9 Current trends in the assessment and management of stocks

Manuel Barange, Miguel Bernal, Maria Cristina Cergole, Luis A. Cubillos, Georgi M. Daskalov, Carryn L. de Moor (formerly Cunningham), José A. A. De Oliveira, Mark Dickey-Collas, Daniel J. Gaughan, Kevin Hill, Larry D. Jacobson, Fritz W. Köster, Jacques Massé,Miguel Ñiquen, Hiroshi Nishida, Yoshioki Oozeki, Isabel Palomera, Suzana A. Saccardo, Alberto Santojanni, Rodolfo Serra, Stylianos Somarakis, Yorgos Stratoudakis, Andres Uriarte, Carl D. van der Lingen, and Akihiko Yatsu

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Summary

The assessment and management of small pelagic fish (SPF) stocks is particularly difficult and uncertain because their short life expectancy, characteristic aggregative behavior, rapid response to climate and environmental signals and large and variable natural mortality make them less tractable through traditional population dynamic models and assumptions. In this review we summarize the assessment and management approaches applied in 29 SPF stocks or management units (12 anchovy, 10 sardine, 4 herring, and 3 sprat). The review demonstrates that the assessment and management of SPF varies substantially in its approach and performance between stocks and regions. Most stocks have a scientific assessment program in place and a management approach that generally takes into account assessment results, but in some stocks management practices deviate substantially from scientific advice and in some, assessment and management processes are largely disconnected. It is concluded that only properly tailored scientific assessment and management programs can provide the speed of response and the flexibility of management that highly variable SPF demand. The most effective monitoring programs are based on fishery-independent surveys (daily egg production or/and hydroacoustics), while analyses based on catch per unit effort offer limited value. Most assessments, defined as what management uses to base its decisions on, rely on catch-at-age or yield per recruit models. Harvest strategies range from those driven by harvest control rules to those derived from outputs of best assessment runs. Some stocks use operating models based on age–structure model outputs or forward VPA[1]. On the issue of scientific uncertainty some practitioners propose reducing it through additional science and measures, while others promote the development of management procedures robust to uncertainty. This difference is particularly evident in relation to the value of recruitment forecasts. Other identified uncertainties include fishing versus natural mortality estimates and fleet catchability estimates. Regarding governance it is suggested that adaptive management practices applied by independent governance structures capable of interacting at ecological, social and economic levels need development for effective stewardship and governance. The review also addresses recent concerns over managing stocks that may be subject to productivity regimes or regime shifts, and whether two-level management strategies are required to address short- and

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long-term resource variability. The straddling nature of some stocks adds complexity to management procedures, and it is feared that this could be accentuated by climate-driven changes in stock distribution. Finally, SPF should be managed under ecosystem considerations, to protect their value as forage species to other fish, mammals and birds, and thus to respect the integrity of ecosystems. While the Ecosystem Approach is yet to be successfully applied to the management of any SPF, this may be the single most important driving force in influencing future assessment and management policies.

Introduction

The assessment and management of small pelagic fish stocks is considered particularly difficult and uncertain because these species do not conform to traditional population dynamics models and assumptions. Some of these invalid assumptions include that the size of the unexploited stock (B_∞ or carrying capacity), as well as the catchability coefficient (the probability of a fish being caught) remain constant, and the assumption that the effect of the environment on population parameters is either constant or generates a random noise (Csirke, 1988). Their particular size-selective shoaling behavior poses problems, not only in terms of catch and effort analysis but also in interpreting age and length frequency data from catches.

Small pelagic species are short lived, fast growing, and are characterized by high and often variable levels of natural mortality. As a result, their stock size is very dependent on incoming recruitments, and thus highly variable, unreliable and less responsive to management measures than other longer lived species. Beverton (1983) classified small pelagic fish populations as the most unreliable and vulnerable to unrestrained fishing, making their exploitation a high-risk activity. Small pelagic species have led to the development of very profitable fisheries, but these have suffered well-known collapses and have experienced partial and slow recoveries. Overall, they are fragile enterprises (Pitcher, 1995).

In recent decades we have experienced the development of new and advanced fisheries assessment methodologies that have contributed to more consistent and responsive fisheries management worldwide (although with limited success because of, among others, poor governance). These developments have also involved small pelagic fisheries. For example, the current support for the so-called Ecosystem Approach to marine resources (FAO, 2003) has been influenced by theories such as the possibility of species replacements in the pelagic marine ecosystem (Lluch-Belda *et al.*, 1992), and the idea that the variability of groups of small pelagic fish stocks may be smaller than individual stocks. This chapter intends to describe the state of the art in the assessment and management of small pelagic fisheries by reviewing current practices in 29 stocks of anchovy, sardine, herring, and sprat, in search of commonalities, successful approaches, and future requirements.

Stock structure, population trends and fishery information

This chapter synthesizes information from 12 anchovy, ten sardine, three sprats, and four herring stocks worldwide (Fig. 9.1). The stock structure, population trends and fishery information are introduced individually, while assessment and management procedures are described and discussed together.

Table 9.1 summarizes information regarding biomass and catch statistics, distribution patterns, and population parameters for the different stocks. Anchovy stocks yield catches between 1200 t (Bay of Biscay) and 7.5 Mt[2] (Central-South Peru), corresponding to biomass levels between 9000 t and 10 Mt (Table 9.1a). Maximum anchovy length is generally 15–19 cm, corresponding to a maximum age of 2–5 years. Spawning generally takes place in spring–summer (but winter in the case of the Peru–Chile anchovy), and 1 year olds (y.o.) are active spawners (Table 9.1a). Sardine stocks yield catches presently between 4500 t (southern Western Australia) and 350 000 t (Chile), corresponding to biomass levels between 102 000 t (Western Australia) and 1.7 Mt (Chile) (Table 9.1b). Maximum sardine lengths range between 23 and 40 cm, corresponding to 4–10-year-old fish. Spawning is generally year round, many stocks displaying a winter spawning peak (except Brazil, California, and Benguela). Some 1 y.o. fish are reproductively active, but in many stocks the age of first spawning is delayed to ages 2 and 3 (Table 9.1b). Herrings and sprats range in catches between the 30 000 t yield by the Gulf of Riga herring to the 1 Mt of the Arcto-Norwegian spring spawning herring, corresponding to biomasses of 165 000 t and 10 Mt, respectively (Table 9.1c). Maximum fish lengths are 30–40 cm in the case of herrings and 15–18 cm for sprats. Of the examples provided, many stocks spawn in spring and autumn, and the age of first spawning is generally 2–4 y.o. (except for the Black Sea sprat, Table 9.1c).

Anchovy stocks

*Japanese Anchovy (*Engraulis japonicus*) – NW Pacific stock*

Three stocks of Japanese anchovy are distributed around Japan: the NW Pacific stock, the Seto Inland Sea stock and the Tsushima Current stock (Kono and Zenitani, 2005; Ohshimo, 2005; Oozeki *et al.*, 2005). These three stocks are distinguished by distribution and migration patterns. The NW Pacific stock accounts for more than 75% of the Japanese anchovy landings of Japan in recent years, and is the focus of this review. Spawning grounds

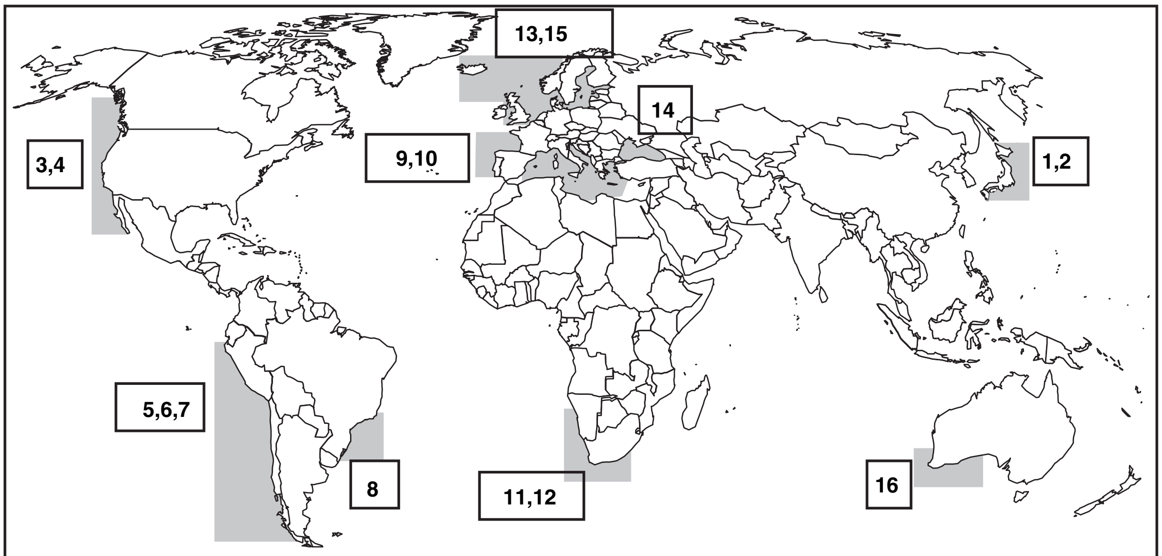

Fig. 9.1. Schematic representation of the location of the 29 stocks reviewed.

are confined between the Kuroshio Current and the coast in the southern Japan and extended offshore as east as 170º longitude in the northern Japan (Oozeki *et al.*, 2005). Spawning takes place mainly from February to September. Juveniles are transported in spring by the Kuroshio and Kuroshio Extension Currents to beyond 170º E longitude. The feeding grounds are mainly located in the Oyashio and Kuroshio/Oyashio Transition Zone in summer and autumn. Major fishing grounds of the purse-seine fishery are located along the Pacific coast of northern Japan. Biological minimum size is 8 cm in standard length, which corresponds to age 1, although half of the age 0 group may mature in favorable conditions. Estimated stock biomass was less than 500 000 t until 1988 but has increased since to reach 1.16 Mt in 2004 (Table 9.1a, Fig. 9.2a). Recruitment of the Japanese anchovy was high from 1997 (Fig. 9.2a). Japanese anchovy are landed mainly by purse seiners and also their larvae and juveniles are landed as “shirasu” (white children) by purse nets around the Pacific coast of Japan. Fishing grounds concentrated only in the coastal area, although the distribution of the Japanese anchovy expanded from the coast of Japan to 160° E as their population size increased. Annual landings have increased from 1990 off the northern Pacific coast of Japan and they now exceed 300 000 t (Table 9.1a). Exploitation rates in recent years were estimated to be around 30%. Although mainly young-of-the-year were historically landed, substantial proportions of age 1 and age 2 fish have been landed since 1989 and 1990, respectively.

California anchovy[3] *(*Engraulis mordax*) – central subpopulation*

The central subpopulation of the California anchovy is distributed along the coast from central Baja California, Mexico (30° N) north to central California (37° N) in the United States (PFMC, 1998). It has a preference for SST in the range 12–21.5 °C. The stock is sedentary relative to other small pelagic stocks. Large individuals venture furthest north and offshore (Parrish *et al.,* 1989) and the stock moves north during El Niño events (Pearcy *et al.*, 1985). Anchovy rarely live to more than 4 years of age, although specimens as old as 7 years old have been

Table 9.1a. *Stock distribution and structure, life history, biomass and population variables: anchovy stocks*

Species	Stock	Biomass and Catch				Distribution		Population variables			
		Max biomass (t)	Present biomass (t)	Max catch (t)	Present catch (t)	Adult distribution	Juveniles distribution	Max length	Max age	Spawning season	Age 1st spawning
Japanese anchovy (*Engraulis japonicus*)	NW Pacific	1.48 M (2003)	1.24 M (2004)	415 437 (2003)	401 158 (2004)	30°–45° N / 130°–180° E	30°–42° N / 130°–160° E	17 cm	4 y.o.	February–September	1 y.o.
	Seto Inland	0.53 M (1985)	0.23 M (2004)	149 953 (1985)	66 000 (2004)	33°–35° N / 131°–136° E	33°–35° N / 131°–136° E	14 cm	2 y.o.	May–September (year round)	1 y.o.
	Tsushima Current	3.80 M (1998)	0.56 M (2004)	1.75 M (1998)[4]	0.26 M (2004)[5]	20° –45° N / 120°–140° E	? –40° N / ? –140° E	15cm	2 y.o.	March–January (year round)	0.5 y.o.
California anchovy (*Engraulis mordax*)	Central sub- population	1.6 M (1974)	0.39 M (1995)	315 000 (1981)	6 900 (2004)	Central Baja California (30° N)–central California (37° N)	Same	23 cm *TL* but typically < 18 cm *TL*	7 y but typically < 4 y	Year round Peaks February–April	1–2 y.o.
Humboldt anchovy (*Engraulis ringens*)	North-Central Peru	22.8 M (1970)	10.5 M (2005)	10.9 M (1970)	7.5 M (2005)	4° S–14°59'S	4° S–14°59'S	20 cm	4 y.o.	Aug/Sep (main) Feb–Mar (secondary)	1 y.o.
	South Peru–North Chile	10 M (2001)	5.7 M (2003)	2.75 M (1994)	2.3 M (2003)	15°–24° S	Same (inshore)	20 cm	5 y.o.	August/October	1 y.o.
Benguela anchovy (*Engraulis encrasicolus*)	Southern (South African)	7.93 M (2001)	3.06 M (2005)	596 000 (1987)	283 000 (2005)	32°–36.5° S / 17°–27° E Inshore to depths >200 m	29°–36° S / 17°–22° E Inshore	16 cm	4 y.o.	October– March	1 y.o.

European anchovy (*Engraulis encrasicolus*)	Bay of Biscay	93 000 (1998)	9200 (2005)	83 600 (1965) 40 500 (2001)	1200 (2005)	South of 47°N and East of 5°W (continental shelf and shelf edge	South of 47° N and East of 5° W	18.5 cm	4 y.o.	April to August	1 y.o.
	NW Mediterranean	GL: 112 000 (2001) CS: 27 000 (2003)	GL:18 000(2004) CS: 26700 (2004) [6]	25 030 (1994)	9379 (2004)	38.5°–43.5°N/0° – 6.5° E	38.5°–43.5° N/0° – 6.5° E	19 cm	4 y.o.	April– October	1 y.o.
	Adriatic Sea	350 000 (1978) VPA	114 000 (2004) VPA	60 000 (1980)	30 000 (2004)	10–200 m	Inshore	19 cm	6 y.o.	March–October	< 1 y.o.
	Aegean Sea	48 000 (2003)[7]	46 000 (2004)[7]	27 227 (1998)	22 007 (2003)[8]	37.5° –41° N/ 22.5° –27° E 10–200 m	37.5° –41° N/ 22.5° –27° E inshore	18 cm	4 y.o.	May–September	1 y.o.
	Black Sea	708 000 (1979)	387 556 (2001)	468 807 (1988)	322 711 (2004)	All Black sea – maximum over shelf	All Black sea – max. in shelf	16 cm	5 y.o.	Summer (Jun–Aug)	1 year

Notes:
[4] Including Korea and China.
[5] Including Korea.
[6] Acoustics estimates.
[7] Acoustics in Greek waters.
[8] Greece and Turkey.

Table 9.1b. *Stock distribution and structure, life history, biomass and population variables: sardine stocks*

Species	Stock	Biomass and catch				Distribution		Population variables			
		Max biomass (t)	Present biomass (t)	Max catch (t)	Present catch (t)	Adult distribution	Juvenile distribution	Max length	Max age	Spawning season	Age 1st spawning
Japanese sardine (*Sardinops melanostictus*)	Pacific	19.5 M (1987)	0.1 M (2004)	2.9 M (1987)	48 000 (2004)	29°–54° N / 130° E–160° W (high-stock period), shrink in low-stock period	29°–38° N / 130° E–160° W (high-stock period)	23 cm	7 y.o.	October–May	1 y.o. (high-stock period), 3 y.o. (low-stock period)
California sardine (*Sardinops sagax caerulea*)	Northern sub-population	3.63 M (1934)	1.06 M (2005)	718 000 (1936)	135 000 (2004)	29°–59° N 115°–136° W Over depths 0–50 m	29°–59° N 115°–136° W Inshore 0–30 m	41 cm	14 y.o.	April–August	1 y.o.
Humboldt sardine (*Sardinops sagax*)	North-Central Peru	11.8 Mt (1984)	0.2 Mt (2000)	3.4 Mt (1988)	0.001 M (2003)	4° S–14°59'S	4° S–14°59'S	39 cm	8 y.o.	Aug/Sep (Main) Jan–Feb (Secondary)	3–4 y.o.
	South Peru–North Chile	9.1 M (1980)	<0.1 M (1996)	3.0 M (1985)	2201 (2003)	15°–24° S	Same (inshore)	40 cm	11 y.o.	Main: August/ September Secondary: February/ March	5 y.o.
Humboldt common sardine (*Strangomera bentincki*)	Central-South Chile	3.09 M (1995/96)	1.65 M (2003/04)	693 833 (1998/ 99)	353 952 (2003/04)	34° S–40° S	34° S–40° S	20 cm	4 y.o.	August/ September	1 y.o.
Brazilian sardine (*Sardinella brasiliensis*)	Brazilian Southeast Bight	1.2 M (1977)	0.5 M (2007)	228 000 (1973)	17 000 (2000)	22°–29° S Over depths <100 m	23°–26° S inshore	27 cm	3.8 y.o.	October–March	1 y.o.
Benguela sardine (*Sardinops sagax*)	Southern (South African)	4.14 M (2002)	0.96 M (2005)	410 000 (1962)	247 000 (2005)	32°–36.5° S / 17°–32° E Over depths 10–500 m	29°–36° S / 17°–32° S Inshore	25 cm	10 y.o.	Year round but mainly August–March	1–3 y.o.
	Northern (Namibian)	11.14 M (1964)	0.27 M (2005)	1.40 M (1968)	25 300 (2005)	15°–25° S / 13°–15° E Over depths 10–200 m	15°–25° S / 13°–15° E Inshore	25 cm	10 y.o.	Year round but mainly August–March	1–3 y.o.
European sardine (*Sardina pilchardus*)	Atlanto-Iberian	0.65 M (2001)	0.4 M (2004)	250 000 (1964)	95 000 (2004)	36°–44° N / 1°–9° W below 100 m	39°–41° N (north Portugal) 6°–7° W (Gulf of Cadiz) inshore	25 cm	10 y.o.	October–April	1–2 y.o.
Australian sardine (*Sardinops sagax neopilchardus*)	Southern Western Australia	0.14 M (1990)	0.10 M	8000 (1988 from one region)	4500 (=TAC, only 1800 tonnes landed)	Part of a continuous distribution around southern half of Australia	Same (inshore)	23 cm	9 y.o.	Summer – winter, variable peak	2 y.o.

Table 9.1c. *Stock distribution and structure, life history, biomass and population variables: herrings and sprats*

Species	Stock	Biomass and catch				Distribution		Population variables			
		Max biomass (t)	Present biomass (t)	Max catch (t)	Present catch (t)	Adult distribution	Juveniles distribution	Max length	Max age	Spawning season	Age 1st spawning
European sprat (*Sprattus sprattus*)	Black Sea	0.58 M (1975)	0.58 M (1999)	0.10 M (1989)	49 446 (2004)	All Black sea with maxima in shelf waters and the northwestern part	All Black sea Upper water layer	14.5cm	5 y.o.	All year with max in Nov–Mar	1 year old
	Baltic Sea	3.1 M (1995)	2.0 M (2005)	0.53 M (1997)	405 000 (2005)	54°–63° N / 10°–29° E	54°–63° N / 10°–29° E depends on drift	16 cm	>10 y.o.	March–June	2 y.o.
	North Sea	n/a	n/a	0.36 M (1995)	0.21 M (2005)	North Sea and English Channel 51–60° N 0–12° E	E.N. Sea and Kattegat, more coastal 51–58° N 0–12° E	18 cm	10 y.o.	February to September	2 y.o.
Atlantic herring (*Clupea harengus)*	North Sea autumn spawning stock	2.18 M (1963)	1.70 M (2005)	1.17 M (1965)	0.66 M (2005)	North Sea and English Channel 51–62° N 0–10° E	E. N. Sea and Kattegat, more coastal 51–60° N 0–12° E	38–39 cm	17–20 y.o.	July to January	2–3 y.o.
	Arcto-Norwegian spring spawning stock	16.2 M (1950)	10.1 M (2005)	1.96 M (1969)	1.0 M (2005)	Norwegian Sea, NE Atlantic 62–78° N 10° W–20° E	Norwegian Sea, Barents Sea 62–79° N 10° E–35° E	42 cm	20 y.o.	March to April	4–5 y.o.
	Central Baltic	3.05 M (1974)	0.93 M (2005)	0.37 M (1974)	92 000 (2005)	54°–63° N / 15°–29° E	54°–63° N / 15°–29° E coastal (spring spawner)	35 cm	>10 y.o.	March–April (spring spawner) Sept.–Oct. (autum spawner)	2–3 y.o.
	Gulf of Riga	0.17 M (2001)	0.16 M (2005)	41 000 (2003)	32 000 (2005)	57°–58.5° N / 22°–24° E	57°–58.5° N / 22°–24° E	30 cm	>10 y.o.	March–April	2 y.o.

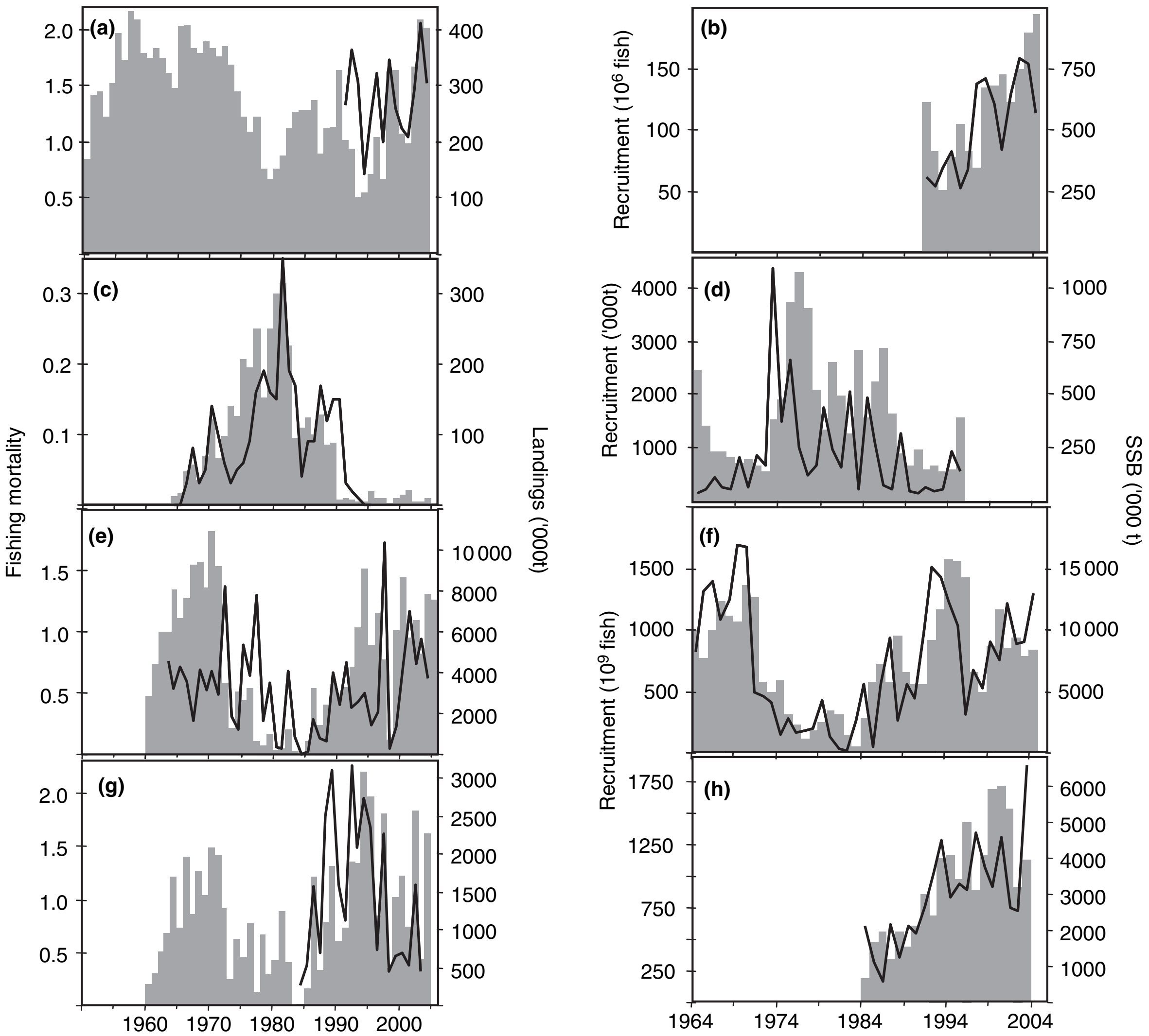

Fig. 9.2a. Time series of fishing mortality (y^{-1}), landings ('000 t), recruitment and spawning stock biomass (SSB, '000 t) for Japanese anchovy (a, b), California anchovy (c, d), and Humboldt anchovy (Central Peru stock, e, f; South Peru–North Chile stock, g, h). Solid lines: fishing mortality and recruitment; grey bars: landings and spawning stock biomass.

recorded (Table 9.1a). Spawning is opportunistic and occurs year round in the main spawning areas off northern Mexico and the southern United States, with peaks during February–April. Maturity is reached by age 2 and the proportion mature at age 1 depends on water temperatures (Methot, 1989; Table 9.1a).

Annual landings in recent years indicated the highest level during the last 25 years (Fig. 9.2a). Anchovy biomass (ages 1+) averaged 326 000 t during 1964–1970, increased rapidly to 1.6 Mt in 1974, and then declined to range 153–392 000 t during 1991–1995 (Jacobson *et al.*, 1995; Table 9.1a). No information is available after 1995. Total catches in the US and Mexico peaked at about 315 000 t during 1981 and then declined to about 10 000 t tonnes during 1991–2005 (PFMC, 2005). Relatively low catch levels since the 1990s have been due to poor market conditions. Anchovy are harvested mainly by purse seines in US waters off California and in Mexican waters off Baja California. They are landed and sold for reduction to fish meal, human consumption, live and dead bait used in recreational fisheries, and as an ingredient in pet food. Live bait, dead bait, and pet food are the most important and economically important uses.

Humboldt anchovy[4] *(*Engraulis ringens*)*

In the Humboldt Current System anchovy is distributed from Zorritos (4°30 S) in northern Peru to about Chiloé (42°30 S) in southern Chile. Three main discrete stocks can be identified (Serra, 1983; Alheit and Ñiquen, 2004). The most productive is located in north-central Peru (4°–15° S), followed by the Peruvian–Chilean stock (16°–24° S), and the most austral and less abundant located off–central–southern coast of Chile (33°–42° S; not considered in this review). The separation between the three stocks is supported by meristic, morphometric, spawning grounds and/or tagging studies (Serra, 1983; Pauly and Tsukayama, 1987; Mendo, 1991). Anchovy is restricted to 100–120 nm offshore, but in spring and summer is limited to 30 nm offshore. Fishing takes place within 60 nm from the coast by an industrial fleet of purse seiners.

Anchovy is a short-lived species with maximum longevity and length of 4 years and 20 cm in total length, respectively. Natural mortality is 0.8–1.0 y^{-1} (Csirke *et al.,* 1994).

For the North-Central Peru (N/C Peru) stock the main spawning areas are between 8–9.5° S and 12–14° S, while for the Peruvian–Chilean (SP/NCh) stock the main spawning area extends from about 17º S to 23º S (Oliva *et al*., 2001; Braun *et al*., 2005). Anchovy spawn almost all year round but the main spawning seasons are between August and September (winter) with a second spawning in late summer (February–March) (Table 9.1a). Spawning is in batches and the population fecundity is indeterminate. Length at first maturity occurs at 11.5 cm in winter and 12.5 cm total length in summer (Simpson and Gil, 1967). It is assumed that all fish at age 1 are mature. The recruitment to the fishery tends to occur from mid November to April (late spring and summer), with a second, less important recruitment period in winter.

Peruvian anchovy fluctuates dramatically in relation to the El Niño phenomenon. In the N/C Peru stock estimated biomass dropped from an average of 14 Mt in 1967–1971 to 4 Mt in 1972 (Tsukayama, 1983), and took years to recover. However, the stock recovered well after the El Niños of 1982–83 and 1998 (Fig. 9.2a). Acoustic estimates agree with monthly VPA analysis (Pauly and Palomares, 1989). Catches have fluctuated with biomass, peaking at 12 Mt in 1970. Current catches are around 7–8 Mt. For the SP/NCh stock the biomass is estimated by stock assessment models. Biomass increased since 1984 and after 1992 it has fluctuated between 6 and 10 Mt. Similar trends were followed by catches, but important variations after 1993 without correlation with the stock biomass are observed (Fig. 9.2a). The catch history of this stock shows coherent and synchronous long-term changes alternating with sardine (*S. sagax*) catches.

Biological information for the assessment of the N/C Peru stock is obtained at all landing sites and, since 1964, through the EUREKA project (Villanueva, 1970). This consists of using the commercial fleet to obtain biological and fishery information in real time. In addition all vessels are equipped with VMS[5], and geo-referenced data are used in the management. From 1965 the assessment consists mainly of spatially separated CPUE from the industrial fleet (from 1965) and fishery independent surveys. For the assessment of the SP/NCh stock the time series from 1984 to 2003 is considered. Catch-at-age matrices from the Chilean and the southern Peru fisheries are computed monthly and grouped by year, and then added to obtain an overall annual matrix for the combined stock. Weight-at-age was obtained by converting the mean length-at-age to weight by an overall length–weight relationship (Serra *et al.,* 2004; GTE, 2003).

*Benguela anchovy (*Engraulis encrasicolus*) – southern Benguela*

Benguela anchovy occur from northern Namibia (around 16° S) to South Africa's east coast (around 27° E), but within this range are thought to comprise two distinct stocks separated by a permanent zone of intense coastal upwelling located at Lüderitz (26° S). Anchovy has been the dominant component of catches made off South Africa but is less important to the Namibian fishery, particularly over the past decade (van der Lingen *et al*., 2006a), hence only southern Benguela anchovy are discussed here.

Spawning of southern Benguela anchovy occurs over the Agulhas Bank (south of the African continent), peaking between October and December (van der Lingen and Huggett, 2003). Historically, spawning was concentrated on the western Agulhas Bank, but in recent years spawners have been concentrated to the east of Cape Agulhas (van der; Roy *et al*., 2007; Lingen *et al*., 2002). Eggs and larvae are transported from the south coast spawning grounds to the west coast nursery areas by a shelf-edge jet current, and juvenile fish undergo a return migration to reach the spawning grounds at an age of around 1 year (Hutchings *et al.,* 1998), by which time they are sexually mature. Anchovy has a maximal recorded age of 4 years (Melo, 1984; Table 9.1a).

The South African pelagic fishery began to target anchovy in the early 1960s following the drastic decline in sardine catches. Anchovy catches increased steadily, peaking around 600 000 t in 1987 and 1988 (De Oliveira, 2003; Fig. 9.2b). Landings decreased to a minimum of 40 000 t in 1996, and then increased sharply to about 287 000 t in 2001. All catches are taken by purse seiners and reduced for fish meal. The fishery commences in January, and operates sometimes well into October/November, with over 80% (by mass and number) of the catch each year comprising juvenile fish of around 6 months old (De Oliveira, 2003). Anchovy spawner biomass was initially estimated using VPA, which is considered to have substantially underestimated stock size; more recent monitoring via acoustic and DEPM[6] surveys has shown large fluctuations in recruitment and spawner biomass, with record levels over the period 2000–2003 (Fig. 9.2b).

As annual age-length keys are not available for the commercial anchovy landings, assumptions about the age structure of the catch are made based on the life-history

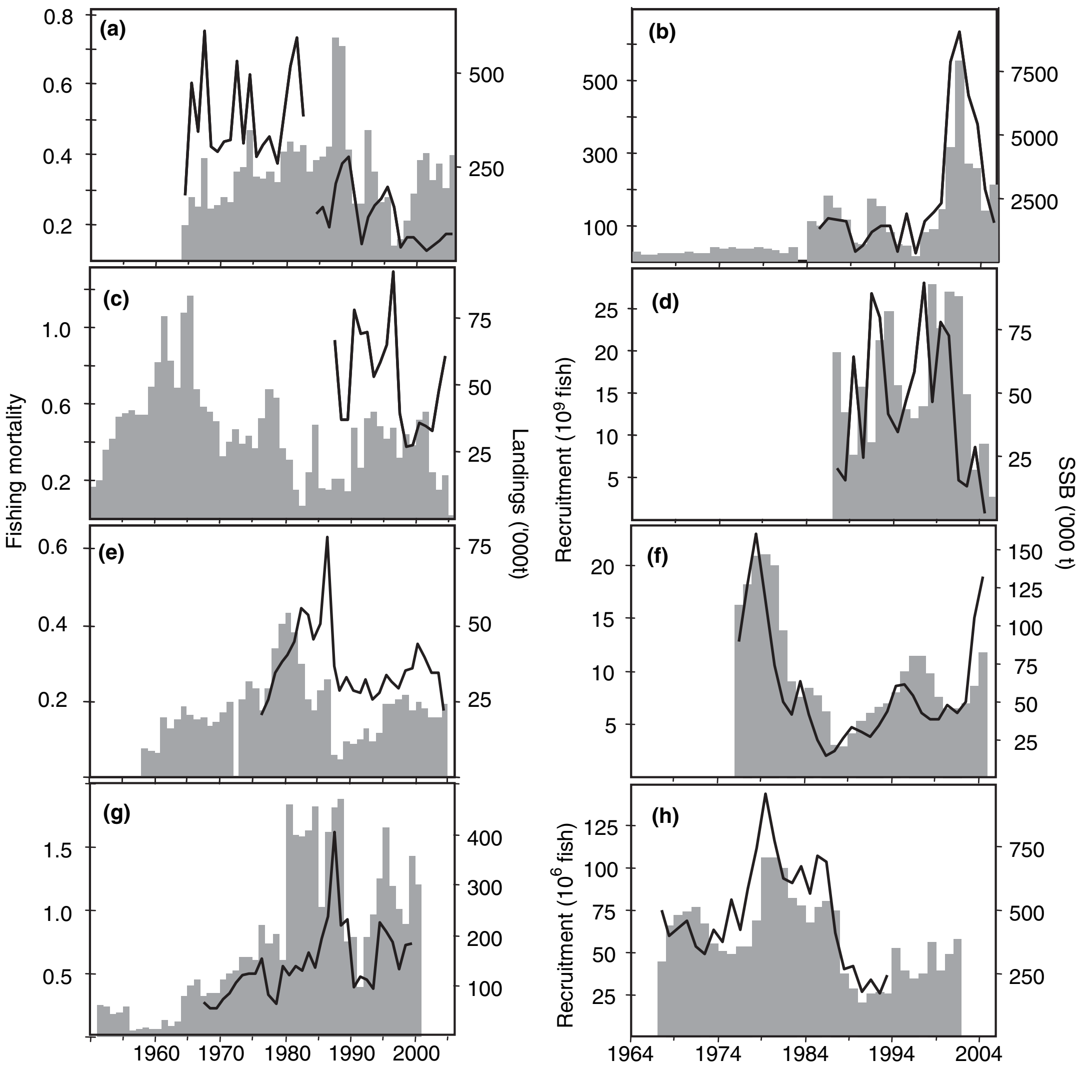

Fig. 9.2b. Time series of fishing mortality (y^{-1}), landings ('000 t), recruitment and spawning stock biomass (SSB, '000 t) for southern Benguela anchovy (a, b), Bay of Biscay anchovy (c, d), Adriatic Sea anchovy (e, f) and Black Sea anchovy (g, h). Notation as in Fig. 9.2a.

characteristics of the stock and general fishing patterns (De Oliveira, 2003). It is assumed that all anchovy landed between November (when spawning is modeled to occur) and March are 1 year olds, and those landed during the remainder of the year are recruits. Corresponding mean masses-at-age are available. Two-year-old and older fish hardly appear in the catch and are not taken into account in the assessment.

European anchovy (Engraulis encrasicolus) –
Bay of Biscay

The European anchovy occurs from the Bay of Cadiz (Southern Spain) in the south to the North Sea and the western Baltic Sea in the north (Reid, 1966; Beare *et al.*, 2004). Anchovy in the Bay of Biscay is considered to be a stock isolated from the small populations either to the north or to the south. Although there is evidence for some heterogeneity inside the Bay, the evident interconnection of the fisheries and homogenous recruitment led ICES to consider the anchovy in the Bay of Biscay as a single unit for assessment and management (ICES, 2006a). Spawning takes place in spring, mainly in the south-east corner, particularly in front of the Adour and Gironde estuaries. Fish mature at age 1, and age-groups 1 and 2 constitute the bulk of the catches and the population. Occasionally, fish of 4 or even 5 years old are recorded, but these are very rare. The anchovy population undergoes seasonal migrations, moving northwards in summer and autumn, after spawning, being matched by the different fisheries in the Bay (Uriarte *et al.*, 1996).

Landings show a high degree of inter-decadal variability, with catches peaking from the mid 1950s to the early 1970s (peak of 83 600 t in 1965, Fig. 9.2b, Table 9.1a). Catches progressively diminished since, followed by a similar decline in the Spanish purse-seine fishery. In the late 1980s French pelagic trawlers entered the fishery and now catch as much as the Spanish purse-seine fleet. During the 1990s catches reached 40 000t, but recently these have declined to historical minima until the recent fishery collapse and subsequent regime of closure periods (ICES, 2006a; Fig. 9.2b). Since 1989, the stock is annually assessed by ICES, surveyed by Spain and France.

The resource is shared between Spain and France: the Spanish fleet is composed of about 211 purse seines that operate in the SE corner of the bay mainly in spring during the anchovy's spawning period. The main anchovy catches from the French fleet are taken in the central-east part of the bay in the first half of the year and the NE during the second half. The fleet consists of 20–30 pair trawlers and about 30 purse seines. No standardization of effort or any series of CPUE is available for the fishery as they are considered unreliable indicators of abundance for small pelagic fishes (Ulltang, 1980; Csirke, 1988; Pitcher, 1995; Mackinson *et al.,* 1997). Total landings are being reported by France and Spain to ICES. Catches are monitored by national institutes for length and age composition. Catches at age are inferred by otolith sampling and serve as input for the assessment. Since 1989, the stock is annually assessed by ICES, supported by the direct monitoring of the resource and the fishery made by Spain and France.

Mediterranean Sea

The European anchovy is the most important pelagic fish resource in the Mediterranean (Lleonart and Maynou, 2002). Three major genetically distinct (Magoulas *et al.*, 2006) stocks exist, with reduced gene exchanges: the NW Mediterranean (Catalan Sea and Gulf of Lions) stock, the Adriatic Sea stock and the northern Aegean Sea stock (Somarakis *et al.*, 2004). Other areas in the Mediterranean are inhabited by smaller and highly fluctuating stocks.

In the NW Mediterranean, anchovy forms a genetically panmictic stock which is distributed and reproduces over the continental shelf areas associated with the runoff from the Ebre river in the Catalan Sea (CS) and the Rhône river in the Gulf of Lions (GL). In the CS, spawning takes place from late April to September, but the reproductive period is shorter in the GL (May–August) (Palomera, 1992). Fish mature at a length of about 11 cm (Palomera *et al.*, 2003), i.e. at the first year of life (Table 9.1a). Maximum length recorded is 19 cm and maximum age, 4 years (Pertierra and Lleonart, 1996; Torres *et al.*, 2004).

In the NW Mediterranean the stock is mainly caught by the Spanish fleet of purse seiners and bottom trawlers and to a lesser extent by a French fleet of mid-water trawlers operating in the GL. Catches show a decreasing trend in the CS from the early 1990s to present (from about 20 000 to 5000 t), while in the GL, catches increase from the mid 1980s to early 1990s and fluctuate thereafter from 1500 to 10 000 t. Mean biomass values are presently at around 15 000 t in the Catalan Sea and 50 000 t in the GL (Table 9.1a).

In the Adriatic Sea, anchovy is distributed all over the northern and central parts, influenced by river runoff (Russo and Artegiani, 1996). Spawning takes place mostly in the western part of the Adriatic, from March to October with a peak in June–July (Regner, 1996). Fish mature at 8 cm, i.e. at less than one year old (Rampa *et al.*, 2005) and young-of-the-year recruit to the fishery during spring. Most individuals fished belong to age classes 0, 1 and 2 (Cingolani *et al.*, 1996) but specimens up to 6 years old have been recorded (Table 9.1a). Most of the catches are taken by the Italian fleet of pelagic trawlers and purse seiners. Biomass was at its highest levels in the late seventies (more than 300 000 t; Santojanni *et al.*, 2003) and is presently estimated to be over 100 000 t (Cingolani *et al.*, 2005; Table 9.1a). Catches reached a peak of about 60 000 t in 1980, decreased steadily thereafter, and now fluctuate between 20 000 and 30 000 t (Fig. 9.2b).

In the Aegean Sea, anchovy is mainly distributed and spawns over continental shelf areas of the northern part of the basin (Somarakis *et al.*, 2005a), influenced by the Black Sea water advection and discharge from large rivers (Somarakis *et al.*, 2002a; Isari *et al.*, 2006). Spawning takes place during late spring/summer (usually from May to September) when the stock is also mainly fished by the purse-seine fleet (Stergiou *et al.*, 1997). Fish mature at a length of about 10.5 cm (Somarakis *et al.*, 2005b), i.e. at the first year of life. Maximum length recorded is 18 cm and maximum age, 4 years (Nikoloudakis *et al.*, 2000; Table 9.1a). The combined Greek–Turkish catch showed an increasing trend from the early 1970s to mid 1980s and fluctuates thereafter between about 15 000 t and 25 000 t. Biomass estimates indicate a stock of about 40 000–50 000 t.

The NW Mediterranean and the Aegean Sea anchovy stocks are not currently being assessed. However, catch records have been collected by the National and Regional Statistical Services of Spain since the early 1940s, France since the early 1970s and Greece since the early 1960s (Stergiou *et al.*, 1997). Additional data on catch-per-day of the involved purse-seine fleet are available in the central database of the Hellenic Center for Marine Research since 1995 (Kapadagakis *et al.*, 2001). Since 2002 the framework of the National Programs for Fisheries Data collection (co-funded by the EC), collects length and age information on a regular basis in both the NW Mediterranean and the Aegean Sea, in order to derive catch-at-age estimates. The first attempt to assess the stock of anchovy in the Catalan Sea has been presented to the General Fisheries Council for the Mediterranean (GFCM; Torres *et al*., 2004).

In contrast to the NW Mediterranean and the Aegean, data on catch, fishing effort and age composition have been routinely collected in the northern and central Adriatic since 1975 (Fig. 9.2b). The activity involves the estimation of length frequency distribution of catches, the ageing of a subsample to obtain catch-at-age numbers and the standardisation of the fishing effort for certain ports using Generalized Linear Models (Santojanni *et al*., 2005).

Black Sea anchovy

Black Sea anchovy is distributed throughout the Black sea but is subject to seasonal migrations. In October–November it moves to the wintering grounds along the Anatolian and Caucasian coasts in the southern Black Sea. In these areas it forms dense wintering concentrations in November–March, which are subject to intensive commercial fishery. The rest of the year it occupies its usual spawning and feeding habitats across the sea with some preference for the shelf areas and the northwestern part of the sea. This area is characterized by the largest shelf area and high productivity due to abundant river runoff (Faschuk *et al*., 1995; Daskalov, 1999).

Two subpopulations of anchovy exist in the Black Sea: the Black Sea and the Azov Sea stocks (Ivanov and Beverton, 1985). The latter reproduces and feeds in the Azov Sea and hibernates along the northern Caucasian and Crimean coast of the Black Sea. The Black Sea stock is of bigger ecological and commercial importance and the information below concerns only this stock.

Anchovy first spawns at about age 1 but precocious maturation and spawning (at age 2–3 months) has been reported in years of stock collapse (Mikhailov and Prodanov, 2002; Table 9.1a). It spawns in the surface layer of warm and stratified areas in summer, the main feeding season (Arkhipov, 1993; Fashchuk *et al.,* 1995). A large convergence zone is formed on the northwestern and the western shelf (the main anchovy spawning area) due to the river Danube inflow, which favors offspring retention.

Anchovy is the object of both an artisanal (with coastal trapnets and beach seines) and commercial purse-seine fishery on the wintering grounds. Decadal fluctuations of abundance are observed and likely to be related to changes in climate (Daskalov, 1999, 2003; Fig. 9.2b). The increasing trend in biomass started in the 1970s and 1980s promoted the expansion of powerful purse-seine fishing fleet and a steady increase in fishing effort (Gucu, 1997). Maxima of catch and fishing mortality were recorded in the late 1980s parallel to the decrease in exploited biomass following recruitment failures in the previous years. Sharp reductions in biomass and catch in the early 1990s were described as stock collapses. In the recent decade the stock partially recovered and catches reached levels of 300 000–400 000 t (Fig. 9.2b, Table 9.1a).

Sardine stocks

*Japanese sardine (*Sardinops melanostictus*) – Pacific stock*

The Japanese sardine fishery targets two stocks, the Pacific stock and the Tsushima Current stock, which are distinguished by distribution and migration patterns (Nishida *et al*., 2006). The Pacific stock, the focus of this review, is distributed along the Pacific coast of Japan; its eastern boundary extends to 160º W during periods of high abundance but in periods of low abundance adults are confined to west of ca. 155º E longitude (Yatsu *et al*., 2003; Yatsu and Kaeriyama, 2005). The western boundary coincides with the spawning ground which at high stock abundance extends from the southern coast of Kyushu Island to the offshore area beyond the Kuroshio axis and to northern Honshu Island (Zenitani and Yamada, 2000).

Spawning takes place from October to May and juveniles are transported in spring by the Kuroshio and Kuroshio Extension Currents to beyond 170º E longitude (Watanabe and Nishida, 2002). The feeding grounds are mainly located in the Oyashio and Kuroshio/Oyashio Transition Zone (KOTZ) in summer and autumn. Major fishing grounds of the purse-seine fishery are located along the Pacific coast of northern Japan. Longevity is about 7 years (Nishida *et al*., 2006). Sexual maturation is in general attained at age 2 (Table 9.1b).

Japanese catch statistics indicate two historic peaks in the 1930s and 1980s (Noto and Yasuda, 2003; Fig. 9.3a). Recruitment increased in the early 1970s and peaked in the mid 1980s, while SSB[7] lagged according to the age of maturity. Consecutive recruitment failures were observed during 1988–1991 when SSB achieved its historical peak (Watanabe *et al*., 1995). The current catch level (48 000 t in 2004) approaches the historic minimum (7000 t in 1965). Purse seine is the major gear used for catching Japanese pelagic fish, including sardine, chub mackerel, anchovy, and jack mackerel, which are targeted alternately according to availability and market demand (Nishida *et al*., 2006; Yatsu *et al*., 2005). During the 1970s and early 1980s, the purse-seine fleets expanded their fishing effort to compensate for the declined global supply of fish meal (Yatsu, 2005). Because of the lower commercial value of anchovy and the cost of adopting anchovy-mesh nets, the purse-seine fishery has been adapted to targeting age-0 and age-1 chub mackerel and sardine since the early 1990s, when chub mackerel and anchovy began to increase (Yatsu *et al*., 2003). In recent years continued intensive Japanese sardine fishing mortality has prevented recovery of sardine despite good recruitment per spawner (RPS) biomass (Yatsu and Kaeriyama, 2005).

*California sardine (*Sardinops sagax caerulea*)*
California sardine comprise three groups: a "northern" (northern Baja California to Alaska), a "southern" (outer Baja California peninsula to southern California), and a "Gulf of California" subpopulation (for review see Smith, 2005). Vrooman (1964) proposed the three subpopulations based on serological evidence, supported by temperature at capture and otolith morphometrics (Felix-Uraga *et al.*, 2004, 2005). Geographic ranges of the three subpopulations partially overlap seasonally, but the degrees of mixing and relative contributions to productivity remain undetermined. The northern stock likely grows larger and lives longer than its counterparts. When the population is large, California sardine are abundant from the Gulf of California to southeastern Alaska. When abundance is low, sardine do not occur north of 34.5° N (southern California). Spawning typically occurs from 50 to 200 n.m offshore (Lo *et al.*, 2005), but sardine have been captured 300 n.m. offshore (Macewicz and Abramenkoff, 1993). Sardine migrate north in the late spring and south in the fall. Movements are more extensive for larger sardine and during El Niño conditions. Spawning occurs year-round in the southern stock (17–21° C), peaks April through August in the northern stock (13–15° C), and January through April in the Gulf of California (22–25° C, Table 9.1b). At low biomass they mature at age 1, at high biomass only some 2-year-olds are mature. They are oviparous multiple-batch spawners with indeterminate and age/size-dependent fecundity (Macewicz *et al.*, 1996). Two-year-olds spawn six times per year and old sardine may spawn up to 40 times in a year (Butler *et al.*, 1993). Maximum age is 14 years, but most commercially caught sardine are less than 5 years old (Table 9.1b). Size-at-age and composition vary regionally, with both increasing in northern and offshore areas (Phillips, 1948; Hill *et al.*, 2006). Recruits first appear in the California fishery between October and April at age 6–12 mo. Adult natural mortality was estimated at $M = 0.4$ yr^{-1} (Murphy, 1966), but M varies by subpopulation.

Paleo-oceanography indicates dramatic changes in California sardine over the past two millennia (Baumgartner *et al.*, 1992, revised in MacFarlane *et al.*, 2002), with biomass peaking from 5 to 6 Mt. At the peak of the historical fishery, biomass was 3.6 Mt (1934) (Murphy, 1966) and the fishery captured 718 000 t (1936). Fisheries progressively collapsed from British Columbia to northern Baja California, and the biomass was only 10 000 t in 1965. Minor fisheries remained in southern Baja California (Radovich, 1982; Lluch-Belda *et al.*, 1989) and a substantial fishery eventually developed in the Gulf of California (Cisneros-Mata *et al.*, 1995). The northern subpopulation began recovering in the early 1980s, and the population grew from 5000 t in 1983 to 1.49 Mt by 1996. The latest assessment indicates current biomass to be 1.06 Mt (Hill *et al.*, 2006). The recovery has fueled fishery redevelopment from Baja California to British Columbia, with a current combined harvest of up to 142 000 t year^{-1} (PFMC, 2006).

Biological data for assessment purposes have been collected since 1919. Fishery sampling resumed for most areas with the onset of the population recovery, and biological data are available from the California and Baja California ports beginning in the early 1980s and from Pacific Northwest ports since the year 2000. The current stock assessment for US management (Hill *et al.*, 2006) includes age composition and weight-at-age data from three fisheries (northern Baja California, California, and Pacific Northwest) for the period 1982–83 to present, aggregated by season (July–June).

*Humboldt sardine (*Sardinops sagax*) – South Peru/ North Chile stock*
Three stock units of sardine exist along the Humboldt Current: from north-central Peru to about 15º S; from southern Peru to northern Chile (15–24º S, the focus of this review) and a third stock off central-south Chile (30–42º S). The latter is only obvious in the expanded phase of sardine's distribution. The Galapagos Islands may have a separate stock (Serra, 1983; Parrish *et al.*, 1989). When sardine is abundant, its distribution ranges from Ecuador (0º S) and the Galapagos Islands down to south-central Chile (42º S) (Serra, 1983; Serra and Tsukayama, 1988; Parrish *et al.*, 1989). When scarce its distribution is from 5–27º S. Its offshore distribution exceeds 200 n.m. but juveniles are found close to the coast (Serra and Tsukayama, 1988). They may reach over 40 cm in total length and sizes over 35 cm were frequent in the Chilean fishery. The maximum longevity of the sardine is 11 years (Table 9.1b). The natural mortality rate has been estimated to be $M = 0.3$ yr^{-1} (Serra and Tsukayama, 1988).

Off Chile the sardines have a long spawning season with two peaks, in winter and summer (Serra, 1983; Serra and Tsukayama, 1988). Sardines are multi-batch spawners with an indeterminate annual fecundity. The size at first spawning is generally 24 cm, equivalent to age 3. All fish > 6 y.o. are mature. Catches are sustained by fish from 5 to 8 years old. In northern Chile, sardines start to be recruited to the fishery at age 1, but are not fully recruited before age 6.

The sardine biomass started to increase in the early 1970s and has declined since 1981 due to intensive exploitation and poor recruitment (Fig. 9.3a) (GTE, 2003; Serra *et al.*, 2004). Meanwhile, catches continued to grow reaching 3 Mt in 1985, decreasing to present levels of about 100 000 t. The catch of this stock shows

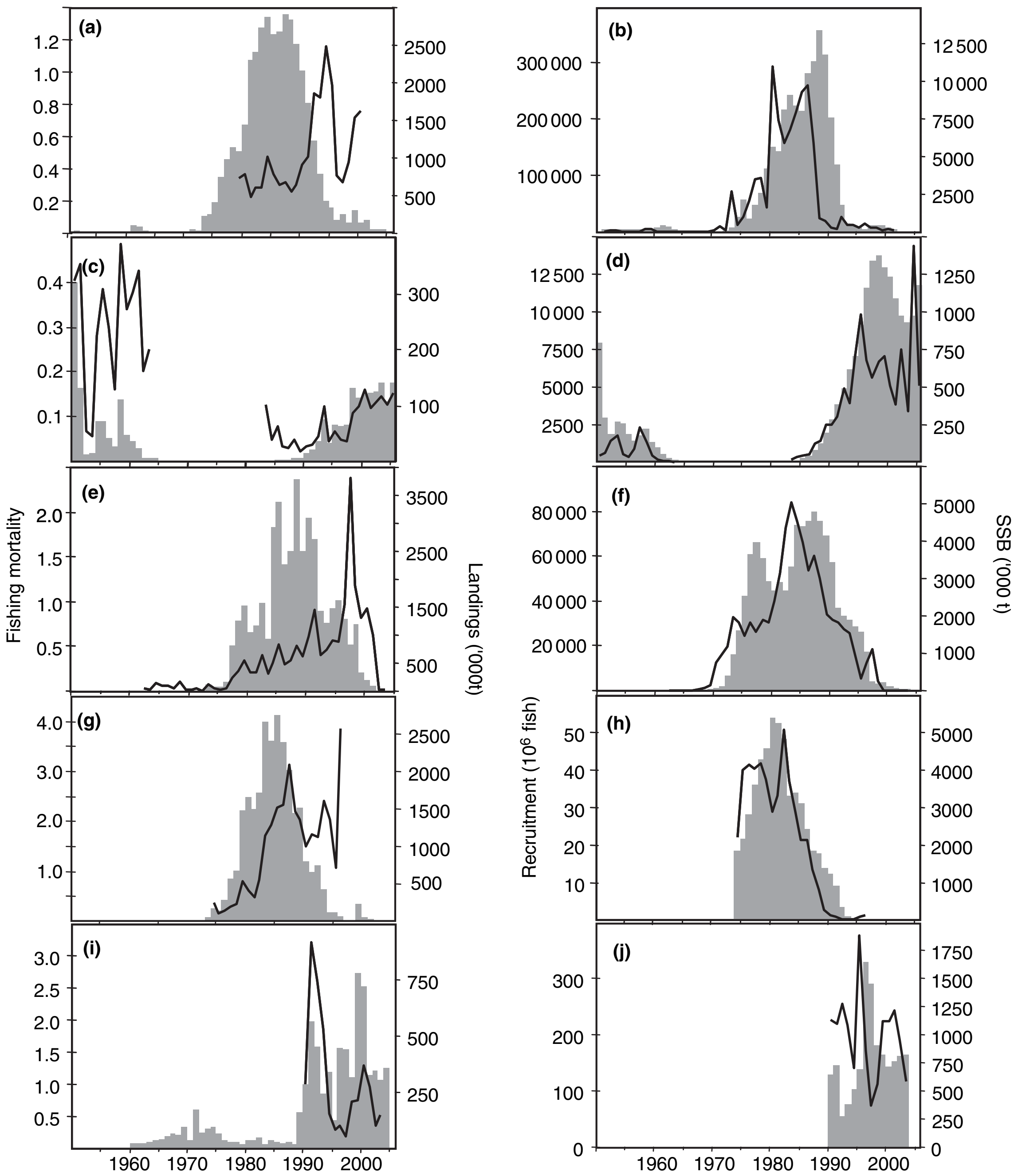

Fig. 9.3a. Time series of fishing mortality (y^{-1}), landings ('000 t), recruitment (106 fish) and spawning stock biomass (SSB, '000 t) for Japanese sardine (a, b), California sardine (c, d), Humboldt sardine (North/Central Peru stock, e, f); South Peru/North Chile stock (g, h) and Chilean common sardine (i, j). Notation as in Fig. 9.2a.

coherent and synchronous long-term changes alternating with anchovy catches (Serra, 1991; Lluch-Belda *et al.*, 1992; Schwartzlose *et al.*, 1999; Figure 9.2a).

The Chilean pelagic fish fishery is monitored by a program of observers and samplers on fixed landing sites. Biological sampling on the landings for species composition, size, weight, sex, maturity and otoliths for age studies has also been taken since 1964. For the assessment of the shared sardine stock between Peru and Chile only the time series from 1974 to 1996 has been considered, as subsequent abundance indices are not reliable. Catch-at-age matrices from the Chilean and the Peru–Chile fisheries are computed monthly and grouped by year, and then added to obtain an overall annual matrix for the combined stock.

Weight-at-age was obtained converting the mean length-at-age to weight by an overall length–weight relationship (Serra *et al*., 2004; GTE, 2003).

*Chilean common sardine (*Strangomera bentincki*)*

The common sardine is a Chilean endemic species, distributed from northern Coquimbo (29º S) to Puerto Montt (42º S; Arrizaga, 1981). Despite morphometric and meristic differences between individuals from different landing ports (Cortes *et al*., 1996) it constitutes a single genetic population (Galleguillos *et al*., 1997).

In the central-south area off Chile, two areas of high abundance of common sardine can be identified between 34°30–37º10 S and between 38–40º S (Castillo *et al*., 2005; Cubillos *et al*., 2005). Eggs, recruits and adults concentrate between the coast and 10–20 nm offshore. The main bays (Golfo de Arauco, Bahía de Concepción, and Bahía Coliumo) are important spawning zones probably through a combination of retention and concentration processes.

The spawning season extends from July to September, with a peak between August and September (Cubillos *et al*., 1999, 2001). Relative fecundity has been estimated by Cubillos *et al.* (2005), ranging between 433 and 535 oocytes per female body weight. Three to four months after spawning, juveniles (5–6 cm total length) recruit to the population (Cubillos *et al*., 2001, 2002). The fishery is heavy dependent on the annual pulse of recruitment, which concentrates in bays and gulfs from January to March (Cubillos *et al*., 1998). The common sardine is a short-lived species and attains a maximum age of 4 years. Size at first maturity is 10 cm total length (1 y.o.; Cubillos *et al*., 1999; Table 9.1b).

Biomass has been estimated by stock assessment models for the period 1990–2003 (Canales *et al*., 2004). Total biomass was highest (3 Mt) at the beginning of the 1995/96 fishing season (Fig. 9.3a).

Total catch and catch per unit effort data from the industrial purse-seine fleet are used in stock assessment from 1991. In addition, catch-at-age and weight-at-age matrices are used in stock assessment models. For the common sardine stock, all age groups are defined to be born on July 1, and the fishery information is pooled by fishing seasons starting the July 1 and ending the June 30 of the following year.

*Brazilian sardine (*Sardinella brasiliensis*)*

The Brazilian sardine inhabits the Brazilian Southeast Bight, between 22 and 29º S. Morphometry, seasonality, and biochemical studies (Saccardo and Rossi-Wongtschowski, 1991) suggest that the species does not constitute a single stock unit. However, it is considered as a single stock for management purposes, as there is not enough information to characterize the different stocks. This species spawns in batches, from October to March, with maximum intensity in December–January (Saccardo and Rossi-Wongtschowski, 1991). Spawning occurs at night in the upper layers of the water mass on the continental shelf (Matsuura, 1983, 1996, 1998; Saccardo and Rossi-Wongtschowski, 1991). Individuals up to 4 y.o. have been recorded and maximum size is 27 cm (Cergole and Valentini, 1994; Table 9.1b). Females mature at approximately age 1, and the whole population is matured by age 2 (Vazzoler, 1962; Isaac-Nahum *et al*., 1983; Isaac-Nahum *et al*., 1988; Wenzel *et al*., 1988; Cergole and Valentini, 1994). Maximum recruitment to the adult stock occurs around July (Cergole, 1993, 1995).

Abundance estimates through indirect methods suggested periods of high and low abundance (Fig. 9.3b): the first period (1977 to 1986) showed a total mean biomass of 668 000 t (SSB 255 000). From 1986 the stock showed two significant declines in 1990 and 2000 (Cergole, 1993, 1995; Cergole *et al*., 2002). First records of sardine catches were collected in 1964, reaching 228 000 t in 1973 before initiating a downward trend until the 1990s. As far back as 1988, it was recognized that the stock was collapsing. Severe management recommendations (Rossi-Wongtschowski *et al.,* 1995; SUDEPE/PDP, 1989) led to some signs of recovery, with catches of 118 000 t in 1997. Since then, a new decline took the total catch to 17 000 t in 2000 (IBAMA, 2005).

The Brazilian sardine purse-seine fishery suffered a 50% reduction by the year 2000, but fishing effort has not decreased because the larger boats with larger fishing power remained in the fishery. From the mid 1990s the fleet has diversified into a multispecies fishery targeting sardines, mackerels and other pelagics.

*Benguela sardine (*Sardinops sagax*)*

Benguela sardine are distributed from southern Angola (approximately 15° S) to South Africa's east coast (around 30° E), and comprise separate northern and southern stocks (Beckley and van der Lingen, 1999). Benguela sardine have historically supported large catches off Namibia and South Africa.

Sardine have a protracted spawning period in both the northern and southern subsystems (although with very low egg concentrations in the austral winter) peaking in September/November and February/March (Beckley and van der Lingen, 1999; van der Lingen and Huggett, 2003). The location of the primary spawning grounds has shown substantial spatial shifts through time in both subsystems (van der Lingen *et al*., 2006a). In the northern Benguela, spawning has moved north with the decline in population size. Off South Africa, intense sardine spawning has been confined to the south coast in recent years (van der Lingen *et al*., 2005). Sardine recruitment in the northern Benguela occurs primarily inshore of major spawning sites, whereas the South African west coast is the principal nursery ground

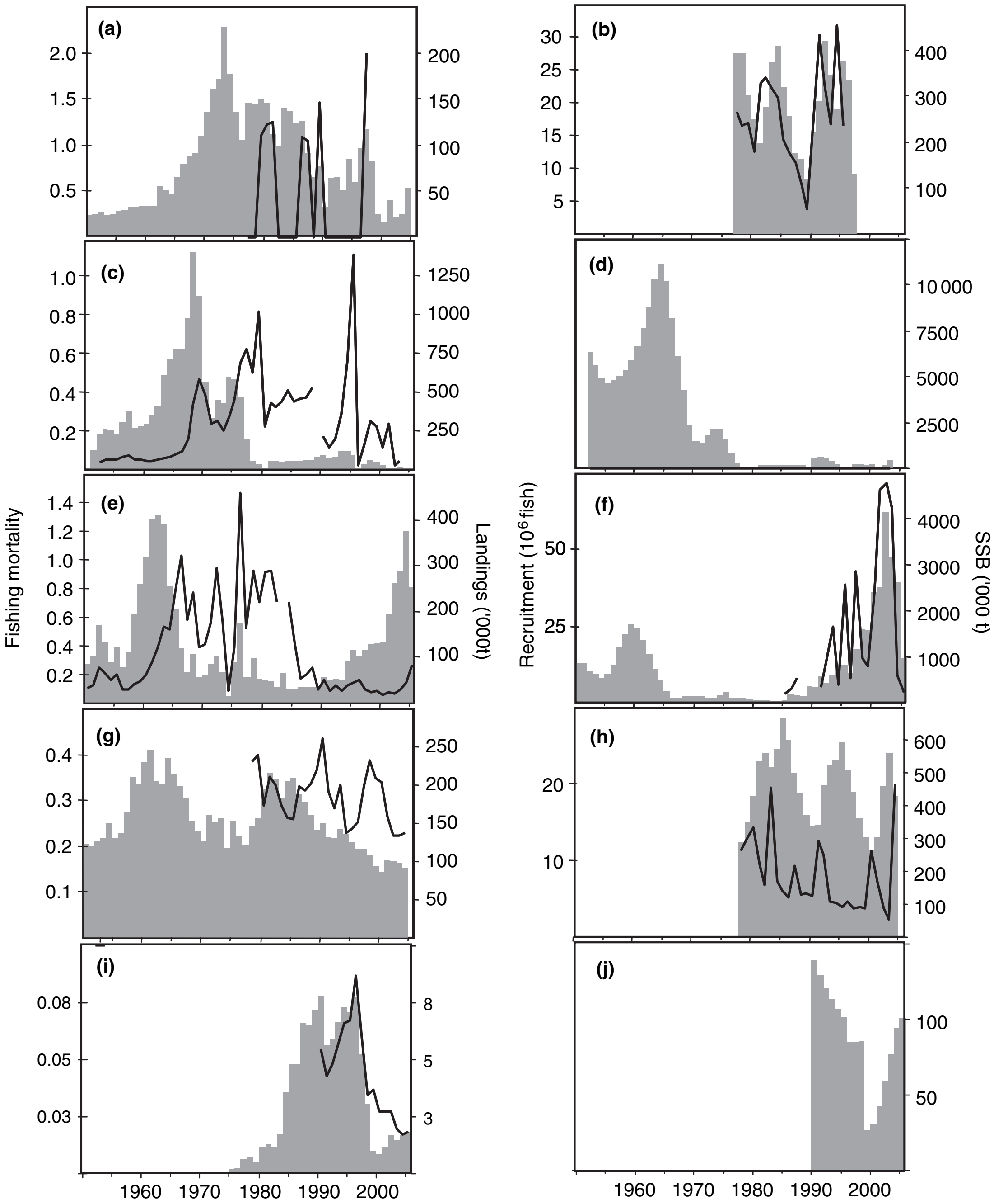

Fig. 9.3b. Time series of fishing mortality (y^{-1}), landings ('000 t), recruitment (10^6 fish) and spawning stock biomass (SSB, '000t) for Brazillian sardine (a, b), northern Benguela sardine (c, d), southern Benguela sardine (e, f), European sardine (g, h) and SW Australian sardine (i, j). Notation as in Fig. 9.2a.

of southern Benguela sardine (Barange *et al.*, 1999). The size at sexual maturity of sardine has varied in both the northern (Thomas, 1986) and southern (Armstrong *et al.*, 1989; Akkers *et al.*, 1996; Fairweather *et al.*, 2006) subsystems, possibly in a density dependent response to stock size (van der Lingen *et al.*, 2006b). Sardine is a relatively short-lived species, with few adults older than 8 years observed and a maximum age of 10 (Le Clus *et al.*, 1988).

Commercial fishing operations targeting South African sardine commenced in 1943 and landings rose dramatically during the late 1950s and early 1960s, peaking at 410 000 t in 1962 and declining rapidly thereafter (De Oliveira, 2003; Fig. 9.3b). The collapse of the southern Benguela sardine stock was ascribed to over-fishing, expansion of fishing grounds, and variable recruitment (Beckley and van der Lingen, 1999), and resulted in increased fishing effort in the northern Benguela, where annual sardine catches rose rapidly to a maximum of 1.4 Mt in 1968 (Boyer and Hampton, 2001; Fig. 9.3b). Thereafter, sardine landings declined and have remained below 100 000 t since the early 1980s. As in the southern Benguela, the collapse of the northern Benguela sardine population was primarily attributed to overfishing, although poor sardine recruitment resulting from adverse environmental conditions exacerbated the decline, both during the 1960s (Boyer and Hampton, 2001) and more recently (Boyer *et al.*, 2001).

Sardine spawner biomass in the southern Benguela has been increasing since the start of the current acoustic monitoring program in 1984, reaching a peak in 2002. A unique phenomenon for this stock is the KwaZulu-Natal "sardine run" (Armstrong *et al.*, 1991), in which large shoals of sardine move up the South African east coast during winter (June) each year, reaching Durban and occasionally further north. Catches (typically < 1000 t) are small in comparison to those on the west and south coasts and are not included in the catch statistics for assessment. It has been proposed that the "sardine run" occurs in response to an expansion of the environment suitable for sardine on the east coast during the cooler water conditions of winter (Armstrong *et al.*, 1991). Off Namibia sardine spawner biomass has remained at a very low level since the start of the current acoustic monitoring program in 1990.

Monthly age–length keys have been compiled for the southern Benguela sardine from 1980 to 1999, giving monthly catch-at-age data (De Oliveira, 2003), and corresponding mean masses-at-age are available. Age readings since 1999 remain scarce at the moment and thus average age–length keys have been used to estimate catch-at-age from 2000 for the most recent assessment. Although a less than ideal scenario, this has enabled a temporary update of the assessment in the absence of such ageing information (Cunningham and Butterworth, 2004b). The catch of juvenile sardine taken between November (when spawning is modeled to occur) and the beginning of the recruitment survey is also utilized in the assessment. The bycatch of juvenile sardine caught in the anchovy directed fishery has also been recorded since 1987, and is used in constructing the management procedure. Length frequencies are generated for sardine caught in the northern Benguela and, as fish are not aged, division into age classes is done according to length, with fish ≤ 16.5 cm *TL*[8] assigned to the 0-group, 16.6–22.4 cm *TL* the 1-group, and ≥22.5 cm *TL* the 2+ fish (Boyer *et al.*, 2001).

*European sardine (*Sardina pilchardus*) – Atlanto-Iberian stock*

The European sardine is widely distributed in shelf waters along the northeast Atlantic (NEA) coast (from 15–20° N to 52–58° N) and the Mediterranean Sea, with residual populations off the Azores, Madeira and the Canary Islands. A recent study integrating morphometrics with genetics (allozymes and microsatellite DNA) revealed five genetic stocks: Azores, Madeira, Mediterranean Sea, and two Atlantic stocks in the area from the North Sea to Mauritania (with an internal boundary possibly at the Bay of Agadir) (Y. Stratoudakis, pers. comm.). For assessment purposes, sardine in European Atlantic waters has always been considered to belong to a single stock, but its geographic limits have changed over time. Sardine assessment under ICES began in 1978 (ICES, 1978), when the stock covered the area from the north of France to southern Iberia. The northern border was revised in 1980, based on a mixture of biological evidence and administrative needs (ICES, 1980). This new delimitation gave rise to what is currently known as the Atlanto-Iberian stock, from the France/Spain border in the inner Bay of Biscay to the Strait of Gibraltar.

Sardine is an indeterminate, batch spawner with high relative fecundity. Spawning activity extends for many months, with regionally varying local peaks (Coombs *et al.*, 2006). Sexual maturation is attained during the first two years of life (Silva *et al.*, 2006). Although sardines up to 14 years of age have occasionally been reported, most fish do not exceed 6–8 years (Table 9.1b). Spawning probably takes place near the bottom towards dusk. Adult fish are widely distributed within the continental shelf, forming characteristically dense schools with marked diel variations in vertical position, shape and integrity. Recruitment demonstrates high spatial fidelity, creating a complex mosaic of adjacent subpopulations of different ages. In general, European sardine shows less pronounced migration patterns than other pelagic species. Sardine maximum size, longevity, and growth are highest at the northern extreme (northern France and English Channel) and lowest in the southern Iberian Peninsula. Upper temperature tolerance to spawning seems also to increase with decreasing latitude. Within the NEA, the largest and most productive stock is situated off Morocco (stock biomass 1–5 Mt; catches around 600 000 t in recent years; FAO, 2003). In the Atlantic waters of the Iberian Peninsula (stock biomass around 500 000 t; ICES, 2005a), catches peaked at 250 000 t in the mid 1960s, but have declined during the past 15 years, and are currently at 100 000 t (ICES, 2005a; Fig. 9.3b). Sardine catches north of the Iberian Peninsula are comparatively small. A dedicated fishery with annual catches of some 10–15,000 t exists in the French waters of the Bay of Biscay, where fisheries interest seems to be increasing. A small seasonal fishery is found in the western

English Channel (3000 – 5000 t), while in recent years sardine distribution is reported to have extended further north, well into the North Sea.

Iberian catches are almost exclusively destined for human consumption, through a complex marketing circuit involving intermediate sales that progressively increase the cost. The average first sale price is 0.61 euros kg^{-1} (2004 in Portugal), with a marked seasonality (price peaking in summer months when sardine are fatter and fresh consumption is highest).

Systematic biological monitoring of sardine catches started in the late 1940s by Spain and Portugal. The first ICES sardine assessment meeting was held in 1978, reviewing existing biological information and sampling plans to provide input data to future assessments (ICES, 1978). Since then, regular sampling of fish catches (length distribution, macroscopic biological information and age-length keys) has been performed, progressively providing a more homogeneous coverage of the Iberian Peninsula and increasing the sampling frequency and intensity. Length distribution samples are obtained fortnightly, biological samples monthly and age readings trimestrally or semestrally. Reviews on sampling and data availability can be found in Pestana (1989), Carrera and Porteiro (2003), Jardim *et al*. (2004), Silva *et al*. (2006).

*Southwestern Australia sardine (*Sardinops sagax neopilchardus*)*

Sardine are continuously distributed around the southern Australian coast from approximately 26° S on both the eastern and western coasts. Highest abundance, and the more significant fisheries, are along the southern coast, with a center of distribution in the state of South Australia, where the fishery has reached annual total catches of 30 000–40 000 t. The fishery along the southwestern region, in the state of Western Australia (WA), has reached an annual total catch of around 12 000 tonnes.

Sardines in WA consist of two breeding stocks: one along the west coast and another along the south coast. The remainder of this summary focuses on the south coast (WA) breeding stock, which has had the longest period of research and monitoring for sardine in Australia. The south coast stock is partitioned into three management units (zones) because of limited alongshore movement of the mature biomass (Edmonds and Fletcher, 1997; Gaughan *et al*., 2002); each management unit has its own TAC[9]. Uncertainty regarding spatial dynamics of recruits has been managed via a "recruit pool" hypothesis, which suggests that each unit of the stock is important for recruitment to the broader breeding stock.

Sardines in Western Australia live to 8–9 years (Fletcher and Blight, 1996) both maturing and entering the fishery at approximately 2 years but full recruitment into the fishery may not occur until 4 or 5 years of age. South coast sardines spawn for most of the year with major peak in winter and a smaller peak in summer; spawning occurs across much of the continental shelf seaward of ~30 m depth (Fletcher and Sumner, 1999; Gaughan *et al*., 2004). These general patterns vary regionally and interannually, depending on size of the SSB and oceanographic conditions. Larvae from the main (winter) spawning are transported east, potentially up to 1000 km (Gaughan *et al*., 2001), requiring a return migration by juveniles.

Purse-seine fisheries on the south coast expanded in the mid to late 1980s at which time the maximum annual catch (8000 t) was recorded. The fleet of relatively small vessels typically fish within 10–15 km of port. Biological assessment and monitoring began in 1988. SSB declined in the 1990s due to poor recruitment, fishing pressure in some regions and a mono-specific mass mortality, but has grown since 1999. However, uncertainty in the estimates of SSB preclude setting harvest rates higher than 10% of SSB.

Herring and sprats

*Atlantic herring (*Clupea harengus*) – North Sea autumn spawning stock*

North Sea herring is made up of a number of spawning components, including Shetland, Buchan, Banks, and Downs (Heinke, 1898; Redeke and van Breemen, 1907; Cushing, 1955, 1992; Zijlstra, 1958). The current stock definition only covers the autumn and winter spawning herring and spring spawners are not included in the stock assessment or the management agreement (Cushing, 1967; ICES, 2006d). Herring spawn benthic eggs and for this stock, spawning occurs in the western North Sea along the coast of the UK (Boeke, 1906; Cushing and Burd, 1957; Burd and Howlett, 1974). Atlantic herring are spatial repeat spawners (McQuinn, 1997) and this behavior is either caused by natal returns to the "home" spawning bed or adopted behavior (Harden Jones, 1968; Wheeler and Winters, 1984; McQuinn, 1997, and references cited therein).

The majority of the larvae drift in an easterly direction towards the German Bight and Skagerrak (Munk and Christensen, 1990; Bartsch, 1993). It is during this life stage that year class strength is determined (Nash and Dickey-Collas, 2005). The juveniles stay in the east until maturity, upon which they then join the adults. Recruits from one spawning will not necessarily mature in synchrony (McQuinn, 1997; Brophy and Danilowicz, 2003). The adults feed in the central and northern North Sea. After feeding, the herring migrate to the spawning grounds. As the majority of North Sea herring are autumn and winter spawners, they exhibit a different energy strategy than Norwegian spring spawning herring (Isles, 1984; Winters and Wheeler, 1996; Slotte, 1999). Genetically, the stock shows no major differentiation between the spawning components, but there is drift with distance between spawning grounds in the genetic make-up of the herring (Mariani *et al*., 2005). There are also strong density dependent effects in the population characteristics (Cushing and Bridger, 1966; Hubold, 1978; Winters and Wheeler, 1996; ICES, 2006d).

In the 1960s, the spawning stock biomass of North Sea herring was over 2 Mt (ICES, 2006d, Fig. 9.4a). However, the stock collapsed in the 1970s through overfishing (Burd, 1985; Cushing, 1992; Nichols, 2001; Simmonds, 2007). The stock slowly recovered to above 1 Mt by the 1990s and a management agreement was brought in to reduce fishing effort. The stock then grew to approximately 1.8 Mt in 2004 (Table 9.1c). This high spawning biomass, however, has not prevented 4 years of poor recruitment (2003 to 2006, at age 0). In 2006, the stock began to decrease in size again in response to this.

Regulated fisheries exist for both the adult and juvenile herring. Juvenile herring are caught as a bycatch in the North Sea industrial fisheries, which usually target sandeel, sprat, and Norway pout. The fisheries on the adults takes place both whilst the adults are feeding in the summer, and during the autumn and winter spawning aggregations. The only comprehensive fishery information collected at present for the management of North Sea herring are the numbers and weights of herring, by age in the catch. These are estimated by each catching nation and combined under the auspices of ICES. Each nation uses different methods, some correct for misreporting and others include estimates

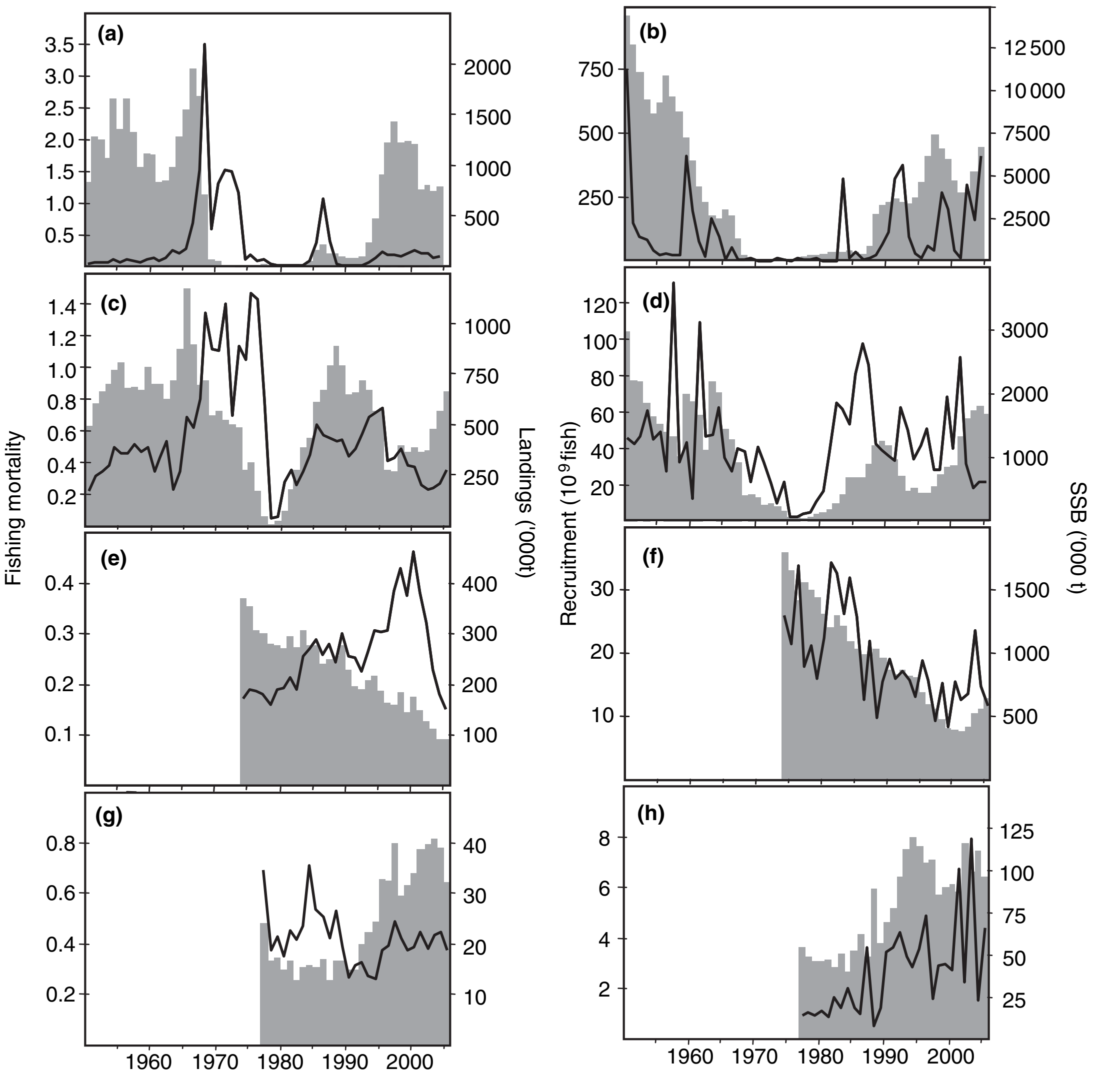

Fig. 9.4a. Time series of fishing mortality (y^{-1}), landings ('000 t), recruitment (10^9 fish) and spawning stock biomass (SSB, '000t) for Arcto-Norwegian spring spawning herring (a, b), North Sea autumn spawning herring (c, d), Central Baltic herring (e, f) and Gulf of Riga herring (g, h). Notation as in Fig. 9.2a.

of discarded fish. No information on the effort, distribution, or efficiencies of the fleets are used in the current management of the stock, as it is thought that over-reliance on catch per unit effort data resulted in the collapse of the stock in the 1970s. Although VMS is now on almost all vessels that fish North Sea herring, these data are not available to scientists. The species make-up of the catches of the industrial fisheries is also monitored and used to determine the total catch of juvenile herring. The herring in these samples are also monitored to determine spawner type (autumn, winter or spring) to ensure that juvenile catches are allocated to the correct stock.

*Atlantic herring (*Clupea harengus*) – Norwegian spring spawning stock*

Norwegian spring spawning herring exhibits large migrations and is distributed throughout the north east Atlantic (from the Shetlands to Iceland to Russia to Spitsbergen). Like other herring it shows high phenotypic plasticity (Jennings and Beverton, 1991; McQuinn, 1997). The maximum size of Norwegian spring spawning herring is large (>40 cm and 700 g) and is considered longer lived (Holst *et al.*, 2004). These herring mature at 4 to 6 years of age (Table 9.1c). The migrations of Norwegian spring spawning herring change, dependent on the size and age profile of the stock and changes in the environment. Norwegian spring spawning herring can also mix with neighboring stocks. This has impacted on the stock definition; whereas Icelandic spring spawners were classed as part of the stock in the past (Johansen, 1919), since the stock recovery in the 1980s the Icelandic spring spawners are considered a separate stock (which has still yet to fully recover). The Norwegian spring spawning herring is also called Atlanto-Scandian herring by some authorities.

Norwegian spring spawning herring spawn benthic eggs on gravel. The spawning grounds are along the Norwegian coast and, unlike North Sea herring, as the adults age they spawn on grounds further and further south along the Norwegian coast (Slotte, 2000; Slotte *et al.*, 2000), i.e. natal homing is thought to be weak. Spawning appears heavily influenced by environmental conditions, with large-scale atresia of the ovaries in some years (Oskarsson *et al.*, 2002). Spring spawning herring exhibit a different strategy in terms of feeding and energy utilization than autumn spawning herring, in that their condition is based on the feeding season of the previous year (Isles, 1984; Winters and Wheeler, 1996; Slotte, 1999). The summer feeding and overwintering areas appear to vary with time; the stock overwintered offshore during the 1950s and 1960s and then moved inshore in the 1970s and 1980s. In the last decade some fish have begun to overwinter offshore again (see Holst *et al.*, 2004).

This stock has had a well-documented collapse and recovery (Toresen and Østvedt, 2000) with an increase in biomass from 1900 onwards to a spawning biomass greater than 15 Mt in the 1940s. In the 1950s the stock declined and then collapsed in the 1960s to less than 20 000 t by 1972 (ICES, 2005b; Fig. 9.4a). The collapse was due to over-exploitation and changes in productivity of the Norwegian Sea (Toresen and Jakobsson, 2002). From 1980 onwards the stock has slowly rebuilt, and is now above 5 Mt (ICES, 2005b). The catches increased from 1900 to the 1960s (from 200 000 t to above 1.5 Mt) and then by the 1970s they were less than 7000 t. Under a program for rebuilding of the stock catches have been reduced to approximately 800 000 t in the mid 2000s.

The fishery now occurs in two main locations: in international waters to the northwest of Norway in the summer and along the Norwegian coast in the winter. The fishery includes vessels from Norway, Russia, Iceland, the Faroes, and the EU[10]. These fleets operate a range of vessel types and fishing gears including pelagic trawls, paired trawls and purse seine, operating in different seasons in different areas. Norway and Russia dominate the catches with approximately 60% and 15% of the annual catch, respectively. The fishery information collected at present for the management of North Sea herring are the numbers and weights of herring, by age in the catch. These are estimated by each catching nation and combined under the auspices of ICES. No information on the effort, distribution or efficiencies of the fleets is used in the current management of the stock.

Atlantic herring (Clupea harengus) – *Baltic Sea*

Herring inhabits the entire Baltic from marine to nearly freshwater habitats. The stock structure is complex with a number of different spawning components, exhibiting variations in spawning period (spring vs. autumn spawners), spawning locations (coastal vs. offshore) and growth rates as well as meristic, morphometric and otolith characteristics (e.g. Ojaveer, 1981; Parmanne *et al.*, 1994). Altogether ICES (2001b) has identified 11 herring stocks in the Baltic. Spring spawning herring dominate in abundance since the early 1970s, with the reasons for the decline in autumn spawner abundance still being unclear (Parmanne *et al.*, 1994). Various spring spawning components mix during their feeding period in summer and autumn in open sea areas of the Baltic (Aro, 1989), which makes stock separation at this time of the year difficult.

Baltic herring is at present assessed in five different stock units (ICES, 2006c), the Western Baltic (ICES Subdivision 22–24 with prolonged feeding migrations into the Kattegat, Skagerrak and the North Sea), the Central Baltic (Subdivision 25–29 inclusive of the Gulf of Finland, i.e. Subdivision 32), the Gulf of Riga (eastern part of Subdivision 28), the Bothnian Sea (Subdivision 30) and the Bothnian Bay (Subdivision 31). In focus of the present review are the Central Baltic and the Gulf of Riga stocks.

Baltic herring is fished by a variety of fishing fleets, for human consumption in trawl, trap-net and gill-net fisheries and in a mixed trawl fishery with sprat for industrial purposes (see Parmanne *et al*., 1989; ICES, 2006c). Central Baltic herring landings and spawning stock biomass decreased from 1980s to 2000 by 60–70% (Fig. 9.4a). Stock abundance decreased as well until the mid 1990s, but less drastically. The decline in landings and biomass was partly driven by a decline in weight-at-age from early 1980s to mid 1990s (ICES, 2006c). Herring in the Gulf of Riga showed an opposite development, with tripled landings and more than doubled spawning stock biomass in the last 20 years (Fig. 9.4a). SSB and stock abundance increased especially until the mid 1990s, while the continued increase in landings also in later years was accomplished by increasing fishing mortalities (ICES, 2006c).

The drastic decline in weight-at-age of the Central Baltic herring has been explained by (1) a reduction in size selective feeding by cod, preying predominantly on smallest individuals within a herring age group (Beyer and Lassen, 1994), (2) different developmental success in sub-stocks exhibiting different growth rates (Sparholt, 1994a), and (3) limitation in food supply (e.g. Cardinale and Arrhenius, 2000). Both the Gulf of Riga herring and the Baltic sprat showed similar reductions in weight-at-age in the absence of predation by cod or size-selective predation, respectively, indicating that the first two hypotheses can not explain the observed changes in weight-at-age alone. In turn, zooplankton data suggest that individual prey availability for both clupeid species declined concurrently with their weight-at-age. Especially the decline of *Pseudocalanus acuspes* affected the nutrional status of herring negatively, while sprat utilizes also other copepods, e.g. *Temora longicornis* and *Acartia* spp., and thus sustained a good nutritional status until density dependent processes started to act as a consequence of the drastic increase in stock size in early 1990s (Möllmann *et al*., 2005). A potential impact of low condition on reproductive success of herring has been hypothesized, but not addressed in specific studies. Herring sexually mature between age 2 and 3, with substantial interannual variability. Yearly maturity ogives have been compiled (ICES, 2002a) but are presently not used in the assessment.

Predation by cod is a major source of clupeid mortality in the Baltic, with especially juvenile herring being preyed upon intensively (Sparholt, 1994b). To account for this, the stock assessment uses age- and year-specific predation mortalities estimated by a Multispecies VPA (ICES, 2005c) as input. A drastic decline in the cod stock throughout the 1980s as the major predator in the system caused a substantial reduction in predation (Köster *et al*., 2003a). While open sea herring of the Central Baltic should have benefited most from the release of predation pressure, this is not obvious from above-described stock dynamics.

Recruitment of herring stocks in the Gulf of Riga, Gulf of Bothnia and Gulf of Finland is highly correlated, but largely decoupled from recruitment variability of the Central Baltic stock (Kornilovs, 1995). Also the longer-term trends are different: while the latter showed a decline in recruitment from mid 1980s to 2000, the reproductive success of Gulf herring, e.g. in the Gulf of Riga, increased (Fig. 9.4a), with recruitment at age 1 being significantly related to SSB, temperature in April and zooplankton abundance in May (Kornilovs, 1995; Barange, 2003). Similarly, winter/spring temperature is correlated with the recruitment of herring in the Bothnian Sea and Bay. The mechanisms affecting year class formation of herring in the Central Baltic are not fully understood. A separate assessment for three popultions performed by ICES (2003a) revealed a similar pattern in recruitment dynamics; however, variability was higher for the southern coastal herring population.

*European Sprat (*Sprattus sprattus*) – Black Sea*

Sprat is distributed in the whole Black sea with maximum abundance in the northwestern part and shelf waters. In spring, schools migrate to coastal waters for feeding and in the summer they stay under the seasonal thermocline forming dense near-bottom aggregations during the day and rising to the surface at night (Ivanov and Beverton, 1985).

Black Sea sprat forms a self-sustaining stock (Ivanov and Beverton, 1985). It is one of the most abundant species in the area with importance for the commercial fishery as well as a food for fish and mammals (Daskalov, 2002).

Sprat matures at age 1 and reproduces during the whole year with a maximum between November and March (Table 9.1c). Spawning can be associated with the winter divergence and spring plankton blooms (Daskalov, 1999). The reproductive niche is limited to offshore subsurface (10–50 m) layers, which are stabilized by the permanent pycnocline. Horizontally, sprat eggs and larvae are concentrated above the shelf edge, and in the central cyclonic areas (Arkhipov, 1993; Fashchuk *et al*., 1995).

Sprat is an object of both artisanal and commercial midwater trawl fisheries. Decadal fluctuations of abundance are observed and likely to be related to changes in climate (Daskalov, 1999, 2003). Maxima of recruitment and biomass were observed in the mid 1970s and mid 1980s (Fig. 9.4b). Maximum catch was in 1989, followed by a stock collapse. In the recent decade the stock partially recovered, with catches of up to 40 000–50 000 t.

*European Sprat (*Sprattus sprattus*) – North Sea*

Sprat is found mostly in the east and south of the North Sea, and also in the coastal areas and lochs of the east of the British Isles. No major studies on stock definition have

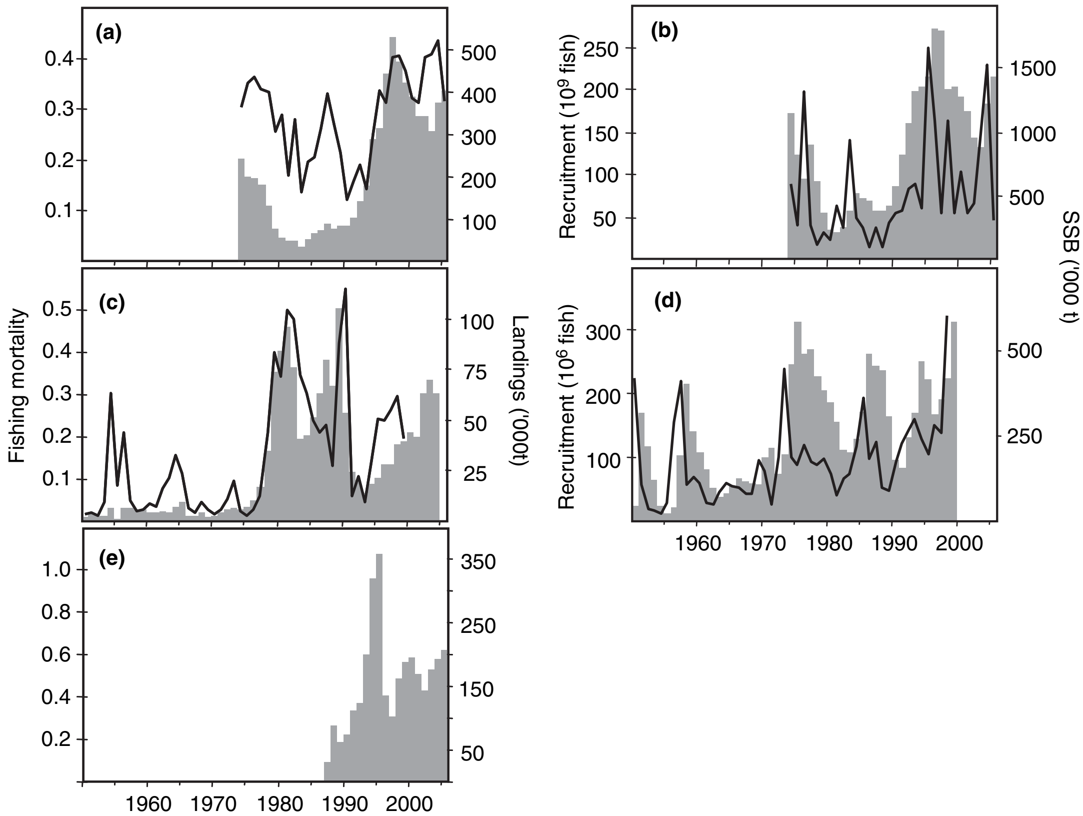

Fig. 9.4b. Time series of fishing mortality (y^{-1}), landings ('000 t), recruitment and spawning stock biomass (SSB, '000 t) for Baltic Sea sprat (a, b), Black Sea sprat (c, d), and North Sea sprat (e). Notation as in Fig. 9.2a

taken place, but at present it is assessed as a separate stock from that in the Skagerrak and the English Channel. In the North Sea, sprat spawns from spring through to late summer (Table 9.1c). This causes problems in aging the fish, as a variable part of each year's new young may not metamorphose that year, and hence a blurring of year classes or cohorts may occur. Sprat spawn pelagic eggs and, in general, larva drift is in a northeasterly direction towards the German Bight and Skagerrak, or towards the coast. Year-class strength appears to be determined during the larval phase, although as this is a short-lived species in the North Sea, there are not many cohorts in the population to support analysis of the consistency of year-class strength. The sprat are not thought to make large migrations, with both feeding and spawning occurring in similar areas. Estimates of absolute biomass are not available. However time series of catch (from 1985 onwards) and research surveys suggest that in 2000–2005, the stock biomass was high compared to earlier in the series (by at least three-fold). As this is a short-lived species in the North Sea, the biomass is strongly dependent on the incoming year-class strength. There are considerable fluctuations in total landings, from a peak in 1975 of 641 000 t to a low in 1986 of around 20 000 t (Fig. 9.4b). Since 1994, landings have varied from ca. 100 000 t (1997) to ca. 300 000 t (1994).

Denmark, Norway, and UK trawlers and purse seiners exploit North Sea sprat. Most of the catch (98%) is used to create fish meal (industrial fishery) and the remainder, mostly from UK and Norwegian catches, is for human consumption. Juvenile herring are also caught as a bycatch. The sprat fishery occurs in the second half of each year. The only fisheries data collected are the total landings, per quarter, from each country and samples of the age and weight of fish in the catch in the industrial fishery. The catch is usually dominated by 1-year-old fish. The bycatch of species other than sprat is monitored in the industrial fishery.

*European Sprat (*Sprattus sprattus*) – Baltic Sea*

Sprat is distributed throughout the entire Baltic Sea, with the exception of the Bay of Bothnia, and it is assessed and managed as a single stock. However, morphometric studies, otolith microstructure and genetic analyses suggest that at least two stocks are present in the western and central to northern Baltic (Parmanne *et al*., 1994). The stock identity of the latter component has been discussed controversially (e.g. Ojaveer, 1989; Sjöstrand, 1989), and the evidence of further separation into different stocks is not generally accepted.

The Baltic Sea is located near the northernmost limit of sprat's geographic distribution (Parmanne *et al*., 1994).

Thus, sprat shows a preference for warmer water layers in winter, specifically the bottom water layers of the deep Baltic basins, for which the pronounced stratification prevents a vertical convection during winter. Spawning takes place in March to June (Table 9.1c) as well in the deep Basins (Parmanne *et al*., 1994), while the distribution in summer and autumn is more widespread, covering coastal as well as deep water areas (Aro, 1989).

Depending on the fishing mortality and predation pressure by cod, sprat of 8–10 years of age can be abundant in the stock. Sexual maturity is reached at the age of 2 (Elwertowski, 1960), but age-group 1 fish sometimes contribute to spawning in significant proportions (ICES, 2002a). The mechanism affecting maturation is not yet clear, but low temperature delays sexual and gonadal maturation.

The stock size of sprat declined during the 1970s, remained at a stable low level throughout the 1980s and increased in the first half of the 1990s to historic high values, reaching more than 3 Mt biomass in 1995 (Fig. 9.4b). This positive stock development was enabled by a reduction in predation pressure by the main predator cod and increasing reproductive success at relatively low fishing mortalities (Köster *et al*., 2003a).

The reproductive success of sprat during the last 15 years is related to high winter temperatures having a positive impact on egg production and survival (Köster *et al*., 2003b; Nissling, 2004), as well as the strong preference of the larvae for the copepod *Acartia* spp. (Voss *et al*., 2003), which has increased since the 1990s in parallel to the increase in temperature (Möllmann *et al.* 2000). This may have led in general to higher survival of larvae, being the critical early life stage in Baltic sprat (Köster *et al*., 2003b). The drastic decline of weight-at-age observed during the 1990s (see section on Atlantic herring in the Baltic Sea) had obviously no impact on the reproductive success of the stock.

As a consequence of the drastic stock increase during the 1990s, an industrial fishery on sprat developed with 300 000–500 000 t catch annually resulting in a reduction in stock size in most recent years. Recruitment has been highly variable since the mid 1990s, but outstanding year classes, on average every second year, sustain the stock on a relatively high level (ICES, 2006c).

Most of the catches are taken by pelagic single and pair trawling, but some demersal trawling exists. The main fishing season is in the first half of the year, but in the northern part of the Baltic ice cover is a limiting factor. Sprat is fished mostly for industrial purposes, but also for human consumption and animal feed (ICES, 2006c). Sprat is also caught as bycatch in some herring fisheries and in turn herring bycatch occurs in several of the sprat fisheries. The latter is regulated by bycatch ceilings and, in fact, use of the TAC for herring has stopped the sprat fishery from fishing out its TAC in recent years. The species composition of the mixed catches is defined from logbooks and, partly, by observers on board of larger vessels. Misreporting of herring as sprat occurs, but the magnitude is unknown. Commercial catch rates are available for some fisheries but not used in the assessment (ICES, 2006c).

Fishery independent monitoring surveys

Until the 1960s most pelagic fish stocks were not assessed or their assessment relied on analysis of data from commercial fisheries. Since then the use of catch data has been compromised by its limitations; CPUE from commercial fleets have proven to be a poor proxy for stock abundance because pelagic fish schooling habits may result in increased catchability as stock abundance decreases rather than the expected opposite (Csirke, 1988). Consequently, the demand for fishery independent estimates of population size increased, either as input to management plans or to provide auxiliary information in catch-at-age analyses.

The main survey programs currently applied in the assessment of pelagic fish stocks are egg and larvae surveys and acoustic surveys. Bottom-trawl and aerial surveys have also been used sporadically (for review, see Gunderson, 1993).

Egg production surveys

Ichthyoplankton sampling started in the North Sea in 1895 (Hensen and Apstein, 1897), initially to identify and describe the biology of eggs and larvae. Currently, most egg and larvae programs are aimed at understanding the processes that determine recruitment fluctuations and to estimate recruitment strength, while some are used to estimate the size of the parental stock. The latter is based on one of three methods: the Annual Egg Production (AEP, Gunderson, 1993) and the Daily Fecundity Reduction (DFR, Lo *et al*., 1992a) methods, both applicable to determinate spawners, and the DEPM (Parker, 1980; Lasker, 1985), which can be used on determinate as well as indeterminate spawners. The DEPM has been most successfully applied to pelagic fish, and is based on daily measurements of egg production and fecundity during a single survey (Alheit, 1993; Hunter and Lo, 1997; Stratoudakis *et al*., 2006). A brief summary of the egg survey programs currently applied in the assessment of anchovy and sardine stocks follows. For sprat, the potential for the application of the egg production method has been demonstrated as well (Kraus and Köster, 2004), but at present the method is not used in any of the assessments.

Anchovy stocks

Monthly egg production surveys have been conducted along the Pacific coast of Japan from 1949 in support of the assessment and management of the Japanese anchovy (e.g., Kubota *et al.*, 1999), using NORPAC-net vertical samples from 150 m depth (or near bottom) to the surface. Monthly egg production is calculated on each 30×30 latitudinal square.

California anchovy are monitored but not actively managed in the US because catches have been relatively low in recent years (PFMC, 1998). The most recent stock assessments used biomass estimates from forward-projecting models (Methot, 1989; Jacobson *et al.*, 1994) with spawning biomass estimates from DEPM surveys, two egg production indices from CalCOFI ichthyoplankton surveys, aerial fish spotter data (see below), and sonar survey data to measure spawning biomass during 1963–1995 (Jacobson *et al.*, 1995).

In the Humboldt anchovy egg surveys have been undertaken in northern Chile since 1992 to assess the spawning biomass applying the DEPM in the area from the border with Peru to 26º S. The Peruvian portion of the anchovy stocks is monitored acoustically twice a year (see below), but the winter survey also includes DEPM estimates of biomass.

Southern Benguela anchovy spawner biomass was estimated using combined DEPM and acoustic summer surveys between 1984 and 1993 (Armstrong *et al.*, 1988; Shelton *et al.*, 1993). The DEPM estimates were considered unbiased and were used to scale acoustic estimates at a time when there was no information on the accuracy or applicability of the acoustic target-strength expression used (Barange *et al.*, 1996; Hampton, 1996). DEPM estimates are no longer provided, and fishery independent monitoring is limited to a program of acoustic surveys (see below).

Similarly, the Bay of Biscay anchovy stock is monitored by annual DEPM spring surveys since 1987 (Santiago and Sanz, 1992; Motos *et al.*, 2005) and acoustics (regularly since 1989, although surveys were also conducted in the early 1970s and 1980s; Massé, 1988, 1994, 1996). Both surveys provide spawning biomass and population-at-age estimates. The DEPM is taken as an absolute indicator of biomass, while the acoustic as a relative one. This survey based monitoring system provides population estimates by the middle of the year, when about half of the annual catches have already been taken; and provides no prediction of the state of the stock in the next year, since the bulk of it will consist of 1-year-old fish being born at the time the surveys take place.

The DEPM has been applied on the NW Mediterranean anchovy stock in 1993 and 1994 (Garcia and Palomera, 1996), and experimentally in the monitoring of the southern Adriatic fishery (e.g. Casavola, 1998). However, neither program is applied routinely. Direct biomass estimates for anchovy in the Aegean Sea through acoustic and/or DEPM surveys have been obtained within the framework of various European and National projects (Machias *et al.*, 1997, 2000, 2001; Somarakis, 2005; Somarakis *et al.*, 2002b, 2004), but lack a regular (in time and space) time series. Since 2003, the Greek National Program for Fisheries data collection monitors the Aegean Sea anchovy stock, using the DEPM and acoustics, which are applied concurrently during the June spawning peak (Somarakis *et al.*, 2005a). Experimental DEPM surveys have also been performed on the Black Sea anchovy for the period 1987–1991 by the USSR, Bulgaria and Romania (Arkhipov *et al.*, 1991). Regular Black Sea anchovy pre-recruit surveys have also been carried out by the former USSR (now Ukrainian) institute YugNIRO, Kerch from early 1960s to 1993 (Tkacheva and Benko, 1979; Arkhipov, 1993).

Sardine stocks

Egg census and pre-recruit surveys are used as predictors of Japanese sardine SSB and recruitment, respectively (Nishida *et al.*, 2006). Monthly egg production census surveys have been conducted since 1949. From these surveys the extension of the sardine spawning area is calculated as a measure of SSB by integrating the areas in which early developmental stage eggs are present (Zenitani and Yamada, 2000). Since 1996, pre-recruit surveys have been carried out in May–June in KOTZ,[13] which is assumed to be the key area of recruitment (Watanabe and Nishida, 2002). A temperature-weighted pre-recruit index corresponds well to the VPA-derived recruitment estimate.

The California Cooperative Oceanic Fisheries Investigations (CalCOFI) survey has collected ichthyoplankton, including California sardine eggs and larvae, since 1951. The present CalCOFI sampling grid is smaller and less frequently sampled than the original design – quarterly surveys now span from San Diego to Avila Beach, California. However, the range is expanded northward each April to sample a significant portion of the sardine's spawning habitat. A NOAA synoptic survey from British Columbia to the US–Mexico border was conducted in April–May, 2006, while Mexico sampled areas off the Baja California coast, in the first tri-national stock assessment cruise. Sampling included oblique bongo and vertical egg net tows, Continuous Underway Fish Egg Sampler (CUFES; Checkley *et al.*, 1997), and trawling for data on reproductive parameters and age composition. Data are used in the DEPM (Lo *et al.*, 1996, 2005). Adult reproductive parameters have been averaged across years, a source of bias caused by limited adult sardines in the sampling. A time series of DEPM-biomass estimates is available from 1986–1988 and 1994–2006, and is updated annually (Hill *et al.*, 2006). The IMECOCAL program (Investigaciones Mexicanas de la Corriente de California) was initiated off Baja California in 1997. Similar to CalCOFI, IMECOCAL conducts quarterly cruises at historic (pre-1985) CalCOFI stations. Sardine are a major focus of this program; however, the ichthyoplankton data are not yet incorporated into the US stock assessment. Lack of adult samples (i.e. reproductive parameters) from the Baja California region has thus far hindered reliable application of the DEPM approach.

The Chilean IFOP[11] monitors the distribution of eggs and larvae of the main pelagic fish stocks. A time series of a Humboldt sardine larval index has been constructed for the winter season, which has been used in stock assessment (Braun *et al.*, 2002). In addition, a conventional DEPM has been applied on the Humboldt common sardine from 2002 (Cubillos *et al.*, 2005). However, because of the short length of the series, DEPM estimates of spawning biomass are not yet integrated into common sardine stock assessment models.

Egg and larvae surveys (Matsuura, 1983, 1996, 1998) and DEPM (Rossi-Wongtschowski *et al.*, 1994; Rossi-Wongtschowski *et al.*, 1995) have been used to monitor the Brazilian sardine stock, but they are not applied in management (Saccardo and Rossi-Wongtschowski, 1991). Instead, catch data have provided the basis for the application of Surplus Production Models and Yield-per-Recruit Models, defining maximum catches since 1974 (Cergole, 1993). Virtual Population Analysis was applied for the period 1977–1997 (Cergole, 1993, 1995; Cergole *et al.*, 2002), including oceanographic and meteorological variables (Sunyé and Servain, 1998; Sunyé, 1999; Jablonski, 2003; Jablonski and Legey, 2004, 2005). Sporadic hydroacoustic surveys have also been conducted (Johanneson, 1975; Rijavec and Amaral, 1977; Castello *et al*, 1991; Madureira and Rossi-Wongtschowski, 2005).

The first application of the DEPM on the European sardine stock was conducted in 1988, although the method did not become a regular tool until 1997. Since 1999 it has been performed triennially (ICES, 2004, 2006e). As for the case of acoustic surveys, the different national sampling efforts are coordinated within the ICES framework (ICES, 2006e). Since 2002, CUFES is used as an auxiliary egg sampler, in addition to the standard station net sampling. Adult sampling is carried out opportunistically. DEPM-based spatially explicit estimates of SSB are experimentally obtained by spatial modeling as the population is known to show large spatial variability which can produce bias if unaccounted for (ICES, 2006e). Spatial estimates are generally consistent with traditional DEPM-based estimates, although their use in assessment is pending a general revision of the methods and data series.

DEPM has periodically been applied to each of the three sardine management units off southern Western Australia since the early 1990s (e.g. Fletcher *et al.*, 1996; Gaughan *et al.*, 2004). Scarcity of adult samples for some surveys has required the application of rules (e.g. sex ratio not to exceed 70%) and the use of reproductive data obtained outside of the survey period. Despite that Fletcher and Sumner (1999) determined the appropriate spatial scales for sampling sardine eggs; patchiness of eggs also remains problematic. Imprecision of SSB estimates has therefore been a problem, particularly since even acceptably low CV[12] can occur by chance. Nonetheless, the time series of DEPM-based SSB estimates have been crucial for managing sardine in Western Australia.

Hydroacoustic surveys

Hydroacoustic winter surveys targeting anchovy recruits have been conducted off Japan in the offshore area 35°–37° N / 141°–145° E to detect and estimate recruitment. From 2004 a second series of hydroacoustic surveys has been conducted to assess the stock biomass outside the fishing ground 35°–45° N / 145°–170° E in the spring season.

The Peruvian anchovy stock is monitored acoustically twice a year, in summer and winter, while in Chile hydroacoustic surveys have been conducted to assess the Humboldt sardine from 1981 to 1995. The surveys estimate the spawning biomass in winter during the spawning process (Castillo *et al.*, 1995; Castillo and Robotham, 2004). Since 1999 annual recruitment surveys are conducted in January for common sardine assessment (Castillo *et al.*, 2005).

Acoustic surveys form the cornerstone of the South African anchovy and sardine assessment program and have been in place since 1984 to monitor the biomass of the spawner stocks (combined with DEPM for anchovy until 1993) in November and recruitment in May/June (Hampton, 1992; Barange *et al.*, 1999). Coetzee *et al.* (2008) and de Moor *et al.* (2008) have updated the SSB time series, accounting for receiver saturation in old-generation echo sounders, more accurate target strength expressions, and acoustic signal attenuation by dense sardine schools. The recruitment survey is conducted at the earliest time possible to provide a reliable survey index of the abundance of the incoming anchovy and sardine recruits for that year, but this estimate is available only 2–3 months after fishing has commenced. The November surveys have also facilitated the collection of data on mean mass-at-age since 1990 for anchovy and since 1988 for sardine, and age–length keys are currently available for surveys conducted between 1992 and 1995 for anchovy and since 1988 for sardine. For other years, a combined 1992–1995 age–length key for anchovy has been used to obtain the proportion (by number) of 1-year-olds in the November surveys (De Oliveira, 2003).

In the northern Benguela hydroacoustic sardine surveys are conducted in autumn (February–April) to estimate adult abundance, and in spring (October–December) to both assess adult stock and provide an index of recruitment. Occasionally surveys have extended into Angolan waters since as much as 50% of the sardine adult stock has been found there in winter (Boyer *et al.*, 2001). In contrast to the southern Benguela where a survey grid of randomly stratified cross-shelf acoustic transects are surveyed by the research vessel only, surveys in the northern Benguela use a two-stage adaptive strategy, whereby systematic zig–zag transects are followed by pelagic fishing vessels that locate sardine school groups which are then assessed using closely spaced parallel transects surveyed by the research vessel.

In addition to the program of acoustic estimation of the size of the Bay of Biscay anchovy stock in spring mentioned above (Massé, 1988, 1994, 1996), an experimental program to survey anchovy juveniles was conducted in 1998 and 1999 in the Bay of Biscay (Uriarte *et al*., 2001, 2002; Carrera *et al*., 2006). This concluded that acoustic surveys at the end of summer and early autumn could provide a good index of pre-recruit abundance. Thus, since 2003 acoustic surveying of anchovy juveniles takes place annually in September–October in order to assess the biomass of anchovy juveniles that will enter the fishery the next year (Boyra *et al.,* 2005; Boyra and Uriarte, 2005). However, the time series is too short to allow a proper evaluation of its performance as a recruitment predictor and it is therefore not yet used for the management of the population.

Routine acoustic surveys of the Mediterranean anchovy have been carried out along the NW Mediterranean coast since 1993 (Abad *et al*., 1998; Liorzou *et al*., 2004; Alemany *et al*., 2002) directed towards recruits in the Catalan Sea (autumn) and to adults in the Gulf of Lions (summer). From 2002, these surveys have been incorporated in the Spanish and French National Programs for Fisheries Data collection. In the Adriatic Sea, anchovy has been acoustically monitored since 1975, covering the Italian territorial waters, i.e. the western half of the Adriatic (Azzali *et al*., 2002). The eastern side has not been covered regularly, but this has improved recently with the participation of Croatian scientists (Tičina *et al*., 2005). Despite this, monitoring of the Adriatic Sea stock has been based mainly on analytical assessments using fishery data. As mentioned above, an acoustic monitoring program for the Aegean Sea anchovy was implemented in 2003.

In the Black Sea regular hydroacoustic or mid-water trawl surveys were performed by the former USSR (in collaboration with Bulgaria and Romania) from 1980 to 1992. Regular pre-recruit surveys have been carried out by the former USSR (now Ukrainian) institute YugNIRO (Kerch, Crimea) from early 1960s to 1993 (Tkacheva and Benko, 1979; Arkhipov, 1993). Sprat biomass has been assessed using mid-water trawl surveys in 1967–1993 (Ivanov and Beverton, 1985; Prodanov *et al*., 1997). The anchovy stock has been monitored using hydroacoustics in Turkish waters (southern Black Sea) for the period 1990–1994.

Acoustic surveys dedicated to the estimation of the European sardine distribution and abundance started in Spain and Portugal during the early 1980s. Off northern Spain, surveys have been performed annually in spring and continue to the present day. Off Portugal, acoustic surveys were carried out every 6 months (spring and autumn) during the period 1984–1988, were interrupted until 1992, and from 1995 onwards are always performed in spring and in most years during autumn. Survey plans and methods from Spain and Portugal have been coordinated in different ICES Planning and Working groups (ICES, 1998b; ICES, 2006e), and since 2000 are also coordinated with the French survey in the Bay of Biscay (ICES, 2006e). In summary, acoustic sampling is carried out following a systematic transect-based survey sampled only during daytime (since the mid 1990s). Opportunistic adult sampling is conducted to identify eco-traces, species biological and demographic characteristics, age–length keys, and so on. Adult sampling is performed using mainly pelagic (Spain) or both pelagic and bottom trawls (Portugal), although opportunistic samples from purse seiners are sometimes used to gather additional information. Following general trends within ICES, acoustic surveys in the Iberian Peninsula have gradually evolved into pelagic community surveys, and a series of pelagic species are now monitored concurrently.

The ICES-coordinated North Sea acoustic surveys began in the late 1980s to estimate the abundance of herring and sprat during summer. The surveys were originally targeted at herring, but from 1996 onwards records of sprat were also processed. However, the coverage of the surveys did not include the complete distribution of sprat, and so they were modified in 2003. These survey appear to give consistent results in terms of the numbers of sprat aged 1+ and herring 2+, but estimates of the younger groups show very high year effects. The time series of sprat is at present too short to be used in any assessment, but herring estimates are used.

Baltic Sea sprat and herring abundances are derived from annual autumn hydroacoustic surveys targeting both species (ICES, 2006b). The hydroacoustic surveys cover the entire stock areas, and all age groups. The open sea survey is internationally standardized; calibration of equipment and ship intercalibrations are performed at each survey. To differentiate between sprat and herring, approximately 100–150 identification trawl sets are carried out. Other abundance indices not used in the present assessment of sprat are hydroacoustic surveys conducted in May/June since 2001 (ICES, 2006b) and egg surveys (Köster *et al*., 2003b) covering both the central stock distribution area. Herring larvae and juvenile surveys have also been conducted more or less regularly in specific areas of the Baltic, but none is used in the assessment.

Finally, the Norwegian spring spawning herring stock is one of the most acoustically monitored pelagic resources worldwide. Surveys are conducted in the overwintering areas (November–December 1992–2004, but also in January 1991–1999), the spawning grounds

(February–March 1988–2005) and the feeding areas (April–June 1996–2005). These surveys target the adult stock (age classes >3–15+). There are also acoustic surveys for the 1–2 y.o. conducted in the Barents Sea in May–June (1991–2005) and August–September (1974–2004) (ICES, 2005b). In addition, a mid-water trawl survey is conducted in August–September (1974–2004) in the Barents Sea to estimate the size of the 0 age class, and an extensive tagging program started in 1975 (ICES, 2005b). A time series of acoustic estimates of 0 group herring from the Norwegian fjords and coastal areas, running from 1975 to 2004, is available, but this is not used in the stock assessment.

Other monitoring surveys

Some pelagic fish, such as herring and sprat, are also surveyed through the ICES coordinated Bottom Trawl Surveys in the North Sea (IBTS) and the Baltic (BITS). The IBTS was originally set up after the collapse of North Sea herring to monitor the recovery of the stock (Heessen *et al*., 1997) but it covers the distribution of sprat as well. The survey uses the GOV (Grande Overture Verticale) trawl in the first and third quarter of the year. The results are used for both following the trajectory of the herring stock, and for catch forecasts of sprat for within-year management. However, catchability fluctuates as a function of hydrographic condition, which is also the reason why catch rates of Baltic herring and sprat available from first and fourth quarter BITS (ICES, 2006b) are presently not used in stock assessment.

Surveys of newly hatched larvae are also used in the assessments of North Sea herring and Norwegian spring spawning herring. The time series run from 1973 and 1981, respectively. Both series are used as indices of spawning stock potential in the assessments. Conversely, surveys of post-larval herring (1992 to present) are used as indicators of recruiting year classes in the North Sea (see Nash and Dickey-Collas, 2005).

In addition to egg production and acoustic surveys, additional surveys are in place to monitor abundance and biology of adult Japanese sardine: driftnet surveys during spring–summer in the Hokkaido area since 1994, longitudinal transects at 155°E, 175°E and 180°E since 1982 (Yatsu *et al.*, 2003), and surface trawl surveys during winter and spring in the Kuroshio, Oyashio and KOTZ since 2001 (Nishida *et al*., 2006).

Aerial fish spotting data have been used in California and Namibia in recent decades. California's fishing fleet uses spotter pilots to locate pelagic fish schools. Data for each flight have been provided to the US National Marine Fisheries Service (NMFS) since 1962 (Squire, 1972). Spotter logbooks contain data on species, school size (tonnes), effort, and location (10 × 10 nm statistical areas). Lo *et al.* (1992b) developed a standardized index of relative abundance, estimating year effects using delta lognormal linear models. The current index for California sardine includes data from 1986 through 2005. After the year 2000, there was a rapid decline in both the number of active pilots and total logbooks returned, as well as a southward shift in effort to offshore areas of Baja California in response to an increase in the tuna net-pen fishery. To remedy this problem and continue the time series, NMFS has contracted professional spotter pilots to survey the Southern California Bight region in 2004 and 2005 and the newly available data have been included in the index. Since the spotter pilots operate in the area of the fishery, i.e. within 30 nm from shore, the index is assumed to represent relative abundance of younger sardine (ages 0–2) and is treated as such in the stock assessment model.

A preliminary program of aerial pelagic fish spotting using aircraft-borne remote sensors was also applied in Namibia in the 1970s, driven by concerns that the extreme patchiness of the shoals and their tendency to avoid vessels may invalidate acoustic surveys of the stock. The program localized and measured shoals at night-time through bioluminescence, while a vessel made synchronous measurements of shoal thickness and packing density (Cram and Hampton, 1976). The expected result was the provision of unbiased absolute estimates of pelagic fish, but it did not survive the experimental phase.

Stock assessment, modeling, and harvest strategies

The assessment and management of small pelagic fish stocks does not fit well into the traditional population dynamics models and assumptions. Beverton (1990) concluded that small shoaling pelagic fish species are the most unreliable and vulnerable to unrestrained fishing. This is mostly because their high natural mortality (around 1 y^{-1} or larger), short lifespan and dependency on annual recruitment pulses results in high population variability and a tendency to go from boom to bust in a short time span. In addition, variability in catchability coefficients means that catch information cannot be reliably used to estimate biomass (Csirke, 1988), thus limiting assessment options. On the other hand, pelagic fisheries are highly targeted, and thus generally do not have the management problems that mixed fisheries pose. In this section we reviewed present approaches to outline the state of the art in small pelagic assessment and management.

Japanese stocks

Japanese anchovy and sardine stock assessments are conducted annually in support of the fishery management process. VPA and survey results are used in this assessment (Table 9.2a,b).

The TAC for the Pacific stocks of Japanese anchovy and sardine for year *t* are determined in November of year *t*–1, on the basis of allowable biological catches (ABC) and socio-economic factors. ABC in year *t* is recommended on the basis of projected stock abundance, SSB, recruitment (age-0 fish) for year *t* and either limiting or targeting fishing mortalities, i.e. F_{limit} or F_{target} for both species. Abundances-at-age in year *t* are predicted using a forward VPA from year *t–2* with prognoses of SSB and recruitment in years *t−1* and *t,* assuming fishing mortality in year *t–1*. In the case of anchovy the relationship between SSB and recruitment is significant and its linear relationship is used to forecast future recruitments. In the case of sardine Recruitment-Per-Spawner (RPS) is negatively correlated with sea surface temperature (SST) in the Kuroshio Extension South Area (KESA) in winter (Noto and Yasuda, 2003). An extended Ricker model has been proposed by including winter SST in the KESA (Yatsu *et al.*, 2005). Suda and Kishida (2003) developed a recruitment model of Japanese sardine by incorporating transportation, prey condition, inter-species competitions and predation. These models, however, have not been applied in management. The sardine management target in recent years has been set to 222 000 t SSB, a magnitude at which relatively stable recruitment has been observed (Nishida *et al.*, 2006). The abundance estimate of pre-recruits from spring surveys in the KOTZ is used to predict recruitment level. As 8–10-month-old recruits are targeted together with older fish by the fishery these estimates are included in catch prognosis and quota determination (Watanabe and Nishida, 2002). Owing to uncertainties in SSB and recruitment predictions, the TACs are reviewed within each season when additional information from surveys and commercial catch becomes available.

Japanese catch statistics indicate a sardine-anchovy replacement, with historical peaks of anchovy around 1960s and the present, and 1930s and 1980s in the case of sardine. The current catch level of anchovy is the maximum during the last 25 years, but given its low landing price, it is unlikely that the Japanese anchovy will become the major target species for the purse-seine industry. The current sardine catch level is approaching the historic minima.

California sardine and anchovy

The California anchovy fishery is monitored at this time but not actively managed because harvest levels and demand are low (< 10 000 t per year during 1991–1995; PFMC, 2005). Under these conditions, a "default" ABC level of 25 000 t is allowed annually in US waters (Table 9.3a). Active management will be required if catch levels rise above the ABC.

In the 1980s, just after stock biomass and fishery landings declined from record highs, the anchovy fishery was managed actively using a harvest control rule that was the first of its kind (PFMC, 1983). The rule allowed very low catches at spawning biomass levels below a pre-specified cut-off level. As spawning biomass increased above the cut-off, catches were allowed to increase up to pre-specified maximum. DEPM spawning biomass estimates were used initially to set harvest levels but proved too expensive to carry out on an annual basis. In lieu of annual DEPM estimates, forward-casting stock assessment models, including the original "Stock Synthesis" model (Methot, 1989), were developed and used to estimate spawning biomass based on a diverse range of data (Jacobson *et al.*, 1994).

Murphy (1966) developed the age-based cohort analysis now known as VPA, first applying it to California sardine. Deriso *et al.* (1996) applied the forward-projecting approach (CANSAR[14]) to contemporary sardine data; Hill *et al.* (1999) modified CANSAR into a two-area model. The population model currently used for assessment and management of the US fishery is called "ASAP" (Age-Structured Assessment Program; Legault and Restrepo, 1999), a forward simulation approach (Table 9.3b). The population dynamics and statistical underpinnings of ASAP are well established (Fournier and Archibald, 1982; Deriso *et al.*, 1985). The current sardine ASAP model includes catch and age compositions for three fisheries (Ensenada, California, Pacific Northwest), as well as the aerial spotter and DEPM indices of relative abundance described above (Hill *et al.*, 2006). Modeled time series begin in 1982–83. The current model does not include SST as a term in the S–R model (as was done with CANSAR): however, the next modeling platform for sardine, Stock Synthesis 2 (Methot, 2005), will likely resume including environmental data.

ASAP's estimation approach is that of a flexible forward simulation allowing for the efficient and reliable estimation of a large number of parameters using the maximum likelihood method. The current ASAP model for sardine includes nine likelihood components and a few penalties: Selectivity in 1st Year, F_{mult} in 1st Year, Catchability in 1st Year, Stock-Recruitment Relationship, Steepness, F_{mult} Deviations, Recruitment Deviations, and Selectivity Deviations.

Periods of warm SST in the California Current Ecosystem are associated with good recruitment and higher productivity for California sardine (Ahlstrom, 1960; Jacobson and MacCall, 1995; Jacobson *et al.*, 2001). Sea-surface temperature (SST-SIO) measured at the Scripps Institution of Oceanography pier (La Jolla, California) since 1916 is a good proxy for environmental conditions influencing positive or negative surplus production in California sardine. The US management approach is unique in that it uses a harvest control rule that depends on a 3-year running average of SST–SIO (PFMC, 1998). In addition, the same SST series has been directly incorporated in the S–R function of the stock assessment model (Deriso *et al.*, 1996; Hill *et al.*, 1999). The ASAP assessment for sardine uses projected catch for the coming season in order to calculate population abundance at the start of the final time step. This provides estimation of B_t for calculating H_{t+1} with the harvest control rule (see OMP[15]).

The harvest control rule used by the Pacific Fishery Management Council (PFMC, 1998) has the following form: $H_{t+1}=(B_t-E)\ U * f$, with $H_{t+1} < H_{MAX}$; where H_{t+1} is the harvest guideline for year $t+1$, B_t is the stock biomass estimate at the beginning of the previous year t, E is a minimum escapement level (150 000 t), U is the exploitation rate at F_{MSY}, f is the fraction of the stock assumed in US waters, and H_{MAX} is the maximum allowable harvest level (200 000 t). In an attempt to make the control rule responsive to environmental forcing, control rules were constructed with U_t for each year based on a regression function relating U_{MSY} to a range of average SST values (Jacobson and MacCall, 1995). $U_{MSY,}$ is constrained to range 0.05–0.15 y^{-1}.

Humboldt fisheries

In Peru anchovy assessments consist of direct methods (acoustics and DEPM) as well as VPAs that incorporate fishery data. VPAs have been applied in the N/C Peru stock for the period 1953–1985 (Pauly and Palomares, 1989), 1960–1994 (Csirke *et al*., 1996), and for the SP/NCh stock for the period 1984–2001 (GTE, 2002; Table 9.2a). For the assessment of the Peruvian–Chilean anchovy stock a statistical catch-at-age model was developed (Serra *et al*., 2004, Table 9.3a). The objective function of this model is the sum of the log-likelihood terms for the catch in tonnes, catch-at-age, spawning biomass, CPUE and penalizations of the parameters of the selectivity model and recruitment. Parameter estimates are found by minimizing the residual sum of squares for each terms of the objective function. The management of the anchovy fishery off Peru aims at maintaining a spawning biomass of 5 Mt at the beginning of the spawning periods in August and February. The management is flexible and short term and comprises disaggregated TAC in time and space, closures during spawning periods, and short-term closures to protect areas with a high proportion of juvenile fish. Anchovy in northern Chile is managed under quota since 2002 through a "maximum catch limit per ship owner" and also by closed periods during the spawning and recruitment seasons. The TAC is estimated under a constant F policy and medium term projections with uncertainty and risk analysis of different scenarios for recruitment (Serra *et al*., 2004; Table 9.3a).

The South Peru–North Chile sardine stock has been assessed by age-structured Sequential Population Analysis, in which an extended version of ADAPT was used (Serra *et al*., 2004; Gavaris, 1988; Conser and Powers, 1990; Table 9.3b). The abundance indices used for the assessment were standardized CPUE and a larval index. The CPUE time series was obtained through a GLM[16] following the approach proposed by Stefánsson (1996). The larval index is from winter surveys from 1985 to 2001 and was obtained from Braun *et al*. (2002). This fishery historically operated under an open access policy. The Chilean fishery in Northern Chile was managed through closed season to protect the spawning stock and size restriction. A catch quota was applied in 1982–1983 but was resisted by the industry. The criterion to estimate the quota was a constant F policy. Since 2002 a catch quota has been applied on the Chilean fishery.

The stock assessment model for common sardine consists of a general population dynamics model to predict catch and age composition, as well as catch-per-unit effort and hydroacoustic biomass (Table 9.2b); likelihood functions for the observed data; priors and penalties to constrain parameterization, and parameter estimation by minimization of an objective function (Table 9.3b). Catch data are expressed by weight and must be available for each time period. In terms of the population dynamics component, standard survival and catch equations are used to describe changes in the age structure of the population. The parameterization used in the models allows for a separation assumption in fishing mortality. Selectivity at age is allowed to vary, but treated by block of years in which they remain constant. The harvest strategy consists of computing a TAC by using a fixed fishing mortality rate defined at the level of 60% of the spawning biomass per recruit. Short-term projections of abundance are conducted under uncertainty, and the performance function is a ratio between the spawning-stock biomass at the end of the projection period and the current spawning-stock biomass. Risk is defined as the probability that fishing mortality in the short term will exceed a target fishing mortality for a range of alternative and equally probable quota options in the short term. A level of 10% of risk is usually taken into account. A second review of the TAC is carried out after the hydroacoustic survey, because of uncertainty in the short-term recruitment projection (Table 9.3b).

The Benguela operational management procedure (OMP) approach

Age-structured production models are used to assess southern Benguela anchovy and sardine stocks. These assume lognormal distributions of recruitment about a hockey-stick (single-sloped) stock–recruitment relationship (base case operating model). Beverton–Holt and Ricker stock–recruitment curves were used in alternative operating models (or assessments), on which the management procedure was tested. The most recent assessment, conducted using data from 1980/81 up to 2003, was Bayesian (Cunningham and Butterworth, 2004a; Table 9.3a,b).

As mentioned above, acoustic estimates of anchovy and sardine SSB are considered relative, while DEPM estimates of anchovy SSB are considered absolute. A probability density function (pdf) for the overall bias in the November survey estimate of sardine SSB has been calculated and the median was used for the base case run, suggesting a 28% underestimation of the stock.

A key feature of the South African pelagic fishery is the problem of juvenile sardine taken as bycatch in the anchovy fishery. This problem arises because the two species school together as juveniles (Crawford *et al.*, 1980) and means that catches of the two species cannot be simultaneously maximized. For this reason, a joint OMP is used for the sardine and anchovy fisheries (Table 9.3a,b). The anchovy fishery is managed through a TAC, with an initial TAC set at the start of the fishing season based on the biomass estimated from the preceding November (SSB) survey, and assuming forthcoming recruitment will be the median of observed values (De Oliveira and Butterworth, 2004). Because the bulk of the coming year's catch consists of incoming recruits, the initial TAC is revised as soon as an estimate of recruitment becomes available from the mid year recruit survey. The computation of the revised or final TAC replaces the median recruitment assumed earlier with the actual estimate from the recruit survey. An "additional sub-season" extending from August until the end of the year was introduced into the anchovy fishery in 1999, with the goal of targeting anchovy that no longer shoal with juvenile sardine. Targeting such "clean" anchovy would minimize the juvenile sardine bycatch, thereby optimizing long-term sardine catch. This sub-season TAC is set using the same data and at the same time as the revised TAC for the normal season, after the mid-year recruit survey.

The southern Benguela sardine fishery is managed by TAC and Total Allowable Bycatch (TAB). The TAC is set at the start of the fishing season based on the biomass estimated from the preceding November (SSB) survey (De Oliveira and Butterworth, 2004). Since anchovy and sardine shoal together during their first few months of life, directed fishing for anchovy is accompanied by a bycatch of juvenile sardine. The initial sardine TAB is based on a fixed tonnage for bycatch (mainly adults) linked to a small fishery for round herring (*Etrumeus whiteheadii*), and a component proportional to the initial anchovy TAC, using a conservative estimate of the ratio of sardine to anchovy juvenile fish. As for the anchovy TAC, the sardine TAB is revised after the recruit survey using an updated estimate of the ratio of sardine to anchovy juveniles. A TAB for the "additional sub-season" (see preceding section on southern Benguela anchovy) from August onwards is proportional to the additional sub-season anchovy TAC, subject to a maximum of 2000 t.

The South African anchovy stock is a worthy candidate for investigating the use in management procedures of environmental indices that predict recruitment, because it is a resource that sustains a recruit fishery for which a measurement of recruitment is not available until after substantial fishing on that recruitment has already taken place (Butterworth *et al.*, 1993). Recent analyses have concluded that, under existing management constraints, an environmental index needs to explain roughly 50% or more of the total variation in anchovy recruitment before a management procedure which takes account of such information starts to show benefits in terms of risk and/or average catch (De Oliveira and Butterworth, 2005).

Assessments for northern Benguela sardine are conducted using an age-structured model in which age classes are either assigned from length frequency data or following cohort analysis, and a Ricker stock–recruitment curve is fitted to the data (Fossen *et al.*, 2001). More recently, simple age-structured models have been investigated, covering a range of uncertainties regarding key assumptions, and these have been proposed as operating models to form the basis for future development of OMPs for Namibian sardine (De Oliveira *et al.*, 2007). Currently though, management of the northern Benguela sardine fishery off Namibia is via a TAC based on a projected fishing mortality ($F = 0.2$ y^{-1}) of the fishable stock (fish >16 cm; Boyer and Hampton, 2001). Fishing is also temporally limited, with the fishing season generally opening in March and closing at the end of August (Boyer *et al.*, 2001). Sardine bycatch taken in fishing directed at anchovy or horse mackerel (*Trachurus trachurus*) was not previously included as part of the TAC but has been included since 1990, a year when a bycatch of 29 000 t was taken in addition to a TAC of 60 000 t. Socioeconomic concerns (i.e. keeping processing factories operating) resulted in TACs being set in the mid 1990s that were higher than had been scientifically recommended, and this, together with permission being granted from 1994 onwards for Namibian vessels to catch unlimited quantities of sardine from Angolan waters (and outside of the closed season), meant that catches of northern Benguela sardine were at unsustainably high levels on several occasions during the past decade (Boyer *et al.*, 2001). This resulted in further decreases, both in TACs set and catches attained (Fig. 9.3b), and culminated in the setting of a zero TAC in 2002, although small TACs of 20 000–25 000 t have since been set.

ICES stocks

ICES advice is based on the work done by research organizations in member countries, which contribute to ICES through data collection and analysis in the institutes and through participation in ICES expert groups. The outcomes of these analyses are translated into operational advice by ICES advisory committees which include scientists from all ICES member countries (ICES, 2005d). The stocks considered in this review are assessed using models in the VPA family. Three (Bay of Biscay anchovy, European pilchard, and North Sea herring) are managed through Integrated-Catch-at-Age models (ICA, Patterson and Melvin, 1996). In contrast to conventional VPA, which assumes that catches at age are measured without error, ICA models consider catches at age to be measured with an error that has an independent lognormal distribution. One advantage of this type of model is that approximate estimates of the variance of many of the estimated parameters can be calculated. ICA also allows catch information to be run against biomass (non-aged) survey data, such as egg, larvae, and total acoustic biomass indices. Baltic Sea sprat and herring are assessed by the Extended Survivor Analysis model (XSA; Shepherd, 1999), a method that partially solves the sensitivity to observation error in the final year when tuning virtual population analysis using abundance indices, while it also makes efficient use of previous estimates of year-class strength. XSA allows for the simultaneous analysis of several sets of abundance indices and for non-linear relationships between abundance indices and population size for the younger age groups. North Sea sprat is assessed using Catch-Survey Analysis (CSA; Mesnil, 2003), an assessment method that aims at estimating absolute stock abundance given a time series of catches and of relative abundance indices, typically from research surveys, by filtering measurement error in the latter through a simple two-stage population dynamics model. Finally, Arcto-Norwegian spring spawning herring stock uses SeaStar, a model specifically developed to use both survey and tagging data to tune long-time catch series.

Bay of Biscay anchovy

The current ICA provides a Maximum Likelihood Estimator (MLE) of recruits and fishing mortality by tuning catch-at-age and direct biomass and population-at-age estimates from surveys (DEPM and acoustics surveys; Table 9.2a). The acoustic estimates are treated as relative and DEPM as absolute. The assessment assumes a constant natural mortality of 1.2 y^{-1} and a separable model of fishing mortality by age, applied over a period of the last 15 years. The most abundant age classes (catches at ages 1 and 2) receive higher weighting factors than other ages. The operating model of this assessment is therefore an age-structured model with the classical survivors and catch equations defining its dynamics within the frame of a separable model of fishing mortality (Table 9.3a). The assessment with ICA is heavily influenced by the surveys because the short lifespan of anchovy does not allow for a proper VPA convergence. In recent years, in addition to ICA, a simpler biomass delay-difference model (Schnute, 1987), based on the model applied to squid by Roel & Butterworth (2000), is being applied with successful results via Bayesian approach (Ibaibarriaga *et al.*, 2005). The model seeks to estimate recruitment at age 1 at the beginning of each year (in mass) accounting for the signals of inter-annual biomass variations obtained from the direct surveys (DEPM and acoustics) and the level of total catches (in tonnes) produced each year. The ICES working group considered that the biomass model is as appropriate as ICA (with less risk of over-parameterization) and it is intended to be adopted as the standard assessment for this species in a next future.

Although a stock/recruitment relationship is not explicitly used, for management purposes it is assumed that a link exists below 21 000 t. This value coincides with the minimum spawning biomass level B_{lim} (set at 21 000 t, the lowest observed biomass in the 2003 assessment) below which the dynamics of this stock is unknown. Management is aimed at keeping biomass above B_{lim} (ICES, 2003b). This led ICES to propose a precautionary level of spawning Biomass (B_{pa}) around 33000 t once assessment uncertainty and natural variability are accounted for.

No proper management plan or harvest control rules are adopted for this fishery. At the end of the year, when the TAC is set and the fishery starts, direct estimates of recruitment are not available. Therefore advice on the short-term development of the fishery has been based on assumptions rather than assessments of the strength of recruitment. Managers have not followed this advice which was usually based on assumed recruitment levels (usually precautionary – low levels of recruitment).

The development of the fishery until the 1970s was conducted without any management. Since 1979 a precautionary TAC for anchovy of about 30 000–33 000 t has always been agreed for the international catches of anchovy (except at the start of 2000 and 2006). Recent scientific advice warning about continuous recruitment failures since 2001 has not been considered. In 2005 the complete failure of the fishery and low levels of spawning biomass led to the closure of the fishery.

In the late 1990s the possibility of forecasting anchovy recruitment on the basis of environmental indices was explored, thus potentially enhancing the TAC setting at a time when year-class strength is still unknown. Using historical data Borja *et al.* (1996, 1998) concluded that recruitment can be favored by the occurrence of spring wind-driven upwelling along the French and Spanish coasts, close to anchovy spawning grounds. The statistical link was supported by Allain *et al.* (2001) using a 3D hydrodynamic model of the Bay in which upwelling intensity was deduced from modeled vertical water velocities rather than from geostrophic wind calculations. Using results from the 3D hydrodynamic model, Allain *et al.* (2001) observed that strong wind events during summer (June–July) provoke a breakdown of the water-column stratification, potentially reducing recruitment. Allain *et al.* (2001, 2004) proposed an environmental model that combines upwelling intensity

Table 9.2a. *Monitoring information and data input: anchovy stocks*

		Fishery data			Fishery-independent data		
Species	Stock	Parameter	Time series	Frequency	Parameter	Time series	Frequency
Japanese anchovy (*Engraulis japonicus*)	NW Pacific	Length frequencies	1978–2004	Monthly	Hydroacoustic surveys	2002–2004	2 × year
		Landings	1978–2004	Monthly	Egg surveys	1978–2004	Monthly
		CPUE	1978–2004	Monthly	SST	1978–2004	Monthly
	Seto Inland Sea	Length frequencies	1981–2004	Monthly			
		Landings	1955–2004	Monthly	Egg surveys	1981–2004	Monthly
		CPUE	1993–2004	Monthly	SST	1981–2004	Monthly
	Tsushima Current	Length frequencies	1991–2004	Monthly	Hydroacoustic surveys		1 × year
		Landings	1960–2004	Monthly	Egg surveys	1991–2004	Monthly
		CPUE	1991–2004	Monthly	SST	1991–2004	Monthly
California anchovy (*Engraulis mordax*)	Central sub-population	Length frequencies			Hydroacoustic surveys	1968–1984	Annually
		Landings	1963–2005	Annually	Egg and larval surveys	1963–2005	Various
					SST (Scripps Pier)	1963–present	Daily
Humboldt anchovy (*Engraulis ringens*)	North-Central Peru	Length frequencies	1953–2005	Monthly	Hydroacoustic surveys	1980–2005	2 × year
		Age-length keys	2000–2005	Quarterly	Egg surveys	1999–2005	1 × year
		Landings	1950–2005	Monthly	SST	1925– 2005	Monthly
		CPUE	1960–2005	Annually			
	South Peru–North Chile	Length frequencies	1984–2005	Monthly	Egg surveys	1992–2005	1 × year
		Age-length keys	1984–2005	Quarterly			
Benguela anchovy (*Engraulis encrasicolus*)	Southern Benguela	Length frequencies	1988–2005	Monthly	Hydroacoustic surveys	1984–2005	2 × year
		Landings	1981–2005	Annually	Egg surveys	1984–1993	1 × year

European anchovy (*Engraulis encrasicolus*)	Bay of Biscay	Catches at age	1987–2005	Quarterly	SSB Hydroacoustic surveys	1989–2005	1 × year
		Landings		Monthly	Egg surveys (DEPM)	1987–2005	1 × year
					Acoustic survey on juveniles (not used in assessment)	2003–2005	1 × year
	NW Mediterranean	Length frequencies	2001–2005	Monthly	Hydroacoustic surveys	GL(summer): 1993–2005 CS (autumn): 1996–2005	1 × year
		Landings	1985–2005	Monthly	Egg surveys	1993–1994	1 × year
		CPUE	2000–2004	Monthly			
	Adriatic Sea	Length frequencies	1975–present	Monthly	Hydroacoustic surveys	1976–present	1 × year
		Landings	1975–present	Monthly			
		CPUE	1975–present	Daily for one port			
	Aegean Sea	Length frequencies	2003–2006	Monthly	Hydroacoustic surveys	2003–2006	1 × year
		Landings	1964–2006	Monthly	Egg surveys	2003–2006	1 × year
		Discards	2003–2006	Quarterly			
		CPUE	1996–2006	Monthly			
	Black Sea	Size and age	1949–2005	Monthly	Hydroacoustic surveys	1980–1994	1 × year
		Landings	1925–2005	Annually	Egg surveys	1987–1991	1 × year
		CPUE	1984–2005		Juvenile fish surveys	1980–1993	1 × year
					SST	1925–2005	Monthly

and stratification breakdown events, which explained about 75% of the inter-annual variability between 1986 and 1997. In 1999 the ICES assessment working group used Borja's upwelling index to predict a failure of recruitment at age 1 for 2000. However the predictions failed and the reduction of the initial TAC adopted by managers turned out to be unnecessary. The TAC was raised to 33 000 t on July 1, after direct estimates of SSB from surveys became available. This experience caused intensive debates among the management bodies and the scientific advisers, not only about the use of environmental indices to predict recruitment but also about the nature of the advice required for this short-living species. The ICES working group decided to abandon the use of the environmental indices given the limited predictive power of the simple model used. In a recent review (ICES, 2005f) the performance of the environmental models up to 2003 were again revisited and concluded that the available models (both Borja's and Allain's) offer poor predictive power.

Of late ICES has proposed a two-stage TAC regime based on a preliminary TAC at the beginning of the year (from an analytic assessment in the fourth quarter), and a revised TAC according to measurements of the stock by acoustic and DEPM surveys in May–June (Table 9.3a). The preliminary TAC at the beginning of the year was aimed at keeping the stock safely above B_{lim} even if the incoming year class is poor (ICES, 2001a, 2003a). However, the two-step TAC regime was not followed by managers before the collapse of the fishery in 2005.

European sardine

Routine assessment of the Atlanto-Iberian stock of sardine has been conducted annually since 1982, currently based on a data series since 1978. In the late 1990s, assessment was based on the ICA model (Table 9.3b), using age-disaggregated catch data from the fishery (0 to 6+ age group), three CPUE series from Galicia (abandoned in 1999 due to difficulties in estimation of changes of effort), three incomplete series of acoustic estimates (relative index of stock abundance from spring series from Spain and Portugal, autumn series from Portugal) and a few point estimates of spawning biomass from the DEPM (absolute index of abundance from combined Iberian spawning biomass estimates; Table 9.2b). A natural mortality of 0.33 (Pestana, 1989) has been used as a fixed value for all ages and years. During the late 1990s, survey indices (acoustics and DEPM) and catch data from Spain and Portugal provided contradictory indications, showing a marked stock decline in Spanish but no indication of change in Portuguese waters.

Although the assessment model used at the time (ICA) allowed for a change in exploitation pattern within the study period, it did not allow for the existence of independent biological units within the stock area or for changes in the relative distribution and abundance within the study period. Finally, there was a bulk of evidence about wider environmental changes in the western coast of the Iberian Peninsula during the 1990s, which in some cases were linked to larger scale inter-decadal changes in hydrological and climatic conditions in the North Atlantic. These issues were reviewed in a dedicated ICES Workshop in 1998 (ICES, 1998*c*) and as part of the ICES Working Group on sardine assessment in 1999 (ICES, 2000) but with inconclusive results.

In the 2002 assessment, a new and more flexible catch-at-age model and software (Assessment Model Combining Information – AMCI) was used to explore the input data and assumptions underlying the ICA assessment. Results from the exploratory analysis detected violations of assumptions regarding survey catchability and fishery selectivity, which pointed to structural uncertainties in the stock assessment. AMCI provided a description of the stock dynamics which was more consistent perceived spatio-temporal variability, and so AMCI was adopted as the standard assessment model in 2003.

Structural uncertainties in the models do not change perceptions of the present level of the stock, but question the evolution of the biomass from the late 1980s to the present. This uncertainty has prevented the setting of reference points for management purposes, as well as the setting of official harvest control rules. However, national management measures have been taken since the decreasing SSB trend observed in the late 1990s (Fig. 9.3b). These national management measures mainly considered the control of fleet size, a reduction in fishing effort and the implementation of stricter catch limits (in many areas translated to a maximum daily catch per vessel).

Catch predictions for management purposes are only carried out for the short term, using a deterministic projection assuming the parameters estimated by the chosen assessment model, with corrections for the estimation of last year recruitment and with subsequent recruitments estimated as the geometric mean of the time series. Nevertheless, results from the predictions are not considered very reliable, both due to uncertainties in the recruitment prediction and to the inadequacy of any equilibrium assumption. Unlike most other European stocks, assessment advice is not translated into TAC and national quotas, but is left to the governments of Portugal and Spain to define annual catch limits for the national fleets and distribute them among producer organizations and non-associated vessels.

North Sea autumn spawning herring stock

The stock assessment is carried out by the ICES herring assessment working group. In 2006, a comprehensive review of the assessment took place (ICES, 2006d), and concluded that the assessment was robust to scrutiny, had little

Table 9.2b. *Monitoring information and data input: sardine stocks*

		Fishery data			Fishery-independent data		
Species	Stock	Parameter	Time series	Frequency	Parameter	Time series	Frequency
Japanese sardine (*Sardinops melanostictus*)	Pacific stock	Length frequencies	1976–2004	Monthly	Egg surveys	1978–2004	Monthly
		Landings	1976–2004	Monthly	SST	1978–2004	Monthly
		Immature fish wintering abundance index	1976–2004	1 × year	Surface trawl survey	1996–2004	2 × year
					Driftnet survey	1994–2004	5 × year
California sardine (*Sardinops sagax caerulea*)	Northern sub-population	Landings	1982–2005	Monthly	Egg survey	1985–87; 1995–06	Each April
		Age composition	1982–2005	Monthly	Aerial survey	1986–05	Year-round
		Weight-at-age	1982–2005	Monthly	SST	1982–05	Daily
Humboldt sardine (*Sardinops sagax*)	North– Central Peru	Length frequencies	1968–2005	Monthly	Hydroacoustic surveys	1975–2005	2 × year
		Age length keys	2000–2005	Quarterly	Egg surveys	1985–2000	1 × year
		Landings	1960–2005	Monthly	SST	1925– 2005	Monthly
		CPUE	1970–2005	Annually			
	South Peru–North Chile	Length frequencies	1974–2005	Monthly	Hydroacoustic surveys	1981–1995	1 × year
		Age–length keys	1974–2005	Quarterly	Egg surveys		
		Landings		Annually	Larval surveys	1985–2005	4 × year
Humboldt common sardine (*Strangomera bentincki*)	Central-South Chile	CPUE	1986–2005	Annually	SST		
		Age–length keys	1991–2005	Quarterly	Egg surveys	2002–2005	1 × year
		Landings		Annually	Larval surveys		
Brazilian sardine (*Sardinella brasiliensis*)	*Brazilian Southeast Bight*	Age–length keys	1981–1995	Annually	Egg surveys	1970–1980s	1 × year
		CPUE	1964–2005	Annually			
		Landings	1964–2005	Monthly	Larval surveys	1970–1980s	1 × year
		Length frequencies	1968–2005	Monthly	Hydroacoustic surveys	1970–1990s	1 × year
					SST	1977–1983	Mean values (Dec–Jan)
Benguela sardine (*Sardinops sagax*)	Southern (South African) stock	Length frequencies	1980–2005	Monthly	Hydroacoustic surveys	1984–2005	2 × year
		Landings	1980–2005	Annually			
	Northern (Namibian) stock	Length frequencies	1980–2005	Monthly	Hydroacoustic surveys	1990–2005	2–4 × year
		Landings	1980–2005	Annually			
European pilchard (*Sardina pilchardus*)	Atlanto-Iberian stock	Length frequencies	1978–2005	Monthly	Hydroacoustic surveys	1984–1988 1995 onwards	2 × year
		Landings	1978–2005	Annually			
					Egg surveys	1988, 1997 1999 – onwards	1 every 3 years
Australian sardine (*Sardinops sagax neopilchardus*)	Southwestern Australia (3 management units)	Length frequencies	1989–2005	Monthly	Egg survey	1993–2005	Every few years for each of 3 management units
		Weight, sex, maturity	1989–2005	Monthly			
		Age composition (age –otolith weight relationship)	1989–2005	Monthly			
		Landings	1975–2005	Monthly			

Table 9.2c. *Monitoring information and data input: herrings and sprats*

		Fishery data			Fishery-independent data		
Species	Stock	Parameter	Time series	Frequency	Parameter	Time series	Frequency
European sprat (*Sprattus sprattus*)	Black Sea	Size and age composition	1945–2005	Monthly	Mid-water trawl survey	1967–1993	1 × year
		Landings	1925–2005	Monthly			
		CPUE	1978–2005	Monthly	Juvenile fish surveys	1980–1993	1 × year
					SST	1915–2005	Monthly
	Baltic Sea	Length and age frequencies	1974–2005	Depends on catches	Hydroacoustic surveys	Autumn: 1983–2005 Spring 2001–2005	2 × year
		Landings	1974–2005	Annually	Egg surveys	1974–2005	3 × year
					Cod stomach contents	1977–1993	Quarterly
		CPUE	1995–2005	Log-book per trip	Winter NAO	1973–2006	Annually
	North Sea	Length frequencies	1984–2005	Quarterly	Hydroacoustic survey	2003–2005	1 × year
		Landings	1984–2005	Quarterly	Trawl survey	1984–2006	1 × year
Atlantic herring (*Clupea harengus)*	North Sea autumn spawning stock	Length frequencies	1960–2005	Quarterly	Hydroacoustic survey	1989–2005	1 × year
		Landings	1960–2005	Annually	Larval	1973–2005	4 × year
		Discards	1996–2005	Quarterly- not all nations	Post larval surveys	1992–2006	1 × year
					Trawl survey	1984–2006	1 × year
	Arcto-Norwegian spring spawning stock	Length frequencies	1950–2005	Quarterly	Hydroacoustic surveys	1988–2005	5 × year
		Landings	1950–2005	Annually	Larvae	1981–2005	1 × year
					0 group surveys trawl	1974–2004	1 × year
					Tagging	1975–2004	1 × year
	Central Baltic	Length and age frequencies	1974–2005	Depends on catches	Hydroacoustic surveys	Autumn: 1982–2005	1 × year
		Landings	1974–2005	Annually			
					Cod stomach contents	1977–1993	Quarterly
	Gulf of Riga	Length and age frequencies	1977–2005	Depends on catches	Hydroacoustic surveys	July–August 1999–2005	1 × year
		Landings	1977–2005	Annually	SST	1977–2005	1 × year (April)
		CPUE (trap-net)	1982–2005	Depends on catches	Zooplankton abundance	1977–2005	1 × year (May)

Table 9.3a. *Assessment and management: anchovy stocks*

		Assessment			Harvest strategy				Operating model	
Species	Stock	Type	Inputs	Outputs	Type	Inputs	Decision criteria	Frequency	Type	Inputs
Japanese anchovy (*Engraulis japonicus*)	NW Pacific	Annual VPA	annual landings, age structure, commercial CPUEs	F, SSB, R, N-at-age	Harvest Control Rules (HCRs) under stable high stock biomass condition	HCRs based on results of annual VPA	$F_{limit}=F_{sim}$ $F_{target}=1.0\times F_{limit}$ F_{sim} can continue the lowest SSB level during the recent five years	Annual	n.a	n.a
	Seto Inland	Monthly VPA	monthly landings, age structure, commercial CPUEs	F, SSB, R, N-at-month	HCRs under stable middle stock biomass condition	HCRs based on results of monthley VPA	$F_{limit}=F_{sim}$ $F_{target}=0.8\times F_{limit}$	Annual	n.a.	n.a.
	Tsushima Current	Monthly VPA	monthly landings, age structure, commercial CPUEs	F, SSB, R, N-at-month	HCRs under increasing middle stock biomass condition	HCRs based on results of monthley VPA	$F_{limit}=F_{current}$ $F_{target}=0.8\times F_{limit}$	Annual	n.a.	n.a.
California anchovy (*Engraulis mordax*)	Central Sub-population	Monitored only	Landings	Trends in landings	Low constant quota	Landings	None	Annual	None	None
Humboldt anchovy (*Engraulis ringens*)	North-Central Peru	Acoustic	Length – frequency sampling, L-W relationship	Length – frequency weighted to biomass, Adult biomass, juvenile biomass	Short-term projections under uncertainty	Adult biomass, juvenile biomass, Fishing and natural mortality	TAC on the basis of minimum spawning biomass	6-monthly	None	None
	South Peru-North Chile	Statistical catch-at-age analysis	Landings, commercial CPUE, and a larval indice	F, SSB, R, N-at-age	Constant F policy and short-term projections under uncertainty	Based on results of annual assessment	TAC based on SSB/R criteria	Annual	None	None

Table 9.3a. *(cont.)*

		Assessment			Harvest strategy				Operating model	
Species	Stock	Type	Inputs	Outputs	Type	Inputs	Decision criteria	Frequency	Type	Inputs
Benguela anchovy (*Engraulis encrasicolus*)	Southern Benguela anchovy	Model-free			OMP – Harvest Control Rules (HCRs), simulation tested on alternative operating models (alternative "states of nature" for the stock)	HCRs based directly on survey indices of abundance	Evaluate summary performance statistics (e.g. risk, average catch, interannual catch variability, stock depletion)	TAC: Initial in January, revised in June. OMP: Multi-annual (3–5 y)	Age-structured production model; stock-recruit model differs between operating models, (Hockey-Stick stock-recruit model assumed for base case)	Catch, survey indices, age composition of catches and survey indices
European anchovy (*Engraulis encrasicolus*)	Bay of Biscay	ICA, (Patterson and Melvin, 1996)	Catches at age, Spawning biomass and population at age indices from Acoustics and DEPM	F, SSB, R, N-at-age	No HCR is adopted. But in practice Fixed TAC of 33 000 t, but within season closure if SSB falls below B_{lim}	In season closure depending on survey indices of biomass at spawning time	No HCR. No projection of the population and the fishery available. In season closure of the fishery if SSB falls below B_{lim}	Annual	No formal operating model is adopted. Draft testing of HCR include age structure dynamic models and biomass Dealy models	n.a
	NW Mediterranean	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.
	Adriatic Sea	Laurec-Sheperd Tuned VPA	Landings, commercial CPUEs	F, SSB, R, N-at-age	n.a.	n.a.	n.a.	Annual	n.a.	n.a.
	Aegean Sea	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.
	Black Sea	VPA Modified Baranov method	Catch at age	N-at-age, F-at-age, SSB, R,	Deterministic short-term projections	Output from "best" assessment	F<=Fpa (Fmsy) Catch <=TAC	Annual	n.a	n.a

Table 9.3b. *Assessment and management: sardine stocks*

		Assessment			Harvest strategy				Operating model	
Species	Stock	Type	Inputs	Outputs	Type	Inputs	Decision criteria	Frequency	Type	Inputs
Japanese sardine (*Sardinops melanostictus*)	Pacific	Annual VPA	annual landings, age structure, trawl survey index, egg abundance	F, SSB, R, N-at-age	Harvest Control Rules (HCRs) under low stock biomass condition	HCRs based on results of annual VPA	$F_{limit}=F_{recovery}$ $F_{target}=0.8\times F_{limit}$ $F_{recovery}$ can recover SSB to the target SSB (1996 level) in 2015	Annual	n.a.	n.a.
California sardine (*Sardinops sagax caerulea*)	Northern sub-population	'ASAP' age-structured forward-projected model, with Beverton-Holt stock-recruit model	Catches, age composition of catches, survey indices	F, SSB, R, N-at-age,	OMP – Harvest Control Rule; simulation tested on alternative states of nature (SST-based F_{msy}) using different thresholds of minimum biomass and maximum catch	Age 1+ biomass output from "best" assessment	F<=Fpa, B>=Bpa	Annual	'ASAP' age-structured forward-projected model, with Beverton-Holt stock-recruit model	Catches, age composition of catches, survey indices
Humboldt sardine (*Sardinops sagax*)	North-Central Peru	Acoustic	Length–frequency sampling, L–W relationship	Length – frequency weighted to biomass, Adult biomass, juvenile biomass	Short-term projections under uncertainty	Adult biomass, juvenile biomass, Fishing and natural mortality	TAC on the basis of minimum spawning biomass. Since 2000, low abundance by environmental effect	Semestral	n/a	n/a
	South Peru–North Chile	ADAPT catch-at-age analysis	Landings, commercial CPUE, and a larval indice	F, SSB, R, N-at-age	n.a	n.a	n.a	n.a	n.a	n.a
Humboldt common sardine (*Strangomera bentincki*)	Central-South Chile	Statistical catch-at-age analysis	Landings, commercial CPUEs, acoustic biomass indices	F, SSB, R, N-at-age	Short-term projections under uncertainty	Output from the age-structured assessment	TAC on the basis of 10% of risk, at F60% fishing mortality reference point.	Annual, reviewed alter the acoustic survey to take into account recruitment	n.a	n.a

Table 9.3b. *(cont.)*

		Assessment			Harvest strategy				Operating model	
Species	Stock	Type	Inputs	Outputs	Type	Inputs	Decision criteria	Frequency	Type	Inputs
Brazilian sardine (*Sardinella brasiliensis*)	Brazilian Southeast Bight	Surplus Yield and Yield/ Recruit Models, VPA	Landings, commercial CPUEs, age structure, survey indices	Estimated CPUE trends, F, SSB, R, N-at-age	Alternative "states of nature" for the stock	Survey indices of abundance (hydroacoustic, eggs and larvae)	Evaluate summary performance statistics (average catch, interannual catch variability, stock depletion)	Multi- annual (~3 years)	Age-structured model, assuming a Beverton-Holt stock-recruit model	Catch, age-composition of catches and survey indices
Benguela sardine (*Sardinops sagax*)	Northern Benguela	Simple age-structured models, accounting for uncertainties related to key assumption	Catch survey indices, length frequencies of surveys and catches	Current stock status	TACs based on projected outcomes from a range of model hypotheses		Reference points are available, but decisions have been heavily influenced by socio-economic factors in recent years of low biomass	TAC: annual	n.a	n.a
	Southern Benguela	Model-free			OMP – Harvest Control Rules (HCRs), simulation tested on alternative operating models (alternative "states of nature" for the stock)	HCRs based directly on survey indices of abundance	Evaluate summary performance statistics (e.g. risk, average catch, interannual catch variability, stock depletion)	TAC: annual TAB: Initial in January, revised in June OMP: Multi-annual (3–5 years)	Age-structured production model; stock-recruit model differs between operating models, (Hockey-Stick stock-recruit model assumed for base case)	Catch, survey indices, age-composition of catches and survey indices
European sardine (*Sardina pilchardus*)	Atlanto-Iberian	AMCI	Landings, survey indices	F, SSB, R, N-at-age	Deterministic short-term projections	Output from "best" assessment	F ≤ Fpa, B ≥ Bpa	Annual	n.a	n.a
Australian sardine (*Sardinops sagax neopilchardus*)	Southern Western Australia (3 management units)	Age-structured model. Simple analysis of age composition	Landings, age-structure, SSB from egg surveys	SSB Mean age	Harvest control rules: SSB relative to SSB and mean age provide a two-way table to objectively set a harvest rate of 0–10% of SSB. Apply a carrying-capacity conceptual model	SSB output from the age-structured assessment. Note: May discard use of model and use annual landings and mean age as HCR indicators	Minimum threshold for SSB; max and min thresholds for mean age. To set a harvest rate of 10% (i.e. maximum allowed) SSB must be > 0. 4 of SSB_{∞}, and mean age must be –5 years	Annual	Age-structured model, assuming a Beverton-Holt stock-recruit model	Catches, age-composition of catches, SSB estimates from egg surveys

Table 9.3c. *Assessment and management: herrings and sprats*

		Assessment			Harvest strategy				Operating model	
Species	Stock	Type	Inputs	Outputs	Type	Inputs	Decision criteria	Frequency	Type	Inputs
European sprat (*Sprattus sprattus*)	Black Sea	XSA, ICA Absolute biomass from mid-water trawl survey	Catch ate age commercial CPUEs, survey indices	N-at-age, F-at-age, SSB, R	Deterministic short-term projections	Output from "best" assessment	F ≤ Fpa (Fmsy) Catch ≤TAC	Annual	n.a	n.a
	Baltic Sea	Extended Survivor Analysis (XSA)	Landings, survey indices, M2 from MSVPA	F, SSB, R, N-at-age	Deterministic short-term projections	Output from "best" assessment, 0-group from hydroacoustic survey, winter NAO	F ≤ Fpa, B ≥ Bpa	Annual	n.a	n.a
	North Sea	Catch Survey analysis (CSA)	Landings, surveys – IBTS	Relative biomass, relative recruits	None. No clear management objectives or evaluated strategy	Output from "best" assessment	None, based on Council of Ministers decision and advice from ICES and STECF	Annual, within year	n.a	n.a
Atlantic herring (*Clupea harengus)*	North Sea autumn spawning stock	Integrated catch age age (ICA)	Landings, misreported catch, discards, surveys	F, SSB, R, N-at-age	Harvest control rule (EU Norway agreement)	Output from "best" assessment	SSB >1.3×10^6 $F_{2-6}=0.25$, $F_{0-1}=0.12$ SSB <0.8×10^6 $F_{2-6}=0.10$, $F_{0-1}=0.04$ If SSB betw. $0.8–1.3\times10^6$ then linear decrease in F	Annual	n.a	n.a
	Arcto-Norwegian spring spawning stock	SeaStar	Landings, surveys, tagging recoveries	F, SSB, R, N-at-age	Harvest control rule (EU Norway Iceland Russia Faroes agreement)	Output from "best" assessment	SSB >5.0×10^6 $F_{5-14}=0.125$ SSB<2.5×10^6 $F_{5-14}=0.05$ If SSB betw. 2.5–5.0 x10^6 then linear decrease in F	Annual	n.a	n.a
	Central Baltic stock	Extended Survivor Analysis (XSA)	Landings, survey indices, M2 from MSVPA	F, SSB, R, N-at-age	Deterministic short-term projections	Output from "best" assessment, 0-group from acoustic survey	F ≤ Fpa	Annual	n.a	n.a
	Gulf of Riga stock	Extended Survivor Analysis (XSA)	Landings, survey indices, CPUE	F, SSB, R, N-at-age	Deterministic short-term projections	Output from "best" assessment, SST in April, Zooplankton abundance in May	F ≤ Fpa, B ≥ Bpa	Annual	n.a	n.a

retrospective change between years and performed well. A survey-based SURBA[17] model (Needle, 2004), a separable period-based ICA model (Patterson, 1998), a VPA-based model XSA (Darby and Flatman, 1994; Shepherd, 1999) and a simple two-stage population model CSA (Mesnil, 2003) gave similar perceptions of the trends in mortality and spawning stock biomass. Due to the higher precision in the estimation of fishing mortality in the terminal year, and the lower retrospective bias between years, the assessment model ICA is used to provide the basis for advice.

The assessment feeds directly into the management agreement between the EU and Norway for Norway Sea herring. The agreement was adopted in 1997 and amended in 2004. According to the agreement, efforts should be made to maintain the SSB of North Sea Autumn Spawning herring above 800 000 t. The agreement states that an SSB of 1.3 Mt acts as a trigger. In years with an SSB above this trigger, the following years TAC will be based on an $F = 0.25\ y^{-1}$ for adult herring (mean ages 2–6) and $F = 0.12\ y^{-1}$ for juveniles (mean ages 0–1). If the SSB falls below 1.3 Mt, the fishing mortality will have to be linearly reduced. If the SSB is below 0.8 Mt a TAC equivalent to $F = 0.1\ y^{-1}$ for adult herring and $0.04\ y^{-1}$ for juveniles should be set. Simulations suggest that this management agreement results in a sustainable fishery within the precautionary approach.

The agreement also states that a TAC deviation of more than 15% between two subsequent years should be avoided, unless the parties consider this appropriate. The regulations on the bycatch of herring in the industrial fishery tend to change between years, but all bycatch should be counted against the TAC.

Arcto-Norwegian spring spawning herring stock

The stock assessment is carried out through the ICES northern pelagic and blue whiting fisheries working group (ICES, 2005b). The SeaStar model is used (www.assessment.imr.no), a statistically based model designed to minimize the use of weighting of time series. It also allows for the inclusion of tagging data into the assessment and only “tunes” the assessment on the bigger year classes. This allows the year-class signals, which are very strong in this stock, to have maximum influence on the assessment.

The assessment feeds directly into the long-term management agreement between the EU, Norway, Faroes, Iceland and Russia (called the Coastal States). The agreement states that every effort should be made to maintain SSB above 2.5 Mt. The TAC should be restricted to ensure that fishing mortality (F) remains less than $0.125\ y^{-1}$. Should SSB fall below 5 Mt, the F should be reduced from $0.125\ y^{-1}$ to allow a safe and rapid recovery of SSB to above 5 Mt. The basis of the reduced F should be at least a linear reduction in F from $0.125\ y^{-1}$ at 5 Mt to $0.05\ y^{-1}$ at 2.5 Mt. So basically the fishery is managed with a target F of $0.125\ y^{-1}$, and a trigger biomass of 5 Mt. This agreement is dependent on the advice from ICES, and simulations by ICES suggest that this management agreement results in a sustainable fishery within the precautionary approach. Despite having this management agreement, the coastal states have not been able to agree on an annual quota for Norwegian spring spawning herring since 2003 but have, however, still kept their quotas close to the scientific advice coming from ICES.

North Sea sprat

The ICES herring assessment working group conducts assessments of North Sea sprat based on indicators derived from a research survey and on a two-stage Catch-Survey Analysis (CSA). The CSA model (Mesnil, 2003) assumes that the population consists of two stages: the recruits and the fully recruited ages. The model results are highly dependent on the assumptions about natural mortality and the IBTS is the survey used with the commercial catch data. The assessment is only used to indicate stock trends. There are no reference points for this stock (either precautionary or target).

ICES has, for many years, recommended “within year” management for North Sea sprat, with the advice based on the results of surveys. The current advice is based on a regression between the annual survey index (IBTS) and annual catches, suggesting a probable catch for that year based on the quarter 1 IBTS, and extrapolations of the CSA assessment with a range of TACs into the forthcoming year. Neither of these methods has been evaluated and can be considered reliable.

Currently, North Sea sprat is managed by a TAC and a by-catch quota, but the scientific rational for the TAC is unclear. Other management measures include that Norwegian vessels are not allowed to fish in the second and third quarters and are not allowed to fish in the Norwegian zone until the quota has been taken in the EU zone. Recently, the sprat fishery has not reached its TAC and the bycatch of herring has been declining (now at <10% of the catch).

Baltic Sea sprat

The standard stock assessment model deployed is an Extended Survivor Analysis (XSA) (Shepherd, 1999) using Multispecies Virtual Population Dynamics (MSVPA) derived predation mortalities as input (ICES, 2005c). Alternatively, ICAs (ICES, 1997a) and multispecies stock production models (Horbowy, 1996) have been tested, revealing similar stock development trends.

The multispecies assessment separates the western and eastern sprat stocks (ICES, 2003c), but as the western stock is comparatively small, the single species assessment uses predation mortalities from the eastern multispecies assessment. Additionally, MSVPA runs for single statistical regions containing major spawning areas. They are presently not used in assessment, but are used in stock recruitment studies (Köster *et al.*, 2003b).

Short-term catch and stock predictions are conducted to recommend TACs, with management constraints being a precautionary fishing mortality (F_{pa}) and a precautionary spawning stock biomass (B_{pa}), below which action is taken to ensure recovery of the stock above B_{pa} and reduction of the fishing mortality below F_{pa}. Furthermore, there exists a biomass limit reference point B_{lim} which if violated result in severe fisheries restrictions, i.e. in case of a reduction below B_{lim} the fishery is stopped. These reference points are either based on stock-recruitment relationships (e.g. F_{pa} equals F_{med}) or medium-term projections (ICES, 1998a).

The short-term prediction conducted in year *t* for the year *t*+1 is based on assessment data from year *t*–1 yielding the population size at the beginning of year *t,* and requires estimation of recruitment for age-group 1 in years *t* and *t*+1. Traditionally, a geometric mean recruitment over a range of recent years has been used as recruitment estimate for all years. However, considering a temperature (or NAO)–recruitment relationship outperformed the standard procedure (MacKenzie and Köster, 2004) and consequently the winter NAO index was incorporated into the short-term prediction by ICES (2006c), together with the 0-group abundance index from the hydroacoustic survey, to predict recruitment in 2006 (year *t*), while recruitment in 2007 (year *t*+1) was still assumed to be the geometric mean. In an exploratory analysis MacKenzie *et al*. (2008) investigated the effect of additionally replacing the geometric mean recruitment for 2007 by an estimate from a NAO–recruitment relationship. The sensitivity of the spawner biomass at the end of the prediction period to the most likely range in NAO values (10th–90th percentiles) was found to be ca. 27%. For the specific case of the NAO in 2008, MacKenzie *et al*. (2006) found a 15% difference in spawner biomass.

The stochastic medium-term projection allows for uncertainty in stock size at the beginning of the projection period, the stock recruitment relationships and weight-at-age. However, the applied stock-recruitment relationship is environmentally insensitive, despite a clear dependence of recruitment on environmental conditions, i.e. temperature (Köster *et al*., 2003b), NAO (Mackenzie *et al*., 2008), and transport (Baumann *et al*., 2006) and a limited variability in recruitment explained by the model. Mean weight-at-age and maturity ogives are kept constant as average over a number of most recent years in both short- and medium-term predictions, although clear density dependent relationships exist for weight-at-age and would, in fact, allow prediction independent of stock size. In single species predictions, natural mortality is kept constant, which appears to be reasonable for short-term predictions, while medium-term projections should include cod as predator, with the methodology of multispecies projections being implemented (ICES, 2005c).

Baltic Sea herring

Herring in the Central Baltic has been assessed by ICES in varying stock units (Sjöstrand, 1989), but these were combined in 1990 due to practical reasons regarding input data. In 2002 preliminary assessments were conducted separating the main Central Baltic stock and the Gulf of Riga herring (ICES, 2002b) and these separate assessments form the basis for the scientific advice given by ICES since 2003.

Advice on both the Central Baltic and the Gulf of Riga stock is based on standard ICES short-term catch and stock predictions against the background of limiting biological reference points. The assessment model deployed is an XSA, tuned by hydroacoustic survey indices and for the Gulf of Riga herring additionally by CPUE from the trap-net fishery (ICES, 2006c). While for the Central Baltic herring MSVPA derived predation mortalities are used as input, the cod abundance in the Gulf of Riga has been low for over two decades and thus natural mortality (M) has been taken as constant at $0.2y^{-1}$ since 1984.

Standard ICES short-term catch and stock predictions are conducted to recommend TACs (see section on Baltic sprat). For the Central Baltic herring recruitment at age 1 in year *t* is estimated from hydroacoustic derived 0-group abundance, while Gulf of Riga recruitment is predicted from a multiple regression of historical recruitment on water temperature in April and zooplankton abundance in May of the year of birth (ICES, 2006c). As the annual assessment is conducted before environmental data for the assessment year are available, recruitment in year *t*+1 is estimated as geometric mean over a range of year classes, similar to the Central Baltic herring stock.

In stochastic medium-term projections stock-recruitment relationships are used for both stocks, for the Central Baltic herring assuming a linear increase in recruitment from origin to a breakpoint, estimated by ICES (2003d), and constant recruitment above this breakpoint. For the Gulf of Riga herring, a Beverton and Holt stock-recruitment relationhip (ICES, 2006c) is applied. Weight-at-age, maturity and natural mortality are kept constant, but uncertainty is considered for stock-recruitment relationships, initial stock sizes, and weight-at-age.

Management constraints for the Gulf of Riga herring include a defined precautionary fishing mortality (F_{pa}) and a precautionary spawning stock biomass level (B_{pa}) as well as a limit spawning stock biomass reference point (B_{lim}). For the Central Baltic herring only F_{pa} is defined. The reference points are either based on stock-recruitment relationships (F_{pa} equals F_{med} for the central Baltic herring, B_{pa} equals

M_{BAL} and B_{lim} derived from B_{pa} for the Gulf of Riga herring; see ICES, 1996) or derived from medium-term predictions (F_{pa} for Gulf of Riga herring; see ICES, 2003e). Other management measures include international (EU and Russia) and national technical regulations. National regulations by Estonia and Latvia for the Gulf of Riga include a limitation on the number and power of trawlers operating in the Gulf, a summer ban in the Estonian part of the Gulf from mid June to September and a 30-day ban for all trawl fishery during the main spawning migrations of herring in April–May.

Mediterranean and Black Sea pelagic fisheries

In the Adriatic Sea, stock assessment has been carried out by means of VPA (Darby and Flatman, 1994). Commercial catch data collection includes Italy, Slovenia and Croatia. Age–length keys are applied to the annual, catch-weighted length frequency distributions to obtain the corresponding age distribution. Age classes used in the VPA are 0 to 3 with the 4 used as a plus group. In most recent years, estimates of fishing mortality rates at age are obtained by Laurec–Shepherd tuning (Laurec and Shepherd, 1983) based on the CPUE at age obtained from the Porto Garibaldi fleet (which lands 25% of total landings). Natural mortality (M) is assumed to be 0.6 y^{-1} according to the age distribution of the catch and the observed maximum longevity of the species in the Adriatic Sea. However, additional VPA runs with $M = 0.8\ y^{-1}$ are usually performed. Recent work points out the importance of environmental factors in determining recruitment strength and the potential of using stock-recruitment models incorporating environmental variables (Santojanni *et al.*, 2006). However, such models have not yet been included in the assessment procedure. Preliminary trials to tune VPA using fishery independent data (echosurveys) are also currently being conducted, showing positive results.

There is no established harvest strategy nor any quota system enforced. The management of the Mediterranean small pelagic fisheries is rather independent from scientific advice.

International stock assessments of Black Sea sprat and anchovy are based on catch-at-age models (Prodanov *et al.*, 1997). Pre-recruit and adult stock abundance indices were used by Daskalov *et al.* (1996) and Daskalov (1998) for tuning catch-at-age assessment models of sprat. In recent years research surveys are performed more rarely because of financial problems of the research institutes. A quasi-decadal pattern dominates the abundance series of both species. Abundance of both stocks increased during the 1970s and 1980s when the highest catches were recorded. A combination of low recruitment and excessive fishing led to stock collapses in the early 1990s (Daskalov *et al.*, 2008), which are illustrated by the peaks in fishing mortality in the late 1980s to early 1990s. At the same time the accidentally introduced ctenophore *Mnemiopsis leidyi* contributed to the decline in recruitment (Grishin *et al.*, 1994). During the 1990s the anchovy and sprat stocks recovered despite the growing *Mnemiopsis* population and catches increased again.

The management of Black Sea stocks has been done separately by the surrounding countries despite the existence of a Joint Commission for the Fisheries in the Black Sea until 1991, which included Bulgaria, Romania and the former USSR, but not Turkey. Studies on stock assessment and management of anchovy and sprat have been published, e.g. by Tkatcheva and Benko (1979), Ivanov (1983), Ivanov and Beverton (1985), Domashenko *et al.* (1985). After the political changes in Eastern Europe, the Commission ceased its function.

At present, there is no routine international assessment and management of Black Sea stocks, although the stocks are shared. In the early 1990s a group of international experts performed stock assessments of 18 commercially important stocks including sprat and anchovy (Prodanov *et al.*, 1997) and the GFCM recommended this study as a background paper for future stock assessment work. In the early 1990s the countries decided to prepare and sign a new convention for the fisheries in the Black Sea under the auspices of which international working groups will prepare assessment and management advice. However, the Convention has not been ratified. At the moment some internationally coordinating functions are carried out by the Commission on the protection of the Black Sea Against Pollution (http://www.blacksea-commission.org) in conjunction with GFCM. Meanwhile, each national government applies its own methodology and legislation in stock assessment and management. Before the collapse of the fisheries in the early 1990s (Daskalov *et al.,* 2008), small pelagics in the Black Sea were considered as under-exploited and no special measures for their protection were adopted. Since 1989, Turkey (which catches 80%–90% of the total sprat and anchovy production) applies restrictions on minimum body size (9 cm total body length for anchovy), minimum mesh size, minimum discard rate, closed season and areas (near shore area of less than 16 m of depth). Regular (annual) stock assessments are used as base for short-term projections (e.g. in Bulgaria, Russia and Ukraine) and F_{msy} and TAC are used as limit reference points (e.g. by Russia and Ukraine). Target harvest is usually recommended as some percentage of the TAC related to the state of the stock. Other regulations on minimum body size, closed areas and seasons are applied on a regular or *ad hoc* basis related to the status of the stock.

Daskalov (1999) used generalized additive modeling (GAM) in exploring non-linear relationships in four Black Sea fish species (sprat, anchovy, horse mackerel and whiting) and different environmental variables (SST, SLP[18], wind, run-off). Despite relatively strong fish–environment linkages found, environmental indices have not been used in operational stock assessment. Pre-recruit surveys have been used for short-term forecasts of recruitment and SST data have been used to predict timing of formation of fishable concentrations of anchovy and sprat by the YugNIRO, Kerch (A. Mihaylyuk, pers. comm.).

Southwestern Australia sardine

The southwestern sardine fishery is managed using TACs and ITQs.[19] TACs for each of three management units are set (recommended) by a formal management advisory committee. An age-structured model with spatial components was developed initially for one region (Fletcher, 1992), with DEPM-based SSB estimates and relative recruitment strength forming the assessment advice for the other two regions. Fletcher's model was later replaced by a more sophisticated age-structured model (Hall, 2000) designed to encompass the entire southern WA sardine breeding stock but incorporating differing parameters for, e.g. recruitment and fishing mortality, between the three regional management units. This model fitted to monthly catches, fishery independent SSB estimates, and age composition. The log-likelihoods for the observed SSB estimates from DEPM surveys, age composition data, and recruitment deviations were combined, together with the penalty functions to form the overall log-likelihood. These objective functions were maximized simultaneously for each region to obtain maximum likelihood estimates of the parameters for that region. Parameters were estimated for each of the three regions including the initial recruitment, estimates of natural mortality (the two parameters of the selectivity curve), and annual recruitment deviations. A Bayesian procedure was used to determine estimates of uncertainty of the mature biomass using Markov-Chain Monte Carlo procedure.

Although the assessment model has successfully tracked trends in SSB, the levels of uncertainty (e.g. in the spatial dynamics of recruits) have negatively impacted the use of the models for providing definitive or "stand-alone" stock assessment advice. Along with the comparatively low natural productivity of the pelagic ecosystem off southern WA, the lack of precise point estimates of SSB has necessitated a conservative harvest strategy, and one that may eventually be based on fishery performance (e.g. annual landings) and age structure.

The current aim of management is to limit the harvest rate to less than 10% of SSB; harvest rate is set objectively using a two-way matrix that considers spawning biomass (0%–100% SSB_∞[20]) and the average age (from 1–8+ years). The age component provides an indication of the health of the stock relative to past and recent recruitment levels. The harvest rate control rules allow the maximum exploitation rate (i.e. 10%) only if $SSB > 0.4\ SSB_\infty$ and mean age is 4–5 years.

A mono-specific mass mortality killed 70% of Australia sardines in 1998/99 (Gaughan *et al*., 2000); SSB has recovered but in WA recovery of the commercial fisheries has been hampered by failure of market-size (i.e. mature, >3+ years) sardines to return to the traditional fishing grounds. Current research activities are focused on bycatch mitigation, particularly quantifying and minimizing interactions with seabirds and dolphin.

Discussion

The present review demonstrates that the assessment and management of small pelagic fish varies substantially in its approach and performance. Most of the stocks reviewed have some sort of scientific assessment program in place and a management approach that takes into account this assessment, albeit to different degrees. California sardine, Humboldt anchovy, southern Benguela, and Baltic stocks are examples where the advice based on scientific assessments is respected reasonably in the management process, while Namibian sardine and Bay of Biscay anchovy are examples of management practices that have deviated substantially from scientific advice on the basis of political or poorly explained socio-economic reasons. Considering aspects not included in biologically based scientific assessments is both understandable and to be expected. For example, issues such as employment, industry and social stability or revenue income should be an intrinsic part of the management process. However, these factors should be subject to the same type of analysis applied in determining biological risk if we are to demonstrate a consistent approach to management (Cochrane, 1999). For example, there was considerable deviation from the scientific advice provided for the southwestern Australia sardine fishery in the mid 1990s, with TACs set higher than recommended for a number of years (which were not filled), which contributed to a stock crash in one region. The development of an objective decision table helped overcome this. The southern Benguela OMP (De Oliveira and Butterworth, 2004) approach appears to have broadened its management objectives without compromising its quantitative, analytical approach. However, it is more common to have rigorous, data-intensive scientific advice modified to account for vague, subjective and untested socio-economic reasons. Such a two-tier approach cannot demonstrate true resource stewardship and use, nor does it allow the implementation of management performance measures.

Surprisingly, this review has demonstrated that many stocks have assessment and management processes that are disconnected, or have scientifically based management advice that is often overlooked. The Bay of Biscay anchovy, for example, has been managed by fairly fixed (30 000–33 000 t for the period 1979–2004) annual TACs set independently of the scientific advice given. Annual catches have been well below this fixed TAC in recent years, indicating that the TAC was unable to constrain the fishery. The main reason for this is that the anchovy TAC responded more to European Union-wide politics, in which quotas can be exchanged between countries for products (e.g. quotas for other fisheries, agriculture products) completely unrelated with anchovy management issues. In the case of the Mediterranean anchovy, as in other fisheries in this basin, management is limited to technical measures on effort and gear regulations that disregard the state of the stock. Despite dwindling transboundary stocks, governments appear to discourage the collection of fisheries data and the development of effective, stock-wide assessment procedures (Lleonart and Maynou, 2002). In another interesting example, the North Sea sprat is managed through an assessment methodology that relies heavily on assumptions about natural mortality and using a historical time series of catches where herring was misreported as sprat. All parties involved accept that there is no reliable scientific basis for the specific numeric TAC advice given. Whilst this is not problematic when exploitation is light and stock productivity high, it is likely to haunt managers if demand increases and/ or productivity declines require a more effective management approach.

In conclusion, in scientific circles it is widely accepted that when science is credible, it has proven to be a strong basis for advice and management of pelagic fish stocks. The examples reviewed here show that the most successfully managed stocks rely on a scientific program of advice, suggesting that science-based management cannot be replaced, and that only a properly tailored scientific assessment and management program can provide the speed of response and the flexibility of management that highly variable small pelagic fisheries demand. However, while reliable science provides the best set of principles, this prerequisite is not sufficient to ensure sustainable long-term use and conservation of fishery resources.

Regarding assessment standards, the most effective monitoring programs are based on fishery-independent surveys that provide precise information of the state of the stocks. Certainly, for short-lived species, catches-at-age analysis based on VPA-like assessments offer little value, given the poor VPA convergence for these species and the poor prognosis of next evolution of the population they can provide. Therefore, direct surveying of the population offers the best alternative to such methods, providing current and ready to apply knowledge about the status of the population. Preferred surveys are either based on ichthyoplankton sampling techniques (e.g. Daily Egg Production Method), hydroacoustics, or both. There are also suggestions that pelagic fish assessment may benefit from incorporating the spatial structure of the population into the estimation procedure. This, for example, has been attempted for southwestern Australia sardine, but the spatial complexities or non-linearities (e.g. interactions between management units change over time) hampered fitting of the model, hence the move back to a simpler management system. The use of catch per unit effort (CPUE) information, particularly in the absence of fishery-independent methods, is questioned because the schooling habits of pelagic fish suggest CPUE may not reflect stock abundance (Ulltang, 1980; Csirke, 1988; Pitcher, 1995; Mackinson *et al*., 1997). However, changes in spawning stock biomass estimated by VPA have been compared to those estimated using annual egg production methods in Japan, indicating similar trends and almost the same stock level although variability in the VPA estimate was lower (Y. Oozeki, unpublished data). As a result, the latter is adopted to estimate the Allowable Biological Catch (ABC). There is also agreement that biological sampling programs directed at the acquisition of catch volumes, length frequency and age data (one of the major sources of uncertainty) must be protected, and that this information should be spatially segregated to allow for spatial management rules.

Overall, scientific uncertainty and lack of governance are the most cited reasons to explain fisheries management failure. Regarding scientific uncertainty two approaches have dominated fisheries science in recent times. One argued that uncertainty needs to be reduced through additional science and measures (Ulltang, 1996, 2003; King *et al*., 2001), while the other assumes that uncertainty cannot be significantly reduced and thus that the development of management procedures robust to it should be promoted (Walters and Collie, 1988). Evidence of both approaches can be found in this review, for example, in relation to recruitment forecasting. Short-lived small pelagic fish species rely on their annual recruitment to secure a healthy adult biomass the following year. In addition, many small pelagic fisheries target these recruits. As a result recruitment estimation has been one of the most debated, conflictive and productive areas of pelagic fish research in recent years (e.g. Shepherd *et al*., 1984; Myers, 1998). Survey-based recruitment estimation in South Africa, for example, is only available after sometimes substantial anchovy catches have been landed (Butterworth *et al*., 1993) on the basis of an initial TAC that assumes median recruitment. Indirect early recruitment estimations based on environmental proxies have been explored because the average annual catch could be increased by up to 48% if a very precise prediction could be made before fishing commences (Cochrane and Starfield, 1992). On the basis of simulation work, De Oliveira and Butterworth (2005) concluded that an environmental index needed to explain 50% or more of the total variation in recruitment before an Operational Management Procedure that uses the index as a recruitment proxy starts to show benefits in terms of long-term catches and risk. However, removing constraints on the extent of TAC changes in the South African OMP would relax the amount of variation that the index needs to explain for comparable benefits, but at the cost of larger interannual variability in TAC. This is because constraints on TAC changes limit the response of an Operational Management Procedure to the environmental index.

In the case of the Bay of Biscay, science cannot produce the advice required by managers because of the lack of a procedure for forecasting the anchovy population and the fishery for the management year. Management needs a reliable indicator of the latest year-class strength prior to its recruitment into the fishery, but the failure of recruitment predictors based on statistical relationships (Borja *et al.,* 1998) led to their abandonment in TAC determination. De Oliveira *et al.* (2005) investigated under what circumstances incorporating environmental-based recruitment indices would lead to management improvements. Results show that precautionary approaches may better ensure successful management than consideration of uncertain or moderate-to-weak (in terms of the relationship to recruitment) environmental effects. Scientists in charge of this stock now consider that any future attempts to use environmental estimates of recruitment would require prior definition of criteria against which success or failure would be judged, a process that should apply to other stocks and species. In the meantime plans are in place to revise the current management regime so as to take into account recruitment fluctuations. This may be achieved by developing a decision rule using directly the information from the existing time series of spring surveys or by taking into consideration the results of a recently established autumn juvenile survey program.

Environment-based recruitment estimators, however, are explicitly used in the management of the California sardine (Jacobson and MacCall, 1995; PFMC 1998). Justification is based on the observation that Maximum Sustainable Yield (MSY) and stock biomass for MSY (B_{MSY}) are dependent on habitat area (Jacobson *et al.*, 2005), and thus monitoring habitat area (proxied by SST) can help anticipate periods of high and low productivity and adjust management accordingly (Jacobson *et al.*, 2005). The value of this index has been consistently proven (Myers, 1998). In recent years more examples have been provided to justify the incorporation of environment–recruitment relationships (e.g. Yatsu *et al.*, 2005), suggesting that this area of work will continue, particularly in relation to developing medium to long term exploitation strategies in the context of global climate change.

Mackenzie and Köster (2004) consider that incorporating a recruitment estimate based on indices for atmospheric NAO and ice coverage outperforms the current assumption of medium recruitment in the assessment of the Baltic Sea sprat and first trials of implementation in assessment are being conducted (ICES, 2006c). The estimate of recruitment in North Sea herring is provided by the survey of post larvae, which explains over 70% of the variance in 0-group fish. This direct method is thus relatively robust and prior to the onset of fishing pressure in this relatively longer-lived species.

Discussing the value of pre-season forecasting, Walters (1989) concluded that improvements in management performance depended strongly on the average productivity of the stock concerned, and on the flexibility of the in-season regulatory system used to manage the stock. The fact is that there is not a single approach to be recommended on the subject of scientific uncertainty, because factors such as the dynamics of the fishery (targeting recruits vs. adults), the type of assessment (monthly, annual, or multi-annual) and the subsequent management measures to be taken, would determine whether resolving uncertainty is cost effective. However, a major stumbling block to using recruitment estimators is the lack of understanding of the dynamics of recruitment, the links between nursery and recruitment grounds, and growth and feeding ecology of young fish, among others. Several recruitment proxies are used, not without problems. For example, the proportion of 2-year-olds in the catch is used as an indicator of southwestern Australia sardine recruitment in different subregions, a figure that is critical in the scientific advice. Indirect methods, in particular, suffer from poor process understanding linking trends in proxy variables with recruitment variability (see De Oliveira and Butterworth, 2005; ICES, 2006f). These aspects are major scientific challenges that are expected to remain research priorities in the future (ICES, 2006e,f).

Another important source of uncertainty relates to the actual levels of fishing vs. natural mortality, as small pelagics are major forage species for other fish, birds and mammals (see Box 9.1). This aspect is influenced by catchability assumptions in the DEPM, bottom trawl surveys and VPAs, posing questions over management strategies based on fishing mortality levels. The catchability of fishing fleets also deserves further attention, particularly in relation to patterns and seasonality, which can be used to assess the effect of different management options and harvest control rules. This is particularly relevant in the case of species that are harvested using both trawlers and purse-seiners, and is linked to behavioral changes in the stocks themselves, as perceived in the Bay of Biscay anchovy of late.

Box 9.1. How is science contributing to the ecosystem approach to South African fisheries?

The ecosystem approach to fisheries (EAF) requires a sound scientific basis to provide the means of assessing the ecosystem effects of fishing and the effectiveness of management options in response to identified risks. South Africa, by virtue of several decades of multidisciplinary studies and monitoring, and a relatively flexible management approach with some stakeholder participation, is well placed for a test case for EAF. Ecological risk assessment (Fletcher, 2005) has been applied to identify and prioritize ecosystem issues in three main fisheries: hake, small pelagics and west coast rock lobster. An example of a high priority issue is given in the table below for small pelagics (SPF) to illustrate how South Africa is moving towards an EAF by identifying required research and/or monitoring, indicators and management actions to address each issue (Shannon *et al.,* 2006). Indicators derived from biological or catch data facilitate the monitoring of ecosystem responses to management actions, implemented to optimize economic and social objectives while ensuring the ecological sustainability. Careful interpretation of these indicators is required such that a decrease in abundance is not misinterpreted as poor management (e.g. catch set too high) if it is rather a consequence of normal fluctuations in the stock (e.g. poor recruitment). Implementation of EAF is regarded as an ongoing process, composed of the following main needs:

(i) Identification of the current status of the resource.
(ii) Examination of concerns regarding single-species, community or ecosystem-based approaches (such as spatial issues or species interactions not currently accounted for in management) and expression of these as ecosystem objectives.
(iii) Identification of indicators related to ecosystem objectives.
(iv) Translation of ecosystem indicators into decision criteria (for example, by defining limit reference points to be avoided).
(v) Identification of research and monitoring needs.
(vi) Development of appropriate management actions to be taken with stakeholder participation.
(vii) Development of evaluation criteria for adopted management measures.

A number of the above needs are already implemented as part of the Operational Management Procedure in South Africa (see Section "The Benguela operational management procedure (OMP) approach", above), while others (points (ii) to (iv)), are specific to EAF.

Issue	*Indicators*	*Research approaches*	*Management actions under consideration*
Impacts of removal of SPF on seabirds bound to breeding sites on land.	Bird population sizes; breeding success (fledgling weight, fledglings raised per breeding pair, breeding proportion); seabird diet composition; spatial indicators (e.g. overlap of seabird foraging and SPF fisheries).	Underway: Routine monitoring of seabird colonies; satellite tracking to assess foraging ranges; minimum realistic models; spatialized models of SPF around seabird colonies; quantification and formalization of the link between the SPF fishery and seabirds; quantification of functional responses of seabirds to SPF and identification of thresholds below which there are serious negative implications for seabirds.	Avoid seabird populations falling below limit reference points set according to IUCN[21] conservation criteria by reducing TACs or closing areas within foraging ranges; allow sufficient escapement of SPF for predators; avoid threshold levels of SPF below which the risk to seabirds is unacceptable

Whether investment is on more complex science, or on more flexible, robust and adaptive management systems (Walters, 1997), or both, a problem most of the fisheries reviewed here have suffered from, at some point in their exploitation history, is inadequate governance. Governance is the sum of the legal, social, economic, and political arrangements used to manage fisheries. It has local, national, and sometimes international dimensions and includes legally binding rules as well as customary social arrangements. Global reviews point out that prevailing systems of fisheries governance have often been ineffective. Governance failures are caused by managers trying to address multiple objectives in the absence of a formal analytical framework that places resource sustainability in its appropriate context. For example, the management of the Bay of Biscay anchovy has been driven by quota trade agreements between countries rather than resource sustainability. Another example of governance failure is the fact that scientific advice is more readily accepted when it implies an increase in fishing effort than when it requires a reduction. Failure to take sufficient action in the face of clear stock declines has been implicated in the collapses of the California and Namibian sardine, for example, even if overfishing was not necessarily the catalyst for the initial stock productivity declines. The impact of governance is clearly shown when comparing North Sea herring in the 1970s and 1990s. From the same biomass (0.5 Mt) failure to act on the advice resulted in a collapse of the stock in the 1970s and action in line with the advice resulted in a recovery in the 1990s (Simmonds, 2007). An example of an adequate governance framework is the South African OMP. This is anchored on fishery-independent surveys, leading to annual TACs which are revised twice through the year and on ground-breaking ongoing consultation with stakeholders to determine the most effective and valuable management solutions. These solutions are taken on the basis of trade-offs and exploitation patterns that are simulated and negotiated (De Oliveira and Butterworth, 2004). Although initially TACs were often overruled or adjusted by political authorities (Cochrane, 1999), over the last decade OMP recommendations have been respected (Plagányi *et al.*, 2007). The South African OMP example provided the basis for the development of objective management criteria in the southwestern Australia sardine fishery, a process for which industry members actively participated. Developing adaptive management practices applied by independent governance structures capable of interacting at ecological, social and economic levels is a necessity for effective stewardship and governance.

The reliance of small pelagic fish populations on annual recruitment explains the boom and bust nature of their fisheries, but there is also increasing appreciation of the existence of low frequency, multidecadal productivity regimes (or regime shifts) underlying their long-term dynamics. Several authors have suggested that these regimes may be synchronized across basins (Kawasaki 1983; Lluch-Belda *et al.*, 1989, 1992; but see Freon *et al.*, 2003). Multi-decadal productivity cycles are poorly understood, although they are observable in centennial records (Baumgartner *et al.*, 1992) as well as in modern catch statistics (Klyashtorin 1998). Among the examples reviewed here, productivity cycles have been detected in the Brazilian and Californian sardine stocks, Humboldt sardine and anchovy, Black Sea anchovy and sprat, Baltic Sea sprat and Arcto-Norwegian spring spawning herring, among others. The existence of regimes poses substantial problems to fisheries managers because the effects of the fishery and its management cannot be separated from natural cycles. They also suggest that, for management decisions to be effective, they must occur in synchrony with these cycles (Fréon *et al.*, 2005). For example, there are current concerns over whether sardine in the northern Benguela can recover from its present low level when the structure and functioning of the system nowadays seems to differ substantially from the way it operated in the 1970s, when the population was at its peak (Heymans *et al.*, 2004), particularly as natural mortality appears to have increased in current times (Fossen *et al.*, 2001). Fréon *et al.* (2005) advocates a two-level management strategy that would combine conventional fisheries management to deal with short-term fluctuations, and a long-term management strategy (for example, driving fleet capacity and investment cycles) driven by decadal fluctuations. Because productivity regimes synchronistically affect different species in opposite ways, single-species management systems are more vulnerable than those that consider the dynamics of competing resources. For example, Baltic sprat and cod have dominated successive productivity cycles in recent decades, with consequences also for associated fish species such as herring (Köster *et al.*, 2003a). Simulations have demonstrated that consideration of species interactions stabilizes the assessment of sprat (ICES, 1997b).

King (2005) and Polovina (2005) conclude that the most appropriate approach to managing fisheries under scenarios of productivity cycles is through the application of regime-specific harvest rates (RSHR). These harvest rates should be part of a decision rule framework, associated with timeframes for management response triggered when there are indications that a regime shift has occurred. Improved results could be achieved even if the switch in harvest rates was delayed some years after the regime shift (MacCall, 2002) either because of detection delays or because stakeholder pressure and institutional stiffness delays implemention. Jacobson *et al.* (2001) noted that the instantaneous rate of surplus production was effective in detecting regime shifts, and therefore useful in adapting management practices in real time. Katsukawa and Matsuda (2003) suggested a management procedure based on non-parametric target switching, i.e. prohibiting catch of the most depleted species in favour of alternating species, which would bring more long-term yield and stable biomass than those obtained from a single-species point of view. However, the different value of the targets and the relationship between value and volume requires further investigation before target switching could be successfully applied.

De Oliveira (2006) investigated different harvesting strategies in the South African fishery for sardine and anchovy and concluded that management procedures designed under the assumption of out-of-phase sinusoidal trends in species abundance could be more effective than traditional modeling approaches. The author observed that, comparing regime shift estimators, a 6-year running mean of the estimator outperforms a precise knowledge of the actual position in the cycle. The performance of these management procedures shows rapid deterioration as the amplitude of regime cycles increases. In the case of marine ecosystems in which exploited small pelagic species play a key role, Fréon *et al.* (2005) recommend specific statistical and socio-economic studies on the feasibility of a two-level management strategy that would combine adaptive management (short-term) and fleet capacity control (long-term) and would take into account ecosystem considerations. In western Australia the current approach to dealing with biomass fluctuations has been to suggest a small pelagic carrying-capacity, which equates to an upper biomass level against which industry should consider investment decisons but also noting that SSB can be much less than the carrying capacity (Gaughan *et al*., 2004). Certainly, the development of exploitation strategies and levels in low and high abundance regimes must be a future research priority.

Regime-specific harvest rates, however, have a potential problem with the long-term investment in fisheries, while regime longevity is uncertain. For example, the Japanese purse-seine fishery – which is the major fishery for sardine, anchovy and mackerels – started its investments in new fishing fleets in the 1980s during the high stock period of Japanese sardine, resulting in over-capitalization when the fishery regime turned unproductive after 1988 (Yatsu, 2005; Yatsu *et al*., 2005). This mismatch in the longevity of fishing fleets vs. productivity regimes is a fundamental problem for sustainable fishing. Fleets expand following increasing trends in fish biomass, usually some time after the start of a new high productivity regime, and the length of this regime is generally shorter than the lifespan of the new fishing fleet, thus leading to overcapacity at the start of a subsequent low productivity regime. Based on a simulation study of a hypothetical sardine-like stock, Harada and Nishiyama (2005) argued that constant harvest rate strategies (CHRS), such as F_{MSY} with a precautionary approach, are preferable to RSHR because (1) CHRS is robust to estimation errors of stock abundance, (2) the resulting long-term catch is only slightly smaller than RSHR, and (3) CHRS is essentially constant for investment and thus can avoid overcapitalization derived from the above mismatch. CHRS also brings less year-to-year variation of catch and biomass compared to RSHR, hence it is preferable for the fishes and probably the surrounding fishery community. The performance of CHRS can be improved by incorporating catch prohibition of immature fish (Harada and Nishiyama, 2005). A CHRS that incorporates annual catch as an indicator of fishery performance is currently being investigated for the south-western Australian sardine fishery in recognition of both the low value of this fishery and the uncertainty regarding point estimates of SSB.

The assessment and management of small pelagic fish is also affected by their straddling nature across political boundaries. This review has noted how the management of Mediterranean and Black Sea fish stocks has been hampered by the international nature of the fisheries and the lack of arrangements for the development of common assessment and management tools. Some of the anchovy stocks in the Mediterranean are shared between Spain and France, others between Greece and Turkey, and others by Italy, Slovenia and Croatia. Each country has its own set of technical management measures, with different degrees of enforcement. Regional assessments and management (as proposed for the Black Sea; Prodanov *et al*., 1997; Zengin, 2003) are essential.

The California sardine is exploited by Mexico, USA, and Canada. Despite the recent creation of a Tri-National Sardine Forum, transboundary management has not been considered. In fact, USA and Mexico have no record of agreeing on a single management system or mechanism for setting or allocating harvest levels. Coordination and data sharing seem to be the most important initial steps, involving annual synoptic surveys, coordinated stock assessments, formalized data exchange, and agreement on the relative contributions of each country to fishing mortality. At a later stage these may allow consistent management guidelines which currently range from the USA's annual assessments with environmentally based harvest control rules, to Canada's limit based on a constant fraction of the USA's harvest guideline, to Mexico's fish size (but no total harvest) limits.

Discrepancies in trans-boundary management approaches often reflect national perceptions. During one of the latest crises in the management of the NE Atlantic sardine, additional management measures were applied to the Spanish and Portuguese fleets. The former did not contest the measures because the reduced abundance was already reflected in lower catches and revenues. In Portugal, however, the reduction in catches was buffered by price increases and as a result they did not see the need for concerted action.

Box 9.2. Management of fisheries under climate change

Climate change will affect, and is already likely to be affecting, fish populations worldwide. There are three generic impacts to be considered:

(a) changes in species distributions due to warming of the oceans and changes in circulation patterns,
(b) changes in species composition as a result of the previous impact, resulting in more tropical species expanding towards the subpolar regions, and
(c) changes in population parameters, such as growth rates, timing and success of reproduction. The latter will be affected by changes in the composition of the food web, temporal mismatch between fish larvae and their prey, and disruptions in the connectivity between spawning and recruitment grounds due to circulation changes.

There is no doubt that improved management of fisheries and marine ecosystems will play an important role in adapting to the impacts of climate change. Many of the improvements require new science and understanding, but particularly require the development of acceptable, effective responsive social institutions and instruments for achieving adaptive management (see this volume, Chapter 11). In the context of climate change, additional and continuous investment in monitoring exploited populations is needed to ensure adequate model parameterization. Most models on which management relies reproduce the dynamics of the stocks using specific population parameters such as average recruitment, growth and natural mortality rates or weight-at-age keys. As these will change with a changing climate, science must be responsive to this reality, for example, by using trends in specific variables rather than averages of a recent past. Well-designed and reliable monitoring programs of exploited stocks and their associated environment remain essential in order to detect changes and give advance warning of alterations in the productivity and structure of the marine ecosystem. In a changing environment, management advice needs to include complete and transparent information on risks and uncertainties which arise from data quality and assessment models' structural deficiencies.

Climate change is another stress on exploited fish resources, which compounds the impacts of overfishing, making the populations less resilient to unfavorable environmental conditions and more vulnerable to excessive exploitation. Under climate change, management will have to be even more precautionary, paying particular attention to uncertainties, and developing structures and institutions capable of applying adaptive management measures on behalf of all stakeholders.

The ICES system is a model for the generation of cross-boundary assessments. Its scientific advice to management bodies (e.g. European Union, Northeast Atlantic Fisheries Commission) is based on the work of ICES expert groups that include government scientists from ICES countries. Government laboratories contribute data and analysis, and the outcomes of their common work are translated into operational advice by ICES advisory committees. The management result is, however, variable. The North Sea herring, which is fished by many nations, assessed by ICES and managed by an EU/Norway management agreement, is an example of a well-managed stock that has recently achieved Marine Stewardship Council (MSC) accreditation, as recognized by FAO. However, ICES advice is not always followed, as noted in the case of the Bay of Biscay anchovy discussed above. In the case of the Norwegian spring spawning stock the lack of international agreement on quotas has maintained fishing mortality above the target value of 0.125 y^{-1}, resulting in catches of approximately 100 000 t more than required in the management plan. This species has a well-documented tendency to change its distributions and behavior, which complicates shared resource usage, particularly under threats of climate change. Hannesson (2006) studied the complex sharing of this stock between exploiting nations, and the role that straddling a hole of international waters can play in determining management strategies. Whether to adopt a competitive versus a cooperative multi-national management solution has substantial consequences. Climate-driven regime shifts can also play a significant role in destabilizing cooperation in the management of transboundary stocks. Shifts of productivity or geography require adaptive and resilient management institutions, based on innovative models of co-operative games in which the players anticipate the possibility of a regime shift (or climate change-driven geographic displacement, Perry *et al.*, 2005) that radically alters the strength of their relative bargaining positions (Miller, 2007).

In recent years nations increasingly have accepted the obligation to consider the impacts of their policies on marine ecosystems. Ecosystem-based fisheries management has been defined as an approach that takes all major ecosystem components and services – both structural and functional – into account in managing fisheries. The EAF (Ecosystem Approach to Fisheries) is now widely accepted (although not yet widely implemented). As a result of the World Summit for Sustainable Development (WSSD), fisheries need to develop and implement an ecosystem approach to their management by 2012. The need to apply ecosystem considerations to fisheries management is unarguable. 30% of the world's marine primary productivity is used directly to sustain current catch levels (Pauly and Christensen, 1995) and about 25% of the catch is discarded (Alverson *et al.*, 1994). The widespread application of single-species MSY policies may have contributed to the evident deterioration in the structure of many ecosystems, in particular regarding the loss of top predator species (Walters *et al*., 2005). Managing small pelagic fish species under ecosystem considerations would be aimed at protecting their value as forage species to other fish, mammals, and birds, and thus to respect the integrity of ecosystems, a need increasingly recognized by fisheries managers (PFMC, 1998). To accomplish this requires quantitative understanding of marine food webs and their dynamics not only with respect to fisheries impact removing large quantities of predators and prey, but also environmental drivers altering structure and functioning of food webs. It has been argued that, while there may be insufficient understanding of the processes involved, a qualitative understanding may suffice to make management arrangements that recognize the pivotal role of small pelagics in marine foodwebs. In the case of the southwestern Australia sardine fishery the carrying capacity concept (including consideration of CHRS) has been applied specifically to account for ecosystem needs, with qualitative descriptions of the importance of small pelagics used as ancillary scientific advice when setting TACs. Besides trophic interactions, direct interactions also need to be managed. Due to political concerns, research activites for sardines in southwestern Australia are currently focused on mitigating interactions with seabirds and dolphins, rather than on stock assessment.

Of the species considered in this review only a small minority are managed under some ecosystem considerations. Baltic Sea clupeids are partially assessed using multispecies models, one of the few multispecies assessments currently implemented. However, assessments neither consider predatory interactions affecting fish stock recruitment nor intra- and interspecific competition, both being evident as important processes affecting Baltic fish stock dynamics. The South African combined anchovy and sardine OMP covers some requirements of an ecosystem approach by including certain risk criteria for both species. These risk criteria relate to low probabilities of depleting each resource below specified levels, and incidentally secure relatively high escapement on average for these species at levels (relative to average pre-exploitation abundances), which are similar to those adopted by CCAMLR[22] to take appropriate account of the needs of natural predators (of krill in CCAMLR's case). Ecological issues that are pertinent to South Africa's small pelagic fishery in terms of an EAF have been discussed by Shannon *et al*. (2006), and the OMP for this fishery is currently being revised to allow for explicit incorporation of the effects of small pelagic fish–predator (African penguin, *Spheniscus demersus*) interactions (Cunningham and Butterworth, 2006). In general, however, the Ecosystem Approach is still to be successfully applied to the management of small pelagic fisheries, and this may be the single most important driving force in influencing future assessment and management policies .

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NOTES

1 Virtual Population Analysis.
2 Million tonnes.
3 Northern anchovy.
4 Peruvian anchovy.
5 Vessel Monitoring System.
6 Daily Egg Production Method.
7 Spawning Stock Biomass.
8 Total Length.
9 Total Allowable Catch.
10 European Union.
11 Instituto de Fomento Pesquero.
12 Coefficient of Variation.
13 Kuroshio-Oyashio Transition Zone.
14 Catch-at-age Analysis for Sardine.
15 Operational Management Procedure.
16 General Linear Model.
17 Survey-Based.
18 Sea Level Pressure.
19 Individual Transferable Quota.
20 Maximal Spawning Stock Biomass.
21 International Union for the Conservation of Nature and Natural Resources.
22 Commission for the Conservation of Antarctic Marine Living Resources.

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