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An approach to the modelling of persistent pollutants in marine ecosystems

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Abstract

An approach for modelling trophic transfer of persistent pollutants within aquatic food webs is described, using radioactivity as an example. This involves constructing a mass-balance trophic model of the ecosystem in question, applying the Ecopath software, which uses the biomass, production/biomass, and food consumption rates of the various functional groups in the ecosystem as its basic inputs, along with a diet matrix. The Ecopath outputs used in this study are the estimates of biomass flow between functional groups, and the corresponding predation mortality matrix, whose columns represent the intake of, and the rows the losses of biomass from a compartment. A set of first order differential equations, relating the intake and loss of biomass to the amounts of radioactivity in the compartments, are then set up. There is additional accounting for loss of radioactivity due to physical decay of the radioisotopes. The equations are integrated over time and calibrated by minimizing the sum of squared deviations between the observed and predicted levels of radioactivity, thus mapping the transfers of radioactivity onto the transfers of biomass. The method is demonstrated through (a) a case study of beta radioactivity in a coral reef ecosystem used as testing ground for nuclear weapons (Enewetak Atoll, Marshall Islands, Micronesia), and (b) preliminary data on ¹³⁷Cs in the upper trophic levels of the Central Baltic Sea ecosystem, following the 1986 Chernobyl accident. The results support the applicability of the approach, for which a general solution, involving an 'importance-sampling' routine, is proposed.

Keywords

Central Baltic Sea, compartment modelling, Ecopath, food web, Enewetak Atoll, persistent pollutants, radioactivity, trophic mass balance models, trophic transfer.

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Introduction

Persistent pollutants, i.e., biologically non-degradable substances such as heavy metals and radionuclides, are of great concern when they occur in aquatic ecosystems, because of their potential to distribute themselves throughout diffuse food webs, and pose both direct and indirect threat to human health and welfare (Clark 1989, Lenssen 1991). The consumption of contaminated seafood can have serious health effects for humans and other top predators such as marine mammals. Tracking the fate of persistent pollutants is thus an important task which, until recently, was rendered difficult by the lack of a standardized approach for describing and verifying aquatic food webs.

Incomplete or thermodynamically unbalanced food webs have often been used to describe the fates of pollutants. Indeed, laboratory experiments structured around simplified food chains are probably among the main reasons for contradictory reports concerning the relative importance of transfers within food webs versus direct uptake (adsorption and absorption) of contaminants by aquatic organisms (Polikarpov 1966, Opel and Judd 1966, Townsley 1966, Hewett and Jefferies 1978, Thomann 1981, Rowan and Rasmussen 1994). Similarly, investigations based on field observations have suffered from difficulties in adequately representing and quantifying the trophic position of the organisms. This problem has impeded studies from examining the importance of trophic transfer in explaining observed patterns of contaminant bioaccumulation (Kiriluk et al. 1995, Vander Zanden and Rasmussen 1996).

Recent studies, based on field data and considering more complex food web structures (e.g., omnivory and fractional trophic levels), have found trophic transfer of contaminants to be significant (Thomann 1981, Rowan and Rasmussen 1994, Kiriluk et al. 1995, Vander Zanden and Rasmussen 1996). However, direct (non-trophic) uptake, no doubt plays an important role, especially for the lower trophic levels including primary producers, but is probably less important as one progresses up the food web, as suggested by Davis (1958).

In this paper we propose that thermodynamically stable food webs, i.e., meeting the criterion of mass-balance, should be used for investigating the fate of persistent pollutants, and further propose that the Ecopath approach and software, initiated by Polovina (1984), and further developed by Christensen and Pauly (1992, 1995), can be used to construct food webs onto which the fate of persistent pollutants can be 'mapped'. This mapping involves back-calibration of preliminary models, initially constructed without reference to the data on persistent pollutants, and subsequent modification of some of the model inputs, until a match is achieved between the food web and the pollutant data. The dissemination of pollutants can then be simulated, using the trophic fluxes determined from the model.

In this study, radionuclides are used as an example with the assumption that they are valid representatives of persistent pollutants. The method is applied to two case studies: (a) the coral reef system of Enewetak Atoll (Marshall Islands, Central Pacific), and (b) the brackish ecosystem of the Central Baltic Sea.

Methods

The modelling approach

Assuming similar conditions over a specified period of time, trophic interactions among the functional groups of an ecosystem ('compartments') can be described by a set of linear equations wherein

$$\text{production} = \text{predation} + \text{non predatory losses} + \text{harvest} + \text{export} \quad \dots 1)$$

In Ecopath, this is represented, for each functional group i , by

$$B_i \cdot (P/B)_i \cdot EE_i = Y_i + \sum B_j \cdot (Q/B)_j \cdot DC_{ij} \quad \dots 2)$$

where B_i and B_j are biomasses (the latter pertaining to j , the consumers of i); P/B_i their production/biomass ratio, equivalent to total mortality under most circumstances (Allen 1971); EE_i the fraction of production ($P = B \cdot (P/B)$) that is consumed within, or caught from the system (usually left as the unknown to be estimated when solving (2)); Y_i is the fisheries catch (i.e., $Y = F \cdot B$); Q/B_j the food consumption per unit biomass of j ; and DC_{ij} the contribution of i to the diet of j .

Solutions for unknowns in Eq. 2, e.g. B_i , are obtained by solving the matrix system in Eq. 2 through a robust inversion routine (MacKay 1981) in Ecopath. [The right side of Eq. 2 can also include a biomass accumulation term (B_{acc}) in cases where the biomass is known to have changed during the period under consideration. This theme is not pursued here (but see Christensen and Pauly 1995).].

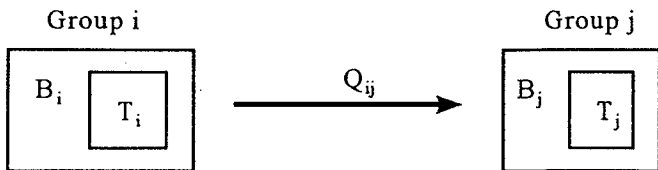
The solution of (2) allows calculation of the energy balance of each compartment, using:

$$\text{consumption} = \text{production} + \text{respiration} + \text{non-assimilated food} \quad \dots 3)$$

Here, mass-balance implies that equations (1-3) applies for all compartments of the ecosystem (typically 15 to 50), i.e., that the estimated EE_i range between 0 and 1 (a diagnostic for mass-balance).

The Ecopath software has a large numbers of outputs (Christensen and Pauly 1992, 1995); those which interest us here are the estimated fluxes of biomass among compartments and the related values.

Mapping the fate of persistent pollutants proceeds by assuming that they are distributed evenly within the compartments, i.e., one may think of the pollutants as 'tagged' biomass (T) that flows from one compartment to another. This can be represented by :



where B_i and B_j are the biomasses ($t \cdot km^{-2}$) of group i and j , respectively, T_i and T_j are the tagged biomasses ($t \cdot km^{-2}$) in group i and j , respectively, and Q_{ij} is the flux of biomass ($t \cdot km^{-2} \cdot year^{-1}$) from group i to j .

The transfer of pollutants per unit time from group i to j , p_{ij} , is proportional to the fraction of 'tagged' biomass to total biomass in group i , T_i/B_i , and the flux of biomass from group i to j :

$$p_{ij} = \frac{T_i}{B_i} \cdot Q_{ij} \Leftrightarrow p_{ij} = T_i \cdot \frac{Q_{ij}}{B_i} \Rightarrow p_{ij} = T_i \cdot M_{ij} \quad \dots 4)$$

where $M_{ij} = Q_{ij}/B_i$. M_{ij} is the transfer coefficient from group i to group j , i.e., that part of the natural mortality of i that is due to j (as output in the Ecopath predation mortality matrix). When dealing with radioactivity, there is an additional loss, δ , within each compartment, resulting from the physical decay of the radioisotopes. Combining the intake, loss and decay terms, the trend in radioactivity in the compartments may be described by a linear differential equation system of the form:

$$\frac{dT_j}{dt} = \overbrace{\sum_{i=1}^n T_i \cdot M_{ij}}^{\text{income}} - \overbrace{T_j \cdot \sum_{i=1}^n M_{ji}}^{\text{loss}} - \overbrace{\delta \cdot T_j}^{\text{decay}} \quad \dots 5)$$

which can be integrated over time.

The Solver routine in Microsoft Excel was used to minimize the sum of squared deviations between the observed and predicted levels of pollutant ($\Sigma(\ln \text{obs}/\text{pred})^2$) by varying the predation and prey mortalities (M_{ij} and M_{ji}). The changes were subsequently incorporated into the underlying Ecopath model by modifying the input biomass, i.e., the inputs directly proportional to the predation mortalities (see Eq. 2), used for the next iteration. This back-calibration of the preliminary Ecopath model ceased when the sum of squared residual was minimized.

First case study: beta radioactivity in the aquatic environment of Enewetak Atoll

From 1948 to 1958, Enewetak Atoll was used for nuclear testing by the U.S. military (Henry and Wardlaw 1990). Concurrently, scientific research was carried out to assess the impact of radioactivity on the biota, including the marine ecosystem (Helfrich and Ray 1987). Some of these results have recently been declassified and released, and a data set on observed beta radiation in various aquatic organisms was compiled, based on Bonham (1958), Palumbo (1959), Welander (1957). This data set forms the basis of our first case study.

An Ecopath model of the windward section of the atoll was constructed, based on a variety of published sources documented in Dalsgaard (1998). The model includes the 27 compartments shown in Table 1.

The theoretical gross beta-decay of mixed slow-neutron initiated fission products of $^{235}\text{Uranium}$ was calculated by Hunter and Ballou (1951). Over a period of 1 to 1000 days the decay is approximated by a straight line on a log-log scale, with an average slope of -1.2 (Bonham 1958). This form of decay can be described by the power function $D = a \cdot t^{-1.2}$, where D is the amount of radioactivity at time t in days after the detonation and a is the intercept. When differentiated, the equation may be re-expressed as:

$$\frac{dD}{dt} = a \cdot b \cdot t^{b-1} = b \cdot (a \cdot t^b) \cdot t^{-1} = b \cdot D \cdot t^{-1} = \left(\frac{b}{t}\right) \cdot D = \delta \cdot D, \quad \dots 6)$$

where $\delta = (b/t)$, which can be incorporated into Equation (5), with b equal to 1.2.

Table 1. Selected parameters^a from the balanced Ecopath model of Enewetak Atoll. The model was balanced using the EcoRanger routine in Ecopath, with acceptable ranges of $\pm 99\%$ around all input parameters (details in Dalsgaard 1998).

Compartment	Trophic level ^b	B (t·km ⁻²)	P/B (year ⁻¹)	Q/B (year ⁻¹)	EE	P/Q	Flow to detritus (t·km ⁻² ·year ⁻¹)
Detritus	1.00	185.00	-	-	0.67	-	0.0
Phytoplankton	1.00	1.32	563.38	0.00	0.78	-	166.9
Benthic primary producers	1.00	3195.14	2.61	0.00	0.07	-	7740.9
Sessile invertebrates	2.00	19.06	3.24	39.59	0.42	0.08	186.6
Bivalves	2.00	27.37	2.01	12.12	0.03	0.17	119.5
Giant clam	2.00	7.38	0.11	3.82	0.00	0.03	6.4
Zooplankton	2.01	28.62	31.65	200.53	0.97	0.16	1751.4
Foraminiferans	2.12	30.63	10.63	21.74	0.37	0.49	337.8
Echinoderms	2.13	56.05	1.44	3.87	0.03	0.37	121.2
Miscellaneous crustaceans	2.13	8.47	4.92	22.13	0.43	0.22	61.4
Surgeon fish	2.15	9.75	1.32	16.87	0.24	0.08	75.6
Shrimps	2.17	2.33	5.17	27.78	0.71	0.19	16.4
Holothurians	2.44	28.84	0.27	4.00	0.12	0.07	76.2
Polychaetes	2.51	26.48	5.88	16.85	0.66	0.35	141.5
Stomatopods	2.75	0.34	1.92	20.49	0.83	0.09	1.5
Corals	2.76	230.56	1.45	4.48	0.72	0.32	298.8
Small omnivorous fish	2.78	1.94	3.23	13.90	0.78	0.23	6.8
Butterfly fish	2.84	6.77	1.98	15.32	0.16	0.13	32.0
Large omnivorous fish	2.86	1.10	1.80	7.78	0.97	0.23	1.8
Gastropods	2.96	20.40	1.44	6.56	0.92	0.22	29.0
Herring	3.11	0.31	2.72	43.85	0.22	0.06	3.4
Small carnivorous fish	3.29	3.56	1.49	9.47	0.93	0.16	7.1
Cephalopods	3.37	0.32	1.24	4.49	0.73	0.28	0.4
Large carnivorous fish	3.37	0.81	0.85	5.19	0.89	0.16	0.9
Parrot fish	3.46	5.62	1.34	15.64	0.41	0.09	39.6
Snapper/groupers	3.48	0.23	0.73	8.78	0.94	0.08	0.4
Miscellaneous piscivorous fish	3.85	5.02	0.25	3.18	0.29	0.08	4.1

a. B = biomass, P/B = production/biomass, Q/B = consumption/biomass, EE = ecotrophic efficiency, P/Q = production/consumption (growth efficiency);

b. Trophic level (TL) = 1 in primary producers, 2 in herbivores, 3 in animals consuming only herbivores, etc. The TLs are computed as: 1 + mean trophic level of preys, and may thus be non-integer values. Corals were considered to be 70% autotrophic and 30% carnivorous (Sorokin 1993) feeding on zooplankton which was assumed to be mainly herbivorous (see Dalsgaard 1998).

Pending a generic method for matching Equation (5) with Equation (2) (see below), the predation mortality matrix from the original Ecopath model was modified, using Solver, to simulate a trend that was more consistent with the observed data: the columns of the predation mortality matrix were scaled up/down by multiplying them with a scaling factor that Solver was preprogrammed to vary within a certain range (0.25 - 1.75), while minimizing the residuals defined above. The rows of the predation mortality matrix (i.e., the prey mortalities) were also modified, by allowing Solver to add an additional mortality to the sum of the prey mortalities (sum of the rows; see below).

Second case study: ¹³⁷Cesium in the Central Baltic Sea

The fate of radionuclides in the Baltic Sea has been monitored since 1981, and a report following the reactor accident at Chernobyl, Ukraine, in April 1986 was published by the Baltic Marine Environment Protection Commission (HELCOM 1995). This report formed the basis of our second case study. According to HELCOM (1995), ¹³⁴Cs and ¹³⁷Cs from the Chernobyl accident and ⁹⁰Sr from the nuclear tests in the 1950s and 1960s are the dominant contaminant radioisotopes in the Baltic Sea, making it one of the most contaminated seas in the world. Due to its relative short half life of two years, ¹³⁴Cs decayed rapidly after the Chernobyl accident, leaving ¹³⁷Cs, with a half life of 30 years, as the major radioisotope. Combined with the morphology of the Baltic Sea (semi-enclosed, shallow, and brackish) and the heterogeneous distribution of the Chernobyl fallout, elevated ¹³⁷Cs concentrations were measured in the water column for a prolonged period of time. In 1991, the radioactivity was more homogeneously distributed. Also, it had declined though mixing and sinking of the water masses, and by removal through adsorption to particles and subsequent sedimentation. However, higher concentrations were still detected in the Bothnian Sea, i.e., in the Northeastern part of the Baltic.

Seasonal mass-balance models of trophic interactions in the Central Baltic Sea, i.e. the part of the Baltic Sea with the highest fishery catches, were constructed by Jarre-Teichmann (1995), using carbon as the unit of energy, and giving emphasis to the upper trophic levels of the ecosystem. The seasonal models were based on biomass from the late 1980s and early 1990s, when the cod stock was low. Data for the commercial fish species (herring, sprat, and cod) were derived from multispecies assessments in the Baltic Sea (ICES 1996) pertaining to ICES subdivisions 26-29. Other functional groups were the primary producers, several groups of planktonic invertebrates and benthos, as well as commercially less important fish. The input data for the invertebrate groups were largely based on Elmgren (1984) and Wulff and Ulanowicz (1989). The seasonal models were combined into an annual model (ICES 1998), used here, and a previous mis-estimate of mysid biomass was corrected, based on Hansson et al. (1997).

The decay of a large population of similar isotopes (in this case ¹³⁷Cs) can be described by a negative exponential function of the form $D = D_0 \cdot e^{-\lambda t}$, where D is the number of isotopes present at time t , D_0 is the number of isotopes present at time zero, and λ is the decay coefficient. When differentiated, the equation may be re-expressed as:

$$\frac{dD}{dt} = -\lambda \cdot D$$

...7)

which can be integrated into Equation (5)

In order to derive the radioactivity in the phytoplankton, observed concentrations of ^{137}Cs (in becquerel per cubic meter, i.e., $\text{Bq}\cdot\text{m}^{-3}$) in the surface layers of the 'Eastern' and 'Western' Baltic were taken from Fig. 8.3.1. in Nielsen et al. (1995), two areas which, combined, correspond to our definition of the Central Baltic Sea. Using surface area and mean depth estimates from Table 8.1.1. in Nielsen et al. (1995), a weighted mean was estimated ($\text{Bq}\cdot\text{m}^{-2}$), encompassing the entire water column (as do the biomass estimates in Ecopath)⁴. Assuming that 1 ml water \approx 1 g, an estimate of radioactivity in $\text{Bq}\cdot\text{g}^{-1}$ water was derived. Further assuming the same density for phytoplankton as for the surrounding water, the radioactivity of phytoplankton was derived as the ratio of phytoplankton biomass stock and radioactivity per g water. A linear regression was performed on the observed concentrations in phytoplankton over time, corresponding to the years 1986-1991 (Fig. 1), and the result used in the simulation.

The same approach was used to derive estimates of the radioactivity in detritus, whose biomass in the Ecopath model had been derived from Eq. (A5) in Pauly et al. (1993). Estimated ^{137}Cs activities in detritus forming the upper layer of sediments (Ilus et al. 1995) were not included, as they could not be straightforwardly converted into estimates pertaining to the water column as a whole. The estimation of ^{137}Cs activity for the other functional groups, and their conversion to $\text{Bq}\cdot\text{m}^{-2}$, are documented in Table 2.

The best fit between the observed and predicted levels of ^{137}Cs was achieved in a similar manner as for the first case study. However, Solver was preprogrammed to vary the scaling factors within a narrower range (0.75 - 1.25).

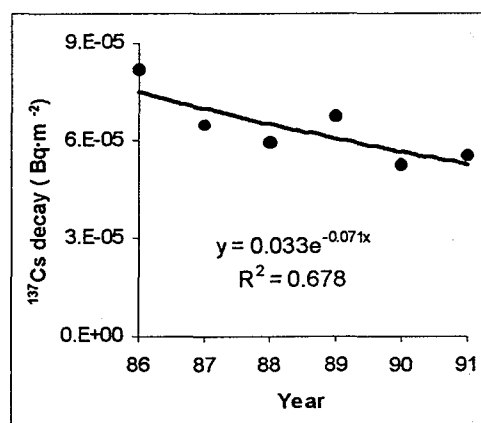


Figure 1. Regression of the observed concentrations of ^{137}Cs ($\text{becquerel}\cdot\text{m}^{-2}$) in phytoplankton over time, in the Central Baltic Sea.

Results.

First case study: beta radioactivity in Enewetak Atoll

The result of the simulation is presented in Fig. 2, which features the trends in beta radioactivity predicted from both the preliminary and the re-calibrated Ecopath model. Table 3 summarizes changes to the predation mortality matrix required for minimizing the residuals and rebalance the back-calibrated Ecopath

⁴ Radioactivity is measured in number of disintegrations per unit of time. ^{137}Cs in the Central Baltic Sea was measured in becquerels (Bq) which are disintegrations $\cdot\text{s}^{-1}$. This combined with the 'currency' used in the Ecopath model resulted in the unit: $\text{Bq}\cdot\text{m}^{-2}$. Beta radioactivity at Enewetak Atoll was measured in disintegrations $\cdot\text{min}^{-1}$, which, combined with the Ecopath model, resulted in the unit: disintegrations $\cdot\text{min}^{-1}\cdot\text{km}^{-2}$.

model. The modifications were justifiable as the initial biomass estimates were associated with high uncertainties, being based either on visual censuses, or generated by the Ecopath program, given the other input parameters (Dalsgaard 1998). The 'additional mortality', set by Solver in the course of the minimization, cannot readily be related to the Ecopath outputs, a point to which we return in the Discussion.

Table 2. Summary of the relationships used to convert observed data on ^{137}Cs in the Central Baltic Sea into $\text{Bq}\cdot\text{m}^{-2}$. All wet weights (ww) were converted to carbon (C) using the conversion: $\text{ww}/10 = \text{C}$.

Compartment	Assumptions
Cod and herring	<ul style="list-style-type: none"> • ^{137}Cs activity in cod filets from the southern Baltic Proper and Bornholm Sea were derived from Fig. 7.2.1. c & d in Kanisch et al. (1995); • ^{137}Cs activity in herring filets from the northern and southern Baltic Proper were derived from Fig. 7.2.2. g & h in Kanisch et al. (1995); • The observations for cod and herring, respectively, from the observation areas was averaged; • Fish filets comprise 63% of the total body weight and ^{137}Cs occurs only in filets (Welander 1957).
Macrobenthos	<ul style="list-style-type: none"> • ^{137}Cs activity in <i>Mytilus edulis</i> can be used as a proxy for activity in macrobenthos; • ^{137}Cs activity in <i>Mytilus edulis</i>, soft parts, were derived from Fig. 7.3.1. b (Bornholm Sea) in Kanisch et al. (1995); • The observed data for <i>Mytilus edulis</i> can be converted from soft parts to whole body assuming that the soft parts comprise 5% of the total weight including the shell; • According to the text, the observed data for <i>Mytilus edulis</i> were in dry weight, and a conversion factor from dry to wet mass of 9 was used (Kanisch et al. 1995, Table 7.3.1.).
Benthic producers	<ul style="list-style-type: none"> • ^{137}Cs activity in <i>Fucus vesiculosus</i> may be used as a proxy for activity in benthic producers; • ^{137}Cs activity in <i>Fucus vesiculosus</i> was derived from Fig. 7.4.1. (Oskarshamm) in Kanisch et al. (1995); • The data for <i>Fucus vesiculosus</i> can be converted from dry mass to wet mass using a factor of 6 (Kanisch et al. 1995, Table 7.3.1.).

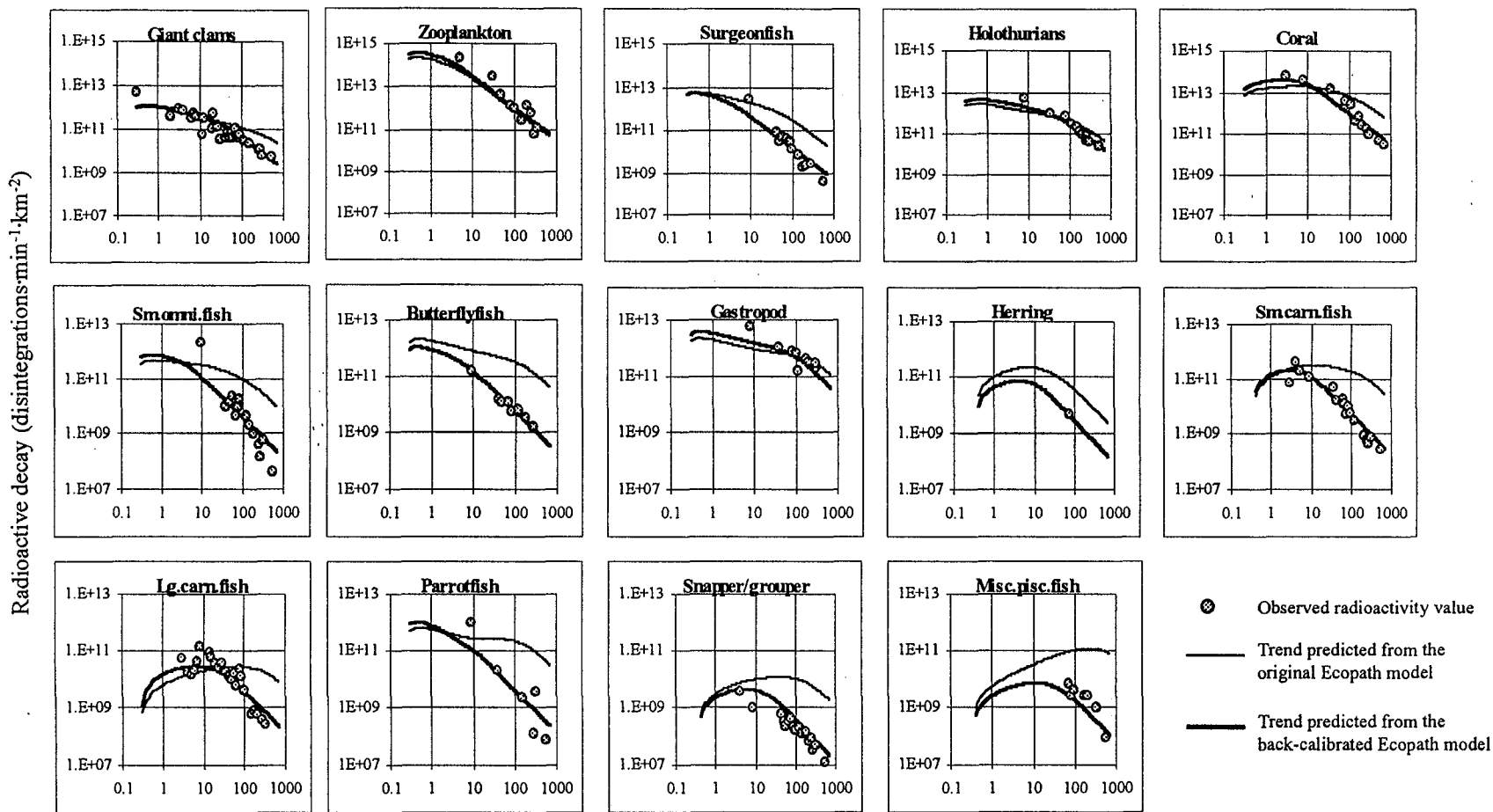


Figure 2. Simulated and observed concentrations of beta radioactivity (disintegrations·min⁻¹·km⁻²) in the compartments from Enewetak Atoll. The graphs are arranged such that the compartments with the lowest trophic level are in the top left corner and the highest trophic level compartments are in the bottom right corner. The dots illustrate the observed values, the thin lines the simulated trends prior to back-calibration of the original Ecopath model, and the thick lines the best fit obtained using Solver. Notice that both axes are logarithmic, and that the scale on the Y-axes varies from the first row of graphs to the second and third row.

Second case study: ¹³⁷Cesium in the central Baltic Sea

A reasonable fit of the modelled to the observed data was attained by running the simulation on the original model (Fig. 3) without major changes to the predation mortality matrix. The predation mortalities generated by the macrobenthos and by cod were lowered by 25%, while the predation mortality generated by herring was increased by 25%. Given the uncertainty about the composition and total biomass of macrobenthos in the Central Baltic Sea, and the large deviations in herring assessments between years (see e.g., Jarre-Teichmann et al. 1998), these changes from the initial inputs are acceptable. In order to re-balance the Ecopath model, the biomass of macrobenthos was therefore lowered by a factor of 0.75, and the biomass of herring increased by a factor of 1.25. The higher biomass of herring made it necessary to lower the fraction of mysids in the diet of herring from 3.0% to 2.7%, with the 0.3% difference assigned to mesozooplankton. Finally, as the gross efficiency (production/consumption, i.e., P/B over Q/B, see Eq.

2) of cod in our original model (20%) was lower than that used for gadoids in adjacent areas (e.g., ICES 1989, 1991), we increased it to 27%. This led to a 25% decrease in the predation mortality generated by cod.

Table 3. Scaling factors used in Solver for the predation mortalities from the Enewetak Atoll system, resulting in the best fit between the observed and predicted concentrations of beta radioactivity.

Functional group	Scaling factor
Misc. piscivorous fish	0.50 ^a
Snapper/grouper	0.50 ^a
Parrotfish	1.75
Large carnivorous fish	1.75
Small carnivorous fish	0.80 ^a
Herring	0.25
Gastropods	1.75
Butterflyfish	0.55
Small omnivorous fish	1.75
Corals	1.75
Holothurians	1.75
Zooplankton	1.75
Giant clams	0.91
Phytoplankton	1.30 ^b

- a. Scaling factor modified from the original Solver-generated value of 1 to 0.5 for miscellaneous piscivorous fish, 0.25 to 0.5 for snapper/grouper, and 0.67 to 0.8 for small carnivorous fish;
- b. The biomass of phytoplankton (originally a model estimate) was increased by 30% to accommodate an increased pressure from zooplankton. The adjusted biomass was still within the ranges of phytoplankton biomass estimates in models of similar reef ecosystem (Opitz 1993, Arias-González 1984).

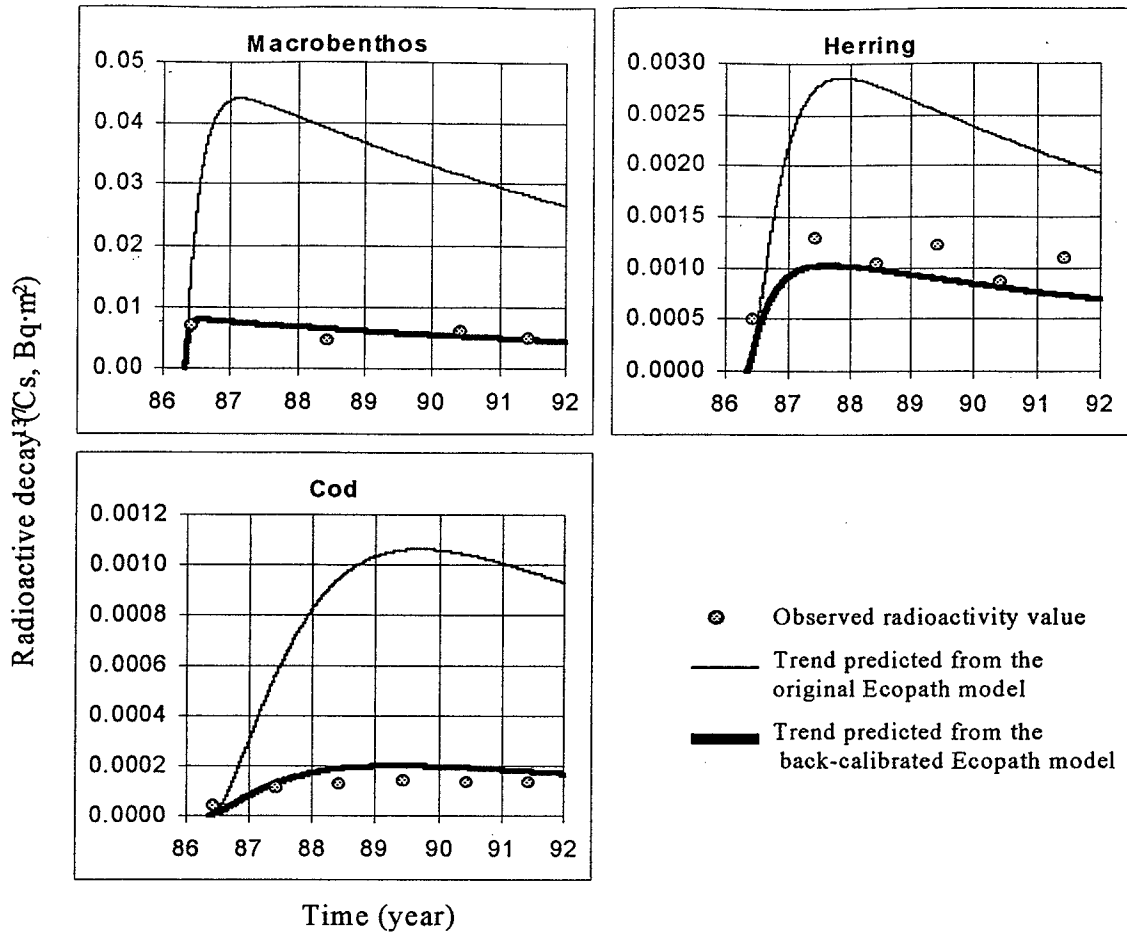


Figure 3. Simulated and observed concentrations of ^{137}Cs ($\text{becquerel}\cdot\text{m}^{-2}$) in compartments from the central Baltic Sea. The dots illustrate the observed values, the thin lines the simulated trends prior to back-calibration of the original Ecopath model, and the thick lines the best fit obtained using Solver. Notice that the scale on the Y-axes is different in all the graphs.

Discussion

Trophic transfer of pollutants within aquatic food webs should, in theory, imply a time lag between the observed maximum and the trophic position of the compartment, reflecting the time required for the pollutant to be moving up the food web. Maximum values should be observed first in the primary producers, then in herbivores, then in higher trophic levels, with top predators reaching their maximum concentrations last (Elliott et al. 1992). Thus, in the Central Baltic Sea, maximum ^{137}Cs values were attained by the macrobenenthos right after the Chernobyl accident, approximately one year later by herring, and three years later by cod. The scenario was also observed in the Enewetak Atoll ecosystem, though, the partly autotrophic (75%, see Dalsgaard 1998) giant clams and parrotfish differed from the expected trend (Fig. 4). In the version of Ecopath used here, trophic levels were computed as: $1 + \text{mean trophic level of preys}$. Since autotrophy is not considered in the calculation, the trophic level of partly autotrophic organisms like giant clams and corals is in fact too high, which in turn, affects the trophic level of the

organisms feeding on them. Parrotfish are an example of this as they were set to feed on both primary producers and corals in the model. Therefore, a detailed analysis of the 'food web time lag' will have to await until autotrophy is explicitly included in the calculations of trophic levels.

Many aquatic organisms are highly mobile, moving in and out of a contaminated area. Under these circumstances, elimination of the pollutant from the organisms and assimilation through predation, as opposed to direct uptake from the water, becomes the dominant factors in determining the levels of pollutant acquired by the organisms (Polikarpov 1966). Migrating organisms, in addition to currents, winds, tides, and rain (Welander 1957), have a diluting effect on the pollutant in the system. The Ecopath model assumes that foraging takes place only within the modelled area (even when 'imports' are included as food items), as defined by the diet composition of the functional groups explicitly included in the model. Violations of this assumption undoubtedly accounts for at least some of the differences between observed and predicted trends (Fig. 2 and 3) both before and after back-calibration of the models. In the present study, the problem caused by 'dilution' was solved by treating it as an additional mortality (M^+), on top of the total mortality (Z , i.e., P/B) to which a functional group is subjected.

In the case of the Enewetak Atoll ecosystem, we were not able to show a correlation between M^+ and the trophic level of the functional groups, although high trophic level groups tend to be more mobile than organisms at lower trophic levels. The lack of such a relationship suggests that physical dilution effects were more important in Enewetak Atoll than effects due to migration. This is not surprising for a relatively small coral reef, swept by strong currents (Atkinson et al. 1981, Atkinson 1987), and where most organisms are either sessile, or have only restricted ranges, with fish showing strong territorial behavior.

Overall, the results for Enewetak Atoll (Fig. 2) show a good agreement between model predictions and observed values. Fig. 2 and the regression in Fig. 5 further suggest that beta radioactivity is not bioaccumulated with higher trophic levels. A possible explanation for this could be that the majority of isotopes produced during a nuclear detonation have very short half lives. Combined with the time lag discussed above, this leads to a reduced amount of radioactivity being transferred up the food web. This, however, does not rule out the possibility that certain isotopes are bioaccumulated with higher trophic levels. Due to data limitation, we were not able to test for specific isotopes, but selective uptake of beta emitting radioisotopes by aquatic organisms is known to occur. For example, herbivorous fish have been shown to accumulate $^{65}\text{Zinc}$ and $^{57,58,60}\text{Cobalt}$, whereas first order carnivorous fish predominantly take up $^{65}\text{Zinc}$ and $^{55,59}\text{Iron}$. Important radioisotopes found in higher order carnivores are $^{65}\text{Zinc}$ followed by $^{141,144}\text{Cerium}$, $^{57,58,60}\text{Cobalt}$, and $^{103,106}\text{Ruthenium}$ (Babiet and Perrault 1987).

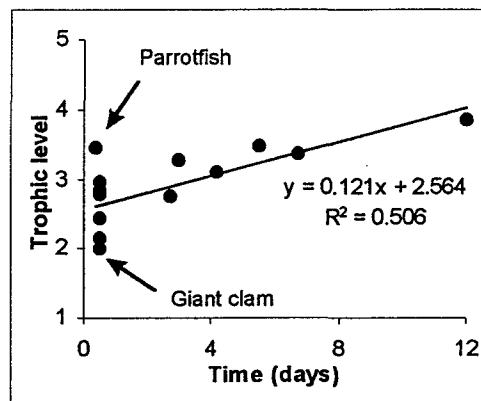


Figure 4. Regression of the trophic level as a function of the number of days required to obtain the maximum level of beta radioactivity in the compartments of the Enewetak Atoll ecosystem.

Given that the Ecopath model of the Central Baltic Sea used here is a preliminary one (Jarre Teichmann 1995, ICES 1998), we are pleased that a reasonable match of observed vs. predicted levels of ^{137}Cs was obtained with relatively few changes to this model, which originally generated ^{137}Cs activities considerably higher than the observed values (Fig. 3).

The measurements of ^{137}Cs available to us were highly aggregated, i.e., estimates by size (or age) classes of fish were not available. Also unavailable was the activity in small crustaceans, a crucial part in the food web of the Baltic (see e.g., Rudstam et al. 1994, Hansson et al. 1997). In addition, cod and herring are highly migratory, implying that for part of the year, they would have been subject to different levels of radioactivity in their environment, and consequently, in their food. As the migrations mainly take place between the Central and the Northern Baltic Sea

(Aro 1989, Bagge et al. 1994, Parmanne et al. 1994), the level of radioactivity to which these fish are exposed when outside of the Central Baltic will tend to be higher than in the area considered here. This may have been more pronounced for herring, a zooplankton-feeder, than for cod, a piscivore, which itself may explain the larger discrepancy between the simulated and observed activities for herring (Fig. 3).

The results from the Central Baltic Sea shows a decline in radioactivity with increasing trophic level. According to Rowan and Rasmussen (1994) it is generally agreed that “food is the major route of cesium uptake by fish”, but there is considerable controversy as to whether it is bioaccumulated or not. This is confirmed by this study, and by Mailhot et al. (1988), who found that cesium undergoes “biodiminution” in aquatic food webs.

Overall, we believe that the two cases studies presented here shows our approach to have considerable potential that will help improve trophic flow assessments and eventually make sound predictions about contaminant transfer. For this to be fully realized, however, this approach for mapping available measurements onto Ecopath-generated food webs should be generalized to any type of contaminant, and in a form so as to require fewer, or better *ad hoc* decisions. [The decision to use scaling factors and fixed relationships between the elements of the predation mortality matrix was such a decision]. We think this can be achieved straightforwardly, as an add-on to the EcoRanger module of Ecopath. This module already includes a Monte-Carlo resampling routine (Christensen and Pauly 1995) which could be used as an ‘importance-sampling’ routine to combine prior distributions from Ecopath, with fits of the observed to the predicted pollutant series providing a likelihood function. This would allow generating posterior distributions of key parameters, thus allowing their interpretation in a Bayesian context (Walters 1996).

Furthermore, Eq. 5 could easily be modified to include, for example, ‘affinity factors’ in cases where one is dealing with fat-soluble pollutants.

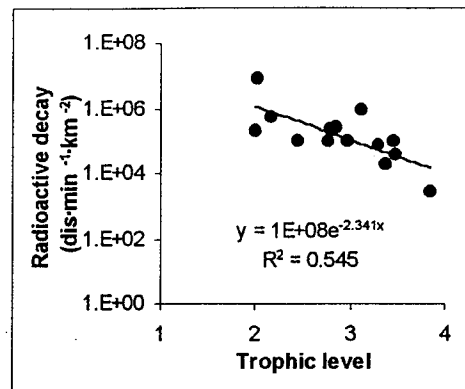


Figure 5. Regression of the maximum level of beta radioactivity (disintegrations·min⁻¹·km⁻²) obtained in the compartments of the Enewetak Atoll model as a function of their trophic level.

A vast database of Ecopath models currently exist, describing freshwater and marine systems in both developed and developing countries (see the Ecopath home page at www.ecopath.org, from which Ecopath may be freely downloaded). These models can be quickly modified to represent different systems with contamination problems, thus providing a basis for ecotoxicological studies such as illustrated here. Alternatively, new models can be constructed for areas and/or system types so far not represented by existing models.

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