

Effects of Polychlorinated Biphenyls (PCBs) on a Marine Plankton Population and Sedimentation in Controlled Ecosystem Enclosures

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ABSTRACT: Effects of polychlorinated biphenyls (PCBs) on a natural marine ecosystem were studied employing controlled ecosystem enclosures (CEEs) which contained 68 m³ of seawater and the natural sea-water biota (Saanich Inlet, British Columbia, Canada). Following addition of PCBs (50 µg l⁻¹) to a CEE, primary productivity was initially reduced by PCBs to less than 30 % of the control CEE to which no PCBs had been added. Settling velocity of particulate matter was initially accelerated by PCB addition, but gross sedimentation decreased due to inhibition of primary productivity. Decomposition activity in sedimented matter was initially inhibited by PCBs, one order of magnitude lower than that of the control, but followed by recovery to the control level within 6–10 d. Drastic and chronic reductions in the size of zooplankton stocks were also attributed to PCB stress.

INTRODUCTION

Polychlorinated biphenyls (PCBs) are widespread and persistent pollutants of the environment (Risebrough et al., 1968). They have been detected in seawater (Bidlman and Olney, 1974), marine sediments (Pavlou and Dexter, 1979) and various species of organisms (Jansson et al., 1975) throughout the world.

Toxicological studies revealed low acute toxicity of PCBs to aquatic invertebrates and vertebrates (Duke et al., 1970; Hansen et al., 1974). But it has been reported that PCBs inhibited the growth in several algal species and changed the species composition of algae in laboratory (Mosser et al., 1972) as well as in natural water (O'Connors et al., 1978). Most of these studies were done with single species in the laboratory, however, little is still known about the overall effects of PCBs on the pelagic plankton community through material cycling.

The Controlled Ecosystem Pollution Experiment (CEPEX) was one approach of reducing the gap between laboratory and field situations with large plastic bags (Menzel and Case, 1977). The present study describes some effects of PCBs on a plankton population in a marine ecosystem, employing controlled ecosystem enclosures. Observations on the behaviour and pathways of PCBs within the enclosure are discussed in a separate paper (Wong et al., 1981).

MATERIALS AND METHODS

Natural plankton populations of Saanich Inlet, British Columbia, Canada were captured on August 19, 1978 in two controlled ecosystem enclosures (CEEs, 68 m³ in volume). Each CEE was constructed of an acrylic floating ring, 2.5 m in diameter, from which a top-open cylinder bag made of polyethylene sheet was hung. CEE deployment and the general design are described in detail by Menzel and Case (1977). On August 22, a mixture of PCBs (Aroclor 1254), dissolved in 50 ml of ethanol and 15 g non-toxic dispersant (Corexit 9527), was mixed with 26 l of seawater pumped from inside

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the CEE and was added to the CEE at a final concentration of $50 \mu\text{g l}^{-1}$ (CEE E) using a technique (a diffusion ring) similar to that described by Topping and Windom (1977). The other enclosure serving as a control (CEE F) received equal amounts of ethanol and the dispersant. Nutrients were added thoroughly in the top 10 m of the water column using the diffusion ring in both CEEs on August 29. Nominal concentrations of nutrients were $10 \mu\text{g-at N l}^{-1}$ of nitrate, $10 \mu\text{g-at Si l}^{-1}$ of silicate, and $1.0 \mu\text{g-at P l}^{-1}$ of phosphate.

Water samples for the measurements of chlorophyll *a* (Chl *a*), photosynthetic activity and particulate organic carbon (POC) were collected by a peristaltic pump at three integrated depth intervals of 0–5, 5–10 and 10–13 m. For nutrient determinations, discrete depth samples of 3 and 10 m were also obtained by pumping.

Fluorometry using a Turner 111-003 fluorometer (Strickland and Parsons, 1972) was applied for Chl *a* determinations. Primary productivity was determined using ^{14}C bi-carbonate for 24 h *in situ* incubation at middle sampling depths in each CEE. POC in particles retained on pre-combusted Whatman GF/C glass-fiber filters was determined with a Perkin-Elmer elemental analyzer (Model 240). Nitrate, phosphate and silicate were analyzed by the methods described by Strickland and Parsons (1972) using a Technicon autoanalyzer II. Salinity was determined with a salinometer (Model 8400, Guildline Instruments).

A sediment trap was hung at 13 m in each enclosure to collect settling materials. The trap was made of two PVC cylinders, 48 cm high and 12.5 cm across the mouth (Fig. 1). A baffle grid (1.3 cm of each compartment and 5.0 cm in height) was placed at the opening of each cylinder in order to reduce distorting effects of currents in or near the upper part of the cylinder and to prevent animals from invading the cylinder. The same type of grid was placed in a sampling chamber made of acrylic in each cylinder, to prevent disturbance and aggregation of collected particles. Two drain spigots on the side of the cylinder allowed the water to escape after trap recovery.

After thorough gentle mixing of sedimented matter, collected in each sampling chamber, the content was divided into 4 aliquots for measurements of dry weight, POC and PON, decomposition rate and for visual observation with a light microscope. PON was determined by the same procedure, described earlier, as POC.

The decomposition rate of the sedimented matter was determined from the increase in total CO_2 in a sedimented-matter/seawater mixture kept in a 125 ml Erlenmeyer flask, which had a flare and was tightly sealed with a ground-glass stopper, during 24 h incubation in dark at surface temperature. Before incubation, 100 mg HgCl_2 were added into a flask as a blank.

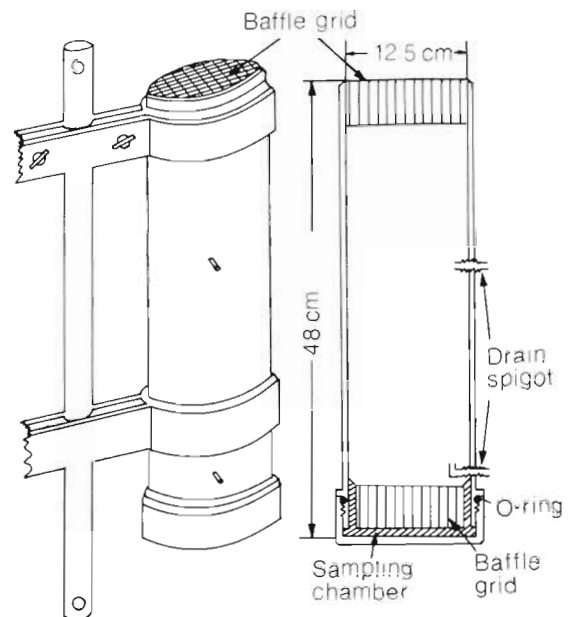


Fig. 1. Sediment trap used in this study

After incubation, a 1 ml sample was removed from each flask with a 1 ml syringe, injected into 1 ml sulphuric acid (pre-purged with nitrogen) in a 5 ml glass ampule through a rubber septum, and purged with nitrogen. The CO_2 released was then analyzed by an infrared analyzer (Total Carbon System, Model 0524B, Oceanography International Corp.). Assuming that decomposition processes are logarithmic with time, the decomposition coefficient of sedimented matter (k , d^{-1}) during the incubation period (t , d) was calculated as

$$k = -\frac{1}{t} \ln \frac{C_0 - \Delta C}{C_0} \quad (1)$$

where C_0 = initial POC amount of sedimented matter, ΔC = decreased amount of sedimented matter determined by the CO_2 released during t d. Assuming that gross sedimentation rate (F , $\text{mg C m}^{-2} \text{d}^{-1}$) and decomposition coefficient of sedimented matter (k , d^{-1}) are constant during collecting time period (t , d), time dependent change (dA/dt) of apparent sedimented matter (A , mg C m^{-2}), collected actually in sediment trap, can be described as

$$\frac{dA}{dt} = F - kA \quad (2)$$

Integrating Equation 2 with time t under the initial condition, $A = 0$ at $t = 0$

$$A = \frac{F}{k} (1 - e^{-kt}) \quad (3)$$

By rearranging Equation 3, gross sedimentation rate F is described as

$$F = \frac{AK}{1 - e^{-kt}} \quad (4)$$

Gross sedimentation during the suspension period (t) can be described as Ft (mg C m^{-2}), and $Ft - A$ defined as total decomposition loss D during the suspension period. Iseki et al. (1980) considered the biochemical decomposition of sedimented matter in a sediment trap in more detail.

RESULTS

Initial Condition of CEEs

Although both CEEs were operated almost simultaneously, the amount of water captured was different; it was 95 and 75 % for CEEs E and F, respectively. The remaining water was pumped from 5 m into the surface of each CEE. This treatment created some modifications in environmental parameters (Table 1). There was an inverse relation between the water capture

Table 1. Initial conditions in two CEEs. All data obtained on August 20, except for salinity (August 21) and nitrate (August 22)

Environmental parameters	Sampling depth (m)	CEEs		Outside
		E	F	
Water capture percentage		95 %	75 %	
Salinity (‰)	0	29.36	29.36	29.17
	5	29.43	29.37	29.40
	10	29.44	29.44	29.48
	13	29.59	29.52	29.51
Chlorophyll <i>a</i> ($\mu\text{g Chl } a \text{ l}^{-1}$)	0–5	3.8	3.3	6.5
	5–10	2.5	2.1	0.9
	10–13	0.9	0.8	0.2
POC ($\mu\text{g C l}^{-1}$)	0–5	171	146	277
	5–10	115	101	61
	10–13	72	54	41
Primary productivity ($\text{mg C m}^{-3} \text{ d}^{-1}$)	0–5	228	242	–
	5–10	51	37	–
	10–13	7	2	–
Photosynthetic rate ($\mu\text{g C } \mu\text{g Chl } a \text{ l}^{-1} \text{ h}^{-1}$)	0–5	2.5	3.1	–
	5–10	0.9	0.7	–
	10–13	0.3	0.1	–
POC: Chl <i>a</i> ratio	0–5	45	45	42
	5–10	47	49	65
	10–13	77	71	274
Nitrate ($\mu\text{g-at N l}^{-1}$)	3	0.2	2.5	–
	10	14.2	12.9	–
– Not observed				

percentage and concentrations of Chl *a* and POC in the top 10 m of the water column.

POC:Chl *a* ratios of CEEs E and F were similar, being 45–49 in the upper 10 m. Initial photosynthetic rates were almost proportional to the amount of water pumped into each CEE. This was explained partly by higher nitrate levels observed in the heavily pumped CEE, 2.5 $\mu\text{g-at N l}^{-1}$ for CEE F, which was compared with 0.2 $\mu\text{g-at N l}^{-1}$ in CEE E.

Changes of Environmental Parameters

Salinity varied from 29.4 to 28.8 ‰ S at the surface but was almost constant (29.4 ‰ S) at 10 m in CEE E throughout the experiment. A similar trend was observed in CEE F. At the beginning of the experiment, nitrate concentrations in both CEEs showed some differences at 3 m due to different pumping rates (Fig. 2). Nitrate was depleted within 3 d after capture at 3 m in CEE E, but was not depleted until 6 d in CEE F. Initial nitrate concentrations at 10 m were all high (12.9–14.2 $\mu\text{g-at N l}^{-1}$); they decreased quickly in the control (CEE F), but slowly in CEE E. By adding nutrients on August 29, final concentrations of 11.6–11.7 $\mu\text{g-at N l}^{-1}$ at 3 m were achieved. Seven days after nutrient addition, nitrate concentrations at 3 m decreased to 2.4–3.3 $\mu\text{g-at N l}^{-1}$ in both CEEs. There was a consistent decrease of nitrate at 10 m between September 5 and 11 in CEE E whereas no remarkable change was observed in CEE F. Overall changing patterns of phosphate and silicate were similar to those of nitrate.

Immediately after launching, Chl *a* started to increase in both CEEs, particularly obvious in the upper 10 m reaching a maximum within 4 d (Fig. 3). Maximum concentrations of Chl *a* in the top 5 m, observed on August 23, were 13.0 and 24.1 $\mu\text{g Chl } a \text{ l}^{-1}$ in CEEs E and F, respectively. Chl *a* increases within the first 3 d were 7.4 fold in CEE F and 3.4 fold in CEE

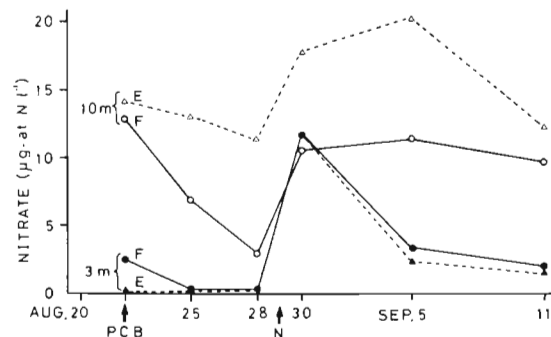


Fig. 2. Changes in nitrate concentrations at 3 and 10 m in CEEs E and F. Arrows marked 'PCB' and 'N' indicate PCB and nutrient addition, respectively

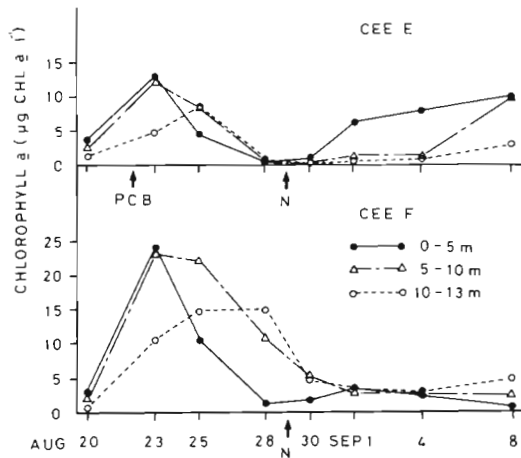


Fig. 3. Changes in chlorophyll a at three different depth intervals in CEEs E and F. Arrows marked 'PCB' and 'N' indicate PCB and nutrient addition, respectively

E. After August 23, Chl a above 5 m in both CEEs declined sharply to the level of $0.5\text{--}1.4 \mu\text{g Chl a l}^{-1}$. Nutrient addition on August 29 reincreased Chl a above 5 m in both CEEs. The increase was pronounced in CEE E which was maintained until the end of the experiment.

Initial Chl a increase was accompanied by an increase in POC but their temporal changing patterns were slightly different (Fig. 4). POC maximum above 5 m occurred on August 25, 2 d after the Chl a maximum, and then declined as did Chl a. Such phase differences in changing patterns between POC and Chl a were not obvious during the later part of the experiment.

On August 20, POC in the water column consisted mainly of small-size particles, less than $35 \mu\text{m}$ (Table 2). The relative abundance of small particles ($< 35 \mu\text{m}$) decreased markedly during the algal bloom between August 23 and 25 in both CEEs and increased again in PCB-treated CEE E after August 28. The small-particle fraction persisted throughout the rest of experiment in PCB-treated CEE E. Contrary, large particles were abundant in the control CEE F between August 23 through September 1, followed by a gradual decrease

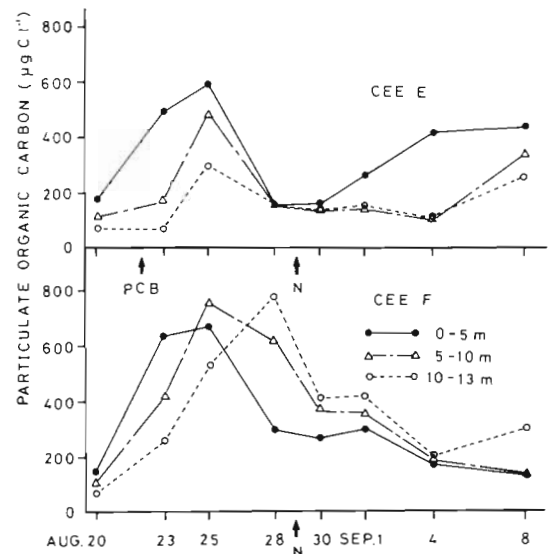


Fig. 4. Changes in POC at three different depth intervals in CEEs E and F. Arrows marked 'PCB' and 'N' indicate PCB and nutrient addition, respectively

to 24 % towards the end of the experiment. The phytoplankton was primarily composed of centric and pennate diatoms on August 23 in both CEEs, but microflagellates became abundant in CEE E after August 28 (152 diatom cells vs. 5817 microflagellate cells ml^{-1} on September 2). In CEE F, microflagellates were also abundant but less significant compared to CEE E (991 diatom cells vs 1682 microflagellate cells ml^{-1} on September 2). Two weeks after PCB addition (September 6) total zooplankton numbers (comprised mainly of *Paracalanus parvus*) in the water column above 10 m were two orders of magnitude lower in PCB-treated CEE E than that in CEE F. CEEs E and F contained 80 and 3000 individuals m^{-3} , respectively.

Changes in Biological Activities

Immediately after PCB addition, primary productivity in CEE E decreased from 1.42 to $0.32 \text{ g C m}^{-2} \text{ d}^{-1}$ (73 % decrease) and maintained a low level until

Table 2. Relative abundance of POC, in two different size fractions, i.e. larger or smaller than $35 \mu\text{m}$ at 0-5 m depth in CEEs E and F

CEE	Size (μm)	Sampling date							
		Aug. 20	Aug. 23	Aug. 25	Aug. 28	Aug. 30	Sept. 1	Sept. 4	Sept. 8
E	$35 <$	28	57	58	8	6	5	7	3
	< 35	72	43	42	92	94	95	93	97
F	$35 <$	26	67	69	60	56	55	42	24
	< 35	74	33	31	40	44	45	58	76

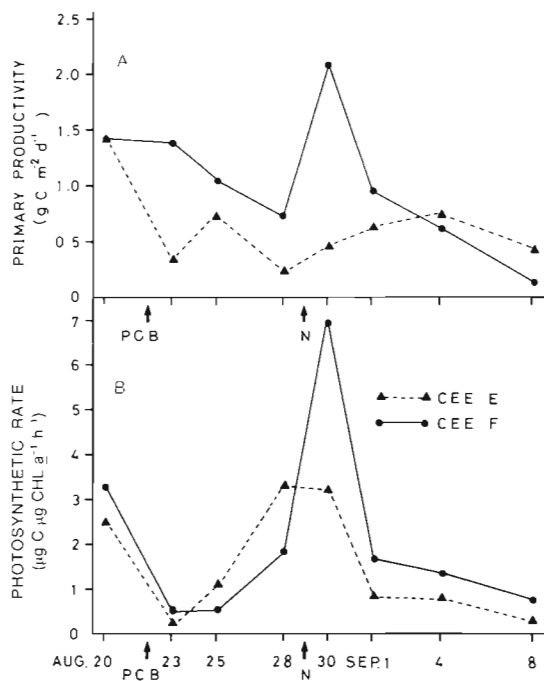


Fig. 5. Changes in primary productivity in the upper 13 m (A) and photosynthetic rate in 0–5 m (B) in CEEs E and F. Arrows marked 'PCB' and 'N' indicate PCB and nutrient addition, respectively

August 28 (Fig. 5A). In contrast, no decline of productivity was noticed in CEE F (control) during August 20 and 23, but thereafter productivity gradually decreased until nutrient addition. After nutrient addition, the productivity in CEE F showed a remarkable enhancement, but in CEE E only gradual increase was observed.

However, there was not much difference between the two CEEs in regard to changes in photosynthetic

rates (Fig. 5B). In both CEEs photosynthetic rates decreased during the first 3 d and then gradually increased during the next 5 d. A sudden increase in photosynthetic rate was observed in CEE F after nutrient addition attained 3.9 times (1.8 to 7.0 $\mu\text{g C } \mu\text{g Chl a}^{-1} \text{ h}^{-1}$). However, enhancement effects by nutrient addition were less significant in CEE E. Photosynthetic rates decreased sharply from August 30 to September 1 in both CEEs and then gradually decreased until September 8. During the last 9 d, photosynthetic rates were always higher in CEE F than in CEE E.

Changes in Sedimentation

The amount of apparent sedimented matter *A* was small and showed almost no variation between CEEs prior to the PCB addition (Table 3). Decomposition coefficients of sedimented matter (*k*), obtained from laboratory incubation experiments, were high – ca 0.3 d⁻¹ in August 21 samples – showing no significant difference between CEEs but decreased down to 0.1 d⁻¹ or lower in samples collected between 23 and 28 August (Table 3). The most significant decrease of *k* (0.01 d⁻¹), was observed in CEE E immediately after PCB addition, whereas 0.09 d⁻¹ was obtained for CEE F in the same period. The decomposition coefficient of CEE E recovered to the same level of CEE F in samples collected between August 28 and September 1. There was not much difference in *k* between CEEs during the subsequent experimental period.

Sedimentation losses from the organic matter produced by primary production, expressed as the ratio of *F/P*, were low and similar in both CEEs at the beginning of experiment; thereafter they increased rapidly (Table 3). The ratio sedimentation loss to primary pro-

Table 3. Primary productivity, sedimentation, and decomposition rates of sedimented matter. *P*, *A*, *k* observed, *F*, *D* calculated values

Sampling intervals	<i>t</i> Suspension period (d)	CEE	<i>P</i> Primary productivity (mg C m ⁻² d ⁻¹)	<i>A</i> Apparent sedimented matter (mg C m ⁻²)	<i>k</i> Decomposition coefficient (d ⁻¹)	<i>F</i> Gross sedimentation rate (mg C m ⁻² d ⁻¹)	<i>D</i> Decomposition loss (mg C m ⁻²)	<i>F/P</i> × 100 (%)	<i>D/Ft</i> × 100 (%)
Aug. 20	1	E	1420	50	0.30	60	10	4	17
Aug. 21		F	1410	60	0.31	70	10	5	14
Aug. 23	5	E	520	1960	0.01	400	40	78	2
Aug. 28		F	1090	2450	0.09	610	600	56	20
Aug. 28	4	E	410	2000	0.02	520	80	127	4
Sept. 1		F	1440	4160	0.02	1080	160	75	4
Sept. 1	4	E	710	1510	0.04	410	130	58	8
Sept. 5		F	740	2770	0.05	760	270	103	9
Sept. 5	4	E	560	1150	0.13	370	330	66	22
Sept. 9		F	300	1350	0.12	420	330	140	20

ductivity revealed a maximum of 127 % in CEE E during August 28 and September 1, followed by rapid decrease. In the control, this ratio showed a gradual increase during the experiment and reached a maximum of 140 % between September 5 and 9.

Total decomposition losses of sedimented matter during suspension, defined as D/Ft , also revealed no significant difference between CEEs prior to PCB addition (Table 3). However, it diminished extensively after PCB addition in CEE E, to 1/10 of the control, between August 23 and 28, and then showed subsequent recovery by August 28 and September 1.

C:N ratios and carbon content (POC per unit dry weight of particles) of sedimented matter varied between 6.6 and 7.2, and 19.5 and 27.7 %, respectively within the two CEEs during the experiment. However, no apparent difference was observed between CEEs.

DISCUSSION

Initial PCB concentrations in the water column were about $40 \mu\text{g l}^{-1}$ at 1 m and 5 m, and $15 \mu\text{g l}^{-1}$ at 10 m in CEE E (Fig. 6). The PCBs were not uniformly distributed throughout the water column, possibly due to uneven PCB addition to the water column using the diffusion ring. Almost exponential PCB decrease was observed with time at 1 and 5 m. At 10 m, however, the PCB concentration increased within the first 2 d after addition, followed by subsequent decrease with time. Six days after PCB addition, the concentrations were below $15 \mu\text{g l}^{-1}$ at all three depths; they were highest at 10 m and lowest at 1 m. It is likely that the rapid removal of PCBs from the water column is directly related to the settling of particulate matter (phytoplankton). Harding and Phillips (1978b) showed that phytoplankton is capable of rapidly accumulating substantial PCB amounts, with equilibration occurring within 2 h.

Extraction of 3 and 10 m samples of bag-wall material revealed that a slight amount of Aroclor 1254 was associated with the bag walls. However, this amount was negligible, less than 0.5 % of the total PCBs added (Wong et al., 1981). It is considered that PCB adsorption on the bag wall and the sediment trap was of minor importance.

After PCB addition, the diatom population initially dominating in the water was quickly replaced by microflagellates. A similar population shift was observed in other pollutant experiments using copper and mercury (Thomas and Seibert, 1977). These findings indicate that small algae are less sensitive or better able to recover from pollutant stress, compared to most of the larger phytoplankters (centric diatoms, in this case). Interspecific differences of phytoplankton in regard to

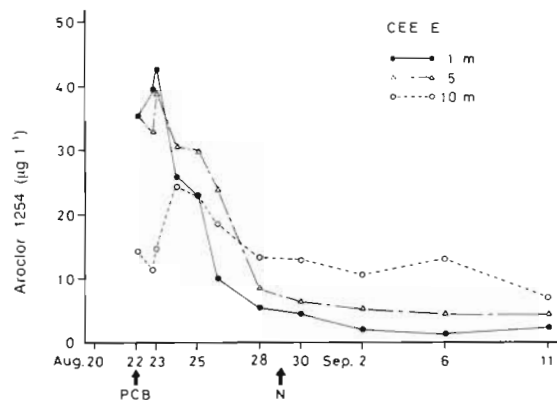


Fig. 6. Changes in Aroclor 1254 at three depths in CEE E. Arrows marked 'PCB' and 'N' indicate PCB and nutrient addition, respectively. PCBs were analyzed using Hewlett-Packard gas chromatography (Model 5830A) equipped with ^{63}Ni electron capture detector. (After Wong et al., 1981)

their PCB susceptibility are known during cell division and photosynthesis (Menzel et al., 1970). Harding and Phillips (1978a) observed that the pennate diatom *Nitzschia longissima* and the green alga *Dunaliella tertiolecta* were relatively insensitive to PCBs, both in regard to photosynthesis and cell division, although substantial inhibition was observed in centric diatoms such as *Thalassiosira pseudonana*, *Chaetoceros socialis* and *Skeletonema costatum*, and in some flagellates such as *Monochrysis lutheri* and *Isochrysis galbana*.

Immediately after PCB addition, an acceleration in sinking rates of materials occurred in CEE E; this was established on the basis of gross sedimentation rate divided by primary productivity, F/P (Table 3); however, this was not obvious in the control. Acceleration in sinking rate may possibly be attributed to direct toxicity effects of PCBs: sinking rates of senescent and dead cells are several times faster than those of growing cells (Smayda, 1970). A similar acceleration in sinking rates of particles has been observed in a copper-treated plankton population by Harrison and Davies (1977).

Following nutrient addition, photosynthetic rate and primary productivity increased significantly in CEE F (Fig. 5). However, in the upper layer of the water column the increase in Chl a and POC was less in CEE F than in CEE E (Figs 3 and 4). During the week following nutrient addition, Chl a and POC persisted in the upper layer in CEE E; there was no rapid settling as at the beginning of the experiment. The above-described situation may be the result of a relative increase in the abundance of small plankters such as microflagellates (as shown in Table 2), which settle slowly, and of the virtual elimination of grazing pressure by herbivores due to PCB toxicity which could

reduce the production of fecal pellets with rapid sinking rates.

PCBs exert a detrimental effect on zooplankton. They eliminate zooplankters from the water column within 15 d after PCB addition. There is no evidence of recovery during the entire experimental period of 20 d. Based on their copper experiment, Harrison and Davies (1977) showed that chronic inhibition of zooplankton feeding and subsequent decline in zooplankton biomass can be attributed to copper stress ($10 \mu\text{g l}^{-1}$); copper-stressed zooplankton contributed only a small fraction to nutrient recycling. Several reports imply that zooplankters play an important role in the recycling of bioelements in a given ecosystem through ammonia and phosphate excretion – leading to rapid and direct exchange of nutrients between phytoplankton and zooplankton (Smith and Whitley, 1977). Where zooplankters assume an important role for nutrient regeneration, a pollutant-caused reduction in zooplankton biomass could have a significant effect on a natural ecosystem.

Bacterial populations and their metabolic activities may recover quickly from mercury and copper perturbations (Vaccaro et al., 1977; Azam et al., 1977). Saylor et al. (1978) also documented a positive relation between PCB concentrations and the number of PCB-degrading bacteria. These results imply that PCB tolerant and PCB degrading bacteria might develop in a PCB-polluted ecosystem and may play an important role in the water column as well as in sediments.

Decomposition coefficients of sedimented matter were significantly inhibited by PCBs added in CEE E, but revealed recovery to the control level at the next sampling within 4 d (Table 3). Decomposition loss during suspension, expressed as mg C m^{-2} , also indicated a significant reduction in PCB-treated CEE E, followed by an increase towards the end of the experiment due to both recovery from PCB stress and variation in gross sedimentation rate.

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