (*e*) Georgia Basin. See text for details.



informative about geographic and temporal patterns in recruitment, (*ii*) it may contribute to a better understanding of postdispersal density dependence and postharvest dynamics, and (*iii*) the data incorporate the effects of poaching.

As discussed earlier, results from Washington's experimental plots obtained during the 1980s and early 1990s appeared to be inconsistent with trends in recruitment reconstructed from AFDs compiled between 1979 and 1983. The age data that were needed to determine whether this was due to a rebound in recruitment became available only two decades later. First a worrisome sign, later the good news: both went unnoticed even when evident in the raw data. The consistency of results from AFDs and from experimental sites illustrates the value of integrating different pieces of information into a monitoring program to capture the dynamics of the harvested stock at different time scales. Yet, it is important to consider that information must be analyzed at a meaningful spatial scale. If long-term regional trends in recruitment are the subject of interest, then information has to be pooled at a large spatial scale to override the effect of small-scale spatial variability. If small-scale density-dependent processes are investigated, then experiments must be tuned to the operating scale of the phenomenon of interest (e.g., through the size of the experimental plots).

Beyond conservative harvest rates

The precautionary approach to fisheries management is often narrowly equated with the adoption of cautious harvest levels, guided by conservative BRPs (Restrepo et al. 1998). This emphasis is a result of fishery regulations articulated after the Sustainable Fisheries Act (in the United States), several international treaties, and other regulatory frameworks. BRPs, and their shortcomings, were discussed earlier in this paper in relation to geoduck fisheries. A well-known consultation conducted by the Food and Agriculture Organi-

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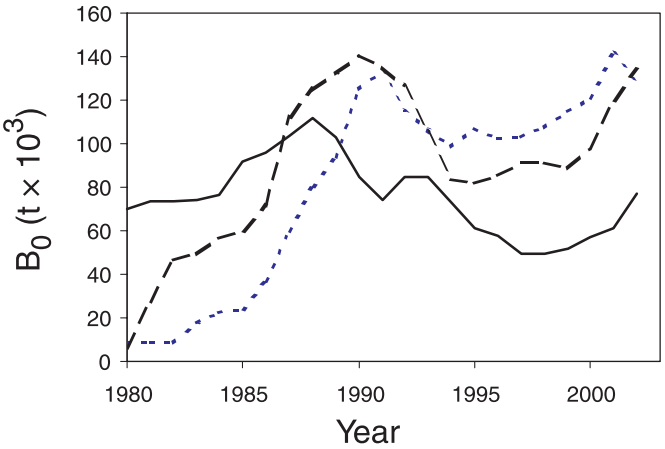
Fig. 10. Virgin biomass (B_0) calculated for three regions in British Columbia, 1980–2002: north coast (dotted line), west coast of Vancouver Island (broken line), and inside waters (solid line).



zation (1995) espoused a much broader view of the precautionary approach to fisheries management, emphasizing institutions, monitoring of the fishery, feedback on regulations, and effective implementation (Hilborn et al. 2001). Ultimately, the structural requisite for sustainability is a management system that offers appropriate incentives to managers, fishers, and other stakeholders. Those incentives take the form of tenure or long-term user rights. User rights are clearly defined in both fisheries, which should prevent the race for fish and overcapacity. The fisheries are not subsidized; instead, the fishery in Washington contributes significant revenues to the State, while in British Columbia, the industry covers the costs of sampling, surveying, and monitoring. Yet, there are significant challenges that still require continued attention, including conflicts between stakeholders, as exemplified by the restoration of rights to Native Americans or First Nations (Fangman et al. 1996), poaching, a problem common to all valuable species from coastal zones, and high-grading (low-quality dark geoducks left on the ground). The latter has been encouraged by the quota system in British Columbia and by uniform taxation of the catch in Washington. Enforcement alone has proved insufficient to deal with poaching and high-grading. These problems pertain mostly to the structure of the management system, institutional frameworks, tenure, and incentives.

The ultimate challenge for scientists and managers is the development of management strategies that rely on monitoring and well-designed indicators. The transition from a BRP-based strategy to one based on monitoring will demand a shift in research focus to the design of practical monitoring programs and the evaluation of management procedures by means of simulations. The simulation models need to represent the process of feedback as well as the dynamics of the resource and the fishing process, as regulated by management. The two different management systems in place in Washington and British Columbia have different monitoring demands: the first requires intensive surveying of tracts that are candidates to be opened and the second a protocol to assess stock status at larger spatial scales, with industry participation. In both cases, the strategies need to provide for timely feedback and thus be sensitive to the state of the resource at a range of time scales, which in the case of geoducks, is dictated by their extraordinary biology.

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Appendix A

In Washington, the biomass of each tract is assessed through direct transect surveys conducted by Washington Department of Fish and Wildlife biologists (Bradbury et al. 1998). Only a limited number of tracts can be surveyed each year. For the others, the most recent estimate is used, subtracting the catches when fishing occurred after the last survey. No adjustments are made to account for natural mortality or recruitment, either separately or pooled, as a recovery rate factor. For a tract to open, a survey must have occurred within the previous 10 years; otherwise, a new survey is required.

Stock assessment in British Columbia is described in detail in Hand et al. (1998). Catch and effort data were initially collected through a mandatory logbook program implemented in 1977, but inconsistencies were found with port validation. This led, after 1996, to a program (funded by the industry) in which port monitors collect harvest information from fishers at the time of landing. Since catch per unit of effort is not accepted as a reliable index of abundance, catch and effort data are not used in stock assessment. Virgin biomass is calculated on a bed-by-bed basis as the product of three factors: bed area (which is acknowledged to be affected by significant mapping errors), virgin geoduck density estimated through direct transect surveys of some beds and extrapolated to unsurveyed beds, and mean geoduck weight.

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