ON THE CAUSES OF THE DECREASE IN THE HARBOUR SEAL (PHOCA VITULINA) POPULATION IN THE DUTCH WADDEN SEA

population dynamics and tissue levels of
heavy metal and organochlorine compounds

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P.J.H. Reijnders

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Proefschrift
ter verkrijging van de graad van
doctor in de landbouwwetenschappen,
op gezag van de rector magnificus,
dr. H.C. van der Plas,
hoogleraar in de organische scheikunde,
in het openbaar te verdedigen
op Woensdag 5 maart 1980
des namiddags om vier uur in de aula
van de Landbouwhogeschool te Wageningen.

" Twenty centuries of progress have brought the average citizen a vote, a national anthem, a car, a bank account and a high opinion of himself, but not the capacity to live in high density without befouling and denuding his environment, nor a conviction that such capacity, rather than such density, is the true test of whether he is civilized. "

Aldo Leopold, 1932.

NNO8201, 795

STELLINGEN

I

De exclusieve rol die Helle et al. (1976), toebedelen aan PCB's als oorzaak van pathologische veranderingen in uteri van zeehonden uit de Botnische Golf volgt niet uit de resultaten van hun analyses naar gehaltes aan bepaalde chemische contaminanten noch uit hun experimenten.

Helle, E., M. Olsson & S. Jensen, 1976. - Ambio 5(5): 261-263.

II

Ten onrechte veronderstellen o.a. Addison et al. (1973) en Holden (1978) dat vetweefsels een inert depôt vormen en dat de daaraan gebonden chloorkoolwaterstoffen buiten periodes van zwangerschap en hongeren geen intoxicatie zullen veroorzaken.

Addison, R.F., S.R. Kerr & J. Dale & D.E. Sergeant, 1973. - J. Fish. Res. Bd Can. 30(5): 595-600.

Holden, A.V., 1978. - Mammal Rev. 8 (1+2): 53-66.

III

Op basis van te verwachten milieueffecten is in Zweden en sommige staten van Amerika de productie en de toepassing van PCB's verboden. Nu er bovendien sterke aanwijzingen zijn dat deze stoffen een rol spelen in de reproductie verstoring bij zeehonden in de Waddenzee, zou men oververwijld moeten besluiten de PCB's in Europa compleet in de ban te doen.

IV

Het aantal bruinvissen in de kustwateren van noordwest Europa is vooral na 1960 drastisch verminderd (Verwey, 1975). Op grond van gevonden PCB gehaltes in weefsels van dieren uit dit gebied (Koeman et al., 1972) en de resultaten uit dit proefschrift rechtvaardigen de conclusie dat PCB's mede verantwoordelijk zijn voor die achteruitgang.

Koeman, J.H., W.H.M. Peeters, C.J. Smit, P.S. Tjioe & J.J.M. de Goey, 1973. - Nature 245: 385-386.

Verwey, J., 1975. - Publ. Versl. Ned. Inst. Onderzoek der Zee (Texel) 17, 1-98, 99-153 (internal report).

V

Dat de zeehond nog niet verdwenen is uit de nederlandse Waddenzee, is grotendeels te danken aan immigratie van dieren uit het duitse waddengebied.

VI

Het nagenoeg verdwijnen van de zeehond uit het deltagebied heeft niet kunnen voorkomen dat deze soort in het nederlandse waddengebied hetzelfde lot dreigt te ondergaan.

VII

Het optisch model dat Rivamonte (1976) van het dolfijnoog berekende wordt in kringen van deskundigen veel te weinig gewaardeerd.

Rivamonte, L.A., 1976. - Neth. J. Sea Res. 10(4): 491-498.

VIII

Voor commerciële kweek van zeevis in Nederland, biedt "ranging" van zalmachtigen in de relatief schone Oosterschelde perspectieven.

Commissie Aquacultures van de Stichting van de Nederlandse Visserij, 1979. Mogelijkheden van de commerciële kweek van zeevis in Nederland.

IX

Omdat door de aflopende conjunctuur de ruimte voor subsidiëring van structurele landbouwoverschotten steeds minder wordt, verdient het aanbeveling het europese houttekort te beschouwen als een grondstoffenprobleem en door emancipatie van de houtteelt binnen de landbouw die visie te effectueren.

X

Gezien het feit dat de kwaliteit van het voedsel voor het edelhert in de nederlandse situatie niet beperkend is voor het terreingebruik, mits deze het leefgebied beter kunnen en durven te benutten, zou de archipel van rasters moeten worden vervangen door één groot eiland.

Van de Veen, H.E., 1979. Food selection and habitat use in the red deer (Cervus elaphus L.). Zoology Dept.Groningen University (thesis).

XI

Het is schrijnend te constateren dat de intensiteit van het zeezoogdierenonderzoek in Nederland parallel loopt met de revenuen die er van de betreffende soort te verwachten zijn.

XII

Er van uitgaande dat het Ministerie van Defensie er is om ons tegen eventuele vijanden te beschermen, lijkt het, gezien de talloze gevaren die de samenleving bedreigen door de voortdurende verslechtering van het milieu, op zijn minst redelijk de begrotingen voor Defensie en het Ministerie van Volksgezondheid en Milieuhygiëne gelijk te trekken.

P.J.H.Reijnders. On the causes of the decrease in the harbour seal (Phoca vitulina) population in the Dutch Wadden Sea. Wageningen, 5 maart 1980.

aan majella, peerke en stijn

Omslag foto : Th.Mutter,Rijks Instituut voor Natuurbeheer.

Het in dit proefschrift neergelegde onderzoek werd mogelijk gemaakt door financiële steun van:

Wereld Natuurfonds Nederland,de Landelijke Vereniging tot Behoud van de Waddenzee,de Stichting Zeehondencrêche Pieterburen,de Nederlandse Organisatie voor Zuiver Wetenschappelijk Onderzoek en de Europese Economische Gemeenschap:Environmental Research Programme,contract nr. 198-77-1- ENV NL.

VOORWOORD

Bij het gereedkomen van dit proefschrift hebben velen een bijdrage geleverd en daarvoor wil ik mijn dank betuigen.

Het zijn allereerst de boeiende cursussen Jacht en Wildbeheer van wijlen J.A. Eygenraam waardoor mijn belangstelling voor een verdere studie in het Natuurbeheer, speciaal het Faunabeheer, sterk werd gestimuleerd. Het is daarna vooral dank zij U, prof. em. dr. M.F. Mörzer Bruyns, dat al in een vroeg stadium van de studie aan de Landbouwhogeschool vorm is gegeven aan mijn ontwikkeling op dat gebied, wat uiteindelijk heeft geleid tot dit proefschrift. Ik ben U dankbaar voor het vertrouwen en de vrijheid die me zijn gegund om dit onderzoek aan zeehonden te kunnen verrichten. Uw persoonlijke wijze van voortdurende belangstelling en discussie tijdens de diverse fases in het onderzoek hebben me steeds zeer aangesproken.

Veel dank ben ik zeker verschuldigd aan dr. J.L. van Haaften. Na als student kennis gemaakt te hebben met het zeehondenonderzoek heb ik, mede op zijn initiatief, dat kunnen voortzetten in het huidige onderzoek. Door zijn jarenlange veldwaarnemingen en ervaring heeft hij me snel en op een enthousiaste manier ingevoerd in de problematiek van het onderzoek naar de achteruitgang der zeehondenstand.

Zeker wil ik ook G.J. de Haan, directeur van het Texels Museum bedanken. In de beginfase van het onderzoek heeft hij de faciliteiten geschapen waardoor er toch ondanks de wankele financiële basis van het projekt gewerkt kon worden. Het contact met hem en de vanzelfsprekendheid waarbij de zeehonden in de bassins van het museum konden worden ingezet ten dienste van het onderzoek stel ik zeer op prijs.

Velen hebben de diverse manuscripten in meerdere stadia kritisch doorgelezen. Mijn oprechte dank hiervoor aan drs.J.W. de Blok, prof.dr.H.Klomp, prof.dr.J.H. Koeman, ir.J.K.Quirijns, drs.C.J.Smit, dr.W.J.Wolff en dr.J.J.Zijlstra.

Vooral prof. dr. J.H. Koeman wil ik in het bijzonder dank zeggen. Allereerst voor de gastvrijheid die me is verleend door de vakgroep Toxicologie van de Landbouw-

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De deskundige inbreng van Aad Droog en bovenal onze zeer vriendschappelijke relatie maken de vliegtellingen letterlijk en figuurlijk voor mij tot hoogtepunten in het onderzoek.

Sinds de komst op Texel van de afdeling estuariene ecologie van het Rijks Instituut voor Natuurbeheer lijkt het pioniersstadium in het mariene onderzoek van het R.I.N. voorbij. Wim Wolff ben ik vooral dank verschuldigd voor zijn positieve opstelling en opbouwende kritiek tijdens perioden waarin er voor het onderzoek weinig perspectieven leken te zijn. Daarnaast ben ik zeer erkentelijk voor de manier waarop ik binnen zijn afdeling kon integreren, hetgeen erg vruchtbaar heeft gewerkt op de realisatie van dit proefschrift. Dank ook voor de geweldige hulp van Michel Binsbergen, die zonder zijn humeur te verliezen, koortsachtig heeft gewerkt om het gehele proefschrift persklaar te maken, en ook voor het vele typewerk door Riny Wielinga en het tekenwerk voor het 3e artikel door Hans Schoonheden. Hun medewerking heeft ertoe bijgedragen dat het publiceren van dit proefschrift binnen de tijdsplanning kon blijven.

CURRICULUM VITAE

Peter Johannes Hubertus Reijnders werd op 29 mei 1944 geboren te Siebengewald (Limburg) en behaalde in 1963 het HBS-B diploma aan het St. Chrysostomus College in Boxmeer.

Na het vervullen van de dienstplicht als adelborst op het Koninklijk Instituut voor de Marine studeerde hij van 1966 tot 1973 aan de Landbouwhogeschool te Wageningen. Tijdens de doctoraal fase van de studierichting Bosbouw koos hij de vakken Natuurbeheer (w.o. wildbiologie), houtteelt en Landinrichting.

Vanaf zijn afstuderen is hij werkzaam als tijdelijk medewerker bij het Rijks Instituut voor Natuurbeheer (R.I.N.) aan het onderzoek naar de oorzaken van de achteruitgang der zeehondenstand in de Waddenzee. Daartoe is hij gedetacheerd op Texel. Aanvankelijk bij het Texels Museum, daarna het Nederlands Instituut voor Onderzoek der Zee en momenteel bij de afdeling estuariene ecologie van het R.I.N., vestiging Texel.

INHOUD

INLEIDING EN SAMENVATTING

Omdat er aanwijzingen waren dat het aantal zeehonden in de nederlandse kustwateren achteruitging, werden de zeehonden in 1954 onder de werking van de jachtwet geplaatst; opdat de jacht zou kunnen worden gereglementeerd. Desondanks ging het aantal door overbejaging verder achteruit en in 1962 werd daarom de jacht geheel verboden. Dank zij die bescherming nam het aantal met 5-10% per jaar toe maar in 1965 zette een nieuwe daling in die duurde tot omstreeks 1974. Sindsdien is de populatie op een laag niveau van ca. 450 dieren gebleven, onder andere door immigratie van zeehonden uit andere delen van de Waddenzee. Die voortdurende achteruitgang was in 1974 de aanleiding om het onderzoek naar de oorzaken ervan sterk te intensiveren.

Dit onderzoek werd ingedeeld in drie fasen: I. Vaststellen van de grootte en de samenstelling van de populatie. II. bepalen welke populatie-dynamische parameter(s) afwijken van die welke bij vergelijkbare stabiele populaties van dezelfde soort worden aangetroffen III. onderzoeken wat de invloed daarbij is van milieufactoren, zoals voedsel, parasieten, verontreiniging en verstoring.

Door geregelde tellingen vanuit vliegtuigen in de periode 1974-1978 werden de totale aantallen bepaald en door het meten der spoorbreedtes van de vluchtsporen werd informatie verzameld over de samenstelling van de populatie. Deze informatie werd gebruikt in een rekenmodel en op die manier werden een gemiddeld productiepercentage en de jeugdsterfte berekend. Door deze resultaten te vergelijken met gelijksoortige resultaten van onderzoek aan een stabiele populatie in Schleeswijk Holstein bleek dat 1) het reproductiepercentage bij zeehonden in de nederlandse Waddenzee lager is dan bij dieren uit het waddengebied van Schleeswijk Holstein; 2) dat de jeugdsterfte in de eerste drie levensmaanden in beide gebieden ongeveer gelijk was; 3) dat immigratie van subadulte dieren uit het duitse waddengebied naar Nederland optrad.

Om de bijdrage van de waterverontreiniging tot de verminderde reproductie in de nederlandse populatie te onderzoeken werden organen en weefsel van dood gevonden zeehonden afkomstig uit Duitsland, Denemarken en Nederland verzameld. Deze werden geanalyseerd op gehaltes aan kwik, seleen, broom en diverse chloorkoolwaterstoffen.

Uit statistische toetsen op mogelijke verschillen in de diverse residu-gehaltes in weefsels van dieren afkomstig uit Schleeswijk Holstein en Denemarken enerzijds en dieren afkomstig uit Nederland anderzijds bleek dat naast dieldrin en αHCH (gehaltes lager dan 0,5 ppm op vet basis) vooral voor PCB's significant hogere gehaltes (gemiddeld 700 ppm op vet basis) werden aangetroffen in weefsel van volwassen nederlandse zeehonden. Dit resultaat, gecombineerd met de informatie die voorhanden is ten aanzien van de effecten van PCB's op de reproduktie bij zoogdieren, ondersteunen duidelijk de hypothese dat PCB's verantwoordelijk zijn voor de verminderde reproduktie bij zeehonden in de nederlandse Waddenzee. Omdat deze hypothese echter gedeeltelijk steunt op een aantal niet bewezen aannames, is voortzetting van het onderzoek onder gecontroleerde omstandigheden nodig om tot resultaten met bewijskracht te komen.

Indien PCB's verantwoordelijk zijn voor de achteruitgang van de zeehondenpopulatie in de nederlandse Waddenzee dan dienen er onverwijld maatregelen te worden getroffen door tenminste alle landen die aan de Noordzee grenzen. Dit is omdat de achteruitgang van de nederlandse zeehondenpopulatie niet slechts een locaal probleem zou kunnen zijn, doch daarnaast een sterke aanwijzing dat PCB's een bedreiging vormen voor alle zeezoogdieren en zelfs voor de menselijke volksgezondheid. De productie en het gebruik van PCB's moeten daarom sterk worden beperkt en de gebruikte PCB's worden verzameld en vernietigd. Maar zelfs indien onmiddellijk tot een compleet verbod van PCB's wordt besloten dan nog zullen de effekten ervan door hun persistente eigenschappen - jarenlang merkbaar zijn. Op grond daarvan mag worden aangenomen dat de zeehondenpopulatie in de nederlandse Waddenzee in de komende jaren een moeilijke periode zal doormaken. De reproductie in de huidige populatie is te laag en het voortbestaan ervan hangt al jaren lang voor een deel af van immigratie uit andere gebieden. Om de Waddenzeepopulatie door deze moeilijke periode heen te loodsen moeten alle mogelijke maatregelen worden genomen die bevorderlijk zijn voor het voortbestaan ervan.

Aangezien de zeehonden in de internationale Waddenzee één populatie vormen is een maximum resultaat te verwachten, indien maatregelen genomen worden voor alle zeehonden in dit gebied en daarom is internationale samenwerking een noodzaak.

Daarbij worden de volgende maatregelen voorgesteld:

- handhaving van het bestaande jachtverbod met als doel een maximaal aantal dieren aan het reproduktieproces te laten deelnemen.
- het instellen van rustgebieden en bovendien het steunen van opvangstations voor jonge zeehonden teneinde de overlevingskansen voor de jonge dieren zo groot mogelijk te maken.

Benadrukt moet echter worden dat deze maatregelen geen effect zullen hebben, als niet tegelijkertijd de hoofdoorzaak van de achteruitgang wordt gevonden en aangepakt.

SUMMARY

The harbour seal population in the Dutch Wadden Sea decreased significantly during the last decades. Calculations based on bounty data revealed that the population decreased from about 2700 in 1950 to about 900 in 1959. Annual aerial surveys - which were carried out since that time - showed a slight increase due to the stop of hunting. However, after 1964 a new decline occurred and since 1974 the population stays at a level of about 450 specimens.

During 1974 to 1978 the population dynamics of the population have been studied. Frequent aerial surveys provided data on the size of the population. These data have been used in a simulation model and the birth rate and the initial juvenile mortality were calculated. During boat trips an indication of the age composition was obtained by measuring track widths. By comparing these results with similar results from a stable population in Schleswig Holstein it appeared 1) that juvenile mortality in the first weeks in the Dutch population is higher than that in Schleswig Holstein but the overall mortality of pups in both areas in their first three months of life is of the same order; 2) that pup production in the Dutch population is low compared to the population in Schleswig Holstein; 3) that apparently immigration from elsewhere occurs.

To investigate the contribution of environmental pollution to the decline of the Dutch seal population tissues of dead, stranded animals originating from Schleswig Holstein, Denmark and The Netherlands were collected and analysed for PCB's, o,p'-DDT, p,p'-(DDT, DDE, TDE), dieldrin, aldrin, endrin, endosulfan, α,β,γ,-HCH, HCB, QCB (pentachlorobenzene), HEPO, total mercury, methylmercury, selenium and bromium.

Tests on possible differences in residue levels between Schleswig Holstein plus Denmark on one hand and The Netherlands on the other revealed that especially PCB levels were significantly higher in Dutch adult seals. Considering epidemiological and experimental data on the effects of PCB's on mammalian reproduction, strong support is obtained for the hypothesis that PCB's are responsible for the decreased reproduction

in seals from the Dutch Wadden Sea.

If PCB's are responsible for the decrease of the Dutch seal population, measures of all North Sea countries are urgently required because these PCB's may not only be a hazard to seals but also to other animals and even man. Production and use of PCB's should be limited and used PCB's should be collected and destroyed.

However, even if an immediate ban on PCB's would be effected, the effects of PCB's - because of their persistent character - are likely to last many years. For that reason it may be assumed that the seal population in the western Wadden Sea still faces a difficult period in the years to come. Presently this population shows an insufficient reproductive rate and is partly dependent on immigration from other populations. In order to lead this population through this difficult period all possible measures enhancing its continued existence should be taken. Since the seals in the Wadden Sea form one population maximum result may be expected from measures applying to all seals in the Wadden Sea. Thus international cooperation is necessary.

The following measures are proposed:

- continued interdiction of hunting in order to obtain a maximum number of animals taking part in reproduction.
- establishment of seal reserves and, additionally supporting seal nursery stations in order to guarantee a maximum reproductive output.

However, it has to be stressed that these measures are not effective if at the same time the main cause of the decrease is not found and under control.

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THE HARBOUR SEAL (PHOCA VITULINA) POPULATION IN THE DUTCH WADDEN SEA: SIZE AND COMPOSITION

by

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(Netherlands Institute for Sea Research, Texel, The Netherlands)

CONTENTS

I. INTRODUCTION

As there were signs that the seal stock in the Dutch coastal waters declined, seals were partially protected by law in 1954 (*cf.* VAN BEMMEL, 1956a, 1956b).

Nevertheless the decrease held on, and hunting was completely banned in 1962. Probably due to total protection there was a slight increase, but since 1965 a new decline set in, which lasts up to the present. This continuing decrease was the motive to start a study of its causes.

The size of a population is determined by reproduction, mortality and migration. Each of these parameters may vary with age and sex. As seals in the Wadden Sea do no longer have any predators since man decided to abstain from hunting, the population size is determined by such factors as food quality and quantity, parasites, diseases, pollution and rate of disturbance.

This study discusses the size and composition of the seal population. Further study is in progress to obtain information about reproduction, mortality and migration.

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The author also acknowledges Mr G. J. de Haan, director of the Texel Museum and Dr D. Dekker, Artis Zoo, Amsterdam, for the opportunity given to experiment with captive seals, the aviator Mr A. Droog, and the crews of M.S. "Eider" of the Institute and M.S. "R.P. 19" of the water police, district Delfzijl.

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II. TRACK WIDTH MEASUREMENTS AS A METHOD FOR AGE DETERMINATION

In comparable studies of Bigg (1969), Smith (1973) and Boulva (1974) information on the size-, age-, and sex-composition of seals was obtained from random samples of animals shot.

Because of the low numbers of seals in the Dutch Wadden Sea this method could not be used, instead another more indirect method was applied. Already van Haaften (1959) in his censuses, used the tracks the seals leave on the flats when they move from their resting places to the water edge. These tracks may be a mean to determine their size. The seal uses the nails of its fore-flippers to crawl over the sand. The width of the track now may indicate the size of the seal and, by a size-age relationship, also roughly the age class (juvenile, subadult and adult animals).

To test the possible use of the track for age group determination the smallest distance between the imprints of left and right fore-flipper nails was measured and is referred to as "track width" (Fig. 1). Seals kept in captivity at the Texel Museum and the Artis Zoo, Amsterdam, provided basic measurements which were carried out in mid-July. The animals were sent one by one through a rather narrow lock with a moist sandy bottom. The track each animal left behind was measured and compared with its age. In addition track widths of seals in representative areas of the Wadden Sea were measured during 4 boat trips, respectively in June 1974 (whelping time), in late August 1974 (mating time), at the end of May 1975 (pre-whelping time) and in

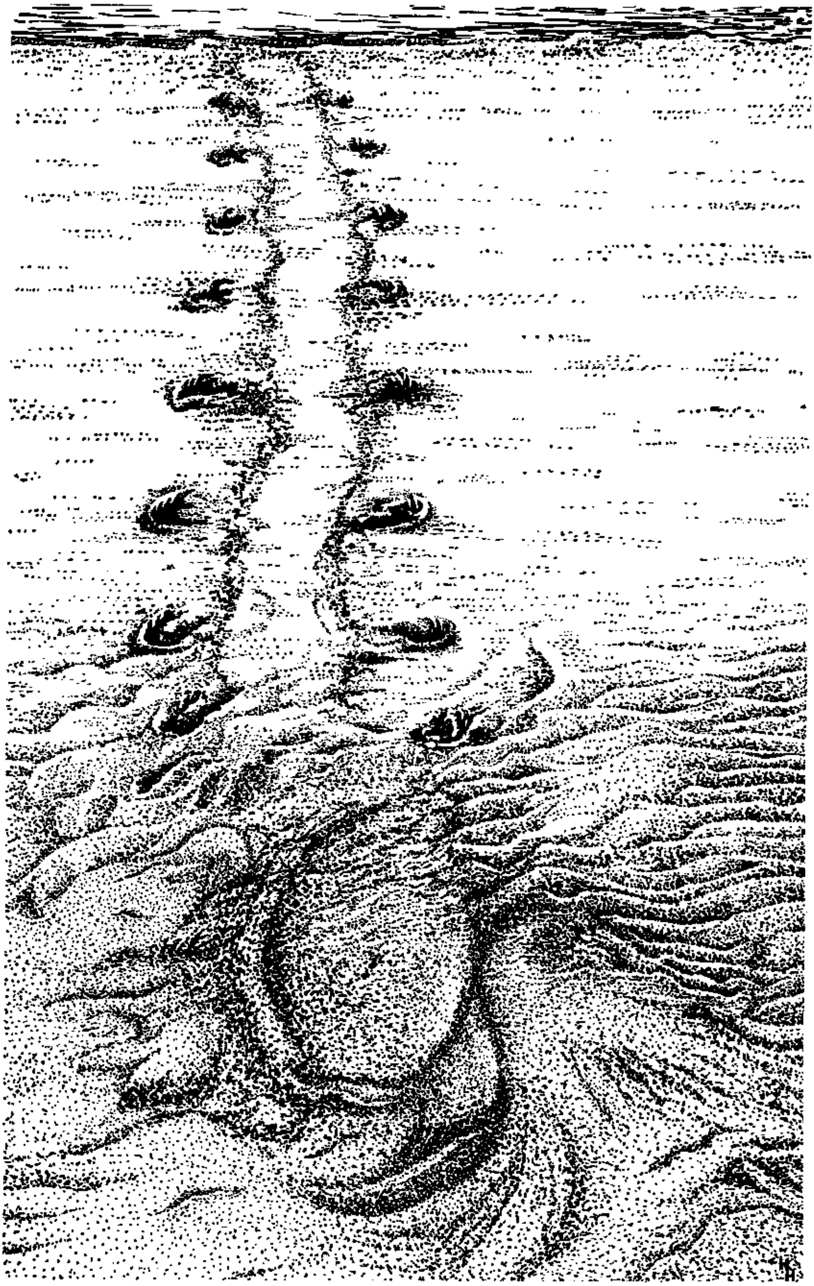

Fig. 1. Resting place and seal track on sand bank.

early August 1975 (post-whelping time).

The relation between age and track width for captive seals is shown in Fig. 2 in which the limits between juveniles, subadults and adults are

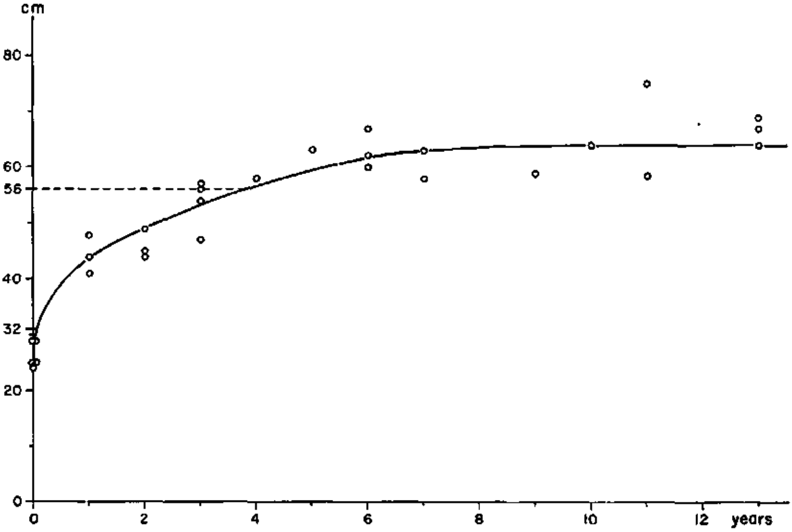

Fig. 2. Relationship between track width (cm) and age (years) of harbour seals based on captive animals measured in July; eye-fitted curve.

indicated. These limits are based on a trinomial fit of the data (REIJNDERS, 1975).

To make sure that the captive seals are representative for seals living in the Wadden Sea—*i.e.* have the same skeletal growth rate—the results were compared with the field observations made in the same period (June-July) as the tracks of the captive seals were measured. When a mother and young were observed, and also single juveniles, the corresponding track widths were measured (Table I).

The mean value for mature captive seals is somewhat higher than that for wild seals. However, in the captive group the mature seals were mainly males (10 out of 15) and full-grown males are about 3% longer (WIPPER, 1974). Thus the mean value for the captive group has to be regarded somewhat too high, the mean value for the wild seals (just females) somewhat too low.

In the captive group, due to low numbers of individuals, no attention could be given to the possibility whether the sexes showed different track widths at certain ages. BISHOP (1968), BIGG (1969) and FISHER (1952) stated that there appears to be no significant difference in length and weight between sexes till the age of 6 years. For seals in the

TABLE I

Comparison between track widths measured for captive harbour seals and for wild ones at ages of less than one month (pups) and of 4 years and older (adults).

	Group	*Mean (cm)*	*SD (cm)*	*Numbers*
pups	captive	26.4	2.4	5
	wild	26.3	3.0	111
adults	captive	63.3	4.7	15
	wild	61.7	4.9	62

Texel Museum a linear regression was established between girth and track width ($r = 0.98$; $p < 0.001$; $n = 22$). FISHER (1952: 6) also reports data on girth behind fore-flippers and age, he found no difference between sexes. Thus FISHER's data offer circumstantial evidence that the sexes, till the age of 6, do not vary in track width.

Fig. 2 shows that 3 age groups have clearly distinguishable ranges in track width; shortly after whelping time juveniles (0.01 year) measured 26 to 32 cm, subadults (1, 2 and 3 year) 36 to 55 cm, adults (4 year and older) above 56 cm.

III. POPULATION SIZE AND COMPOSITION

1. METHODS

The harbour seals in the Wadden Sea do not constitute one continuous population, but are distributed in several small groups, apparently isolated from one another, at least during summer time when they stay in the Wadden Sea. Fig. 3 shows the subareas between which little or no exchange takes place. It was found that the same animals returned every low tide to one specific sand bank. It could not be determined whether the same animals returned to that specific sand bank the next year after having left the Wadden Sea during winter time (REIJNDERS, 1973).

Estimates of the population size were made by regular aerial surveys from May till September 1974 and 1975 during low tide, when the seals are resting on the edges of the tidal flats. All counts were made at heights of about 150 m and each survey lasted about 2½ hours.

Size and composition of the population were established during the 4 boat trips made in 1974 and 1975, both years about whelping time (June) and about mating time (August). Juveniles, subadults and adults were distinguished by measuring the track widths of the animals.

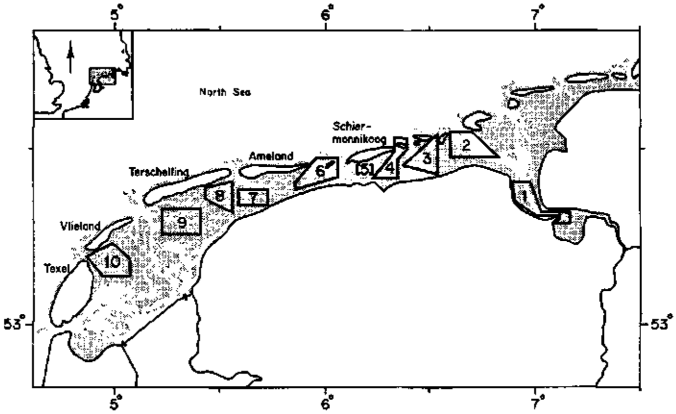

Fig. 3. Studied area of the Dutch Wadden Sea with localities occupied by harbour seal.

2. RESULTS

a. POPULATION SIZE

Since 1959 estimates of the size of the seal population are available from regular aerial surveys, they are given in Fig. 4. Such aerial censuses are thought to be fairly reliable because in restricted areas counts by plane gave nearly the same results as counts by boat (Table II). Moreover, the whole Dutch Wadden Sea can be covered during one low tide, thus avoiding double counts.

b. POPULATION COMPOSITION

The track width measurements provide the information for an estimation of the age group composition of the total wild population established in the aerial surveys. The age group distribution was estimated per area (Table II) which needed some calculation.

Not all seals in each area left a well defined track behind because the substrate was unsuitable (*e.g.* too soft) or because seals moved by lateral undulation rather than by the usual dorso-ventral way or because seals did not haul out at the moment their hauling area was visited. The assumption is made (based on Table II, post-whelping time 1975) that the animals measured represent a fair sample of the animals present in the area. In some areas no measurements were made, in that cases age group splits were made on basis of the total of the areas where measurements were carried out. A better estimate was possible during whelping time (1974) and the post-whelping time

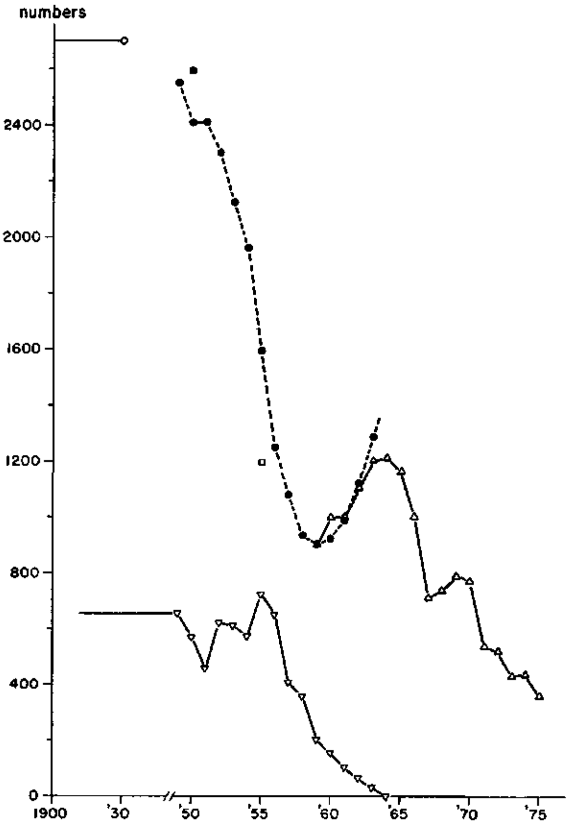

Fig. 4. Population sizes of harbour seals present in the Dutch Wadden Sea in September, according to the aerial surveys (△), estimates of HAVINGA (1933) (○), MOHR (1952) (■) and VAN BEMMEL (1956b) (□), and to own calculations backward and forward from 1959 (●). Also indicated are numbers killed by hunting according to MEYER (1964) (▽).

(1975), as it could be demonstrated (REIJNDERS, 1973; VAN HAAFTEN, personal communication) that, due to suckling and their apparently higher need for rest, all pups are detected during the aerial counts in these periods. Therefore, the ratio achieved between subadults and adults for the areas with measurements was applied to the numbers counted by the aerial survey at the nearest date in the other areas, after subtraction of the known number of juveniles.

The data treated in this way, resulted in estimates for the whole Dutch Wadden Sea population as shown in Table II (last column).

TABLE II

Calculation of the population composition of seals in the Dutch Wadden Sea based on track width measurements in part of the subareas, and on aerial census. In 1974 for whelping time (end of June) and mating time (early September); in 1975 for pre-whelping time (end of May) and post-whelping time (early August). Data between brackets are estimates obtained from ratios in the subareas sampled (for location of subareas see Fig. 3). Resulting total numbers of juveniles, subadults and adults are given with 95% confidence limits.

Group	*Subarea*										
	I	*II*	*III*	*IV*	*V*	*VI*	*VII*	*VIII*	*IX*	*X*	*Total*
1974, whelping time											
Juveniles	0	12	9	2	1	0	0	0	0	2	26 ± 13
Subadults	(7)	31	24	27	12	21	18	13	(10)	17	180 ± 24
Adults	(5)	20	26	21	9	4	0	0	(7)	22	114 ± 23
Measured	0	34	34	32	15	18	7	8	0	22	170
Boat census		34	47	43	15	18	10	8		22	197
Aerial census	12	63	59	50	22	25	18	13	17	41	320
1974, mating time											
Juveniles	(4)	(7)	5	5	4	6	(3)	(1)	(2)	(3)	40 ± 27
Subadults	(24)	(40)	38	42	9	25	(16)	(4)	(13)	(22)	233 ± 49
Adults	(24)	(40)	54	21	20	13	(15)	(3)	(12)	(21)	223 ± 49
Measured	0	0	38	29	18	14	0	0	0	0	99
Boat census			45	45	22	15					127
Aerial census	52	87	97	68	33	44	34	8	27	46	496
1975, pre-whelping time											
Juveniles	0	0	0	0	0	0	0	0	0	0	0
Subadults	(3)	26	16	5	9	19	(2)	(2)	4	7	93 ± 18
Adults	(4)	14	40	15	3	5	(3)	(3)	5	14	106 ± 18
Measured	0	25	35	19	4	18	0	0	5	14	120
Boat census		40	56	20	7	24			5	15	167
Aerial census	7	40	56	20	12	24	5	5	9	21	199
1975, post-whelping time											
Juveniles	2	12	22	9	2	2	4	4	6	1	64 ± 17
Subadults	20	30	34	12	15	28	10	14	16	9	188 ± 23
Adults	11	39	62	33	11	5	22	19	16	19	237 ± 23
Measured	32	69	114	39	23	34	30	34	34	20	429
Boat census	33	81	117	47	23	34	32	36	40	20	463
Aerial census	33	81	118	54	28	35	36	37	38	29	489

About the sex ratio no data could be obtained with the present method but juvenile harbour seals caught in the Dutch Wadden Sea in the period 1965 to 1975 had a sex ratio of 1, 100 females against 102 males (Table III).

For subadults, data collected from stranded seals by VAN HAAFTEN (personal communication), a sex ratio of nearly 1 was found (25 females out of 47 animals).

TABLE III

Sex ratio (females : males) of harbour seals estimated by various authors (numbers sampled between parentheses).

Author	*Area*	*Sex ratio*			
		Juv.	*Subad.*		*Adult*
Present study	Dutch Wadden Sea	1(202)	1.1(47)		
WIPPER (1974)	West German Wadden Sea	1.1(180)			0.9(42)
HAVINGA (1933)	Dutch Coast	1(37)		1(38)	
BONNER & VAUGHAN (1972)	England	1(308)			
BOULVA (1971,1974)	Eastern Canada	0.9(188)		1.1(246)	
BIGG 1969	British Columbia	1(64)		1.1(245)	
BISHOP (1968)	Alaska	1(197)		0.9(50)	

IV. DISCUSSION

Before 1959, the year VAN HAAFTEN (1974) started his aerial censuses (Fig. 4), data on the population size of *Phoca vitulina* in the Dutch Wadden Sea are sarce. BROUWER (1928) estimated the total numbers at 1500, HAVINGA (1933) at 2700 and VAN BEMMEL (1956b) at 1200 specimens. None of these authors explains exactly how and when the estimates were obtained. From data available on annual numbers killed (MEYER, 1964), and by using rates known from stable populations of harbour seals, it is tried to calculate population sizes earlier than 1959.

Since the population size varies with the season, and the counts of VAN HAAFTEN (1974) were done in September, calculations for earlier years have been made for the same time of the year.

Assuming pups to constitute, on an average, about 30% of the September population (VAN BEMMEL, 1956b; VAN HAAFTEN, personal communication) and an annual mortality of 17% (BOULVA, 1974), the annual hunting data (K_{t+1}) can be used to obtain the September size of the population (N_t) out of the September size of the population in the next year (N_{t+1}), following the equation:

$$N_{t+1} = \frac{100}{70}(1 - 0.17)\, N_t - K_{t+1}$$

Starting with the value of 900 seals for 1959 of VAN HAAFTEN (1974) and using the annual kills according to MEYER (1964) (Fig. 4), the September population sizes (N_t) are calculated backwards from year to year (Fig. 4). In this way a population size of 2550 specimens is obtained for 1949.

HAVINGA (1933) estimated the harbour seal population in the Dutch Wadden Sea from 1900 till 1930 on about 2700 specimens. For his estimate he assumed the percentage of subadults to be 15% of the total population which is certainly too low as BOULVA (1974), BIGG (1969) and WIPPER (1974) arrive at percentages of 30%. At the other side he added a certain number of seals to his estimate (based on bounty data) without further explanation.

Also on bounty data MOHR (1952) estimated the population size in the Dutch Wadden Sea on about 2600 specimens for 1950.

Though based on several implicit assumptions, the calculation over the period from 1949 to 1959 seems useful to bridge the gap between the time of the stable population and the period of registered decline. It may imply that between 1900 and 1950 the population in the Dutch Wadden Sea fluctuated around 2600 specimens and that from that time on a steady decrease set in. The start of this decrease may be caused by the fact that from 1949 bounties were not paid any longer. The result will have been that nearly only pups were killed for the fur industry. The interest for adults was diminished not only because the payment of bounty was stopped but also because the demand for train-oil had ended (MEYER, 1964; 'T HART, personal communication).

It should be mentioned that in the foregoing calculation no attention was paid to migration. A considerable migration to or from especially other parts of the Wadden Sea (Denmark, Germany) may have had its influence on the population under consideration. Also in these areas bounty killing occurred.

From 1959 on the aerial surveys provided more reliable data (Fig. 4), though they are quite probably underestimates as some animals did not haul out. The seals still decreased in numbers, with only slight peaks in 1963–1964 and 1969–1970. The first increase, setting in with the count of 1959, probably resulted from the fast decrease in kills up to the year 1962 when hunting was forbidden and the seals got totally protected; compare the continuation after 1959 of the calculated population size estimate (Fig. 4). This extra quantity of pups not shot in this way may not only explain the first increase in numbers but perhaps also the slight second increase beginning in 1968, as these pups started a new wave of births when they became mature 4 to 5 years later.

When the calculated curve for the population, with its decrease before 1959 and its initial increase after that year, is of real importance this would also imply (Fig. 4) that after 1963 another factor than hunting has to be responsible for the further decrease in numbers.

The frequency distribution (Fig.5) of the track widths measured is fairly different for the 4 measuring series. The first boat trip was made during whelping time, whilst the third trip was made before the whelping began. The second trip was carried out late in the mating time, so some seals might already had left the Wadden Sea (VAN HAAFTEN, 1974), whilst the fourth trip was made just when whelping

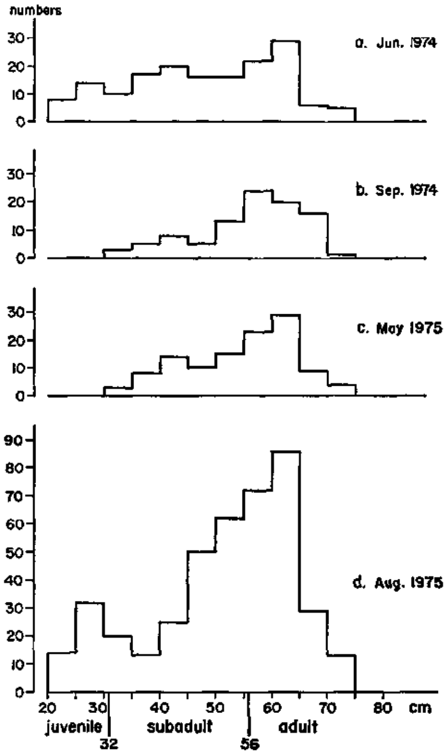

Fig. 5. Track width frequency distributions of harbour seals in the Dutch Wadden Sea: a. at whelping time (end June 1974); b. at mating time (September 1974); c. at pre-whelping time (end May 1975); d. at post-whelping time (early August 1975).

was finished and the mating time started. The estimates during the post-whelping time of 1975 are obviously the best ones as 88% of the population present was measured. The distribution of these 1975 measurements is clearly bimodal. One peak, at 25 to 30 cm, reflects the pronounced age group of the newborn pups. The second peak, at 55 to 65 cm, demonstrates the decreasing length growth from their fourth year forward (Fig. 2) of the adults.

Table II gives more details on the population composition of the seals. Due to the high variance in the age composition between subareas it was necessary to sample as many subareas as possible to obtain by extrapolation a more or less reliable picture of the total composition during the 4 measuring series. The results of 1974 should be considered with caution.

The results indicate that all three groups distinguished increased between June and August. The largest increase is found for the adults, in both years about 120. This may signify migration of males into the Wadden Sea for the mating season (VAN BEMMEL, 1956a; VAN HAAFTEN, 1974). Also the increase of subadults in 1975 is remarkable. Information about migration is not available but the general impression is that first the pregnant females migrate to the Wadden Sea and soon afterwards the subadults and other adults.

Data on population composition from other Wadden Sea areas are not available, only maximum numbers of pups counted and maximum population sizes. WIPPER (1974) found for the western German Wadden Sea the pups to constitute about 27% of the total population. In the Dutch Wadden Sea only 11% pups were found in 1974, and 16% in 1975. As the East-Frisian population is considered to be fairly stable, it is clear that in the Dutch Wadden Sea population either reproduction is abnormally low or juvenile mortality is abnormally high, or both. This low recruitment in the Dutch Wadden Sea will be the main cause for the decline of the population.

Table III shows that the sex ratio for juvenile and older harbour seals in the Dutch Wadden Sea agrees well with that of other regions.

V. SUMMARY

The numbers of harbour seals (*Phoca vitulina*) in the Dutch Wadden Sea decreased from about 2700 in 1950 to 350 in 1975. The age groups juveniles, subadults and adults in the population were estimated by measuring the width of the tracks the animals leave when they move to the edge of the tidal flat. It appeared that the recruitment of the population was much lower than in the adjoining German part of the Wadden Sea. This will be the main reason for the drastic decrease in numbers of the population.

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RECRUITMENT IN THE HARBOUR SEAL (PHOCA VITULINA) POPULATION IN THE DUTCH WADDEN SEA

by

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CONTENTS

I. INTRODUCTION

The harbour seal population in the Dutch Wadden Sea has decreased significantly in the last decades (REIJNDERS, 1976). Apart from the possibility of an increased migration from the area, this decline might either be induced by a higher mortality or by a reduced recruitment. In this paper recruitment to the seal stock in the Dutch Wadden Sea is studied.

Attention is paid to a possible reduction in the reproductive rate and to an enhanced juvenile mortality, as both factors would lead to a reduced recruitment. Reproductive rate and juvenile mortality are compared to those found in more stable seal populations.

Ideally such a comparison should be made with the use of LESLIE (1945, 1948) matrices or based on life-tables as constructed by DEEVY (1947) and CAUGHLEY (1966), integrating information on age-specific mortality, reproductive rates, mean life expectancy at birth and mean generation time. Unfortunately studies on population parameters in the case of the harbour seals in the Wadden Sea are hampered. Firstly, by the fact that the usual sampling techniques applied in studies on populations of other seals cannot be used. Mortality rates *e.g.*, are usually estimated from animals obtained by shooting (CAUGHLEY, 1967). The harbour seal population in the Dutch Wadden Sea is too small (about 400 animals) to make this procedure practicable.

Secondly, in harbour seals no reliable data on breeding success can be obtained directly (BONNER, VAUGHAN & JOHNSTON, 1973). This contrasts the situation in the grey seal (*Halichoerus grypus*) as their pups remain on land till they are about 5 weeks old. In this species information on reproduction and juvenile mortality can be obtained either by aerial surveys (HEWER, 1964) or by counts on the breeding grounds (SUMMERS, BURTON & ANDERSON, 1975; RADFORD, SUMMERS & YOUNG, 1978). Harbour seal pups (newly born juveniles) in the Wadden Sea behave almost like adult seals, moving in the water during a large part of the day. Only at low tide most animals, juveniles (0 to 1 year old), subadults (1 to 4 year) and adults, alike, are found outside the water on the edges of tidal sand flats. This makes it impossible to count births in a population and also the mortality among the juveniles cannot be registrated as the dead pups will be washed away by the tides.

During the reproduction period the number of pups present at any given moment is a function of the numbers born and the numbers that died already. Therefore, estimates of birth rate and juvenile mortality rate have to be approached indirectly. This has been done by frequent aerial countings at low tide during the breeding period. Those data have been used in a recruitment model which provided estimates of the number of live pups present.

Acknowledgements.—I am much indebted to H. G. Fransz for his assistance in mathematics and the recruitment model he developed for this part of my study on the harbour seal. I am grateful to J. W. de Blok, H. Klomp, W. J. Wolff and J. J. Zijlstra for critically reading the manuscript and to M. F. Mörzer Bruyns and J. L. van Haaften for their continuous interest and discussions .I also want to acknowledge the aviator Mr A. Droog and the crew of the R. V. Eider of the Netherlands Institute for Sea Research.

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II. METHODS

The material collected consisted of counts of both juvenile and older animals, made by means of aerial surveys carried out during low tide when the animals were resting on the edges of the tidal flats. The surveys were made with intervals from one to two weeks. Pups could easily be recognized by their steel-grey colour, small size and lying aside of the mother. Counts were made at an altitude of about 175 metres. Within $2\frac{1}{2}$ hours the whole Dutch Wadden Sea could be covered. The animals were localized in a number of subareas (Fig. 1). The observations on the abundance of pups in the course of the whelping season, carried out in 1974 to 1978, yielded the basic material used in a recruitment model. The method is described by FRANSZ (1979) and used for the Dutch seal population by FRANSZ & REIJNDERS (1978).

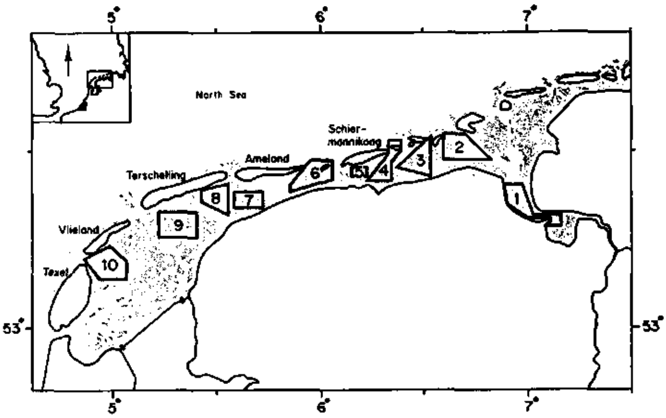


Fig. 1. Studied area of the Dutch Wadden Sea with subareas occupied by harbour seals (after REIJNDERS, 1976).

In this recruitment model the time from the first birth to the moment that the number of pups present (those born minus those already died) reaches its maximum (t_{max}) and the total length of the reproduction time (l_r) are used, because the ratio (t_{max}/l_r) provides a first estimate of the juvenile mortality (M) by means of Fig. 2 (which is redrawn from

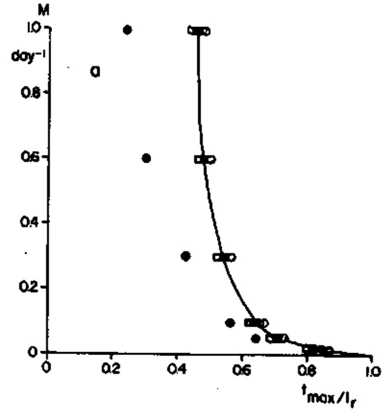


Fig. 2. Relationship between mortality rate (M) and the ratio of time to reach the maximum numbers of pups counted (t_{max}) over timelength of whelping period, l_r, at values for V ($= l_r/25$) of 2 (●), 10 (○), 16 (■) and 25 (□) (after FRANSZ & REIJNDERS, 1978).

FRANSZ & REIJNDERS, 1978). Then, around this first estimate, a number of values for M are tried in the model to check the possibility of an age-dependent mortality and to generate the curve with the best fit to the field data. For each year this procedure is followed.

Information on age class composition is obtained by measuring the width of the tracks that seals leave on the tidal flats and are a measure for their size, and therefore for their age. Track width measurements were carried out during the whelping time in 1976 and 1977, the mating time in 1977 and the post-mating time in 1978. The data obtained were treated in the same way as described earlier (REIJNDERS, 1976).

III. RESULTS

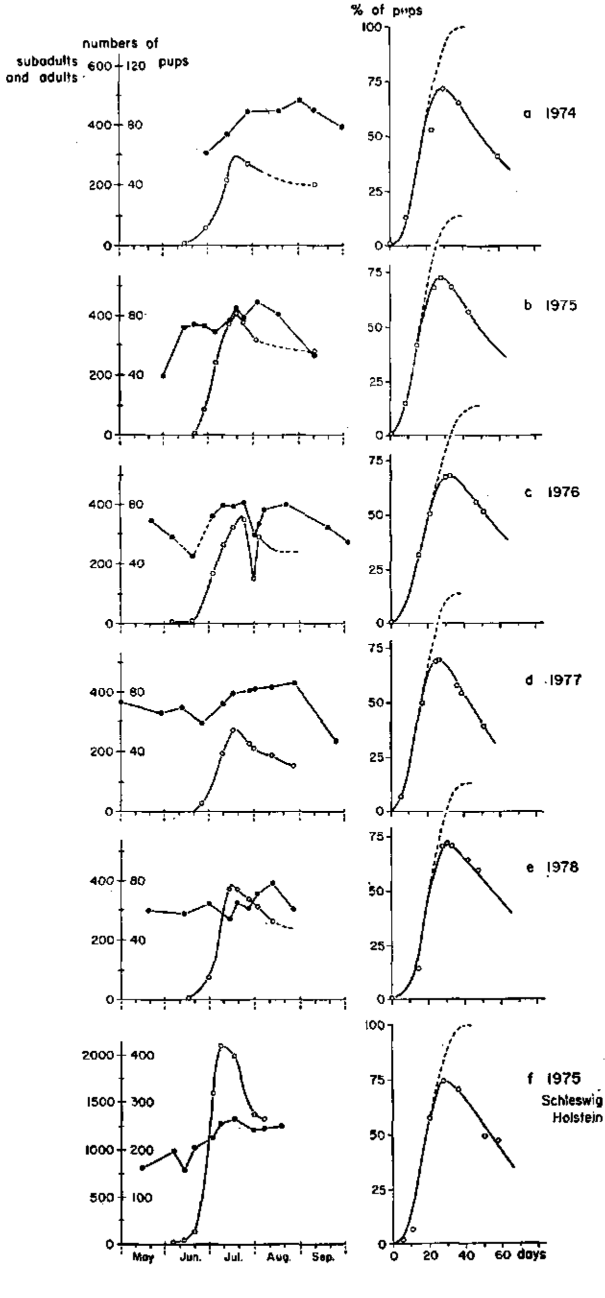
1. AERIAL SURVEYS

Fig. 3a to e (left-hand diagrams) show the results of the aerial surveys in the Dutch Wadden Sea for 1974, 1975, 1976, 1977 and 1978. The survey from 31 July 1976 was affected by bad weather conditions resulting in an underestimate of the numbers present. Fig. 3f shows comparable results of aerial surveys in the Wadden Sea of Schleswig Holstein for 1975 (DRESCHER, 1975). Combination of boat observations and aerial surveys gave the lengths of the whelping periods in the Dutch Wadden Sea (Table I). In all years whelping started about the second half of June. In the Wadden Sea of Schleswig Holstein the whelping period lasted about 37 days (DRESCHER, personal communication). The increase in numbers of pups due to births and the subsequent decrease, assumed to be caused by the mortality among the pups,

TABLE I

Dates upon which the whelping period started and the maximum number of pups was sighted; the length of the whelping period (l_r), the time (t_{max}) and relative moment (t_{max}/l_r) to reach the maximum numbers; and the corresponding relative mortality rates (M) of the seal population in the Dutch Wadden Sea for the years 1974 to 1978 and in Schleswig Holstein (S.H.) for 1975.

Area	*Year*	*Dates pups observed*		l_r	t_{max}	t_{max}/l_r	M
		First	*Maximum*	*(days)*	*(days)*		*(days^{-1})*
Neth.	1974	29 June	27 July	37	29	0.78	0.022
	1975	21 June	19 July	37	29	0.78	0.022
	1976	19 June	24 July	46	36	0.78	0.022
	1977	20 June	16 July	36	27	0.75	0.026
	1978	16 June	17 July	40	31	0.78	0.020
S.H.	1975	6 June	7 July	37	31.5	0.85	0.015


numbers of
subadults and adults
pups
% of pups
a 1974
b 1975
c 1976
d 1977
e 1978
f 1975
Schleswig Holstein
May
Jun.
Jul.
Aug.
Sep.
days

appear from the left hand diagrams of Fig. 3. After about 4 to 6 weeks the pups get weaned and can no longer be recognized with certainty by aerial counting. By this difficulty some points in the declining phase of the curves may have been affected, underestimating the number of pups, and consequently the juvenile mortality may be overestimated if these data were used for a calculation of juvenile mortality.

2. TRACK WIDTH MEASUREMENTS

The track width measurements resulted in estimates of the population composition of the whole Dutch Wadden Sea seal population as shown in Table II (which is an extension of table II in REIJNDERS, 1976). The age group distribution was estimated per subarea (for subareas see Fig. 1).

3. DETERMINATION OF INITIAL JUVENILE MORTALITY RATE AND RECRUITMENT

The initial juvenile mortality rate (M) was estimated with the model indicated (page 166). Table I gives the data needed for the calculations. Applying the values for t_{max}/l_r found for the different years in the model, several recruitment curves around each M value have been tried out. It appeared that the initial juvenile mortality rates can be considered as constant and the values for M which provided the curves with the best fit are given in Table I. With this result the corresponding recruitment curves (1974 through 1978) have been constructed (Fig. 3a to e; right-hand diagrams). The maximum numbers counted are given in Table III, the maximum numbers of pups observed as fraction of the numbers of pups born in Table IV.

The same procedure has been carried out on the data obtained in 1975 from the seal population in Schleswig Holstein (DRESCHER, 1975); the figures computed are also listed in Tables I, III and IV and the recruitment curve was constructed (Fig. 3f).

4. JUVENILE MORTALITY

After the suckling period but before migration occurs (early Septem-

Fig. 3. Number of pups (○) and of subadult plus adult harbour seals (●) counted during aerial surveys, and assumed curves (full line), in the Dutch Wadden Sea for 1974 to 1978 (the last count of subadults plus adults should be read as 430) and in the Wadden Sea of Schleswig Holstein in 1975 (left hand diagrams). Number of pups born (broken line) and number of pups surviving (full line) in percentages, according to the recruitment model and as based on the aerial surveys (○) for the same years and populations (right hand diagrams).

TABLE II

Population composition of seals in the Dutch Wadden Sea based on track width measurements in subareas, and on aerial census in 1976,1977 and 1978. For location of subareas see Fig. 1. Resulting total numbers of juveniles, subadults and adults are given with 95% confidence limits.

Group	*Subarea*										*Total*
	1	*2*	*3*	*4*	*5*	*6*	*7*	*8*	*9*	*10*	
1976, whelping time (end of June)											
Juveniles	6	21	23	6	2	0	4	3	3	2	70 ± 19
Subadults	8	20	38	22	10	13	14	21	15	15	176 ± 24
Adults	23	36	68	33	12	8	15	14	12	9	230 ± 23
Measured	21	75	103	39	22	19	28	29	17	17	370
Boat census	24	77	106	45	23	20	31	34	24	18	402
Aerial census	37	64	129	61	24	21	33	38	30	26	463
1977, whelping time (July)											
Juveniles	1	7	32	7	1	1	1	1	2	1	54 ± 14
Subadults	23	47	25	31	11	11	17	12	6	6	189 ± 22
Adults	10	36	72	21	13	19	12	18	14	8	223 ± 23
Measured	32	76	122	63	17	10	29	31	19	8	407
Boat census	32	77	129	68	22	12	30	32	20	11	433
Aerial census	34	90	129	59	25	31	30	31	22	15	466
1977, mating time (mid-August)											
Juveniles	1	4	9	9	2	1	2	1	0	1	30 ± 12
Subadults	22	42	38	40	11	13	21	12	7	6	212 ± 23
Adults	8	47	64	28	15	15	12	15	9	8	221 ± 23
Measured	33	83	123	65	21	22	18	21	12	10	398
Boat census	34	88	146	68	27	22	24	27	12	10	448
Aerial census	31	93	111	77	28	29	35	28	16	15	463
1978, post-mating time (mid-September)											
Juveniles	2	7	9	4	1	1	1	1	2	0	28 ± 13
Subadults	19	28	58	30	8	8	8	31	20	8	218 ± 24
Adults	17	33	59	21	20	12	15	12	11	12	212 ± 24
Measured	27	55	102	48	22	17	19	36	16	17	359
Boat census	31	56	114	49	26	17	19	36	17	17	382
Aerial census	38	68	126	55	29	21	24	44	33	20	458

TABLE III

Maximum number of pups, and subadults plus adults in different Wadden Sea areas; pups also as mean percentage of subadults plus adults. Data for Niedersachsen from WIPPER (1974), KRAFT (1976) and MANNINGA (personal communication) for Schleswig Holstein from DRESCHER (1975; personal communication); for Denmark (mean estimate) from JOENSEN, SØNDERGAARD & HANSEN (1976; personal communication) and for The Netherlands read from Fig. 2a to e.

Year	*The Netherlands*		*Niedersachsen*		*Schleswig Holstein*		*Denmark*	
	Pups	*Subad. + ad.*	*Pups*	*Subad. + ad.*	*Pups*	*Subad. + ad.*	*Pups*	*Subad. + ad.*
1974	60	470	221	784	±360	±1140		
1975	82	440	201	848	420	1329	±150	±550
1976	72	410	253	803	404	1249		
1977	55	430	198	942	488	1318		
1978	77	430	232	996	469	1326		
mean %	16		26		34		27	

ber) the numbers of pups present were derived from aerial counts combined with measurements of track widths. With this latter method juvenile seals can easily be discriminated from older individuals (REIJNDERS, 1975). Early September counts are available for 1974 and 1977. Due to bad weather the measurements in 1978 could not be completed before the end of September. As emigration already occurs in this period these data have not been used in the following calculation. In 1974 the numbers of pups recorded in this way amounted to 40 and in 1977 to 30. Compared with the numbers born (Table IV), the early September estimates in 1974 and 1977 indicate a mean decrease of 57%.

TABLE IV

Total number of pups counted and of births calculated; ratios of pup numbers to birth number and of birth numbers to number of subadults plus adults; 1974 to 1978 in the Dutch Wadden Sea and 1975 in the Wadden Sea of Schleswig Holstein (S.H.).

Area	*Year*	*Pups counted*	*Births calculated*	*Pups/ births*	*Births/ subad. + ad.*
Neth.	1974	60	83	0.72	0.18
	1975	82	114	0.72	0.26
	1976	72	106	0.68	0.26
	1977	55	80	0.69	0.19
	1978	77	107	0.72	0.25
S.H.	1975	420	560	0.75	0.42

IV. DISCUSSION

The method used to relate numbers counted to numbers born is based on the assumption that the births are normally distributed over the whelping season. Though the right-side parts of the curves (Fig. 2a to f; left-hand) do not show the normally distributed births since these curves are strongly affected by mortality, it appeared that they are in agreement with this important provision.

In the Introduction it is stated that the decline in the seal population may be caused by a reduction in the reproductive rate as well as an enhanced juvenile mortality.

The second possibility may certainly play a role according to the results quoted in Table I. The initial juvenile mortality in the Dutch population is higher than that in the population of the Wadden Sea in Schleswig Holstein which is considered to be in a stable state (DRESCHER, personal communication).

The first possibility—reduction in the reproductive rate—is more difficult to evaluate. An indication can be obtained from the ratios between the maximum numbers of pups observed and the total numbers of subadults and adults present (Table III). The percentage of pups in the Dutch Wadden Sea is clearly lower than in the other parts of the Wadden Sea with a stable population. More reliable information can be obtained from the ratio between the total numbers of births and the maximum numbers of subadults and adults observed (Table IV). Again it seems that the reproductive rate in the Dutch Wadden Sea is lower than that in the stable population of Schleswig Holstein. Possible causes of the noted lower number of pups may be: (1.) difference in sex ratio, with a lower share of adult females in the Dutch population; (2.) a difference in the average number of pups per adult female with a lower number in the Dutch population; and (3.) different share of subadults with a higher number of subadults in the Dutch population.

It is impossible to distinguish between sexes with the methods to be used in the Wadden Sea. Some data were presented (REIJNDERS, 1976: table III) on the sex ratio of juvenile, subadult and adult seals from which it may be concluded that sex ratio in harbour seals is normally close to 1, also in older seals. Direct information on the sex ratio of adults in the Dutch Wadden Sea is not available. Although a sample of 33 dead adult seals, washed ashore in the Dutch Wadden Sea area in the period 1964 to 1974, consisted of 6 males and 27 females, it is believed that this reflects a difference in distribution of males and females in time and space rather than an imbalance in sex ratio. This is supported by a sample of 46 shot adult seals from the Wadden Sea of Niedersachsen, mentioned by WIPPER (1974) and consisting of

19 males and 27 females. For the Schleswig Holstein population more females in the ratio of 1.35 were derived from Table V, which will be discussed lateron (page 175). In spite of the restricted information it is assumed that sex ratio of adult seals in the Dutch Wadden Sea is similar to that of Schleswig Holstein and that the lower number of pups is not caused by a low percentage of females in the adult population. Assuming for the Dutch population also a sex ratio of 1.35, the average number of pups per female can be calculated. The ratio of subadults to adults is taken from Table II and earlier data (REIJNDERS, 1976: table II), the maximum number of subadults and adults from Table III. In 1974 about 230, in 1975 about 245, in 1976 about 232, in 1977 about 233 and in 1978 about 212 adults were present which results in numbers of about 132, 141, 133, 134 and 122 females, respectively. From these data and the number of births in the corresponding years (Table IV) it is derived that the average pup production per adult female was 0.63, 0.81, 0.80, 0.60 and 0.88, respectively; the average value is 0.74.

Literature data on pup production per adult female result in an average value of 0.85 for *Phoca vitulina concolor* (BOULVA, 1973), 0.88 for *Phoca sibirica* and 0.85 for *Phoca fasciata* (USSR-reports, 1976), 0.80 for *Phoca hispida* (SMITH, 1973), 0.86 for *Phoca hispida* (JOHNSON, FISCUS, OSTENSON & BARBOUR, 1966) and 0.85 for *Pagophilus groenlandicus* (SERGEANT, 1971). MCLAREN (1958) arrives for *Phoca hispida* at 0.67 but this value is neglected as he arbitrarily reduced the fertilities in the older age classes (SMITH, 1973).

These figures suggest a reduced pup production per female in the seal population of the Dutch Wadden Sea. Since the pup production per female is not known for the stable Schleswig Holstein population it has been tried to determine this parameter indirectly for this population by means of the construction of a life table.

A life table is a concise summary of certain vital statistics of a population. Life table parameters for a population in a stable state, including survivorship (l_x), mortality (d_x), mortality rate (q_x), mean life expectation (e_x), mean number of seals in the age interval x to $x + 1$ (L_x) and the proportion of females (k), have been calculated according to formulas outlined by DEEVY (1947) and CAUGHLEY (1977). The following assumptions have been made:

(1.) longevity of females 35 years and males 25 (WIPPER, 1974; data obtained by VAN HAAFTEN and the author);

(2.) mortality in the first year 60%;

(3.) age-independent mortality rate in the year classes 1 to 35 (SERGEANT & FISHER, 1960; BIGG, 1969; BOULVA, 1973; SMITH, 1973);

(4.) different mortality rates for males and females resulting in the proportion of females as given by BOULVA (1973);

(5.) sexual maturity at the age of three (MOHR, 1952; TIKHOMIROV, 1966; BISHOP, 1968; BOULVA, 1973), thus females produce their first pup at the age of four;

(6.) average number of pups per female at the age of four 0.33, five 0.59, six 0.81, seven and eight 1.00 and nine to thirtyfive 0.92 (BOULVA, 1973).

Concerning the average number of pups per female (assumption 6.), it was stated by BIGG (1969, for harbour seals), BOULVA (1973, for harbour seals), HEWER (1974, for grey seals), SERGEANT (1971, for harp seals), SMITH (1973, for ringed seals) and TIKHOMIROV (1966, for ringed seals, harbour seals, ribbon seals and bearded seals) that reproduction in these species is age-independent after 7 to 8 years and remains constant till the end of life. So there is no reason to assume that the harbour seal displays sexual senility.

The first year mortality (assumption 2.) is deduced from the Schleswig Holstein population. There, DRESCHER (personal communication) estimated this mortality at 45 to 50%. Taking into account the underestimation of the total numbers born (see Section III.3) his figure results in an overall first year mortality of about 60%. HEWER (1964) found for grey seals *(Halichoerus grypus)* in their first year, a mortality of about 60% and CHAPMAN (1961) for the Pribilov-fur seal (*Callorhinus ursinus*) also about 60%.

An estimate of the mortality amongst subadults and adults (assumption 3.) is derived in relating the size of the prewhelping population (N) in the year $t + 1$ to the prewhelping population in year t, using the numbers of adults and subadults dying ($D \cdot N_t$), the numbers of pups born ($p \cdot N_t$) and the numbers of pups dying ($d \cdot p \cdot N_t$):

$$N_{t+1} = N_t\{1 - D + p\,(1 - d)\} \tag{1}$$

Based on this equation, a general one allows calculation of the seals present after t years (N_t) when the size of the prewhelping population in year 0 (N_0) is known:

$$N_t = N_0\{1 - D + p\,(1 - d)\}^t \tag{2}$$

or:

$$N_t = N_0\, e^{t \cdot \ln\{1 - D + p\,(1 - d)\}} \tag{3}$$

This equation is similar to the equation of population growth, $N_t = N_0 e^{rt}$, where r is the realized intrinsic rate of increase (WILSON & BOSSERT, 1971) parallel to the ln rate of increase in equation *(3)*.

As the population in Schleswig Holstein is considered to be in a stable state the realized intrinsic rate of increase is zero and $p = D/$

$(1 - d)$. Since d is assumed to be 0.60, p is 2.5 D. For p a value of 0.42 (Table IV) was calculated, but according to DRESCHER (personal communication) this value is too high because the number of subadults and adults is clearly underestimated. For that reason several values for p ranging from 0.30 to 0.40 were used to calculate corresponding

TABLE V

Life table for a stable harbour seal population (Schleswig Holstein). Under certain assumptions (page 173) and according to formulas outlined by DEEVY (1947) are calculated for each age class: survivorship (l_x), mortality (d_x), mortality rate (q_x), mean number of seals in the age interval x to $x + 1$ (L_x), mean life expectation (e_x), ratio females to total (k) and number of females ($k \cdot 1000\ l_x$).

Age in years	*1000* l_x	*1000* d_x	*1000* q_x	*1000* L_x	e_x	k	$k \cdot 1000\ l_x$
0	1000	600	600	700	3.55	0.50	500
1	400	52	130	374	7.12	0.50	200
2	348	45	etc.	325.5	7.11	0.50	174
3	303	40		283	7.09	0.51	155
4	263	34		246	7.08	0.51	134
5	229	30		214	7.07	0.52	119
6	199	26		186	7.05	0.52	103
7	173	22		162	7.03	0.53	92
8	151	20		141	7.01	0.54	82
9	131	17		122.5	6.98	0.55	72
10	114	15		106.5	6.95	0.55	63
11	99	13		92.5	6.92	0.56	55
12	86	11		80.5	6.89	0.57	49
13	75	10		70	6.85	0.58	44
14	65	8		61	6.80	0.59	38
15	57	7		53.5	6.75	0.60	34
16	50	7		46.5	6.67	0.61	31
17	43	6		40	6.59	0.62	27
18	37	5		34.5	6.51	0.63	23
19	32	4		30	6.42	0.64	20
20	28	4		26	6.33	0.66	18
21	24	3		22.5	6.22	0.68	16
22	21	3		20	6.10	0.70	15
23	19	2		18	5.94	0.73	14
24	16	2		15	5.72	0.77	13
25	14	2		13	5.52	0.82	12
26	12	1		11.5	5.26	1.00	12
27	11	2		10	5.00	1.00	11
28	9	1		8.5	4.67	1.00	9
29	8	1		7.5	4.32	1.00	8
30	7	1		6.5	3.90	1.00	7
31	6	1		5.5	3.42	1.00	6
32	5	0		5	2.91	1.00	5
33	5	1		4.5	2.26	1.00	5
34	4	0		4	0.99	1.00	4
35	4	4		2	0.56	1.00	4

mortalities. Starting with a cohort of 1000 (July), these data plus the other assumptions were used in building several life tables. From each life table the net reproductive rate (*cf.* PARIS & PITELKA, 1962; KREBS, 1972) was calculated according to the age specific reproductive rates (assumption 6). The life table where p was set at 0.32, implying D to be 0.13 (Table V), fitted the best to a net reproductive rate of 1 occuring in a stable population.

The number of females in the year classes 4 to 35 (Table V), combined with the reproductive rates mentioned before (assumption 6.), results in an average pup production per female of 0.87. It is concluded that the reproductive rate of the seal population in the Dutch Wadden Sea, also based on the assumptions 1 to 5, and calculated at 0.74, is lower than that of the population in the Wadden Sea in Schleswig Holstein.

Finally attention has to be paid to the question whether the low ratio of births to the total numbers of subadults and adults in the Dutch Wadden Sea (Table IV) is caused by an overrepresentation of subadults. Table VI gives the percentages of births, subadults and adults

TABLE VI

Population structure of a stable seal population (*e.g.* Schleswig Holstein) and the population in the Dutch Wadden Sea in the years 1974 to 1978; percentages calculated for the stable population on base of Table V and for the Dutch population on base of REIJNDERS (1976: table II), Tables II, III and IV.

Seals	*Stable population*	*Dutch population*				
		1974	*1975*	*1976*	*1977*	*1978*
Juv. born	24.7	15.0	20.6	20.5	15.7	19.9
Subadults	26.0	43.4	35.2	34.5	38.6	40.6
Adults	49.3	41.6	44.2	45.0	45.7	39.5

in a stable population derived from the life table (Table V) and the same parameters determined in the field for the Dutch seal population (REIJNDERS, 1976: table II; Tables II and IV). It demonstrates that the number of subadults in the Dutch population is relatively high, compared to the number of births, as well as to the number of adults.

The same conclusion can be reached in another way. From the number of pups in the preceding years, the number of subadults in 1977 is calculated. It is based on the last count of pups in each year (Fig. 3) and, from that date on, a constant mortality rate of 13% per year—which rate probably is an underestimate for the mortality during the rest of the first year. In this way 144 subadults are found for 1977; 27 born in 1974; 49 in 1975 and 48 in 1976. To this number, the number of juveniles returned to the Wadden Sea by seal nursery stations (Pieterburen and Texel) has to be added. Applying the same mortality rate to these 50 young seals, 40 subadults will have to be added.

These 164 subadults calculated to be present in 1977—which is probably an overestimate—are clearly less than the 189 subadults observed in the Wadden Sea at whelping time in 1977 (Table II). This observation indicates an immigration of subadults. It is known that migration amongst subadult marine mammals is quite common (WADA, 1969; JOHNSON, 1975). As there was a higher share of subadults in the age-group composition of the subareas adjacent to the German Wadden Sea compared to the other subareas (Table II) and as disturbance was much heavier in the German part of the area, it is believed that migration takes place from the German part to the eastern part of the Dutch area. Apart from the question whether an over-representation of subadults exists in the Dutch population or not, its birth ratio is too low. Table VI shows again that during 1974 to 1978 in the Dutch population the ratio juveniles-adults (mean 0.43) is lower than in the stable population of Schleswig Holstein (0.50).

The Dutch Wadden Sea population has declined strongly in the period 1965 to 1975 (REIJNDERS, 1976). On the other hand Table III gives the impression that the numbers of subadults and adults remained more or less constant in the period 1974 to 1978. For that reason it is investigated whether the ratio of births over subadults plus adults was lower in 1965 to 1974 than in the period 1974 to 1978. Again the equation of population growth *(3)* was used under the assumptions that D is 0.13 and d is 0.60. For N_t is taken 470 (Table III) and N_0 is based on a count on 10 July 1965 (REIJNDERS, 1976: fig. 4), a value that is certainly an underestimate of the total number of subadults and adults present in 1965. Substituting these values in the equation *(3)* an average share of pups born (p) of 0.09 is found over the period of 9 years instead of the 0.32 calculated for a stable population (page 175). This either indicates that the mean reproductive rate from 1965 to 1974 was much lower than from 1974 to 1978, or that an increased juvenile mortality operated during 1965 to 1974.

V. SUMMARY

During 1974 to 1978 population dynamics of the harbour seal population in the Dutch Wadden Sea have been studied. Frequent aerial surveys provided data upon the size of the population. During boat trips information about the age composition was obtained by measuring track widths. These data have been used in a simulation model to calculate the birth rate and the initial juvenile mortality.

It was found (1.) that the initial juvenile mortality in the Dutch population is higher than that in Schleswig Holstein; (2.) that pup production in the Dutch population is low compared to the population in Schleswig Holstein if the ratio adult females to males in the Dutch population is supposed to be equal to that in Schleswig Holstein, or that mortality amongst adult females in the Dutch population is higher than in Schleswig Holstein; (3.) that the number of subadults in the Dutch population is relatively high compared to a stable population (Schleswig Holstein) and that apparently immigration from elsewhere occurs; (4.) that the reproductive rate was higher or the juvenile mortality lower during 1974 to 1978 than in the preceding decennium.

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ORGANOCHLORINE AND HEAVY METAL RESIDUES IN HARBOUR SEALS FROM THE WADDEN SEA AND THEIR POSSIBLE EFFECTS ON REPRODUCTION

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I. INTRODUCTION

The harbour seal population in the Dutch Wadden Sea (fig. 1) has decreased strongly during the last decades (Reijnders, 1976; Summers et al., 1978). The causes were unknown and by studying its population dynamics it has been established (Reijnders, 1978) that pup production in the declining Dutch seal population was lower than that in the stable population of Schleswig-Holstein and Denmark. In fact an increasing trend in pup production rates was found going along the Wadden Sea coast from the Netherlands via Niedersachsen to Schleswig-Holstein and Denmark.

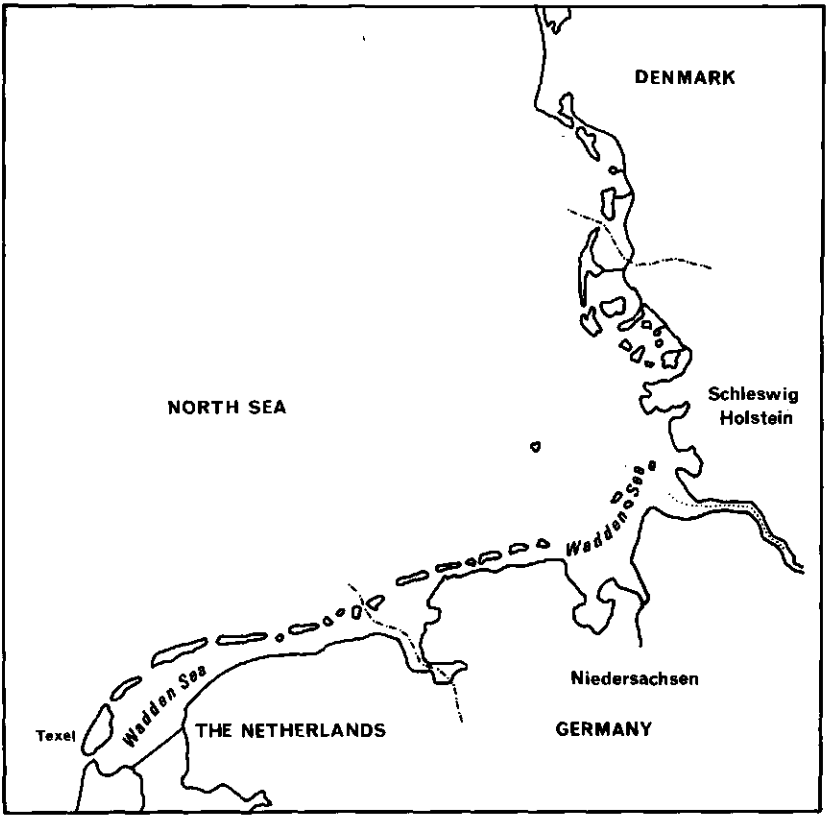

Fig. 1. Map of the North Sea and the Wadden Sea with the adjacent countries.

It was suggested that some environmental factors play a role and the aim of the present study was to investigate the contribution of environmental pollution.

It is well known that marine mammals including seals, are subject to environmental contaminants such as organochlorine pesticides, PCB's and certain metals. Extremely high levels of mercury were detected in 1967 in seals in Finland (Helminen et al., 1968). The presence of relatively high levels of organochlorines in grey seals and dolphins from the British coast was reported by Holden & Marsden (1967) and Robinson et al. (1967). For harbour seals from the Dutch coast Koeman & Van Genderen (1966), Koeman et al. (1972, 1975) report data on heavy metals and organochlorines. For harbour seals from the Schleswig-Holstein coast, similar reports were given by Drescher, Harms & Huschenbeth (1977).

More recently, organochlorines and metals have been associated with reproductive failure in marine mammals (Le Boeuf & Bonnell, 1971; Delong et al., 1973; Olsson et al., 1975; Gilmartin et al., 1976; Helle et al., 1976, 1976a). Furthermore there are experimental data concerning studies with PCB in mink, which support the hypothesis of the effects of PCB's on reproduction performance of seals (Aulerich & Ringer, 1977; Jensen et al., 1977).

However these experiments clearly showed species-specific responses to PCB and total DDT, i.e. pathological changes of the uteri found in seals did not occur in minks. Besides Koeman et al. (1973) found a perfect linear correlation between total mercury and selenium in livers of several marine mammal species. They concluded from their and other studies that selenium might be involved in a mechanism which protects seals against the toxic action of mercury.

This study was undertaken to compare residue levels of several pollutants in different tissues of seals from Schleswig-Holstein plus Denmark on one hand and the Netherlands on the other and besides to achieve data on the present state of contaminant burden in the total Wadden Sea seal population.

It is stressed that sample treatments differ, analytical techniques used are different and therefore quantitative data may vary considerably. Especially when residue levels in tissues of seals originating from two areas have to be tested on significant differences, results taken from literature stand a large chance to lead to unreliable conclusions. In this study the material was collected in an identical procedure and all analyses for heavy metals, organochlorines and bromium have been carried out each with the same techniques in the same laboratories.

II. MATERIAL AND METHODS

The material used in this study was collected during 1975 and 1976. It consists exclusively of harbour seals found dead on the beach, in total 16 animals from the Schleswig-Holstein plus Danish Wadden Sea and 14 specimens from the Dutch Wadden Sea. The animals were autopsized and brain, liver, kidney and blubber tissue was collected and deepfrozen. The age was determined by counting annual growth layers in teeth cementum, according to Laws (1962).

Mercury in the inorganic and organic form and selenium was analyzed by Atomic Absorption Spectophotometry. Mercury according to the procedure devised by Magos (1971) and modified by Van de Ven, Koeman & Svenson (1979). For analysis of selenium, tissue samples were digested according to the procedure devised by Watkinson (1966), measured according to the method of Duncan & Parker (1974) as modified and applied by Van de Ven, Koeman & Svenson (1979).

Organochlorines were analyzed by Gas Liquid Chromatography. The samples were treated firstly according to the clean-up procedure of column chromatography, devised by Holden & Marsden (1969). The PCB components were converted to decachlorobiphenyl (DCB) according to Berg, Diosady & Rees (1972). This procedure appears to be acceptable as can be deduced from a report of an international intercalibration programme on monitoring organochlorine and mercury residues in wildlife (Holden, 1973). Organochlorines analyzed were PCB's, o,p'-DDT, p,p'-(DDT+DDE+TDE), dieldrin, aldrin, endrin, endosulfan, α,β,γ,-HCH, HCB, pentachlorobenzene and HEPO (hepta chloro epoxide).

Bromium was analyzed by a colorimetric method according to Hunter (1955), as modified and applied by Rauws & Van Logten (1974).

The results from both Schleswig-Holstein plus Danish and Dutch seals were divided in two age classes: juveniles and subadults on one hand and adults on the other. Several possible correlations between age and residue levels of pollutants were analyzed, and interrelationships between residue levels in several tissues have been computed (Tektronix, 1975). For both age classes the results were statistically tested on significant differences between both areas. The equality tests were Wilcoxon, Student's t-test and de Weir's (1960) standardized t-test. Since the total DDT group consists of DDT and its metabolites DDE and TDE, - each of which is related to the others by metabolizing processes -, a multivariate statistical technique according to Kramer & Jensen (1969) has been used for DDT.

Acknowledgements

First of all I want to express my sincere thanks to my collegues Dr. B. Clausen (State Veterinary Laboratory, Copenhagen) and Dr. H.E. Drescher (Institut für Haustierkunde, Kiel) who provide me with tissues of harbour seals found dead in the Danish and Schleswig-Holstein parts of the Wadden Sea. I want to express my gratitude to the Department of Toxicology at the Agricultural University in Wageningen, for rendering me hospitality and facilities to carry out the analyses on mercury and selenium and especially Mr. W.S.M. van de Ven who made me familiar with and discussed the analytical procedures. I also want to acknowledge Mr. J.K. Quirijns (Central Institute for Nutrition & Food Research TNO, Zeist) and Dr. A.G. Rauws (National Institute of Public Health, Bilthoven) for carrying out the analyses on respectively organochlorines and bromium. I am gratefull to Mr. W.G. van Arkel (Netherlands Institute for Sea Research, Texel) for his assistance in mathematical affairs and Dr. W. van Utrecht (Institute for Taxonomic Zoology, Amsterdam) who kindly carried out the age determination of the Schleswig-Holstein seals.

Finally I want to thank Prof. Dr. J.H. Koeman for his critical reading of the manuscript and the discussions upon the subjects in this study.

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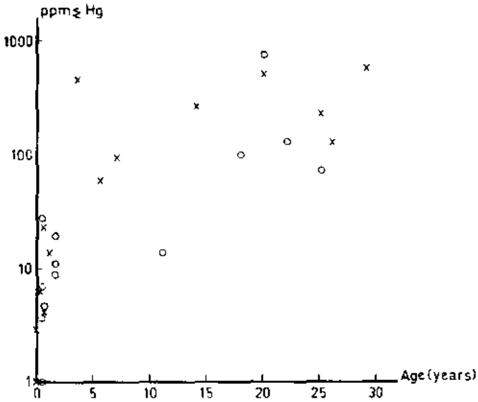
III. RESULTS

The results are presented in Tables 1-8 and Figs. 2-32. All values for mercury, selenium and bromium are expressed in ppm on wet weight basis. The data on chlorinated hydrocarbons in blubber are expressed in ppm on extractable lipid basis whereas the residues found in kidney and liver are on wet weight basis.

1. MERCURY, SELENIUM AND BROMIUM

It is clear from these data that total Hg is not distributed equally over the different organs. For total mercury the tissue concentrations increased with age as shown in Figs. 2, 3 and 4. This increase is most pronounced in the liver, however the same pattern can be observed in the brain and kidney. The maximum level found in the liver was 751 ppm Hg. This age dependent increase of the mercury content in tissue of seals has been demonstrated by several authors (Hellminen, Karppanen & Koivisto, 1968; Henricksson, Karppanen & Helminen, 1969; Koeman et al., 1972; Freeman & Horne, 1973; Heppleston & French, 1973; Sergeant & Armstrong, 1973; Koeman et al., 1975; Smith & Armstrong, 1975; Roberts, Heppleston & Roberts, 1976; Drescher, 1977).



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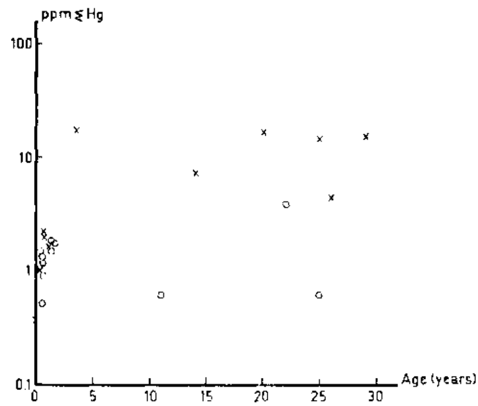

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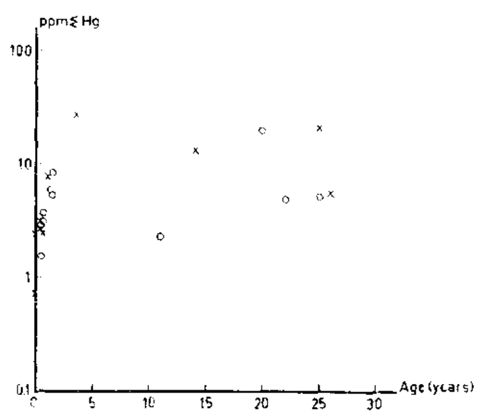

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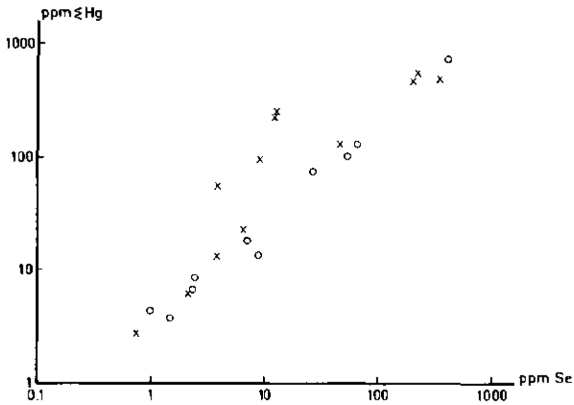
Figs. 2, 3 + 4.
Relationships between total mercury (ΣHg) content and age in respectively liver, brain and kidney tissue of harbour seals from different areas in the Wadden Sea. Specimens from The Netherlands (X) and Schleswig Holstein plus Denmark (O). Levels expressed in mg/kg wet weight.

Figs. 5 and 6 show in the material surveyed a strong correlation between mercury and selenium in both liver and brain. The highest Se levels were found in the liver (maximum was 409 ppm) and age relationships were similar to those for mercury. These data confirm the previously reported findings of Koeman et al. (1973, 1975) who demonstrated an almost perfect linear correlation between mercury and selenium in several marine mammals.



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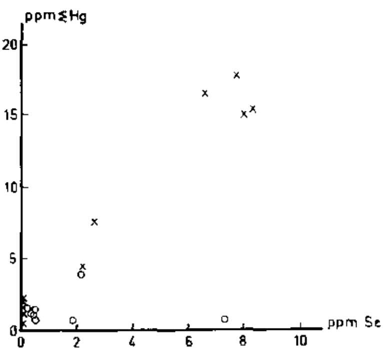
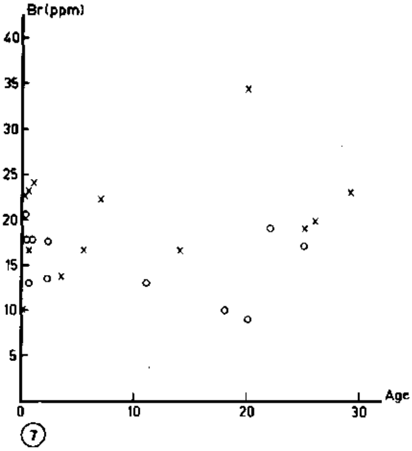
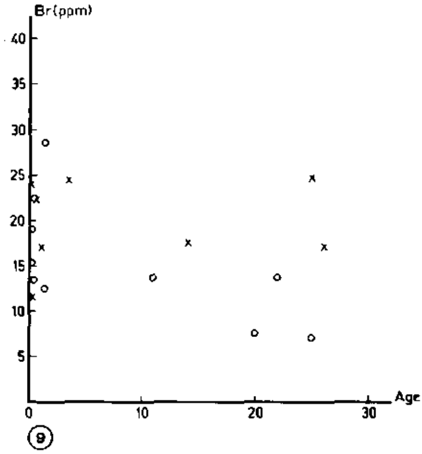


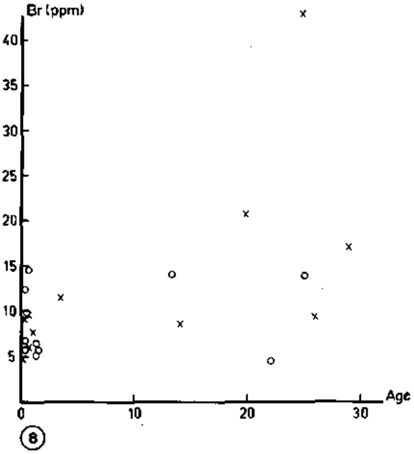
6

Figs. 5 + 6.
Relationships between total mercury (ΣHg) and selenium (Se) contents in respectively liver and brain tissue of harbour seals. Specimens from The Netherlands (X) and from Schleswig Holstein plus Denmark (O). Levels expressed in mg/kg wet weight.

From the material available, no dependency of bromium content on age could be deduced (Figs. 7, 8, 9).







Figs. 7, 8 + 9.
Relationships between bromium (Br) content and age in liver, brain and kidney tissue of harbour seals. Specimens from The Netherlands (X) and from Schleswig Holstein plus Denmark (O). Levels expressed in mg/kg wet weight. Age in years.

2. ORGANOCHLORINES

PCB, Σ DDT(p,p'-DDT, DDE, TDE), α HCH, β HCH, HEPO and dieldrin levels in the blubber, liver and kidney tissues are shown in Tables 2 and 3. The maxima for PCB and Σ DDT were respectively 1447 ppm and 178 ppm. It is obvious that the highest levels of all contaminants are found in seals obtained from the Dutch area. In general lower levels occur in juveniles than in adults.

The results of the statistical tests on possible significant differences in residue levels between the German plus Danish seals on one hand and Dutch seals on the

		Schleswig Holstein + Denmark		Netherlands	
		juveniles and subadults	adults	juveniles and subadults	adults
Kidney	Σ HG	4.28 1.58 - 8.28 (8)	7.54 2.24 - 17.90 (4)	3.25 0.71 - 3.22 (5)	17.04 5.44 - 28.16 (4)
	CH_3Hg	2.21 0.62 - 3.97 (8)	3.48 0.96 - 9.15 (4)	1.70 0.26 - 4.66 (5)	7.93 2.12 - 15.42 (4)
	Se	0.56 trace - 1.33 (7)	3.54 1.87 - 7.25 (4)	1.22 0.55 - 1.55 (5)	7.10 2.28 - 9.95 (4)
	Br	18.5 12.4 - 28.6 (6)	10.6 7.1 - 13.9 (4)	14.92 11.4 - 23.9 (5)	21.05 17.1 - 24.9 (4)
Liver	Σ HG	10.19 1.07 - 27.32 (8)	214.16 13.6 - 751.4 (5)	8.41 0.87 - 22.96 (6)	292.87 58.58 - 573.32 (8)
	CH_3Hg	3.49 0.46 - 6.64 (8)	31.09 3.61 - 103.47 (5)	3.25 0.46 - 7.62 (6)	60.57 16.03 - 134.42 (8)
	Se	2.57 trace- 10.82 (7)	112.09 9.21 - 408.65 (5)	2.84 0.74 - 6.53 (5)	109.17 3.92 - 350.36 (8)
	Br	16.6 12.9 - 20.7 (6)	13.6 8.9 - 19.0 (5)	19.28 10.0 - 24.0 (5)	20.68 13.4 - 34.4 (8)
Brain	Σ Hg	1.23 0.51 - 1.71 (8)	1.67 0.61 - 3.79 (3)	1.39 0.37 - 2.12 (5)	12.58 4.32 - 17.52 (6)
	CH_3Hg	0.92 0.44 - 1.18 (8)	0.50 0.22 - 0.99 (3)	0.99 0.24 - 1.52 (5)	3.19 1.04 - 5.16 (6)
	Se	1.12 trace- 7.38 (8)	1.50 0.53 - 2.16 (3)	traces (5)	5.85 2.17 - 8.25 (6)
	Br	8.3 5.1 - 14.6 (8)	10.83 4.6 - 14.0 (3)	7.54 5.5 - 9.7 (5)	18.28 8.6 - 42.8 (6)

Table 1. Means (underlined) and ranges of total mercury (ΣHG), methyl mercury (CH_3Hg), selenium (Se) and bromium (Br) contents in kidney, liver and brain of juvenile plus subadult and adult harbour seals from Schleswig Holstein plus Denmark and The Netherlands. Number of specimens analyzed, between brackets. Levels expressed in mg/kg wet weight.

	Schleswig Holstein + Denmark		Netherlands	
	juveniles and subadults	adults	juveniles and subadults	adults
PCB	113.8 / 23 - 340 (8)	76.4 / 40.5 - 122.9 (6)	134.0 / 5.0 - 680 (6)	701 / 87 - 1447 (8)
Σ DDT	16.9 / 4.0 - 47.1 (8)	8.5 / 3.7 - 15.2 (6)	29.8 / 3.3 - 127.1 (5)	47.3 / 6.6 - 178 (8)
DDT	5.5 / 1.3 - 13.0 (8)	2.0 / 0.7 - 6.1 (6)	9.0 / 0.9 - 36 (5)	11.7 / 1.0 - 64 (8)
DDE	10.8 / 2.6 - 30.0 (8)	5.0 / 2.4 - 8.8 (6)	16.1 / 1.2 - 83 (6)	26.6 / 3.8 - 72 (8)
TDE	0.6 / 0.1 - 2.1 (8)	0.7 / 0.2 - 3.0 (6)	1.8 / 0.2 - 8.1 (5)	9.0 / 0.3 - 42 (8)
α HCH	0.5 / 0.2 - 1.4 (8)	0.2 / 0.1 - 0.3 (6)	0.3 / n.d.- 0.6 (5)	0.4 / n.d.- 0.7 (8)
β HCH	0.3 / n.d.- 1.4 (8)	n.d. (6)	* / n.d.- 0.3 (5)	* / n.d.- 0.1 (8)
HEPO	0.4 / 0.1 - 1.4 (8)	0.2 / 0.1 - 0.4 (6)	1.0 / n.d.- 3.5 (5)	1.0 / 0.2 - 3.1 (8)
Dieldrin	* / n.d.- 0.2 (8)	n.d. (6)	0.5 / n.d.- 0.9 (5)	0.4 / n.d.- 0.4 (8)

Table 2. Organochlorines contents in blubber tissue of juvenile plus subadult and adult harbour seals from Schleswig Holstein plus Denmark and The Netherlands.
*: only one detection; n.d. not detectable, γ HCH (Lindane) was not detectable (detection limit 0.1 ppm). Levels expressed in mg/kg fat weight. Number of specimens analyzed, between brackets.

	Schleswig Holstein + Denmark				Netherlands			
	juveniles and subadults		adults		juveniles and subadults		adults	
	kidney	liver	kidney	liver	kidney	liver	kidney	liver
% lipid content	2.93 / 2.5 - 3.2 (8)		3.48 / 1.7 - 7.5 (5)	7.32 / 2.1 - 17.8 (5)	2.66 / 2.1 - 3.3 (5)	4.8 (1)	4.90 / 1.8 - 6.5 (4)	6.75 / 3.9 - 9.9 (4)
PCB	1.81 / 0.3 - 7.2 (8)		3.69 / 0.26 - 13.0 (5)	4.42 / 0.67 - 9.4 (5)	1.54 / 0.22 - 5.79 (5)	1.66 (1)	27.4 / 3.26 - 72.34 (4)	31.47 / 22.43 - 46.30 (4)
ΣDDT	0.20 / 0.01 - 0.85 (8)		0.21 / 0.02 - 0.64 (5)	0.43 / 0.03 - 0.97 (5)	0.38 / n.d. - 1.61 (5)	0.11 (1)	0.88 / 0.17 - 2.14 (4)	2.11 / 0.93 - 4.70 (4)
DDT	0.02 / n.d. - 0.08 (8)		* / n.d. - 0.08 (5)	n.d. (5)	0.02 / n.d. - 0.8 (5)	n.d. (1)	0.03 / n.d. - 0.06 (4)	n.d. (4)
DDE	0.15 / 0.01 - 0.62 (8)		0.15 / 0.02 - 0.45 (5)	0.23 / 0.02 - 0.61 (5)	0.28 / 0.02 - 1.2 (5)	0.08 (1)	0.64 / 0.14 - 1.5 (4)	1.60 / 0.80 - 3.3 (4)
TDE	0.03 / n.d. - 0.15 (8)		0.05 / n.d. - 0.11 (5)	0.18 / 0.01 - 0.52 (5)	0.08 / n.d. - 0.33 (5)	0.03 (1)	0.23 / 0.02 - 0.58 (4)	0.49 / 0.13 - 1.3 (4)
αHCH	* / n.d. - 0.01 (8)		n.d. (4)	n.d. (4)	* / n.d. - 0.01 (5)	n.d. (1)	* / n.d. - 0.01 (4)	0.015 / 0.01 - 0.03 (4)
βHCH	n.d. (8)		n.d. (4)	n.d. (4)	n.d. (5)	n.d. (1)	n.d. (4)	n.d. (4)
HEPO	* / n.d. - 0.02 (8)		* / n.d. - 0.01 (5)	0.02 / n.d. - 0.04 (5)	* / n.d. - 0.05 (5)	n.d. (1)	0.02 / n.d. - 0.04 (4)	0.08 / 0.04 - 0.16 (4)
Dieldrin	n.d. (8)		n.d. (5)	n.d. (5)	0.01 / n.d. - 0.03 (5)	0.02 (1)	0.02 / n.d. - 0.03 (4)	0.02 / n.d. - 0.03 (4)

Table 3. Organochlorines and % lipid contents in kidney and/or liver tissues of juvenile plus subadult and adult harbour seals from Schleswig Holstein plus Denmark and The Netherlands.
*: only one detection; n.d.: not detectable, γ HCH was not detectable (detection limit 0.01 ppm). Levels expressed in mg/kg wet weight. Numbers of specimens analyzed, between brackets.

other are presented in Tables 7 and 8.

In the juvenile plus subadult group the residue levels of total mercury in kidney and selenium in the brain were significantly higher in the Schleswig-Holstein and Danish seals whereas selenium and bromium were significantly higher for respectively kidney and liver in the Dutch seals. In the adult group both total mercury and selenium levels were significantly higher in brain and liver and bromium in kidney and liver of Dutch seals.

With respect to organochlorines, in the juvenile plus subadult group dieldrin levels in blubber were significantly higher in Dutch seals. In the adult group significantly higher levels for DDT, TDE, dieldrin, α HCH and PCB's were found in blubber of the Dutch seals.

IV. DISCUSSION

1. MERCURY, SELENIUM AND BROMIUM

a. accumulation, tissue distribution and metobolism

Figs. 2-9 show that considerable amounts of mercury, selenium and bromium are already present in organs of very young seals. So transport of these chemicals across the placenta results in accumulation in the embryo during the gestation period. Mercury levels obviously reach a maximum at the age of 5-8 years in liver tissue as well as in brain and kidney. Heppleston & French (1973) concluded that in brain tissue levels reached a maximum at the age of 12-18 months, but concentrations in liver continued to increase with age. This result was not confirmed by any other study. Figs. 2, 3, 4 and 10, show that also in the material surveyed such a phenomenon does not exist. There is a definite positive correlation between the total mercury content in the liver and that in the brain; high levels in the liver coincide with high levels in the brain.

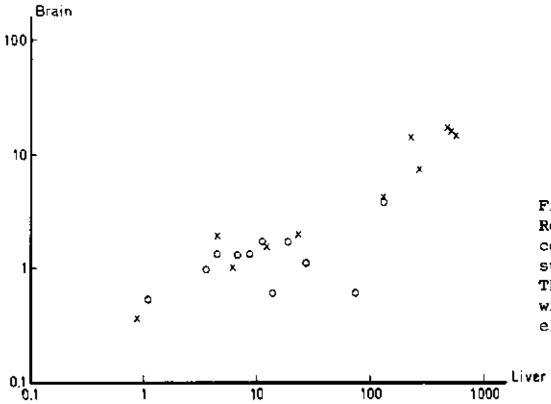


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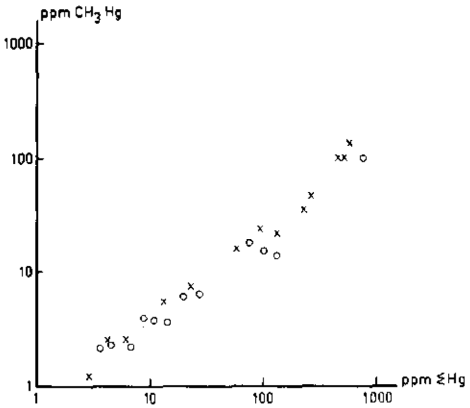
Fig. 10.
Relationship of total mercury (ΣHg) content between brain and liver tissue of harbour seals. Specimens from The Netherlands (X) and from Schleswig Holstein plus Denmark (0). Levels expressed in mg/kg wet weight.

Most of the mercury in seal tissue can be recovered in the inorganic form (figs. 11-15). This is quite remarkable as in fish (the principal food of the seal) most of the mercury is generally present as methylmercury (Westoö & Rydalv, 1971; Berglund et al., 1971; Koeman, 1972, 1975; Rissanen & Miettinen, 1972). This implies that most of the methylmercury taken up by seals is converted into inorganic mercury.

Jones et al. (1976) took blood samples from pups which were caught shortly after birth and fasted for 2 and 3 days and analyzed them for methylmercury. The methylmercury residue level of the fasted pup was twice as high as that observed in a normally fed pup. They conclude that methylmercury penetrates the placental barrier and is already present at a level of 0.5-1.2 ppm at birth. They further showed that only small amounts of methylmercury were passed from mother to pup in milk. Kim et al. (1975) confirm this observation in a study of newly born northern fur seal pups and it has also been reported in humans (Wallace et al., 1971). However, Freeman & Horne (1973) concluded that in harp



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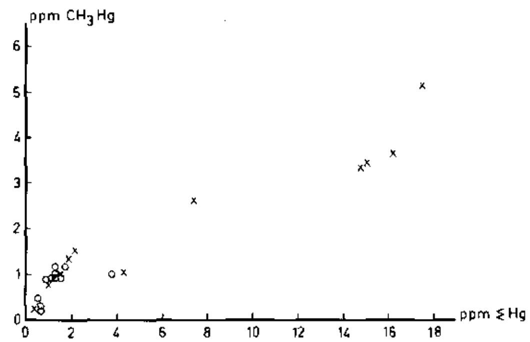

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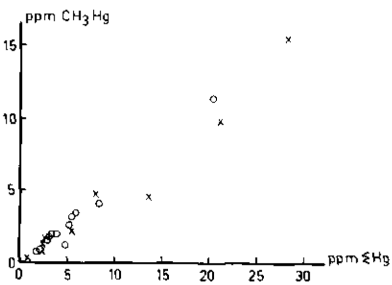

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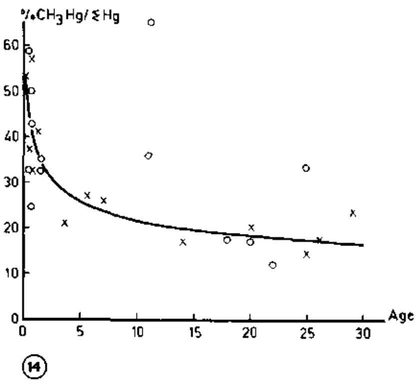
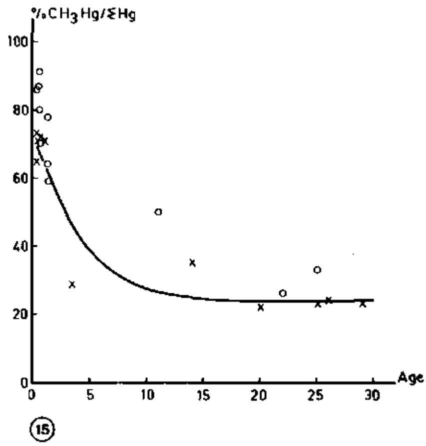
Figs. 11, 12 + 13.
Relationships of methyl mercury (CH_3Hg) and total mercury (ΣHg) content in respectively liver, brain and kidney tissue in harbour seals. Specimens from The Netherlands (X) and from Schleswig Holstein plus Denmark (O). Levels expressed in mg/kg wet weight.





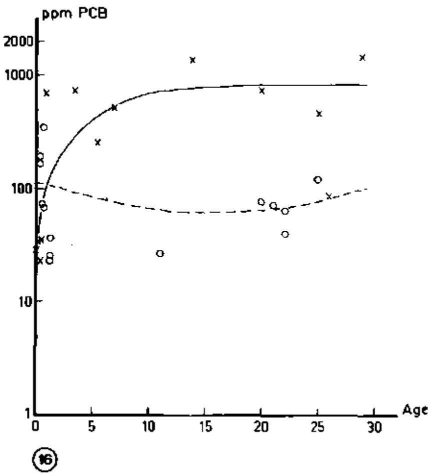
Figs. 14 + 15. Relationships (curves handdrawn) between percentages of methyl mercury per total mercury (% $CH_3Hg/\Sigma Hg$) and age, in respectively liver and brain tissue of harbour seals. Specimens from The Netherlands (X) and from Schleswig Holstein plus Denmark (O). Age in years.

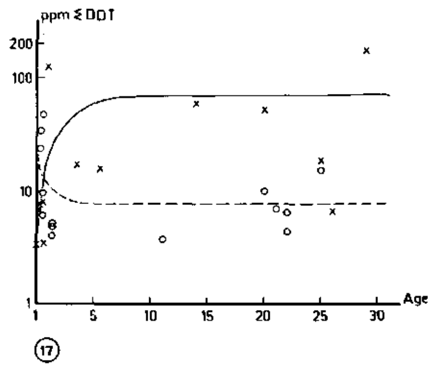
seal pups the stomach content (milk) contained about 70.5% methylmercury. Experiments carried out by Tillander et al. (1972) revealed that methylmercury was excreted rapidly (half-time three weeks) in older seals. Ronald et al. (1977) found evidence of demethylation in both kidney and liver and also at low dosage in the small intestine. There is general support for the hypothesis that in older seals a mercury demethylating mechanism exists and that those processes are likely to take place in the liver and kidney.

In our material, the ratio organic/inorganic mercury in older seals is more or less stable and fluctuates at a level of about 20%. It was already reported by Koeman et al. (1972, 1973) that only 2-14% of mercury occurs in the methyl form in harbour seals. Freeman & Horne (1973) found for several Canadian seals 5.6% of mercury in the methyl form in the liver. Sergeant & Armstrong (1973) found only 5% of mercury as methylmercury in the livers of seals in eastern Canada and Gaskin et al. (1973) reported that mercury in the methyl form varies from 1.8 to 36.6% in the livers of harbour seals from the same area. Anas (1974) stated a value of 2% of the mercury in the livers of Californian sea lions while Holden (1975) found values often as low as 1-2% in livers of seals from coastal waters in north, west and east England. In liver tissue of common seals from West Scotland and East Anglia less than 15% of the total mercury was in the form of methylmercury (Roberts et al., 1976). Also in livers of seals from brackish and fresh waters in Finland only 10% of the total mercury consisted of methylmercury (Kari & Kauranen, 1978). Several authors suggested a demethylation mechanism for mercury ingested with fish. There have been made tentative efforts to identify the nature of the demethylation process. However, Van de Ven et al. (1979) have not been able to clarify whether that transformation process is of an enzymatical, bacteriological or a chemical nature.

b. possible toxicological implications

In a large number of studies the possible effects of mercury on several organisms have been reported (a.o. Bidstrup, 1964; Berglund et al., 1971; OECD, 1974). These reports will not be discussed here but the conclusion from those reports is that it is not possible to indicate to what extent the environmental contamination with mercury contributes to intoxication of seals. Koeman et al. (1972) discovered that high levels of selenium accompanied high levels of mercury in the livers of marine mammals and they postulated possible mutual protection mechanisms of both elements. This conclusion was also based on experimental data presented by Parízek (1969, 1971) and other workers who demonstrated that selenium compounds can protect animals against the toxic action of mercury compounds. In a further report Koeman et al. (1975) suggested that selenium might play a role in that specific mechanism. They found in a large sample of seals and other marine mammals from widely distributed parts of the world an equimolecular relationship for the concentration of mercury and selenium in both liver and brain. Kari & Kauranen (1978) also found in livers of seals in brackish and fresh waters in Finland a 1.16:1 molecular ratio for Hg and Se. However it is still unclear whether selenium really is involved in the demethylating process and/or binding of mercury (Roberts, Heppleston & Roberts, 1976). Experiments with rats have shown that small amounts of selenium compounds given during pregnancy or lactation decreased the passage of





Figs. 16 + 17. Relationships between respectively PCB, total DDT (ΣDDT) and age in blubber tissue of harbour seals. Specimens from The Netherlands (X) and from Schleswig Holstein plus Denmark (O). Levels expressed in mg/kg fat weight. Curves are handdrawn. Age in years.

mercury from the maternal organism into the offspring (Pařízek et al., 1969 and Pařízek et al., 1969). It was also found that administration of mercuric compounds showed the same effect for selenium (Pařízek et al., 1971).

Martin et al. (1976) report about a mercury-selenium-bromine imbalance in premature parturient Californian sea lions. They found atomic ratios of approximately 1Hg:1Se:1Br in livers of normal parturient mothers. In premature parturient mothers the Hg:Se hepatic ratios were also near unity in contrast to Br concentrations which were always severely depressed. In view of those data it was tried to establish from Table 1 atomic Hg:Se:Br ratios for the Schleswig-Holstein plus Danish material, as that population is supposed to be reproducing normally. Considering Figs. 7, 8 and 9 it is clear that the negative results were not quite unexpected. It is concluded from these results that the suggested detoxifying role of bromium, in combination with the mercury-seleniumbalance found in Californian sea lions cannot be supported by the data obtained for this study from harbour seals from the Wadden Sea area.

A general conclusion with respect to the role of mercury in a possible intoxication of the seals is impossible to give. The simultaneous presence of selenium makes it difficult to assess the ecological risks for the harbour seals on basis of the present data available.

2. PCB'S AND DDT

a. accumulation, tissue distribution and metabolism

Most organochlorine compounds, because of their lipophilic nature tend to accumulate in lipid tissues and the highest concentrations in seals also in this study have been found in blubber tissue (see Tables 2 and 3).

As can be seen in Figs. 16-19, especially for PCB's and total DDT, considerable amounts already occur in young seals. As for mercury, this shows again that these contaminants are taken up by the foetus during gestation. In this study two still-born pups were found to contain already respectively 5 and 29 ppm PCB in blubber. Transplacental transfer of organochlorine residues is known to occur in a large number of mammals (Hayes, 1966; Hunter & Robinson, 1967; Robinson & Roberts, 1967; Acker & Schulte, 1970; Clark & Lamont, 1976; Orberg, 1977) and especially in seals (Frank et al., 1973; Gaskin et al., 1973; Jones et al., 1976 and Addison & Brodie, 1977).

The data in Figs. 16 and 17, show that after approximately four years both for Schleswig Holstein plus Denmark and the Netherlands a plateau is reached but at different levels. As seals in both areas consume the same kind of fish (flatfish mainly), these different residue levels must have originated from different levels of contaminants in their diets. This fits well in the statements of Robinson & Roberts (1967), Hunter & Robinson (1967), Hayes & Curley (1968) and Hunter et al., (1969) who studied pharmacodynamics of some organochlorines. They all agree that the amount of residues results from an equilibrium between rate of intake and excretion. The residue levels reach an equilibrium value and this plateau level might increase if the amount of residues in the environment continues to increase.

Holden (1972), Heppleston (1973), Olsson et al., (1975), Drescher (1977) and Harms et al. (1977) found no specific relationships between levels of organochlorines in blubber and age and/or sex.

Frank et al., (1973), Addison et al., (1973), Addison & Smith (1974) and Helle et al., (1976) found a positive correlation of organochlorine levels and age in seals. It also appeared that this was more pronounced in males as in females. Some authors (Gaskin et al., 1971, 1973; Addison et al., 1973 and Addison & Smith, 1974) suggest that females may loose residues through parturition and/or lactation when fat is mobilized from blubber. This might explain the variation in organochlorine residue levels with age and sex they found.

Addison & Brodie (1977) gave some data on organochlorine residues in maternal blubber, milk and pup blubber and they concluded that the loss of the organochlorine burden is balanced by its estimated annual intake of these residues from food. Jones et al. (1976) found no relationship between residue levels in lactating adults and their pups. It can be deduced from data of both Addison & Brodie and Jones et al. that mothers with high levels of organochlorines did not have pups with relatively high levels and reverse. They gave no explanation for this phenomenon. However, this is in contradiction with the results of Olsson et al., (1975), Drescher et al., (1977) and Harms et al., (1977) who did find neither an age accumulation nor a sex difference in organochlorine residue levels. The first report deals with data from a population with quite a low reproduction (Helle, 1975; Helle et al., 1976) whereas the latter ones refer to a fairly stable population with an estimated reproduction of about 30% (Drescher, 1978). In fact, Helle et al., (1976) tested pregnant and non-pregnant (but normal) seals on PCB and DDT content and found no significant differences in residue levels of both groups. Obviously at those high residue levels parturition and lactation may only have a slight temporary effect on the contaminants burden.

Anas (1974) and Harms et al., (1977) stated in their reports that high values of contaminants are probably correlated with feeding behaviour. That is in agreement with the results in this study when feeding behaviour is understood as differences of contamination levels in the food available.

Comparing the means of levels of PCB and total DDT (Table 2) for juvenile plus sub-adult (0-3 years) and adult (4 years and more) seals of Schleswig-Holstein and Denmark there is a decreasing trend when seals grow older. Figs. 16 and 17 show high levels for young seals, less for subadults and an increased level again for adults, which however do not exceed the juvenile levels. This phenomenon has also been observed by Jones et al., (1976).

Apparently the balance between uptake and excretion becomes negative for some period and changes later on. The difference between the German and Dutch seals might be explained by the fact that the uptake of these contaminants by young seals through their diet is lower in the German Wadden Sea area than in the Dutch area. This effect might be reinforced by the increasing lipid content of the blubber tissue in maturing pups. After some years levels increase until a certain plateau is established.

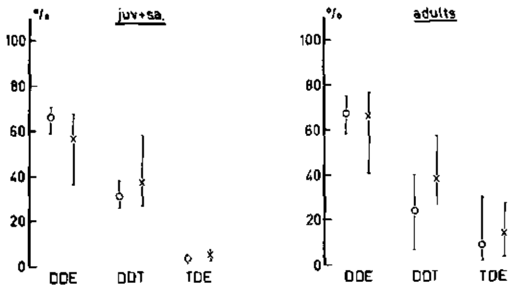


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Fig. 18. Percentages of DDT, DDE, TDE in total DDT content of juvenile plus subadult and adult harbour seals. (X) and (O): means and ranges (vertical bars) of respectively seals from The Netherlands and from Schleswig Holstein plus Denmark.

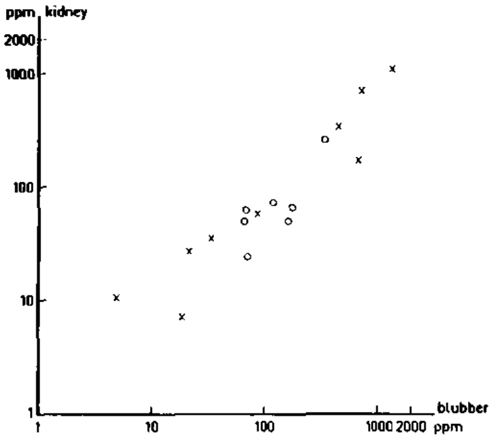
Fig. 18 shows the composition of the DDT group in blubber of juveniles plus subadults and adults from both areas. The main compound is DDE, followed by DDT and a relatively small amount of TDE. The values lie in a range mostly found in a seal blubber by other authors (Johnels, 1970;

Addison et al., 1973; Addison & Brodie, 1977; Drescher et al., 1977 and Olsson, 1977). Much research has been done on the pathway of the degradation of DDT (Bridges et al., 1963; Kallmann & Andrews, 1963; Finnley & Pollmore, 1963; Miskus et al., 1965; Mendel & Walton, 1966; Wedemeyer, 1966; Johnson et al., 1967 and Ko & Lockwood, 1968). There is general agreement about the degradation of DDT either to DDE and thereafter to TDE or directly to TDE. Perry and Hoskins (1950) state a breakdown by enzymes in houseflies and Miskus et al., (1965), Mendel & Walton (1966) and Cliath & Spencer (1972) report a bacteriological breakdown in soil. Also aquatic micro-organisms are able to metabolize DDT to DDE (Cope & Sanders, 1963 and Matsumura et al., 1971). It is not quite certain whether most of the degradation occurs in the predator itself or already in its prey. Frank et al., (1973) and Olsson & Jensen (1975) report of their analyses of fish on the DDT-metabolites. Though their data show great variation in the DDE-DDT-TDE ratios a microbiological metabolisation of DDT was demonstrated. The principal storage metabolite in mammals is DDE (O'Brien, 1967 and Menzie, 1969). In mammals the breakdown of DDT occurs mainly in the liver and to a smaller extent in the intestine (Miskus et al., 1965; Fukuto, 1971 and Jansson et al., 1975). Heath et al. (1970) and Metcalf (1971) give evidence that DDE is less toxic than DDT and suggest that a high DDE percentage might indicate a possible protective mechanism. But Olsson (1977) states that DDE is the main metabolite because of its higher persistence compared to the others.

It was stated before that the highest concentrations of organochlorines are found in the blubber.

Figs. 19 and 20 show the relationships between residue levels of PCB and total DDT in kidney and blubber. The regression equations for DDT and PCB were respectively $ij = 0.41x + 1.6$ ($r = 0.94$) and $ij = 0.78x - 21.3$ ($r = 0.94$). These results confirm the tendency that high PCB and total DDT residue levels occur together, both in kidney and blubber. The highest concentrations are established in blubber tissue.

Several authors report considerable amounts also in kidney, liver, heart, muscle, spleen and brain of seals and porpoises, although kidney and liver ranked next to



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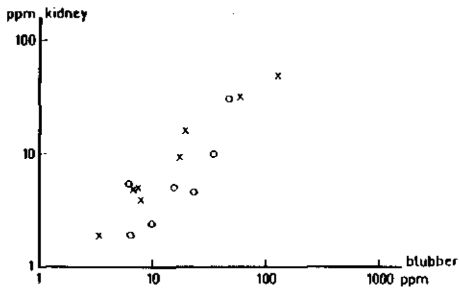

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Figs. 19 + 20.
Relationships of respectively PCB and total DDT (ΣDDT) contents between kidney and blubber tissue of harbour seals. Specimens from The Netherlands (X) and from Schleswig Holstein plus Denmark (O). Levels expressed in mg/kg fat weight.

blubber with respect to residue level (Koeman & Van Genderen, 1966; Holden & Marsden, 1967; Gaskin et al., 1971 and Alzieu & Duguy, 1978). Frank et al., (1973) suggests a blood-brain barrier in harp seals, as in brain tissue total DDT and PCB residue levels on extractable fat basis were much lower than those in blubber and liver. Whether this is correct or not, the differences in PCB and DDT concentrations in brain and blubber are also influenced by the fact that brain tissue consists to a large proportion of phospholipids. PCB's and DDT are less soluable in phospholipids than in normal blubber. Apart from the existence of a blood-brain barrier it is quite possible that lipids in several other organs differ in composition.

Also Holden (1975) reports PCB and total DDT concentrations in blubber, kidney, muscle, heart, spleen and brain on wet weight as well as on extractable lipid basis. He showed that the distribution of organochlorines may not be entirely related to lipid content only.

b. possible toxicological implications

Organochlorine compounds are reported to cause (during in vitro experiments) reproductive failure in fish (Cope, 1963; Burdick et al., 1964), birds (Peakall, 1967 and 1970; Stickel, 1973) and mammals (Barnard & Gaertner, 1963; Korschgen & Murphy, 1967; Gilbert, 1969 and Aulerich et al., 1971).

Chlorinated hydrocarbons as DDT and PCB are known to have effects on the steroid hormone balance, i.e. to induce microsomal enzyme activity in the liver (Pardini, 1971; Villeneuve et al., 1972). This results in an increased rate of hydroxylation - virtually accelerated destruction - of normal body steroids as androgens and estrogens including progesterone and estradiol (Kupfer, 1967; Peakall, 1967; Bitman & Cecil, 1970; Lincer & Peakall, 1970; Platonow & Funnell, 1972; Platonow et al., 1972; Vos, 1972).

The mechanisms by which reproductive processes were affected, vary considerably and therefore only information on mammals with reproductive physiology similar to seals is considered. A review of experimental and epidemiological effects of PCB's and DDT on reproduction in respectively mink and marine mammals is given in tables 4 and 5.

In Californian sea lions increased abortion has been reported by LeBoeuf & Bonnell (1971). This report was followed by studies of DeLong et al., (1973) and Gilmartin et al., (1976). The former group found that in aborting females significantly higher amounts of DDT and PCB compounds were present, compared to normally breeding females. The latter group suggested that besides PCB and DDT contamination also disease agents, i.e. a virus and some bacteria, have to be considered as the cause of premature parturition. Independent studies on high PCB and DDT levels in seals from Swedish waters (Olsson et al., 1974) postulate also effects on reproduction in those seals. A study on the very low reproductive rate among ringed seals in the Northern

REFERENCES	*EFFECTS*	*PCB's*	*DDT*
Aulerich & Ringer,1973	reproductive failure: mating occurred,less whelps born/female	+	-
Aulerich & Ringer,1977	ibid	+	-
Jensen et al.,1977	mating and implantation occurred,number of whelps born/female decreased	+	-

Table 4. Experimental data on effects of PCB's and DDT on reproductive performance in mink. + = effect demonstrated, - = effect not demonstrated.

REFERENCES	*EFFECTS*	*SPECIES*	*PCB's*	*DDT*
LeBoeuf & Bonnell,1971	premature pupping	Californian sea lions	not considered	+
DeLong et al.,1973	ibid	ibid	+	+
Gilmartin et al.,1976	ibid	ibid	+	+
Helle et al.,1976	low reproduction	ringed seal	+	+
Helle et al.,1976	pathological changes in seal uteri:stenosis and occlusions	ringed seal grey seal harbour seal	+	-

Table 5. Epidemiological data on the effects of PCB and DDT compounds on reproduction in different marine mammals.+ = effect demonstrated, - = effect not demonstrated.

Bothnian Bay (Helle, 1975) led to a publication by Helle et al., (1976) inwhich they report that in ringed seals significantly higher levels of PCB and total DDT have been found in the non-pregnant females compared to the pregnant ones. Comparing their findings with data on Californian sea lions they postulate that PCB was the substance responsible. In a later report (Helle et al, 1976) they were able to support their hypothesis by the discovery of pathological changes in seal uteri. The uterine horns of about 40% of a sample of Baltic ringed seals were closed by stenosis and occlusions. The pathological changes were correlated with PCB and DDT levels, because for both biocides significantly higher levels of both DDT and PCB substances were found in seals showing those changes. Again PCB was strongly indicated to be the responsible factor. However, their report was not complete. They tested, both for total DDT and PCB residue levels in extractable fat, the pregnant females versus the non-pregnant ones with stenosis and occlusions and the pregnant females versus the non-pregnant ones with normal uteri. They found significantly higher levels of both PCB and DDT in the non-pregnant group with pathological changes in the uteri and the levels in the normal non-pregnant group were about the same as those in the pregnant group. Unfortunately, they did not test the non-pregnant females with stenosis and occlusions versus the non-pregnant ones with normal uteri. After carrying out the same test as they used (Student's-t-test) it was found that between these groups no significant differences in levels, neither for PCB nor for total DDT existed. This implies that in the non-pregnant group high levels of PCB and total DDT are not linked to the occurrence of pathological changes. This means that it cannot be demonstrated with certainty that non-pregnancy can be solely attributed to the significantly higher levels of PCB and total DDT. Unfortunately, the number of specimens in the non-pregnant group females with normal uteri was rather small compared to the other (8:29) and this may have influenced the standard errors. Otherwise it might be possible that within the non-pregnant group conditions are more favourable to lead to pathological changes. Anyhow, their conclusions about pathological changes caused exclusively by PCB's should be considered more tentative-

(Addison, Kerr, Dale & Sergeant, 1973; Addison & Smith, 1974; Addison & Brodie, 1977; Holden, 1978; Duinker & Hillebrand, 1979).

Some data have been reported on the composition of fatty acids in blubber of seals but little is known of the dynamics (Jangaard & Ke, 1968; Ackman et al., 1971; Ackman et al., 1977). The chemical nature of fatty acids in adipose tissue is in the form of triglycerides. There are continuous processes of uptake and lupolysis from adipose tissue to the blood. Triglycerides are hydroxylated and transfered by the enzyme lipase to the fatty acids and glycerol (lipolysis). A considerable part of the free fatty acids is resynthesized to triglycerides. The relevance of this apparently less economic functioning lies in the regulation of the secretion viz. availability of fatty acids as an energy source by their oxidation and as initial substratum for the biosynthesis of steroids (Prosser, 1973; Newsholme & Start, 1974; Beenakkers, 1978).

In adipose tissue probably at least two lipid-metabolizing compartments seem to exist, one in a constant exchange with the plasma pool and with a high turnover, the other with a very slow turnover as evident from halflife times (Dole, 1961; Stein & Stein, 1962).

Little is known with regard to the distribution of organochlorine biocides stored in each compartment. Nevertheless in adipose tissue there is obviously a permanent turnover and due to this process organochlorines are presumably released. Therefore a continuing interference of organochlorines with the steroid hormone balance will take place.

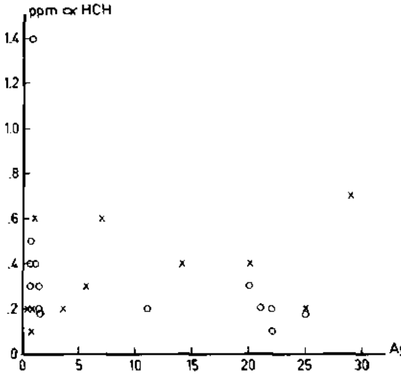
Fluctuations in the steroid hormone balance are met by feed back mechanisms but there always will be a time delay. Therefore it is assumed that the impact of the high PCB levels is especially severe at the end of the implantation delay period in the harbour seal. In a very short period adequate concentrations of circulating hormones (progesteron) have to be available to bring the uterus in the right condition. Contrary to the other phases in the reproductive cycle where feed back mechanisms have due time to react, at least to reestablish adequate concentrations, this might not be possible around the time of implantation. This hypothesis is supported by the experiments of Jensen et al. (1977) who found in mink given different doses of PCB decreased numbers of whelps born per female by an interruption of the pregnancy at an early stage of gestation. The suspected role of PCB in this respect is further strengthened by the discovery that PCB's on a weight basis have about five times the hormone biotransformation potential of DDE (Risebrough et al., 1968).

Besides the presumable direct effects of PCB on the reproductive system it should not be neglected that PCB's are known to have a strong immunosuppressive capacity. In experiments fish (Hansen et al., 1971), birds (Friend & Trainer, 1970) and mammals (Vos et al., 1970; Vos & Van Driel-Grootenhuis, 1972; Vos & de Roy, 1972; Vos, 1977) have shown an increased susceptibility to diseases. Gilmartin et al. (1976) gives evidence for an interrelationship of environmental contaminants and disease agents. Drescher (1978) reports skin lesions found on the harbour seal in the Wadden Sea and discusses its causes. He states that one of the important factors involved is disturbance during the first weeks of the seal's life, resulting in inflammation in the umbellical region. Considering the rather high residues of PCB's already present in those pups it is reasonable to assume that due to their reduced immunity they are more vulnerable to infection after mechanical injury.

3. OTHER COMPOUNDS STUDIED

For α + β HCH, hepo and dieldrin residue levels in blubber of seals from both areas no relation with age was found (Figs. 21, 22 and Table 2). All values are of the same order as found in other recent studies (Clausen & Berg, 1975; Drescher et al., 1977).

Chlorinated dibenzofurans are impurities in the manufacture of PCB's. These extremely toxic chemicals are capable of causing teratogenic effects at low levels. In the literature surveyed in this study was found that until now, these residues have only been detected in commercial PCB-mixtures and not in marine organisms (Vos et al., 1972; Zitko, 1972 and Bowes et al., 1973).



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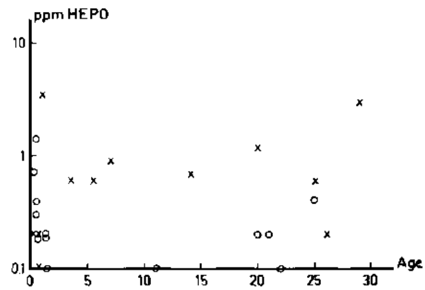

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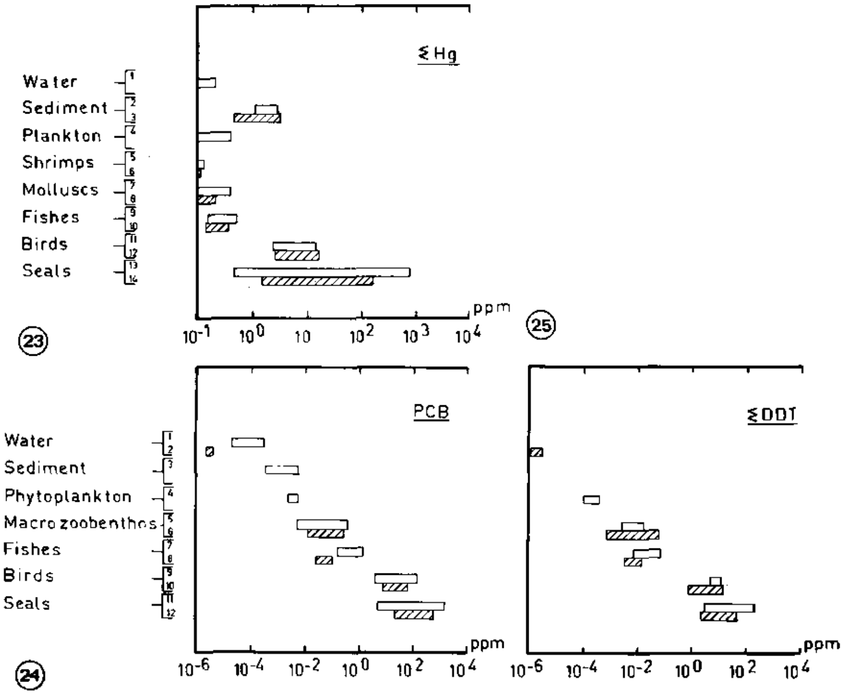
Figs. 21 + 22. Relationships between respectively α HCH, HEPO and age in blubber tissue of harbour seals. Specimens from The Netherlands (X) and from Schleswig Holstein plus Denmark (O). Levels expressed in mg/kg fat weight. Age in years.

4. ROUTES OF TRANSPORT IN THE MARINE ENVIRONMENT

As discussed before, different levels of contaminants in tissues of nearly equally old seals, have to a large extent to originate from differences in body-burden of their food. Via biomagnification during the transport of contaminants from prey to predator, levels of these substances show considerable increase in food chains.

A higher position in the food chain is not indissolubly related to higher concentrations in the organisms (Jernelöv & Lann, 1971; Knauer & Martin, 1972; Olsson, 1976, 1977). Levels of accumulated substances are the net result of species-specific processes of intake, degradation and excretion influenced by abiotic factors as well (Robinson et al., 1967; Olsson, 1977; Cember et al., 1978). However, residues of organochlorines and heavy metals tend to be greater in marine organisms of the higher trophic levels. This is demonstrated for total mercury, PCB and total DDT by data available from literature on the analyses of biota from the Dutch, German and Danish Wadden Sea areas (Figs. 23, 24, 25).

Presence of organochlorines and metals has been reported in marine food chains - including man, from the Arctic, Subarctic and Antarctic (Tatton & Ruzicka, 1967; Addison & Smith, 1974; Bowes & Jonkel, 1975 and Charlebois, 1978) as well as from industrialized areas (Holden & Marsden, 1967; Risebrough et al., 1968; Jensen et al., 1969; Ackefors et al., 1970; Nuorteva, 1971; Jones et al., 1972; Williams & Weiss, 1973; Frank et al., 1973; Anas, 1974; Andersen & Neelakantan, 1974; ICES, 1974; Bühler et al., 1975; Wright, 1976, Koeman & Stasse-



Figs. 23, 24, 25. Prevailing ranges of total Hg (ΣHg), PCB and total DDT (ΣDDT) contents in biota of several trophic levels in marine food chains. Levels for Hg espressed in mg/kg wet weight; PCB plus ΣDDT, up to macrozoobenthos, expressed in mg/kg wet weight, for fishes, birds and seals PCB and DDT values expressed in mg/kg fat weight. Hatched colums: Schleswig Holstein plus Danish specimens, open colums: Dutch specimens.
References: total Hg, 1: Fonds(1971); 2: de Goey et al. (1971), de Groot et al. (1971); 3: Gadow & Schäfer (1974), Müller & Förstner (1974), Dörjes et al. (1976); 4: Koeman et al. (1971); 5: Koeman et al. (1971), Quirijns et al. (1975); 6: Günther et al. (1972), Haar (1974), Quirijns et al. (1975); 7: Koeman et al. (1971), de Wolf (1975); 8: de Wolf (1975), Karbe et al. (1978); 9: Koeman et al. (1975), Quirijns et al. (1975); 10: Günther et al. (1972), Harms (1975); 11: Koeman et al. (1971); 12: Vauk et al. (1979); 13: Koeman et al. (1972), present study; 14: Schmid et al. (1976), Drescher et al. (1977), present study.
PCB and total DDT, 1: Duinker (1974), Duinker (1976); 2: Stadler & Ziebarth (1975); 3: Duinker (1974); 4: ten Berge & Hillebrand (1974); 5: ten Berge & Hillebrand (1974), Quirijns et al. (1975); 6: Schaefer et al. (1976); 7: ten Berge & Hillebrand (1974), Quirijns et al. (1975); 8: Buschenbeth (1973), Schaefer et al. (1976); 9: Koeman et al. (1972); 10: Figge et al. (1976); 11: Koeman et al. (1972) present study; 12: Drescher et al. (1977), present study.

Wolthuis, 1978).

Although aerial transport of metals and organochlorines occurs even over long distances (Abbot et al., 1966; Risebrough et al., 1968; Tarrant & Tatton, 1968; Goldberg, 1970; Södergren, 1972; Young et al., 1976) the main route of pollution in the Wadden Sea is by riverine transport. Approximately 10% of the water and 50% of the suspended matter supplied by the highly polluted Rhine river to the North Sea enter the Dutch Wadden Sea. Several authors have reported on the transport of heavy metals and organochlorines from the Rhine to the Wadden Sea (de Groot et al., 1971; Weichert, 1973; Duinker et al., 1974; Müller & Förstner, 1975; Duinker & Nolting, 1976; Poels et al., 1978). Koeman (1971) and Quirijns et al., (1979) reported on decreasing levels of chlorinated hydrocarbons in mussels going along the coast from the Rhine estuary to the eastern part of the Dutch Wadden Sea. De Wolf (1975) analyzed mercury content of mussels from West European coasts. He found the highest levels to occur in mussels taken from the vicinity of the Rhine, Ems and Dollard which implies not a complete west-east trend but regional differences.

The obviously different rates of accumulation of organochlorines and heavy metals in marine organisms was confirmed in figures 23, 24 and 25. The analyses shown in these figures have been carried out during several years by several institutes and consequently mostly different techniques. Because of differences caused by only the latter factor as well as the fact that in some cases only few specimens were analysed, the data are given as ranges without means or standard deviations. For the different trophic levels even these rough data already indicate a more pronounced increase for PCB and total DDT residue levels compared to total mercury. Besides, these figures indicate for PCB and total DDT in fishes obvious differences between the Schleswig Holstein plus Danish Wadden Sea on one hand and the Dutch Wadden Sea on the other whereas these differences are absent with respect to total mercury. Consequently this would result in differences for PCB and total DDT residue levels in seals originating from both areas, which was confirmed by statistical tests, discussed earlier.

Olsson et al., (1975) found in Baltic seals for PCB an accumulation ratio of 7 to 9 and for total DDT a ratio of 10 to 12. Also Holden & Marsden (1967); Jensen et al., (1969) and Frank et al., (1973) when going from fish to seals found for PCB and total DDT a tenfold increase.

The accumulation ratios (content in seal/content in fish) (in both extractable fat) for PCB and total DDT in Schleswig Holstein plus Denmark are respectively about 10 and 4 and in the Netherlands about 30 and 50. However data on organochlorine residue concentrations in fishes of the Schleswig Holstein plus Danish and Dutch areas are highly diverse and of an incidental character. They are too limited to provide proper information on the contents of compounds in the diets of seals over a long period of time. Therefore they are too small to prove that numerical differences in accumulation ratios are statistically significant. It has to be stressed that the accumulation ratios given have to be considered just as indications; for reliable, accurate data much more analyses would be required. For total mercury it can be deduced from data given by Heppleston & French (1973) and Roberts et al., (1976) that the accumulation ratios (content in seal liver/total fish) were at least one hundred. In the Schleswig Holstein plus Danish on one hand and the Dutch seals on the other these ratios were respectively about 1000 and 1500.

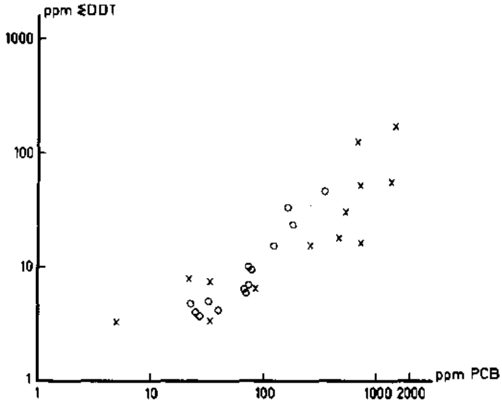
5. POSSIBLE INTERRELATIONSHIPS BETWEEN THE CHEMICALS STUDIED

There is a tendency for high PCB and high DDT residues to occur together as can be deduced from fig. 26. The regression equation for PCB and total DDT concentrations in blubber is PCB (ppm) = 12.5 total DDT (ppm) - 36.3; the correlation coefficient amounts to 0.81. Also Holden (1972) and Drescher et al., (1977) found a similar relationship. Holden (1972) considered it to be coincidental but the results of Drescher et al., (1977) and this study are in such a close range (Drescher et al. found the equation: PCB (ppm) = 16.1 total DDT (ppm) + 5.27) that this explanation does not fit here. Considering ratios of PCB and total DDT found in other areas the tendency of high values occurring together generally exists but the ratios are different, i.e. total DDT residues exceed the PCB values in these areas.

Total DDT/BCB ratios in blubber of seals were e.g. 1.2-9.2 (Johnels, 1970); 1.0-6.4 (Frank et al., 1973) and 0.5-2.4 (Addison et al., 1973). For Californian sea lions Delong et al., (1973) found a mean ratio of total DDT/PCB in blubber tissue of 6.6. Considering the geographical distribution of both organochlorines the highest PCB levels are found in areas with high industrialization i.e. South-eastern Canada, east and west coast of Great Britain, Wadden Sea area and the Baltic. The PCB levels are low in the Arctic both on the Canadian and the Norwegian side.

High DDT levels also indicate great human activity but of a different origin. Whereas PCB is mostly discharged by effluents of industries, DDT was generally used as agricultural treatment, especially in forested areas. Therefore in e.g. Scandinavian countries and around the Gulf of St. Lawrence due to the use of DDT in forest insect spraying, and in the Baltic area also due to the enormous agricultural activities in the German Democratic Republic and Poland, the total DDT concentrations greatly exceed the PCB concentrations, contrary to areas as S.W. England, Wales and the Wadden Sea. This opinion is supported by studies of Nicholson (1967), Edwards (1970), Zitko & Choi (1971) and Sanderman (1974) who generally agree that sources of pollution through PCB are industrial wastes and for DDT surface runoff after agricultural use.

The similarity in DDT/PCB ratios (fig. 26) in seal blubber from the German, Danish and Dutch Wadden Sea areas also suggests a distribution achieved by an uniform national c.q. regional use of those contaminants and identical source(s) by which the seals got contaminated. This opinion is supported by data of ICES (1974) showing that both organochlorine pesticide residues and PCB's in fish and shellfish are higher in the Southern Bight of the North Sea than in the German Bight. Besides, Husschenberg (1972) found in fresh water fish (especially those from the Mosel) higher amounts of total DDT and PCB's than in sea fish. However the strongest support is given by Duinker & Hillebrand (1979) who



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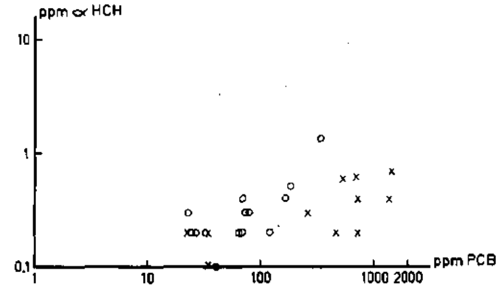
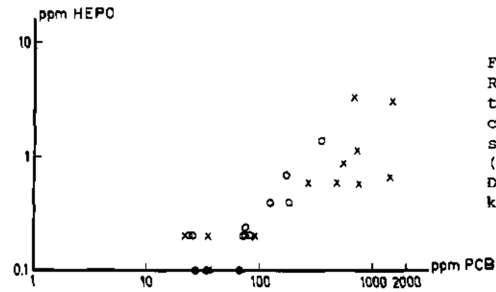


(27)



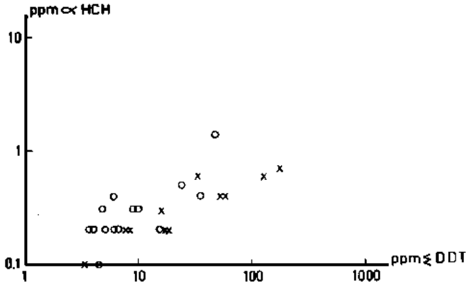
(28)

Figs. 26, 27, 28.
Relationships between respectively total DDT (ΣDDT), α HCH, HEPO and PCB contents in blubber tissue of harbour seals. Specimens from The Netherlands (X) and from Schleswig Holstein plus Denmark (O). Levels expressed in mg/kg fat weight.

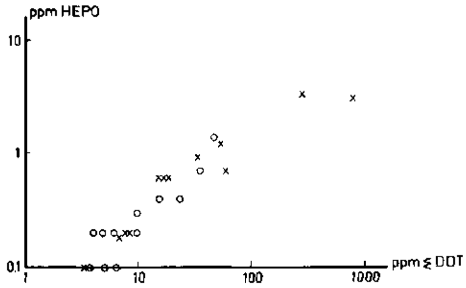
carried out organochlorine analyses of sea water during a survey over the total Wadden Sea, from the Rhine estuary to Esbjerg in Denmark. They state that the contributions of the German and Danish rivers to the PCB and DDT pollution in the Wadden Sea is very small compared to the input by the river Rhine.

Peakall & Lincer (1970); Moilanen & Crosby (1973) and Maugh (1973) report of a mechanism by which DDT is converted under influence of uv-light into PCB's. Though their theory did not get much support so far (Zepp et al. 1977) it anyhow makes the establishment of the DDT/PCB ratios still more complicated.

For the other organochlorines also the interrelationships of the respective residue levels in seal blubber were studied; these are demonstrated in Figs. 27-31.

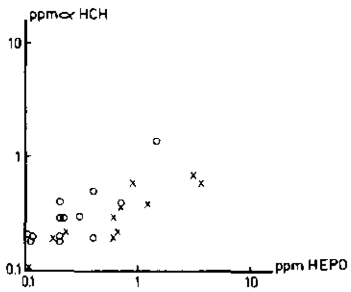


(29)



(30)

Figs. 29 + 30.
Relationships between respectively α HCH, HEPO and total DDT (ΣDDT) contents in blubber tissue of harbour seals. Specimens from The Netherlands (X) and from Schleswig Holstein plus Denmark (O). Levels expressed in mg/kg fat weight.



(31)

Fig. 31. Relationship between α HCH and HEPO content in blubber tissue of harbour seals. Specimens from The Netherlands (X) and from Schleswig Holstein plus Denmark (O). Levels expressed in mg/kg fat weight.

Every combination (Hepo versus α-HCH versus PCB versus total DDT) showed a tendency identical to PCB and total DDT; high values occurring together. Since it can be deduced from these Figs. that PCB and total DDT concentrations compared to the other contaminants are higher in the Dutch seals than in those from Schleswig Holstein plus Denmark, it seemed not correct to pool the data of both areas in calculating regression equations. In Table 6, for different organochlorines the correlation coefficients are given separately for Schleswig Holstein plus Denmark and the Netherlands. It can be deduced from this table that the best correlations for PCB versus the other ones are found in the German plus Danish material but for total DDT versus Hepo and versus α-HCH there are no considerable differences.

The differences in blubber residue levels between seals from Schleswig Holstein plus Denmark on one hand and Dutch seals on the other are tested on their significancy for all organochlorines and will be discussed later on.

6. COMPARISON OF RESIDUE LEVELS IN SEALS ORIGINATING FROM DIFFERENT PARTS OF THE WADDEN SEA

a. mercury, selenium and bromium

The significantly higher total mercury and selenium levels (Table 7) in juvenile and subadult seals from Germany plus Denmark are already reversed in the adult group, i.e. significantly higher in Dutch seals. This implies that the possible effects of both contaminants with respect to the reproductive capacity of both populations have to be considered in the adult group. Comparing the levels of total mercury in kidney and brain of the proportionally higher contaminated Dutch seals with those obtained by Ronald et al. (1977) in their experiments, the levels found in this study are still far below those levels where Ronald et al. did not yet find any clinical signs of neurological, renal and hepatic dysfunction. The combined higher residues of both mercury and selenium in brain and kidney tissue of the Dutch seals was not quite unexpected because of the already earlier mentioned equimolecular Hg-Se relationship.

	Schleswig Holstein + Denmark	The Netherlands
PCB - total DDT	0.96	0.77
PCB - alpha HCH	0.90	0.67
PCB - HEPO	0.94	0.61
total DDT - alpha HCH	0.84	0.84
total DDT - HEPO	0.95	0.94
alpha HCH - HEPO	0.92	0.82

Table 6. Correlationcoefficients of lineair relationships between residues in blubber of Schleswig Holstein plus Danish (n=14) and Dutch seals (n=12).

In liver the greatest amounts of mercury and selenium are accumulated and in the liver tissue no significant difference in residue levels was found between the Schleswig Holstein plus Danish seals on one hand and Dutch seals on the other. In view of this result and the formerly mentioned findings of Ronald et al. (1977) it is assumed that with respect to the decreased reproduction in the Dutch population the mercury and selenium contamination observed in this study is of minor importance.

The expected suppression of bromium residue levels in the Dutch seals (discussed above) was not confirmed; on the contrary both bromium and PCB residue levels are significantly higher in the Dutch seals compared with the Schleswig Holstein plus Danish ones.

b. organochlorines

PCB levels in blubber of adult seals from the Dutch Wadden Sea are significantly higher than those in seals from the Schleswig Holstein and Danish area. This is shown in fig. 32 and statistically tested (Table 8).

The multivariate analyses of the DDT group revealed significantly higher levels for DDT and TDE in the Dutch seals (Fig. 20 and Table 8).

With respect to the possible effects of the total DDT group this finding is less important as the DDE metabolite is the most prevalent one and moreover both for total DDT and DDE no significant differences have been determined. The presence of DDT (Tables 2 and 3) implies that in both areas DDT uptake has happened rather recently.

The observed higher amounts of dieldrin and α-HCH in the blubber tissue of the Dutch seals are of the same order of magnitude as Drescher, et al. (1977) found in seals of a stable population, with normal reproduction percentages. The rather low residue levels of these contaminants obtained in this study for both areas therefore seem of less importance when compared to the striking difference in PCB-contamination burden for adult seals.

	ΣHg		Se		Br	
	j.+s.a	ad.	j.+s.a	ad.	j.+s.a	ad.
brain	–	☆☆	☆☆(1)	☆☆	–	–
kidney	☆☆(2)	☆	☆☆	☆	–	☆☆
liver	–	–	–	–	☆☆	☆☆

Table 7. Statistical tests on significant differences in total mercury (ΣHg) , selenium (Se) and bromium (Br) contents in juvenile plus subadult as well as adult harbour seals from Schleswig Holstein plus Denmark versus The Netherlands. ☆☆☆: $p < 0.01$; ☆☆: $0.01 < p < 0.05$; ☆: $0.05 < p < 0.10$; — : not significant; (1+2): contents significant higher in Schleswig Holstein plus Danish seals. (Tests used: Wilcoxon, Students t-test and de Weir's standardized t-test).

	ΣDDT	DDT	TDE	DDE	DIELD.	αHCH	HEPO	PCB's
juveniles + subadults	–	–	–	–	★★★	–	–	–
adults	–	★★	★	–	★★★	★★	–	★★★

Table 8. Statistical tests on significant differences of organochlorines contents in blubber of juvenile plus subadult as well as adult harbour seals from Schleswig Holstein plus Denmark versus The Netherlands.
★★★ p< 0.ok; ★★: 0.01< p; ★: 0.05 p <0.10; —: not significant. (Tests used: Wilcoxon, Students t-test and de Weir's Standardized t-test).

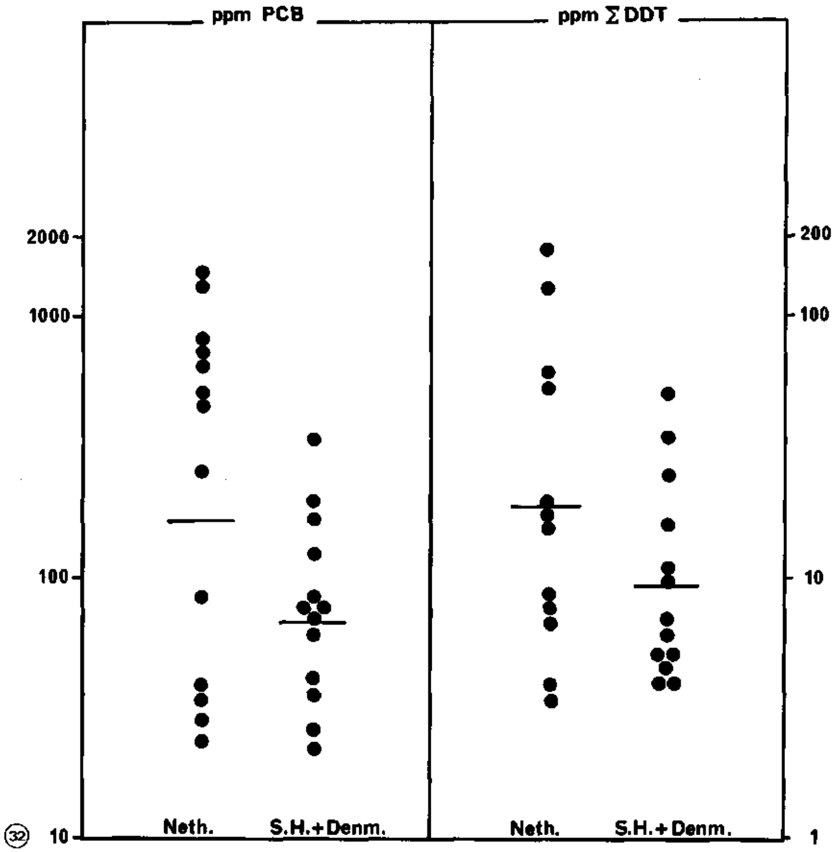


Fig. 32. Concentrations of PCB and total DDT in blubber of harbour seals from Schleswig Holstein + Denmark and The Netherlands. Concentrations in ppm, on lipid weight; ——— geometric means.

c. other contaminants

In this study no analyses on cadmium, arsenic and lead have been carried out. Comparing data available from literature reveals no differences between levels found in tissues of seals from Schleswig Holstein plus Denmark on one hand and from the Netherlands on the other (Koeman et al., 1972; Drescher, 1976 and Van Haaften, pers. comm.).

VII. CONCLUSIONS

Considering the several residue levels in tissues of Schleswig Holstein, Danish and Dutch seals it is concluded that without neglecting the other high pollutant levels, the highly significant difference found for PCB is the most remarkable finding.

Comparing the results of 1) a decreased seal population caused by diminished reproduction, 2) in the total Wadden Sea seal population a west-east trend in increasing reproduction percentages, 3) an inverse trend in contamination of the total Wadden Sea, mainly caused by input from the river Rhine and the diluting effect, going east, of the North Sea and without ignoring the other pollutants studied the conclusion can be drawn that the observed decrease in the reproductive success of the Dutch seal population correlates strongly with an elevated concentration of PCB's in the tissues. Considering the existing epidemiological and experimental data indicative for the ability of these compounds to inferfere with mammalian reproduction, the present finding supports the hypothesis that PCB's might be responsible for the low rate of reproduction in the Dutch seal population. However, it should be recognised that the symptoms observed are not similar to the symptoms observed with reproductive failure found elsewhere e.g. in the Bothnian Gulf.

Within the limits of the material available it is assumed that either implantation does not occur or is shortly followed by resorption or abortion.

VIII. SUMMARY

Since 1950 the harbour seal population in the Dutch Wadden Sea has decreased significantly. Study of the population dynamics revealed that pup production in the Dutch population was lower compared to the more stable population in Schleswig Holstein, Germany. The observed west-east trend of increasing pup production was correlated with a possible inverse trend of residue levels in tissues of seals from the western part of the Wadden Sea (The Netherlands) and the north-eastern part (Schleswig Holstein and Denmark).
Deadstranded animals were collected in both areas and blubbe

liver, brain and kidney were analyzed for PCB; o,p'-DDT; p,p'-(DDT + DDE + TDE); dieldrin, aldrin; endrin; endosulfan; α, β, γ, -HCH; HCB, QCB (pentachloro benzene); HEPO; total mercury and methyl mercury; selenium and bromium.

The amount of residue levels already present in stillborn pups indicates transplacental transport of all organochlorines and metals analyzed.

Age specific increase was found for mercury and selenium in liver and brain tissue of seals both from Schleswig Holstein plus Denmark and the Netherlands. The highest ratios of methylmercury/total mercury were found in juvenile seals decreasing to a certain plateau level in older ones.

An equimolecular relationship was found for mercury and selenium but not for bromium and mercury and bromium and selenium.

For PCB and total DDT an age specific accumulation in residue levels was found in Dutch seals whereas this phenomenon failed in the Schleswig Holstein plus Danish specimens. The main metabolite in the DDT family was DDE.

A clear cut correlation could be demonstrated for mercury and selenium in liver and brain tissue and for PCB and total DDT in blubber and kidney. Deposition of contaminants was generally higher in older seals, high levels of all compounds except bromium, occurring together. Tests on differences in residue levels between Schleswig Holstein plus Denmark on one hand and the Netherlands on the other revealed that especially PCB levels were significantly higher in Dutch adult seals.

Without ignoring the other pollutants studied the conclusion can be drawn that the observed decrease in the reproductive success of the Dutch seal population correlates strongly with an elevated concentration of PCB's in the tissues. Considering the existing epidemiological and experimental data indicative for the ability of these compounds to interfere with mammalian reproduction, the present finding strongly supports the hypothesis that PCB's might be responsible for the low rate of reproduction in the Dutch seal population. However, it should be recognised that the symptoms observed are not similar to the symptoms observed with reproductive failure found elsewhere e.g. in the Bothnian Gulf.

Within the limits of the material available it is assumed that either implantation does not occur or is shortly followed by resorption or abortion.

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MANAGEMENT AND CONSERVATION OF THE HARBOUR SEAL, *PHOCA VITULINA*, POPULATION IN THE INTERNATIONAL WADDEN SEA AREA

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ABSTRACT

Population assessment of the harbour seal in the international Wadden Sea Area has been carried out. The results show that the populations in the different areas have to be considered as one entity. Therefore management and conservation measures have to be placed in an international context. Because of the low pup production in the Dutch area the survival of a harbour seal population in the Dutch Wadden Sea depends on migration from the adjacent areas as long as adverse environmental factors i.e. pollution and disturbance are not under control.

1. INTRODUCTION

The Wadden Sea is located along the coasts of The Netherlands, Germany (Niedersachsen and Schleswig Holstein) and Denmark (Fig. 1). It is a part of the North Sea separated by a row of barrier islands and characterized by tidal flats.

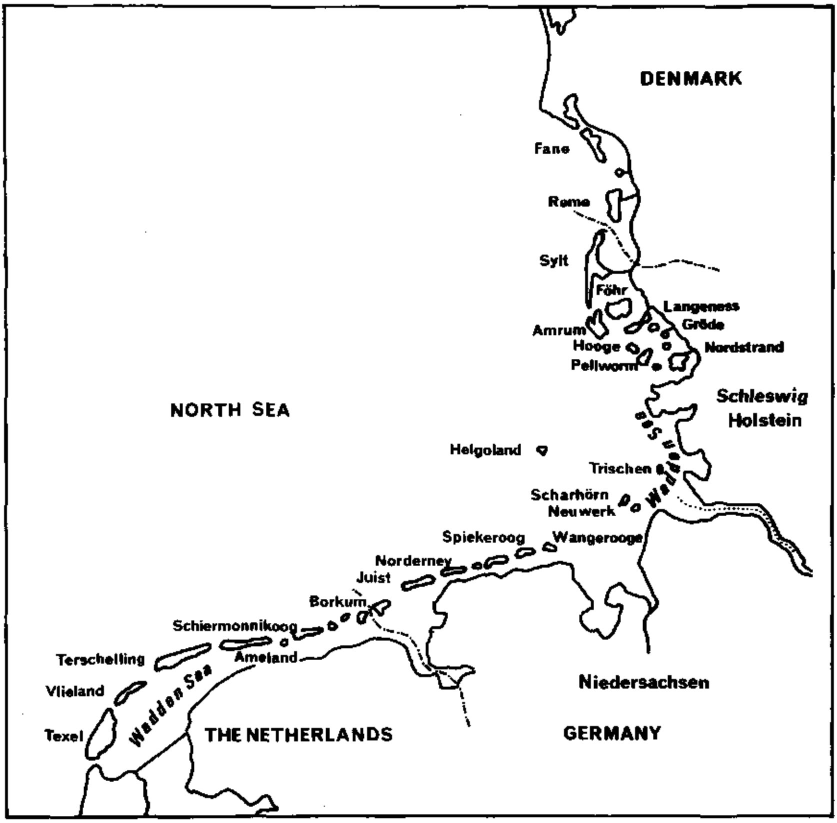

Fig. 1. Map of the international Wadden Sea and the adjacent countries.

The harbour seal *(Phoca vitulina)* in this area has decreased drastically during the last decades, especially in The Netherlands observed and reported by Van Haaften (1974) and Reijnders (1976). As seals do not acknowledge state boundaries it is rather obvious to consider the stock of seals as one population with respect to conservation and management strategies. This has been done by population assessment and evaluating its potential threats.

2. POPULATION ASSESSMENT

a. size

The number of seals counted in the Wadden Sea fluctuates throughout the year (Wipper, 1974; Reijnders, 1976, 1978). At the end of July the highest numbers are counted on the tidal flats. The number of pups born per mature female fluctuates each year and therefore the maximum number of subadults and adults should be taken and regarded as an important character when comparing population sizes in subsequent years.

Censuses of the different Wadden Sea (sub)-populations have been carried out in the past mostly by boat and only since about 1960 in each of the four Wadden Sea countries by plane. As boat censuses nearly always provide underestimates and take more time to cover the whole area only aerial surveys are considered here. Unfortunately seals are counted in The Netherlands until 1974 around mid-September and at that time numbers present on sandbanks already decrease (Wipper, 1974; Reijnders, 1978; Drescher, 1979). Therefore these numbers have to be revised to compete the apparent underestimates. Because the mentioned decrease does not occur each year at the same time and at the same rate, a mean converting factor was calculated based on the data from 1974 till 1978 (Reijnders, 1978). These more correct numbers have been added to the data available for Niedersachsen (Wipper, 1974 and Manninga, pers. comm.), Schleswig Holstein (Behnke, 1974 and Drescher, 1979), Denmark (Søndergaard et al., 1976 and Drescher, 1979) and Helgoland (Vauk, 1973 and pers. comm.) resulting in fig. 2. It is clear from this figure that the total population

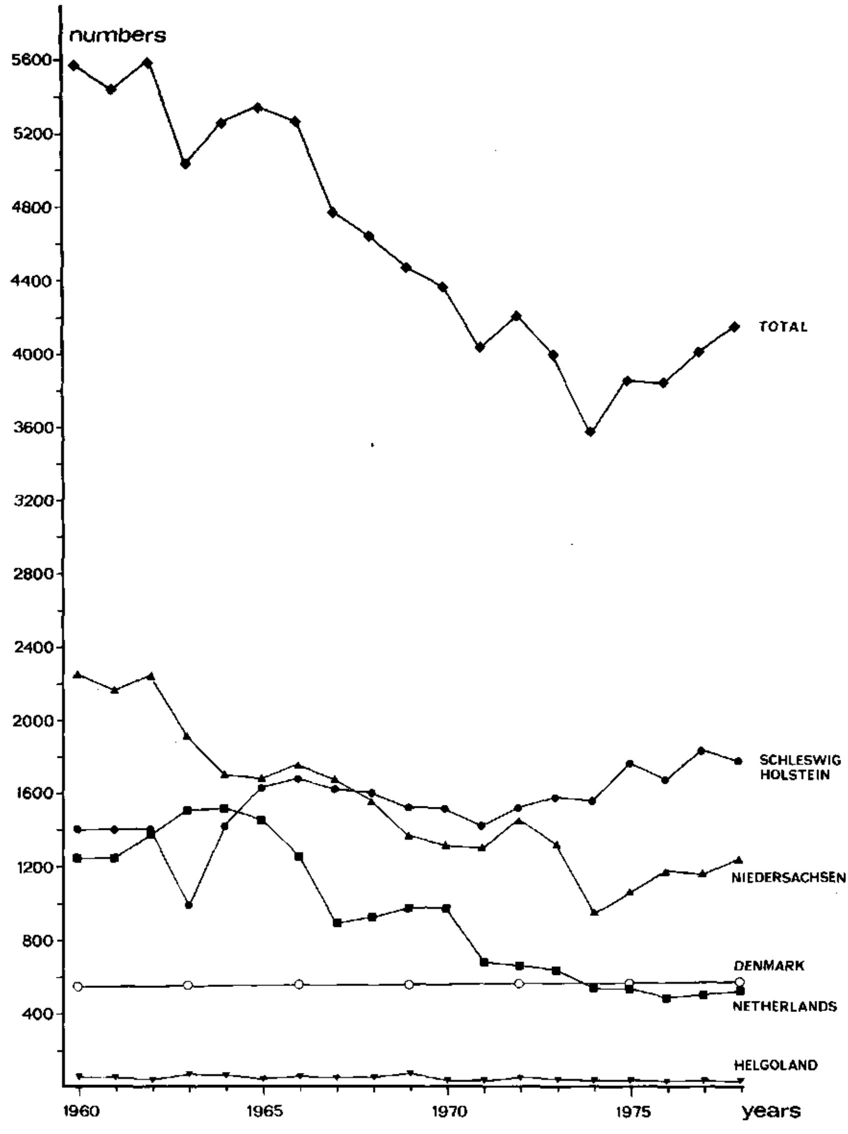

Fig. 2. Total numbers of harbour seals (incl. juveniles) in different areas of the Wadden Wadden Sea and for the whole area. The numbers are calculated on surveys carried out at the end of July each year.

tendency to disperse into all directions from the point of tagging. Drescher (1979) states this to be unexpected as generally the harbour seal is thought to be rather stationary. However it is well known in many species that especially young animals tend to disperse. One of the explanations that migration among young marine mammals is quite common (Wada, 1969; Johnson, 1975). Older seals on the contrary, are assumed to be stationary as for years the same sand banks are chosen as resting, whelping and nursing sites. Apart from some banks which are transformed by heavy currents, nearly every year the very same spots on specific, probably suitable sandbanks are preferred. Drescher (1979) even states that certain age classes prefer some specific sandbanks but agrees that till now, it is unknown whether the same seals return to the same places after winter time.

Mohr (1952) and Van Haaften (1974) concluded that the harbour seal leaves the Wadden Sea during winter. This conclusion is based on the fact that during winter time much less seals are counted on the tidal flats than during summertime. The question arises whether this conclusion is correct. It may well be possible that seals stay in the water during winter and are not counted for that reason. The needs to haul out, - as whelping, nursing, moulting and may be some social functions - are met all during summer months. Furthermore seals are capable of sleeping and resting in the water for months (Wada, 1969; Wipper, 1975 and Drescher, 1979). Therefore we assume that in wintertime seals spend time on sandbanks only when weather conditions are favourable. Under this assumption seals are still present in the Wadden Sea but not all of them are observed because they stay in the water. This assumption is supported by monthly aerial surveys by the author during October till May, 1977-1978. More than half of the maximum numbers present on the tidal flats in that past summer were observed during these surveys. Two further independent data strengthen the assumption. The first is an observation by the author on two subsequent days in February. The same area in the Wadden Sea was visited. The first day only 6 animals were observed, the water and air temperature were both 2^{o} C. The next day 28 specimens were observed, the water temperature naturally still at 2^{o} C but a bright sun warmed up the air temperature to nearly 8^{o} C. The second is a statement by Vauk (1973) that seals staying on the rocks at the east side of Helgoland do not haul out for weeks during stormy weather and return immediately when the sea is calm again. These data indicate that most seals, although not observed in aerial surveys, during winter time still stay in the Wadden Sea but generally aggregate to sand banks with ready access to the North Sea. Only during ice-forming in the Wadden Sea they disappear to the North Sea. This has beer noted before but also in the past winter the author observed that within a week after icefloes drifted to the North Sea and/or melted, an equal number of seals to that before the ice came, was already present again. So the animals must have stayed in the North Sea but relative close to the Wadden Sea. This aspect will be studied in more detail by means of telemetry.

3. CAUSE OF FLUCTUATIONS

a. hunting

Seals have been persecuted by man for centuries. In all of the countries adjacent to the Wadden Sea hunting pressure has been rather heavy as can be deduced by bounty statistics as discussed by Wipper (1974), Drescher (1975) and Reijnders (1976). Especially in the Dutch Wadden Sea the collapse in numbers of seals was initially caused by an uncontrolled hunt for only juvenile seals (Reijnders, 1976). Hunting was definitely prohibited in The Netherlands in 1962, in Niedersachsen in 1971, in Schleswig Holstein in 1973 and in Denmark in 1977. Hunting did not seem to effect the population in Schleswig Holstein as these group of seals stayed fairly stable over the last years. Drescher (1979) already suggested, without giving evidence, that this is not true because the actual decrease may only be visible in those regions where conditions are suboptimal. Evidence however for the impact of hunting in e.g. Schleswig Holstein on the total population can be deduced from data on dispersal of juveniles given by Wipper (1974) and Drescher (1979). Assuming that tagging does not influence the dispersal behaviour of juveniles and that the chances of tag recovering are the same in each country, tagging data show that a balance must have existed between the numbers of seals that moved eastwards and westwards before the severe decrease started. The authors mentioned gave data from which it is possible to calculate that both from the Schleswig Holstein and the Niedersachsen population about 15-20% of juveniles move westwards and the same quantum north-eastwards. Van Haaften (pers. comm.) states that the same percentage from the Dutch population moves eastwards. This means that in the past about an equal number of juveniles moved from Schleswig Holstein to Niedersachsen and inversely. The same holds for Niedersachsen and The Netherlands. Because pup production in The Netherlands decreased, more seals moved from Niedersachsen to The Netherlands. This caused, together with the hunting pressure, a decrease in Niedersachsen although at a slower rate than in The Netherlands.

Since hunting was stopped in Niedersachsen and Schleswig Holstein the population in the Dutch Wadden Sea decreased at a much slower rate and in fact remained fairly stable (Reijnders, 1978). It must be stressed however that this "improvement" is only due to the fact of an immigration from Niedersachsen and probably to a less extent from Schleswig Holstein as was proved by Reijnders (1978) by assessing age class composition in the Dutch population. So it is clear that hunting, even occurring in Schleswig Holstein inhibits the unrestricted dispersal of juvenile seals and indirectly inhibits the possible birth surplus from that area to replace the gap in pup production in The Netherlands.

d. disturbance

During the whelping and nursing period seals highly depend on sandbanks during low tide and it is rather obvious that the growing demand of tourists to recreate on the Wadden Sea can interfere with that. Seals often are forced to leave the tidal flats disturbed by boats, air planes and people walking from their boats on those tidal flats. This causes less time for nursing and resting and possibly increased pup mortality. It was mentioned earlier that pup mortality was nearly equal in all the Wadden Sea areas and estimated at about 60%. It is in-correct to conclude from this figure that disturbance does play a minor role. The tourist pressure in the total Wadden Sea area might be considered to be of the same magnitude,but it is not unlikely that the pup mortality of 60% found nowadays has been considerably lower in earlier times. Moreover Drescher (1978) found that disturbance played a significant role in the occurrence of skin lesions on seals. When seals are often chased into the water, a mechanical injury of the nearly healed umbelliculus may occur and an entrance for all kinds of disease agents is present.

c. pollution

Koeman et al. (1975) and Drescher et al. (1977) show considerable amounts of toxic chemicals accumulated in seals. Reijnders (1979) related pup production to the amounts of biocide residues present in seals of different Wadden Sea areas and found higher levels especially of PCB's present in seals from The Netherlands which exhibit a low pup production. Comparing that result with the findings of Olsson et al. (1974) and Helle et al. (1976a and b) in Baltic seals it is tentatively suggested that PCB's may be held responsible for the noted seal decrease in the Dutch Wadden Sea.

In relation to Drescher's work (1978) on skin lesions in seals it is likely that PCB's play a role too. It is well-known that PCB's have a strong immuno suppressive capacity. Keeping in mind the rather high residue levels of PCB's already present in young seals, it is assumed that after a mechanical injury due to disturbance, the PCB-weakened pups are more vulnerable to infections which leads to the process Drescher describes.

4. MANAGEMENT AND CONSERVATION

Summarizing the parameters and the environmental factors which determine the several (sub)-populations it is clear that the seals in the Wadden Sea have to be considered as one population.

Consequently, management of the harbour seal population has to be carried out in an international context based on internationally calibrated research programs. The international seal working group has already started to work but unfortunately it is still doubtful whether its composition and respective budgets are guaranteed in the future.

Furthermore, conservation measures must be taken soon, i.e. undisturbed rearing areas for pups have to be settled. In Niedersachsen some sanctuaries already exist and in The Netherlands several areas are established as or planned to become reserves for seals as well as for shore birds. In Schleswig Holstein however, the promising plans for a National Park (Erz, 1972) are cancelled and a few areas nominated as kind of reserves are still free for people to enter.

The voluntary cooperation of the several seal nursery stations along the Wadden Sea coast works fruitfully and it is hoped that their work at least offers the opportunity to gain time to favour the conditions of settling a long run management strategy. It is rather obvious that the impact of pollution, the major environmental stress on the harbour seal population, cannot be stopped by sanctuaries. Therefore international cooperation is needed of people concerned with environmental management and conservation and those concerned with clean water like farmers, water companies and industries. They will have to consult together with politicians of all countries located along the Wadden Sea and not at least those along the large rivers (especially the river Rhine). It is hoped that this might happen soon, the seals will do the rest.

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