ICES Journal of Marine Science, ■■: ■■–■■ (2005)
doi:10.1016/j.icesjms.2005.01.008

Developing an operational reference framework for fisheries management on the basis of a two-dimensional index of ecosystem impact

Sergi Tudela, Marta Coll, and Isabel Palomera

Tudela, S., Coll, M., and Palomera, I. 2005. Developing an operational reference framework for fisheries management on the basis of a two-dimensional index of ecosystem impact. – ICES Journal of Marine Science, 62: 000–000.

A composite quantitative index for the percentage of primary production required to sustain fisheries and the average trophic level of catch (%PPR–TL_c) was employed to develop ecosystem-based reference functions suitable for fisheries management. Established ecosystem models, characterized by pairs of %PPR–TL_c, were classified as either sustainably exploited or ecosystem overfished, on the basis of the results of factorial correspondence analysis applied to selected ecological indices, and on information from various sources. Canonical discriminant analysis of these pairs was applied to establish the discriminant function to separate the two exploitation classes. Next, reference functions related to different probabilities of ecosystem overfishing were developed to obtain an operational framework for ecosystem-based fisheries management. Values of ecosystem-based maximum sustainable catches associated with different probabilities of belonging to a sustainable situation were calculated. Overall, results show that most current fishing scenarios entail high risks of ecosystem overfishing.

Keywords: ecosystem approach to fisheries, ecosystem index, ecosystem overfishing, fisheries management, primary production, reference function, trophic level of catch.

Received 1 April 2004; accepted 3 November 2004.

S. Tudela: WWF Mediterranean Programme, Canuda 37, 08002 Barcelona, Spain. M. Coll and I. Palomera: Institute of Marine Sciences (ICM-CSIC), Passeig Marítim de la Barceloneta 37-49, 08003 Barcelona, Spain. Correspondence to S. Tudela: tel: +34 93 3056252; fax: +34 93 2788030; e-mail: studela@atw-wwf.org.

Introduction

The primary production required to sustain fisheries (PPR; typically expressed as t km^{-2} y^{-1}) has been conceived as an ecological footprint that highlights the role of fishing, in channelling marine trophic flows towards human use (Pauly and Christensen, 1995), and is routinely deduced by Ecopath modelling (Christensen and Walters, 2004). The trophic level (TL) identifies the position of organisms in the food chain (Lindeman, 1942; Odum and Heald, 1975). By convention, primary producers and detritus have TL = 1; values for other groups are determined using mass-balance models, gut content analysis, or isotope data (Briand, 2000). The average trophic level of the catch (TL_c) reflects the strategy of a fishery in terms of foodweb components selected, and is calculated as the weighted average of TL of harvested species (Christensen and Walters, 2004).

Tudela (2003) proposed the use of %PPR (relative PPR, calculated by dividing by the total primary production available) in combination with TL_c as a quantitative ecosystem index to capture the effect of fisheries. The reason given was that, for a given %PPR, a fishery with a higher TL_c would be intrinsically less disruptive than a fishery with a lower one, and for a given TL_c, a lower %PPR would be less disruptive than one with a higher %PPR. Therefore, the sensitivity of a system to the effects of fishing depends on both indicators, and it cannot be deduced from either metric independently. Accordingly, Tudela (2003) defined a theoretical framework to relate the %PPR–TL_c parameter space to concepts of sustainable exploitation and ecosystem overfishing (EO; Figure 1). The challenge then is to establish the appropriate limit, as well as target reference functions (Caddy and Mahon, 1995) based on available ecological information.

Our objective here is to define a quantitative boundary for EO in this framework, based upon available information on %PPR–TL_c pairs and evidence of disruptive exploitation for different marine ecosystem types. In addition, the

1054-3139/$30.00

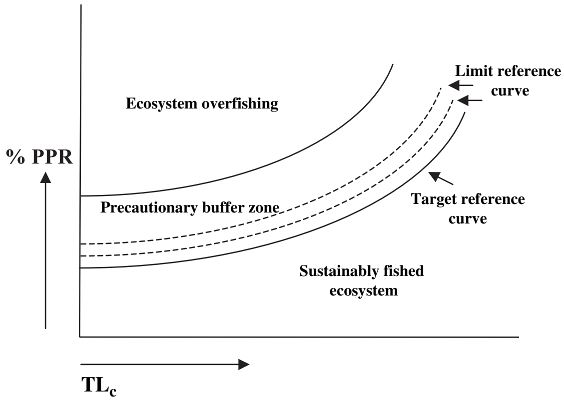

Figure 1. General theoretical framework based on the composite index %PPR−TL_c and ecosystem overfishing status (after Tudela, 2003).

framework is used to evaluate ecosystems for which direct evidence of EO status was lacking, and to calculate the ecosystem-based maximum sustainable catches (EMSC) related to the chosen reference functions.

Material and methods

Pairs of %PPR and TL_c were compiled on the basis of published Ecopath models (Christensen and Walters, 2004) for different areas and periods (www.seaaroundus.org). Models for upwelling regions were excluded, because of difficulties in distinguishing between effects of fishing and those caused by environmental factors. Because %PPR values for primary producers only and for primary producers and detritus combined are highly correlated and yield conceptually similar results, we restrict ourselves here to analyses for primary producers only. Each Ecopath model was characterized by a %PPR−TL_c pair, and classified if possible either as a sustainably fished, non-disrupted ecosystem, or as an overfished, structurally and functionally degraded ecosystem. Ecosystems were given an over-exploited status when cumulative fishing impacts manifested at least one overexploitation symptom, based on the EO definition proposed by Murawski (2000). The classification was based on specific information obtained from published and unpublished sources, on personal communication with experts on the ecosystem concerned, and on previous analysis of Ecopath-derived indices of ecosystem development, *sensu* Odum (Odum, 1969; Christensen, 1995b; Christensen and Pauly, 1998; Table 1). Factorial correspondence analysis (FCA), based on Pearson correlation coefficients and the WARD hierarchical method as aggregation algorithm (statistical package SPAD, version 5.2), was applied to a selection of Ecopath-derived indices (Table 1) from a data set comprising 27 models for different periods and six ecosystems. Table 2 lists the 49 ecosystem models included in the analysis that could be classified in

Table 1. Ecopath-derived ecological indices applied to factorial correspondence analysis, the expectedly/supposedly increasing (≫) or decreasing (≪) trend of fishery development according to ecological theory (Odum, 1969; Christensen, 1995b; Christensen and Pauly, 1998), and the percentage of analysed models behaving as expected.

Index	Description	Change (%)
B/T	Biomass/throughput ratio	≪60
P/R	Production/respiration ratio	≫87
PP/B	Total primary production/biomass	≫73
B/P	Biomass/production ratio	≪67
R/B	Respiration/biomass ratio	≪60
GE	Gross efficiency of the fishery	≫87
RO	Relative overhead	≪67
FCI	Finn cycling index (proportion of system throughput recycled)	≪67
PL	Path length index (measure of the complexity of ecosystems)	≪73
SOI	System omnivory index (related to variance in the TL of prey groups)	≫67

terms of sustainability/overexploitation criteria, and those that could not be classified.

Because of heteroscedasticity in the data, %PPR−TL_c pairs were standardized by log-transformation. The discriminant functions associated with the classified ecosystems were then obtained through canonical discriminant analysis (CDA; using SPSS, version 11) to check classification efficiency. Limit reference functions (following the nomenclature of Caddy and Mahon, 1995) were established on the basis of 50%, 70%, and 90% probabilities that %PPR−TL_c pairs belonged to a sustainably fished situation. Graphically, these functions can be used to identify the parameter space corresponding to different probabilities of EO. The discriminant functions were then applied to categorize previously unclassified models (Table 2b), and the classification obtained was evaluated against available information on exploitation effects.

Following Pauly and Christensen (1995), catches can be expressed in terms of %PPR by

$$\text{Catch} = \left(\%\text{PPR} \times \text{PP}/\left(10^{\text{TL}_{c-1}}\right)\right) \times 9$$

Calculating %PPR from the limit reference functions defined above allows us to estimate the ecosystem-based maximum sustainable catch (EMSC) for each ecosystem type, knowing the associated TL_c and primary production (PP) values.

Results

FCA results for historical series of exploited ecosystems confirm that, among different %PPR−TL_c pairs from a given ecosystem, the suite of ecological indices chosen

Table 2. Ecosystem models (listed by main ecosystem types) that (a) could, and (b) could not be classified according to ecosystem overfishing criteria (Murawski, 2000; EO status: +: overexploited, −: not overexploited).

Ecosystem model (a) Classified	TL_c	%PPR	EO	Reference
Temperate shelves and seas				
1 Faroe Islands (1961)	3.9	14.7	−	Guénette *et al.* (2001)
2 Icelandic fisheries (1950)	3.4	1.9	−	Guénette *et al.* (2001)
3 North Sea (1880)	3.8	0.3	−	Guénette *et al.* (2001)
4 North Sea (1963)	3.9	62.6	+	FC database
5 North Sea (1974)	3.9	62.0	+	FC database
6 North Sea (1981)	3.4	21.8	+	Christensen (1995a)
7 Newfoundland (1900)	3.5	27.2	−	Heymans (2003)
8 Newfoundland (1985−1987)	3.9	6.3	+	Heymans (2003)
9 Newfoundland (1995−2000)	3.1	15.1	+	Heymans (2003)
10 Norwegian and Barents Sea (1950)	3.6	11.7	−	Guénette *et al.* (2001)
11 Eastern Bering Sea (1950s)	3.4	30.9	−	Trites *et al.* (1999)
12 Eastern Bering Sea (1980s)	3.3	15.2	+	Trites *et al.* (1999)
13 Northern British Columbia (1750)	3.5	4.0	−	Ainsworth *et al.* (2002)
14 Northern British Columbia (1900)	3.3	23.3	−	Ainsworth *et al.* (2002)
15 Northern British Columbia (1950)	3.4	12.3	+	Ainsworth *et al.* (2002)
16 Northern British Columbia (2000)	3.3	9.8	+	Ainsworth *et al.* (2002)
17 Northern Gulf of St Lawrence (1985−1987)	3.6	22.9	−	FC database
18 Lancaster Sound Region (1980s)	4.1	3.3	−	Guénette *et al.* (2001)
19 Georgia Strait (1950)	3.3	7.0	−	Pauly *et al.* (1998)
20 West Greenland shelf (1997)	3.2	20.2	+	Guénette *et al.* (2001)
21 Scotian shelf (1980−1985)	3.7	8.0	−	Pitcher and Cochrane (2002)
22 South Catalan Sea (1994−2000)	3.1	43.9	+	Coll *et al.* (2004)
23 Azores archipelago (1997)	3.8	0.3	−	Guénette *et al.* (2001)
24 Cantabrian Sea (1994)	3.8	82.3	+	Sánchez and Olaso (2004)
Tropical shelves and seas				
25 Gulf of Thailand (1963)	3.0	1.8	−	Christensen (1998)
26 Gulf of Thailand (1980s)	3.1	16.1	+	Christensen (1998)
27 Gulf of Thailand (1993)	3.0	6.7	+	FC database
28 Southwest coast of India (1994)	2.6	13.2	+	Silvestre *et al.* (2003)
29 Southwest coast of India (1995)	2.6	10.7	+	Silvestre *et al.* (2003)
30 Southwest coast of India (1996)	2.6	11.7	+	Silvestre *et al.* (2003)
31 Southern shelf of Brazil (1975−1979)	3.6	27.7	+	Vasconcellos and Gasalla (2001)
32 Southern shelf of Brazil (1990−1994)	3.6	27.6	+	Vasconcellos and Gasalla (2001)
33 Southeastern shelf of Brazil (1977−1980)	2.8	33.1	+	Vasconcellos and Gasalla (2001)
34 Southeastern shelf of Brazil (1990−1995)	3.1	52.7	+	Vasconcellos and Gasalla (2001)
35 Venezuelan northeastern shelf (1980s)	2.8	20.7	+	Christensen and Pauly (1993)
36 Gulf of Mexico continental shelf (1990s)	2.6	31.6	+	Christensen and Pauly (1993)
37 Brunei Darussalam (1980)	3.2	7.4	−	Christensen and Pauly (1993)
38 Vietnam−China shelf (1980)	3.3	9.7	−	Pauly and Christensen (1993)
39 South China deep sea (1980s)	3.5	10.0	−	Pauly and Christensen (1993)
40 Hong Kong (1990s)	3.0	27.2	+	Pitcher *et al.* (2002)
41 Bay of Bengal (1984−1986)	2.7	8.3	+	Silvestre *et al.* (2003)
42 San Pedro Bay (1994−1995)	3.3	3.1	−	Silvestre *et al.* (2003)
Coastal areas and coral reefs				
43 Bay of Revellata, Corsica (1998)	3.8	11.9	−	Pinnegar (2000)
44 Prince William Sound, Alaska (1994−1996)	4.1	4.5	−	Okey and Pauly (1998)
45 Coast of Western Gulf of Mexico (1990s)	3.4	89.5	+	Christensen and Pauly (1993)
46 Gulf of Lingayen (1990s)	3.3	51.6	+	Pauly and Christensen (1993)
47 Maputo Bay (1980s)	2.5	20.3	+	Christensen and Pauly (1993)
48 San Miguel Bay (1992−1994)	3.0	14.7	+	Bundy and Pauly (2001)
49 Boliano reef flat (1991)	2.2	2.8	+	Christensen and Pauly (1993)

(*Continued on next page*)

Table 2 (*continued*)

Ecosystem (b) Not classified	TL_c	%PPR	Partial information available	Reference
Temperate shelves and seas				
1 Faroe Islands (1997)	3.7	33.2	Recovering from heavy exploitation	Guénette *et al.* (2001)
2 Icelandic fisheries (1997)	2.9	1.2	Decrease in exploited biomass; cod heavily exploited	Guénette *et al.* (2001)
3 Norwegian and Barents Sea (1997)	3.5	17.9	Medium-heavily exploited	Guénette *et al.* (2001)
4 Central North Pacific (1998)	3.8	20.4	Fishery on top predators	FC database
5 West Greenland shrimp trawling area (1994)	3.3	3.6	Regime shift; large cod bycatch	Guénette *et al.* (2001)
6 Gulf of Maine, Georges Bank (1982)	3.5	6.5	Heavily exploited	Guénette *et al.* (2001)
7 Atlantic coast of Morocco (1980s)	3.2	5.5	Heavily exploited	Guénette *et al.* (2001)
Tropical shelves and seas				
8 Campeche Bank of Yucatan shelf (1990s)	4.1	50.4	Heavily exploited; some overexploited resources	Christensen and Pauly (1993)
9 Gulf of Thailand (1973)	2.7	3.3	Moderately exploited	FC database
10 Kuala Trengganu (1980s)	3.4	26.1	Moderately exploited	Pauly and Christensen (1993)
11 Bali Strait (1990s)	2.9	6.9	Heavily exploited, sardine possibly overexploited	Pitcher and Cochrane (2002)
Coastal areas and coral reefs				
12 Schlei fjord (1984)	3.1	43.4	Moderately exploited	Christensen and Pauly (1992)
13 Beach-seine fishery area of Gulf of Mexico (1990s)	3.1	52.0	Fishing pressure has declined	Christensen and Pauly (1993)
14 Shallow areas of Gulf of Thailand (1979)	2.6	2.6	Small-scale fisheries	Pauly and Christensen (1993)
15 North coast of central Java (1979–1980)	3.0	11.4	Heavily exploited	Silvestre *et al.* (2003)
16 Great Barrier Reef, prawn ground in the far north (1994–1996)	2.4	7.9	Fully exploited; no evidence of overexploitation	FC database

was useful in tracking structural and functional changes within the ecosystem driven by changes in exploitation scenarios (Table 1), so were broadly consistent with the theory that resilience decreases if fishing pressure increases.

Figure 2 presents paired estimates of %PPR–TL_c for classified cases (Table 2a). Sustainably fished ecosystems are limited by values of TL_c ranging from 3.0 to 4.1, and of %PPR ranging from 0.3 to 14.7, with few exceptions (Newfoundland 1900; Eastern Bering Sea 1950s; northern British Columbia 1900). Overfished ecosystems have a wider TL_c range of 2.2–3.9 and %PPR of 2.8–89.5, except in the case of Newfoundland (1985–1987), which has a high TL_c and a low %PPR.

CDA applied to the %PPR–TL_c pairs from previously classified models correctly reclassify 88% of the cases (Figure 3). Figure 2 also shows limit reference functions associated with probabilities of the EO status, using the discriminant function from CDA. Sustainably fished ecosystems involve a $TL_c > 3.0$ and a low to moderate %PPR, and the range of sustainable levels for %PPR increases with TL_c. The likelihood of EO status clearly increases as TL_c decreases, and models located within this area are characterized by fisheries exploiting small fish and invertebrates (Table 2a).

Pairs of %PPR–TL_c for unclassified models (Table 2b) were plotted within the general framework defined (Figure 4). Faroes (1997) and Norwegian and Barents Sea (1997) models were located above the (50% probability) threshold curve for EO. The rest of the temperate models fitted within the range for sustainable exploitation. All models from tropical shelves and seas and coastal areas and coral reefs were located above the threshold curve,

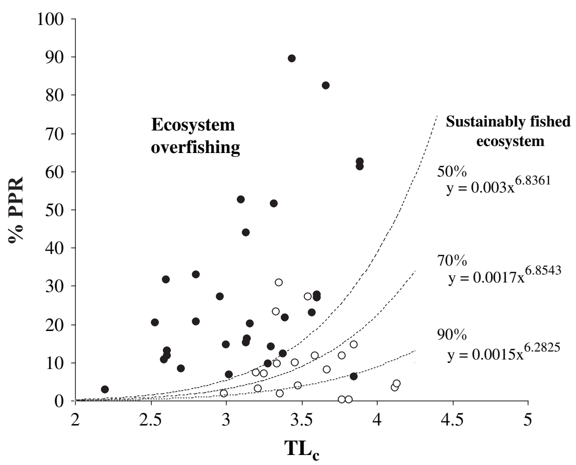

Figure 2. Ecosystem-based reference framework based on %PPR–TL_c, with reference functions related to 50%, 70%, and 90% belonging to a sustainably fished situation.

ARTICLE IN PRESS

DTD 5

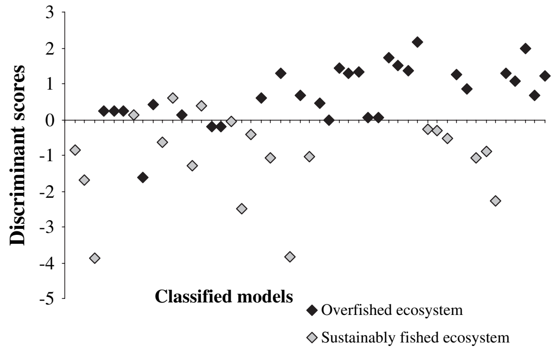

Figure 3. Discriminant scores from canonical discriminant analysis of pairs of %PPR−TL_c (the sequence of models following that in Table 2a).

indicating that present exploitation implies a high risk of EO status. In general, the results fit the fragmentary information available from these ecosystems (Table 2b).

Table 3 lists a compilation of estimates of ecosystem-based maximum sustainable catches (EMSC) for tropical and temperate shelves and seas and coastal areas and coral reefs, in comparison with current annual catches. Reported catches are far in excess of $EMSC_{50}$ and $EMSC_{70}$ estimates, by factors of 1.7 and 2.9, respectively, in temperate shelves and seas, 2 and 3.4 in tropical shelves and seas, and 2.7 and 4.6 in coastal areas and coral reefs.

Discussion

Some limitations of the data set need to be considered when it comes to interpreting the ecosystem-based framework developed for fisheries management, and to applying it to new pairs of %PPR−TL_c characterizing exploited ecosystems. First, more information was available for temperate areas than for other marine ecosystem types, and modelled

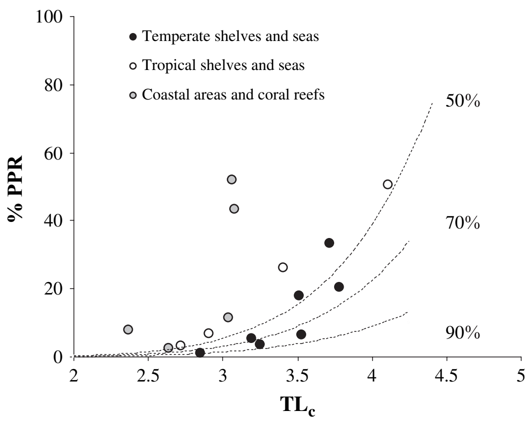

Figure 4. Previously unclassified pairs of %PPR−TL_c (Table 2b) plotted within the ecosystem-based operational reference framework.

ecosystems included in the analysis referring to reconstructed past situations are mainly from temperate regions. The models for historical situations are fundamental to this approach (Jackson *et al.*, 2001), even if their precision tends to be lower than for more recent models (Pitcher, 2001). Furthermore, catches are subject to high uncertainty, owing to difficulties in estimating discards, bycatches, and illegal, unreported, or unregulated catches. However, the proposed ecosystem-based framework for fisheries management and the related reference functions can be modified easily whenever more accurate data become available.

The initial hypothesis on the behaviour of %PPR and TL_c with respect to ecosystem overfishing (Tudela, 2003) has been confirmed, and can now be described mathematically, allowing limit reference functions to be established. Any new pair of %PPR−TL_c for any exploited ecosystem can now be directly assessed in terms of its deviation from a situation desirable to preclude EO status. Overall, the framework appears to represent a clear step forward in the difficult process of translating concepts of "ecosystem overfishing" into operational definitions that can be used in fisheries management (Larkin, 1996), and it provides a holistic ecosystem-based evaluation of a harvesting strategy, taking into account marine foodweb structure and functionality.

Discriminant analysis correctly reclassified a large number of models. Exceptions refer to reconstructed Newfoundland (1900), eastern Bering Sea (1950s), and northern British Columbia (1900) systems, which were characterized by great fishing pressure on marine mammals and a high %PPR. Newfoundland (1985−1987) was classified as an overfished ecosystem (Heymans, 2003), but it is located within the range of sustainable exploitation. This may be due to a transitional state of the ecosystem, caused by a supposed regime shift (Steele, 1998). This stresses the importance of detailed and wide-ranging historical information when interpreting ecosystem state.

Results highlight the fact that ecosystem overfishing is reached sooner following an increase in %PPR operating on low-TL species. Clearly, disrupting energy flows lower in the trophic web has the farthest-reaching effects on the whole ecosystem. This is important in the context of evidence about the rapid worldwide depletion of predator fish, and the consequent decrease of the TL_c (Christensen *et al.*, 2003; Myers and Worm, 2003).

According to our results, %PPR values of $\geq 20\%$ are only compatible with sustainable exploitation if fisheries are harvesting high in the foodweb ($TL_c > 3.5$). The sustainable ranges for TL_c and %PPR values identified contrast with estimates reported by Pauly and Christensen (1995). This raises serious concern about the sustainability of marine ecosystems under current exploitation regimes, mainly characterized by fisheries focusing on low-TL species and a high harvesting intensity. If reconstructed situations from the past are eliminated, Figure 2 provides a good graphic overview of the overall impact of current

Table 3. Ecosystem-based maximum sustainable catches (EMSC) at a 50% and 70% probability of being exploited sustainably for different ecosystem types, compared with current levels of catch and discards.

Ecosystem type	PP* ($gC\,m^{-2}\,y^{-1}$)	TL_c*	Catch and discards ($t\,km^{-2}\,y^{-1}$)*†	$EMSC_{50}$ ($t\,km^{-2}\,y^{-1}$)	$EMSC_{70}$ ($t\,km^{-2}\,y^{-1}$)
Tropical shelves and seas	310	3.3	2.87	1.46	0.84
Temperate shelves and seas	310	3.5	2.31	1.38	0.80
Coastal areas and coral reefs	890	2.5	10.5	3.96	2.28

*Pauly and Christensen (1995).
†1988−1991.

fishing practices on marine ecosystems: clearly, overexploited systems now dominate.

Estimated EMSC values below current catch levels agree with previous assessments of exploited ecosystems dominated by EO situations, and point to current catches being well beyond sustainable levels. Moreover, EMSC estimates could be overestimated because they are theoretical maxima, considering a mean TL_c. If the fishery is operating on a broad spectrum of TLs, sustainable maxima will be lower than those proposed, owing to the exponential relationship relating %PPR and TL_c.

Catches exceeding EMSC might be maintained within chronically degraded ecosystems, although the long-term viability of such a situation is doubtful, and large-scale resource collapses may continue (Jackson *et al.*, 2001). This conclusion reinforces the need to apply a more generous precautionary approach to fisheries management.

Acknowledgements

We thank Villy Christensen for providing the database and useful advice, various scientists for supplying essential information about, and personal experience with, the ecosystems modelled, and Maria Lourdes Palomares and Niels Daan for constructive comments on the draft manuscript.

References

Ainsworth, C., Heymans, J. J., Pitcher, T., and Vasconcellos, M. 2002. Ecosystem models of northern British Columbia for the time periods 2000, 1950, 1990 and 1750. Fisheries Centre Research Reports, 10(4). 41 pp.

Briand, F. (Ed). 2000. Fishing down the Mediterranean food webs? CIESM Workshop Series, 12. Kerkyra, Greece, July 2000.

Bundy, A., and Pauly, D. 2001. Selective harvesting by small-scale fisheries: ecosystem analysis of San Miguel Bay, Philippines. Fisheries Research, 53: 263−281.

Caddy, J. F., and Mahon, R. 1995. Reference points for fisheries management. FAO Fisheries Technical Paper, 347. 83 pp.

Christensen, V. 1995a. A model of trophic interactions in the North Sea in 1981, the year of the stomach. Dana, 11: 1−28.

Christensen, V. 1995b. Ecosystem maturity − towards quantification. Ecological Modelling, 77: 3−32.

Christensen, V. 1998. Fishery-induced changes in a marine ecosystem: insight from models of the Gulf of Thailand. Journal of Fish Biology, 53(Suppl A): 128−142.

Christensen, V., Guenette, S., Heymans, J. J., Walters, C., Zeller, D., and Pauly, D. 2003. Hundred-year decline of North Atlantic predatory fishes. Fish and Fisheries, 4: 1−24.

Christensen, V., and Pauly, D. 1992. Ecopath II. A software for balancing steady-state models and calculating network characteristics Ecological Modelling, 61: 169−185.

Christensen, V., and Pauly, D. (Eds). 1993. Trophic Models of Aquatic Ecosystems. ICLARM Conference Proceedings, 26: 390 pp.

Christensen, V., and Pauly, D. 1998. Changes in models of aquatic ecosystems approaching carrying capacity. Ecological Applications, 8(Suppl 1): 104−109.

Christensen, V., and Walters, C. 2004. Ecopath with Ecosim: methods, capabilities and limitations. Ecological Modelling, 172: 109−139.

Coll, M., Palomera, I., Tudela, S., and Sardà, F. 2004. Mass-balance approach applied to the north western Mediterranean Sea: the case study of the Delta of Ebro river area. Fourth World Fisheries Congress Abstracts. The American Fisheries Society, Vancouver, Canada. May 2004. p. 172.

Guénette, S., Christensen, V., and Pauly, D. 2001. Fisheries impacts on North Atlantic ecosystems: models and analyses. Fisheries Centre Research Reports, 9(4). 344 pp.

Heymans, J. J. 2003. Ecosystem models of Newfoundland and Southeastern Labrador: additional information and analyses for "Back to the Future". Fisheries Centre Research Reports, 11(5). 81 pp.

Jackson, J. B. C., Kirby, M. X., Berger, W. H., Bjorndal, K. A., Botsford, L. W., Bourque, B. J., Bradbury, R. H., Cooke, R., Erlandson, J., Estes, J. A., Hughes, T. P., Kidwell, S., Lange, C. B., Lenihan, H. S., Pandolfi, J. M., Peterson, C. H., Steneck, R. S., Tegner, M. J., and Warner, R. R. 2001. Historical overfishing and the recent collapse of coastal ecosystems. Science, 293: 629−638.

Larkin, P. A. 1996. Concepts and issues in marine ecosystem management. Reviews in Fish Biology and Fisheries, 6: 139−164.

Lindeman, R. L. 1942. The trophic-dynamic aspect of ecology. Ecology, 23: 399−418.

Murawski, S. A. 2000. Definitions of overfishing from an ecosystem perspective. ICES Journal of Marine Science, 57: 649−658.

Myers, R. A., and Worm, B. 2003. Rapid worldwide depletion of predatory fish communities. Nature, 423: 280−283.

Odum, E. P. 1969. The strategy of ecosystem development. Science, 104: 262−270.

Odum, W. E., and Heald, E. J. 1975. The detritus-based food web of an estuarine mangrove community. *In* Estuarine Research, 1. Ed. by L. E. Cronin. Academic Press, New York.

Okey, T., and Pauly, D. 1998. A trophic mass-balance model of Alaska's Prince William Sound ecosystem, for the post-spill

period 1994−1996, 2nd edn. Fisheries Centre Research Reports, 7(4). 146 pp.
Pauly, D., and Christensen, V. 1993. Stratified models of large marine ecosystems: a general approach and an application to the South China Sea. *In* Stress, Mitigation and Sustainability of Large Marine Ecosystems, pp. 148−174. Ed. by K. Sherman, L. M. Alexander, and B. D. Gold. AAAS Press, Washington, DC.
Pauly, D., and Christensen, V. 1995. Primary production required to sustain global fisheries. Nature, 374: 255−257.
Pauly, D., Pitcher, T., and Preikshot, D. 1998. Back to the future: reconstructing the Strait of Georgia ecosystem. Fisheries Centre Research Reports, 6(5). 99 pp.
Pinnegar, J. K. 2000. Planktivorous fishes: links between the Mediterranean littoral and pelagic. PhD thesis, University of Newcastle-upon-Tyne, UK. 213 pp.
Pitcher, T. J. 2001. Fisheries managed to rebuild ecosystems: reconstructing the past to salvage the future. Ecological Applications, 11: 601−617.
Pitcher, T., Buchary, E., and Trujillo, P. 2002. Spatial simulations of Hong Kong's marine ecosystem: ecological and economic forecasting of marine protected areas with human-made reefs. Fisheries Centre Research Reports, 10(3). 75 pp.
Pitcher, T., and Cochrane, K. 2002. The use of ecosystem models to investigate multispecies management strategies for capture fisheries. Fisheries Centre Research Reports, 10(2). 156 pp.
Sánchez, F., and Olaso, I. 2004. Effects on fisheries on the Cantabrian Sea shelf ecosystem. Ecological Modelling, 172: 151−174.
Silvestre, G., Garces, L., Ahmed, M., Stobutzki, I., Santos, R. A. V., Luna, C., Lachica-Aliño, L., Christensen, V., and Pauly, D. 2003. Assessment and management of tropical coastal fisheries in developing Asian countries. ICLARM Conference Proceedings, 67. 952 pp.
Steele, J. 1998. Regime shifts in marine ecosystems. Ecological Applications, 8(Suppl 1): S33−S36.
Trites, A. W., Livingston, P. A., Mackinson, S., Vasconcellos, M. C., Spriner, A. M., and Pauly, D. 1999. Ecosystem change and the decline of marine mammals in the eastern Bering Sea: testing the ecosystem shift and commercial whaling hypotheses. Fisheries Centre Research Reports, 7(1). 106 pp.
Tudela, S. 2003. The pair %PPR and $TL_{fishery}$: a possible composite indicator of ecosystem impact aligned with a holistic vision of the ecosystem-based fisheries management approach. Ecofish. In Workshop on Sustainability Indicators in Fisheries, Dakar, April 2003.
Vasconcellos, M., and Gasalla, M. A. 2001. Fisheries catches and carrying capacity of marine ecosystems in southern Brazil. Fisheries Research, 59: 279−295.