EFFECTS OF LOG STORAGE ON ZOOPLANKTON AND JUVENILE SALMONIDS IN BABINE LAKE, BRITISH COLUMBIA

by

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ABSTRACT

Effects of log storage on water quality, zooplankton and juvenile salmonids were investigated at Babine Lake, British Columbia in a series of enclosure, field and laboratory experiments. Enclosures were stocked with lake zooplankton and treated with lodgepole pine (Pinus Contorta) and white spruce (Picea glauca) logs for two 25 day periods. Oxygen depletion, to levels as low as 2.5 mg/l, and increased lignin and tannin (L-T) concentration (a measure of wood leachate) occurred in log treated enclosures. Zooplankton density significantly decreased with increased log number, but changes in community diversity were not consistent.

In field studies at Morrison Arm, Babine Lake, extreme oxygen depletion (<1 mg/l) was observed in localized surface waters within a log storage area. Dye tracer studies within the log bundles implied reduced water movement, which may be involved in oxygen depletion. Local zooplankton abundance was usually lower at log storage sites than nearby undisturbed littoral sites and sockeye fry held in situ for 24 h periods acquired fewer and/or a lower diversity of prey items in log storage areas.

Laboratory toxicity studies indicated that spruce bark leachates were more toxic than pine, but lethally toxic

bark leachates had higher L-T values than those measured in the Morrison Arm log storage area. In chronic Daphnia bioassays, mortality rates significantly increased and fecundity rates significantly decreased during long term exposure to low concentrations of bark leachates. Results of enclosure experiments, field studies and laboratory bioassays provide evidence that zooplankton are reduced in abundance by conditions which accompany log storage, possibly through chronic toxicity or reduce fecundity. Because fry diet was sensitive to small changes in food abundance, there is potential for reduced survival of sockeye fry exposed to low oxygen concentrations and reduced food levels.

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1. GENERAL INTRODUCTION

Over 90% of the timber logged from British Columbia's forests spends time in water-based systems of transportation and storage (Edgell and Ross 1983) en route to processing mills. Log handling activities represent one of the largest costs to the forest industry.

Historically, wood has always been harvested from the most accessible places first, such as along the shores of seas, rivers and lakes; consequently water-based log handling systems were developed (Sedell and Duval 1985). Log driving (transportation of wood using the power of river water flow) was a common practice which caused extreme damage to river habitats, particularly spawning and juvenile rearing areas (I.P.S.F.C 1966). In the late 1800s, river improvement consisted of removing obstacles from rivers, straightening out crooked streams and blocking off sloughs and marsh areas with cribbing to keep water and logs in the main channel (Sedell and Duval 1985).

Presently, water-based log handling centres around dumping, sorting and storage activities. Most of the site leases are concentrated in shallow, protected areas such as bays and estuaries which are perceived as sensitive to manipulation by man. This concern is evidenced by a number of recent studies commissioned to look at the ecological aspects of log storage,

particularly in estuarine environments (Anon. 1980a, Anon. 1981, Levy et al. 1982).

There are many aspects of log handling in water which may adversely affect the environment. Recognized effects of log storage on fish habitat falls into three categories: physical effects, chemical effects and biotic effects. The most obvious and well documented impacts of log handling on the aquatic environment are physical changes (eg. bark deposition, sediment compaction, and increased water turbidity) (Toews and Brownlee 1981). Chemical changes to water quality induced by log storage have been quantified in laboratory studies (Graham 1970, Atkinson 1971). Leachate extraction occurs in water and is accompanied by increased C.O.D. (chemical oxygen demand), B.O.D. (biological oxygen demand) and oxygen depletion. Literature on impacts of log storage on biota is mainly descriptive; several reviews and summary publications have been produced, all of which contain the same information (Anon. 1980a, Edgell and Ross 1983, Sedell and Duval 1985).

In a recent review of log handling in British Columbian estuaries, the Environmental Review Panel (Anon. 1980a, p 6) stated:

> Particularly critical is the need for information on the precise interaction between fishes and environments used for log handling.
>
> Lack of data on the impact of log handling on coastal ecosystems cannot be overstated.

There are data on effects of log storage on benthic invertebrates (Conlan 1975, Conlan and Ellis 1979, Levy et al. 1985b) from which predictions about the food supply of benthically feeding fish can be made. However, with the exception of a single study on the Fraser River estuary (Levy et al. 1982), no published information exists on direct effects of log storage on fish populations (Ainscough 1979, Anon. 1980a).

This lack of information places habitat managers in an awkward position. In Coos Bay, Oregon, U.S.A., government biologists set stringent log handling water policies in 1979, and were placed under considerable pressure from the forest industry because they lacked data to support their belief that log rafts were affecting fish production detrimentally (M. Brownlee, pers. comm.). In the Nanaimo estuary, federal biologists found themselves in a similar position, which resulted in formation of a task force to examine the situation (Anon. 1980b).

When Houston Forest Products (H.F.P.) obtained rights to harvest timber infected by spruce budworm near Morrison Arm, Babine Lake, B.C. (55 deg.N, 123 deg. W), they applied to install a log storage and transportation system on the lake. Officials from the Department of Fisheries and Oceans expressed concern about potential deleterious effects on Babine Lake and

gave conditional approval, provided that a 3 year study be implemented to determine the effects of log handling on the fish habitats and population of Babine Lake. The Westwater Research Centre received a grant from H.F.P. (funded under Section 88.2 of the B.C. Forest Act) to undertake the study. The main objective of the study was to "determine the effects of log dumping, storage, and dewatering on fish rearing and spawning habitat and migration routes of fry, smolts, and adults at specific use sites, including those to be used by H.F.P., and the potential impact on Babine Lake fisheries" (Levy et al. 1984, p. 1).

Babine Lake is the largest natural lake in British Columbia and one of the major drainage basins for the Skeena River system (Johnson 1965, Levy and Hall 1985). Sockeye salmon (Oncorhynchus nerka

the Babine system contribute over 90% of the Skeena sockeye fishery (Larkin and McDonald 1968). Average production (over past 30 yr.) of adult sockeye has been about 1.5 million annually (McDonald and Hume 1984).

H.F.P. installed two log handling facilities at Babine Lake (Figure 1). At the head of Morrison Arm a log dump/storage site was installed which is used from the time of freeze up to early summer. Log bundles from winter logging are slid down a ramp into the water and then stored in a small bay kept ice free with an air bubbler system until spring ice break up. Then the log

bundles are towed in large rafts to the dewatering site near Fulton River (Figure 1). There, log bundles are loaded onto logging trucks and taken to a mill.

Impacts of logging activities along the shore of Babine Lake will potentially be greatest in sheltered littoral areas of Morrison Arm where logs are stored. Poor water circulation and wood leachate extraction may combine to produce deleterious water conditions for sockeye salmon fry and their food supply, zooplankton. To examine this prediction, enclosure, field and laboratory experiments were designed to examine effects of log storage conditions on zooplankton and fish.

A localized reduction in food supply for recently emerged fry may restrict their growth during this "critical" period, according to Hjort (1914) and Braum (1967). To test the hypothesis that zooplankton abundance decreases under simulated log storage conditions, enclosures with stocked zooplankton populations were treated with logs. Then water quality and zooplankton abundance and diversity were monitored over time (Chapter 2).

To complement the enclosure studies of water quality and zooplankton, comparative monitoring was carried out for water quality and zooplankton abundance at log storage and undisturbed littoral field sites (Chapter 3).

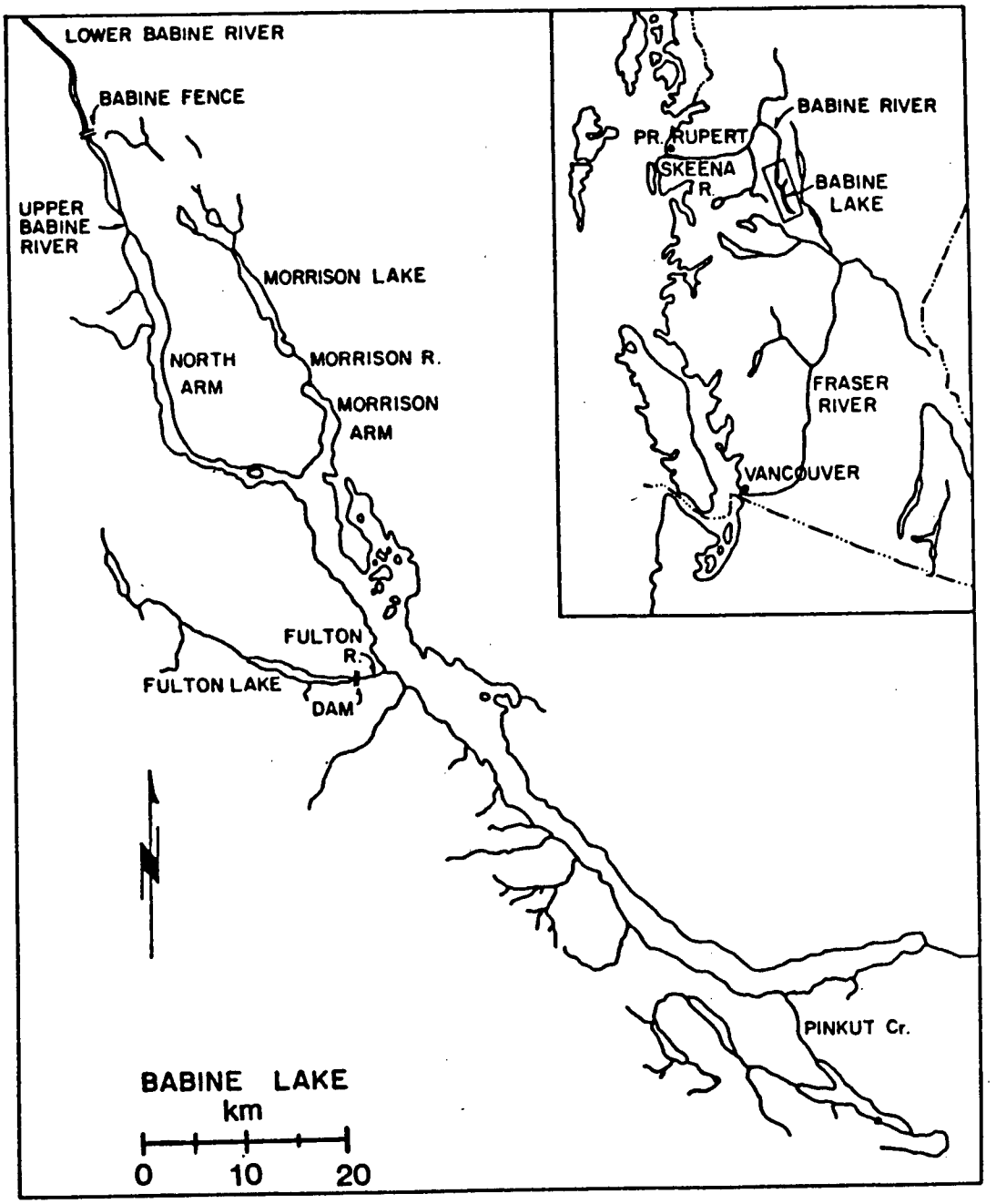

Figure 1. Map of Babine Lake, B.C. showing Morrison Arm and Fulton River (after West 1983).

I examined feeding by sockeye fry held at log storage and undisturbed littoral sites to test for differences in food intake over 24 h periods between the two areas (Chapter 3). Ration level and growth rate in sockeye salmon are correlated (Brett et al. 1969) and it was expected that small sockeye fry would experience mortality and hence higher mortality rates (West 1983) in disturbed sites.

To determine the wood leachate levels necessary to affect zooplankton negatively, lethal and chronic laboratory bioassays were conducted (Chapter 4). It is known that wood and bark leachates are toxic (Atkinson 1971, Buchanan et al. 1976) but at levels higher than those which usually occur in log storage areas. Therefore, chronic bioassays for Daphnia were used to test for sublethal effects of low concentrations of leachate.

2. ENCLOSURE EXPERIMENTS

2.1 INTRODUCTION

Most research on the biotic effects of log storage is mainly descriptive or qualitative and concentrates on benthic invertebrates. Physical impacts, such as sediment compaction and bark deposition, as well as chemical changes in the environment (reduced oxygen, hydrogen sulphide and toxic wood leachate production) are responsible for depletion of benthic communities (Toews and Brownlee, 1981). However, no research has been done on the effects of log storage on pelagic invertebrates such as zooplankton in marine, estuarine or fresh water systems. One might not expect these organisms to be affected by log storage because zooplankton move with water masses. Yet almost half of British Columbia's log handling leases are for sites with negligible tidal currents (FERIC 1980) which increases the potential for chemical impacts on invertebrates in the water column. Certainly, Levy et al. (1985) and the present study (Chapter 3) have demonstrated that severe oxygen depletion can occur in a log storage area within a very short time.

There is concern about localized reduction of food supply for planktivorous fishes, particularly commercially important species such as sockeye salmon, whose juvenile life stages depend largely upon lake zooplankton (Narver 1970, Rankin 1977). Sockeye fry from Morrison Creek seem to stay inshore for a few

days as they move down the arm and then offshore (Levy et al. 1985b). When littoral postlarval sockeye enter the Morrison Arm log storage grounds, they are often just beginning to feed on zooplankton (pers. obs.). Hjort (1914) and Braum (1967) suggest that the most critical period for larval fish is during the transition from yolk sac to external feeding. A food shortage during this time may induce starvation because young fish are least resistant to low energy reserves.

While bioassay data are indispensible in assessing relative toxicities and providing baseline data, they do not adequately assess the long term impact of a toxicant on a natural community assemblage (Leeuwangh 1978, Stephenson et al. 1984). Alternatively, field studies encounter the inherent difficulty of repetitively sampling the same population of organisms in the same water mass over time (Gamble and Davies 1982) and they often lack adequate controls. The need in toxicology, and aquatic ecology in general, for well controlled experiments on complex systems has led to the use of in situ enclosures which isolate part of the water column and allow well controlled manipulations. Artificially impounded populations are held under controlled conditions so that treatment effects can be discerned (Grice and Menzel 1978, Grice and Reeve 1982, Stephenson et al. 1984). Buikema et al. (1982) point out that since the goal of most environmental studies is to examine effects on the entire community, microcosm (<10 cubic meters, Banse 1982) tests may be the most appropriate method for prediction.

Microcosms have been widely used to evaluate the environmental impacts of aquatic pollutants (Banse 1982). Large variation in zooplankton population density and species composition make it difficult to detect long term effects of pollutants in the field (Kuiper 1982). The use of enclosures in addition to short term bioassays permits the inclusion of effects from natural biotic factors which may influence the outcome of the toxicity experiment (Kaushik et al. 1985, Salki et al. 1985), but cannot be included in laboratory situations.

Experimental enclosures were used to examine effects of bark and log treatments on water quality and zooplankton density in the littoral zone of Babine Lake. Deterioration of water quality and reduced zooplankton density under log treatment, relative to control enclosures, will be accepted as evidence that log storage can detrimentally affect the localized environment that sockeye salmon fry inhabit for their first weeks of life. Decreased diversity in zooplankton populations exposed to toxicants has been used as an indicator of deleterious effects (Washington 1984, Kaushik 1985). I predict that zooplankton diversity will decrease over time in log treated enclosures and 5 log treatment populations will have the lowest diversities.

More specifically, four main question were addressed: (1) are there differences in the effects of the two main commercial

tree species, Pine (Pinus contorta) and spruce (Picea glauca) on water quality and zooplankton density? and (2) at what point does the ratio of wood to water, or logs per enclosure, significantly reduce lake zooplankton densities? (3) are there differential effects on the different zooplankton species and life history stages? and (4) do log treatments reduce the species diversity of zooplankton in enclosures, indicating a change in community structure?

This chapter is based mainly on log treatment experiments conducted during the 1985 field season. Results from bark addition experiments will be briefly presented and discussed with respect to general trends.

2.2 METHODS

Enclosure experiments were conducted in waters of a depth similar to that of a shallow boom site (z=3 m), within a protected littoral area at Granisle, Babine Lake. Eight identical enclosures were built during the summer of 1984.

The enclosures consisted of two parts: a plastic bag and a float from which the bag was suspended (Figure 2). The bags were made of a woven polyolefin fabric (Fabrene R Type "T.M.") and were sewn by False Creek Industries Ltd. into a cylinder 1.5 m in diameter and 2.1 m in depth (Volume = 3700 l). The enclosures had a solid bottom; there was no water exchange or contact with

lake sediment. The float was of plywood construction; buoyancy was provided by styrofoam blocks. The bag was suspended from the inside perimeter of the float and laced into place. The tops of the enclosures extended 0.3 m above the surface of the water to minimize water spillover.

Procedure: Enclosures were filled by water pump with lake water taken over a 0 to 2.0 m depth range. During the 1984 field season, zooplankton were pumped with this water directly into the enclosures and difficulties were encountered in initially stocking the enclosures with similar zooplankton populations. Therefore, during the 1985 field season, the enclosures were stocked with zooplankton independently in a method suggested by W.E. Neill (pers. comm. 1985). The water pumped into each enclosure was filtered through a 100 micrometer mesh net to remove zooplankton. Enclosures were then stocked with zooplankton obtained from vertical hauls from 4.0 m to surface. The following formula was used to calculate the number of vertical hauls required per enclosure:

$$\text{no. hauls} = \frac{\text{volume of enclosure}}{\text{volume of haul X net efficiency}}$$

The efficiency of the zooplankton net used to stock the enclosures was compared to a PAR diaphragm bilge pump and found to be (1) equally efficient for sampling both nauplii and copepodites (2) 30% more efficient for sampling Diacyclops

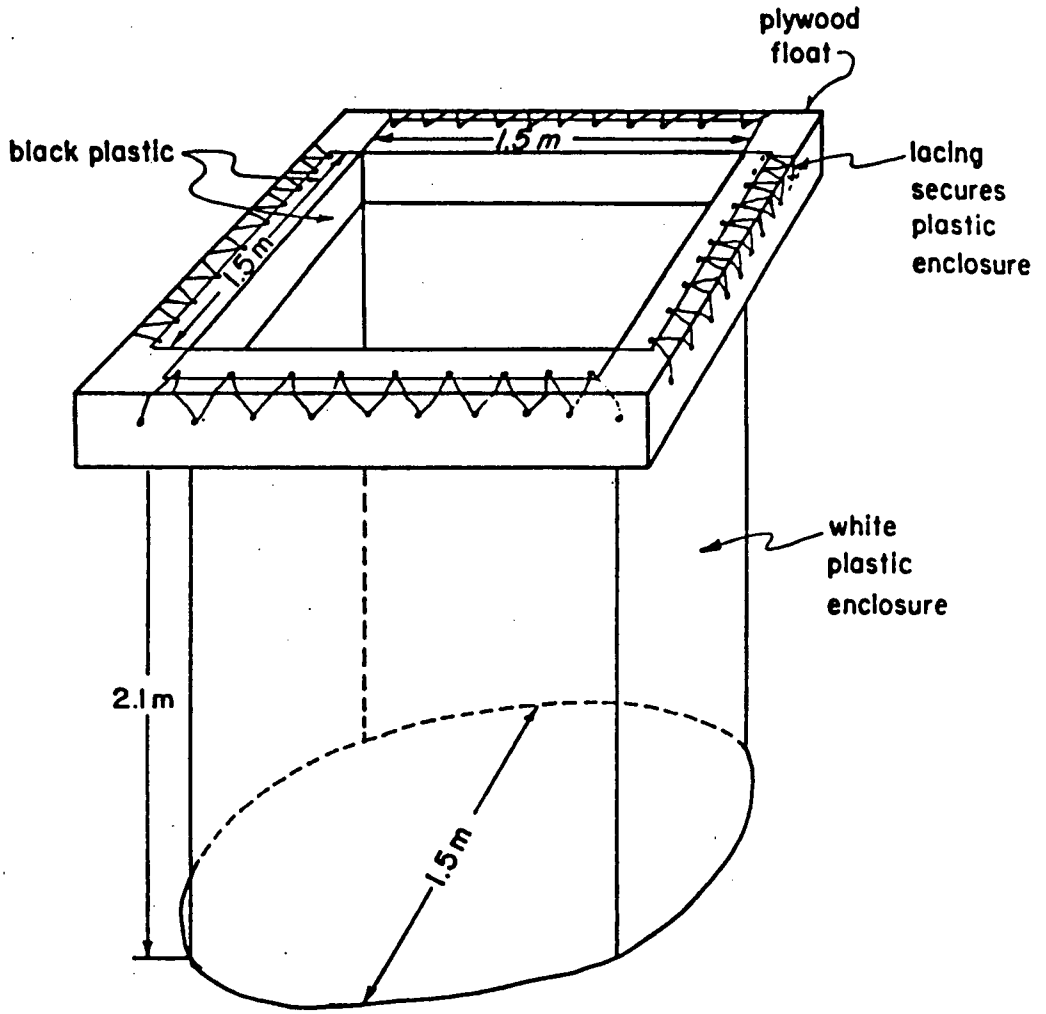

Figure 2. Design of enclosures (volume = 3700 l).

adults and (3) 20% less efficient for sampling Diaptomus adults. Overall, there was no net difference in efficiency between the zooplankton net and the pump for sampling the early May zooplankton population. Therefore, assuming this to be true, the number of hauls to stock one enclosure was calculated to be 20 hauls from 4 m to surface. To test this result, a trial zooplankton stocking experiment was conducted; the trial enclosure was understocked, particularly for adult life stages. Therefore, to reach lake densities of zooplankton, 100 vertical hauls (4 m to surface) were mixed and randomly allocated over the four enclosures used for each experiment.

The percentage similarity between enclosures was calculated to assess relative initial conditions. I was aiming for a percentage similarity of 80% or better. Renkonen's index, modified with a natural log transformation, as suggested by Wolda (1981), was used.

RENKONENS'S INDEX OF SIMILARITY

$$P.S. = \min (p_{1j}, p_{2j})$$

after $\ln (n_{1j}, n_{2j})$

where p_{1j} = proportion of total no. of individuals consisting of the j^{th} type

n_{1j} = no. of individuals of the j^{th} type

Water quality and zooplankton were sampled prior to the introduction of log rafts (day 0) and then consecutively, on days 5, 10, 15, 20, and 25 of each experiment. Dissolved oxygen (D.O.) and temperature were determined *in situ* with a YSI (model 57) dissolved oxygen/temperature meter. The oxygen probe was calibrated at the beginning of each sample day by the air saturation method and altitude corrected. All measurements were taken at 1 m depth in the centre of the enclosures. Lake samples of water quality and zooplankton were taken adjacent to the enclosures on every sample date. Water samples were frozen and then later analyzed for lignin and tannin (L-T) concentration. Lignins and tannins contain aromatic hydroxyl groups that reduce tungstophosphoric and molybdophosphoric acids to form a blue colour. The colour absorbence determined with a spectrophotometer (APHA *et al*. 1985) gives a measure of lignin and tannin concentrations. Water quality data were obtained cooperatively with Paula Wentzell, a civil engineering graduate student.

To assess the zooplankton populations in the enclosure experiments and lake, samples were taken by pumping for one minute (37 l) from 1 m in depth, with the pump outflow passing through a 100 micrometer net. Duplicate samples were taken for each treatment. Samples were preserved in a sucrose formaldehyde solution (Haney and Hall 1975) and enumerated and identified under a stereo dissecting scope using Edmondson (1959) and Smith

and Fernando (1978) for taxonomic classification. Most samples were completely counted; subsampling was used for abundant organisms only.

The logs used in the experiments were approximately 1.4 m long and 0.15 m in diameter. They were cut two weeks prior to the start of each experiment and tied together in rafts before being introduced into enclosures. After completion of an experiment, the logs were measured and then the bark was removed for dry weight determination. Enclosures were emptied and scrubbed clean between experiments.

Experimental Design: In 1984, only four enclosures were constructed initially. Pairs of enclosures were used for treatment/control combinations. Treatments were randomly applied to single enclosures and consisted of amounts of bark (60:40 ratio of pine:spruce by weight); 20 kg, 5 kg and 1 kg bark weights were used for consecutive 31, 14 and 12 day experiments, respectively. Also, a raft of 8 logs was used as an experimental treatment for three weeks in a single enclosure to simulate a log bundle. Water quality and zooplankton were monitored over the 2, 3 and 4 week experiments. These experiments provided information which was used for experimental design of the 1985 field season.

The 1985 enclosure experiments were designed to test effects over time of logs of two tree species on the littoral

zooplankton population of Babine Lake. Treatments of tree species (pine and spruce) and log number (0, 1, 3 and 5 per enclosure) were examined using eight experimental enclosures with one replicate for each log number and tree species combination. Log treatments corresponded to loading densities of 0 (control), 0.0067, 0.0201 and 0.0335 cubic meters wood/ cubic meter water. Log number treatments were randomly assigned to the four enclosures used for each of pine and spruce treatments. Experiments were run twice, each for a length of 25 days, as outlined in Table 1. This experimental design did not provide true replication because experiments were repeated at different times (Hurlbert 1984).

Table 1. Experimental design and times of 1985 enclosure

Tree species	Duplicate	Experiment dates
pine	1	May 26 - June 20, 1985
pine	2	July 2 - July 27, 1985
spruce	1	June 2 - June 27, 1985
spruce	2	July 9 - August 3, 1985

Statistical analysis: The effects of the log treatments on enclosure zooplankton populations were assessed using SPSS:X MANOVA which is a generalized multivariate analysis of variance program that can analyze repeated measures designs (SPSS:X User's Guide 1983).

When the same experimental unit (in this case, zooplankton individuals within enclosures) is observed repeatedly under all treatments, the design is called repeated measures (Winer 1962). This approach provides control for individual differences among experimental units in their responsiveness to treatments (which may be a result of initial conditions) and provides a more powerful test of the effects of treatment factors when intersubject variability is high (Harris 1975).

I used a three-way MANOVA with repeated measures to test the June and July, 1985 data for the main effects of tree species, log number and zooplankton species and their interactions. Sample date is considered a repeated factor and this analysis does not depend on these dates being independent of one another (Winer 1962). Essentially, although all sample dates are used, the number of deg. of freedom does not increase. All data on zooplankton density were log(x+1) transformed before I conducted MANOVA tests; this reduced heterogeneity among groups and normalized distributions as determined by SPSS:X subcommands. A priori tests for differences between levels of significance for main effects were examined using the contrast (p=0.05) subcommand.

Simpsons's Diversity Indices (Washington, 1984) were calculated for the zooplankton in each enclosure for every sample date using the formula:

SIMPSON'S DIVERSITY INDEX (D)

$$D = 1 - \sum_{i=0}^{s} (p_i)^2$$

where s = no. of species

p_i = proportion of total no. of individuals of i^{th} species

2.3 RESULTS

2.3.1 Bark treatments (1984)

2.3.1.1 Water quality

Dissolved oxygen concentrations were dramatically reduced over time in bark treated enclosures (Figure 3), and this effect increased with the amount of bark added. The eight log treatment also depressed oxygen to < 1 mg/l within 14 days.

Lignin and tannin concentration of enclosure water increased over time in all bark treatments, particularly for 20 kg (Figure 4). The eight log treatment enclosure water reached a L-T concentration slightly above 4 mg/l which was lower than

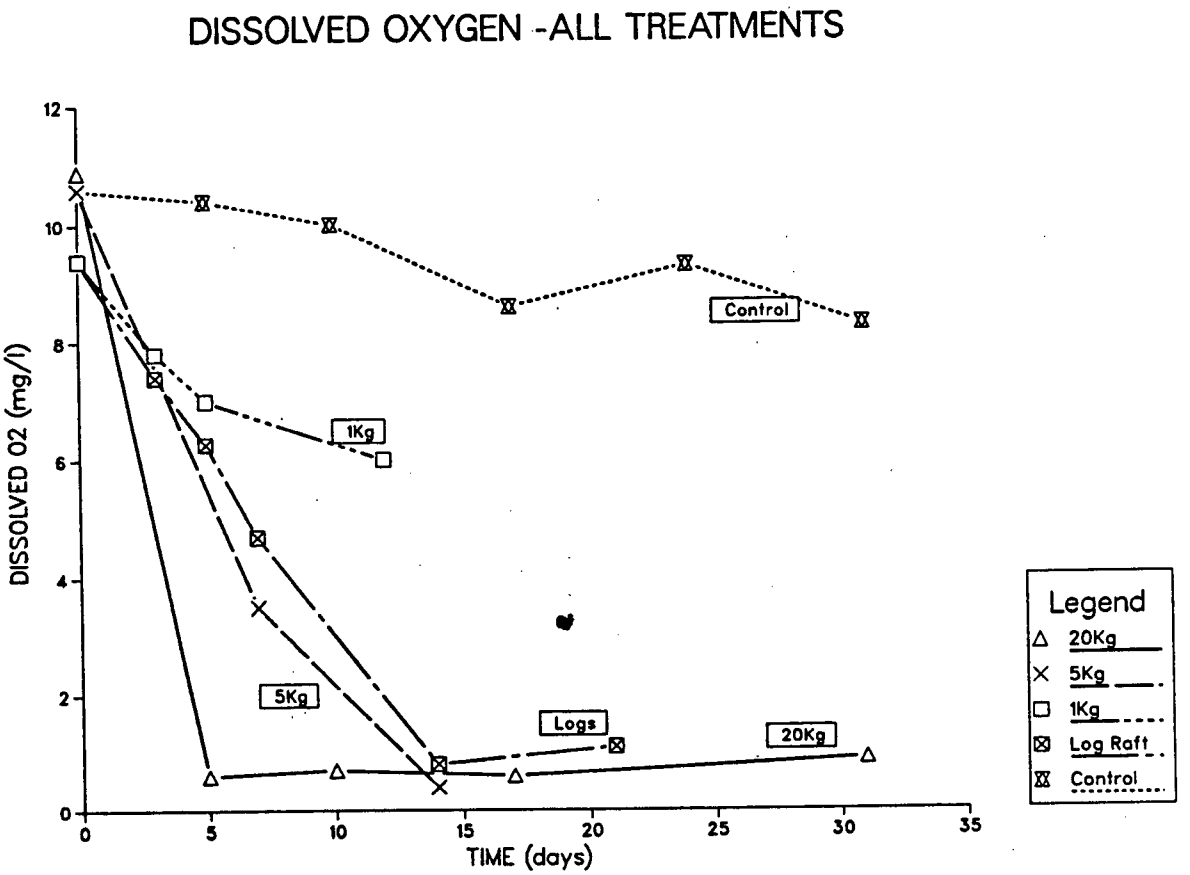

Figure 3. Dissolved oxygen concentrations (mg/l) in 1984 enclosure experiments under bark and log treatments (see legend).

measured for all bark treatments.

2.3.1.2. Zooplankton

Initial zooplankton densities among enclosures and between lake and enclosure populations were dissimilar, which makes comparisons difficult. However, several trends were clear from which decisions about the 1985 field season could be made.

Under the 20 kg bark treatment, total zooplankton abundance decreased significantly within the first week of exposure to the point where no zooplankton survived the last two weeks of the experiment (Figure 5). In the control enclosure, zooplankton populations were maintained throughout the experiment. Zooplankton densities decreased in the 5 kg bark treatment to levels below those measured in the control. However, in the 1 kg treatment, where initial conditions were dissimilar, control zooplankton populations declined substantially during the experiment and the treatment population increased slightly. Zooplankton populations exposed to the 8 log treatment declined in density to zero, but control populations also decreased to extremely low levels.

2.3.2 Log treatments (1985)

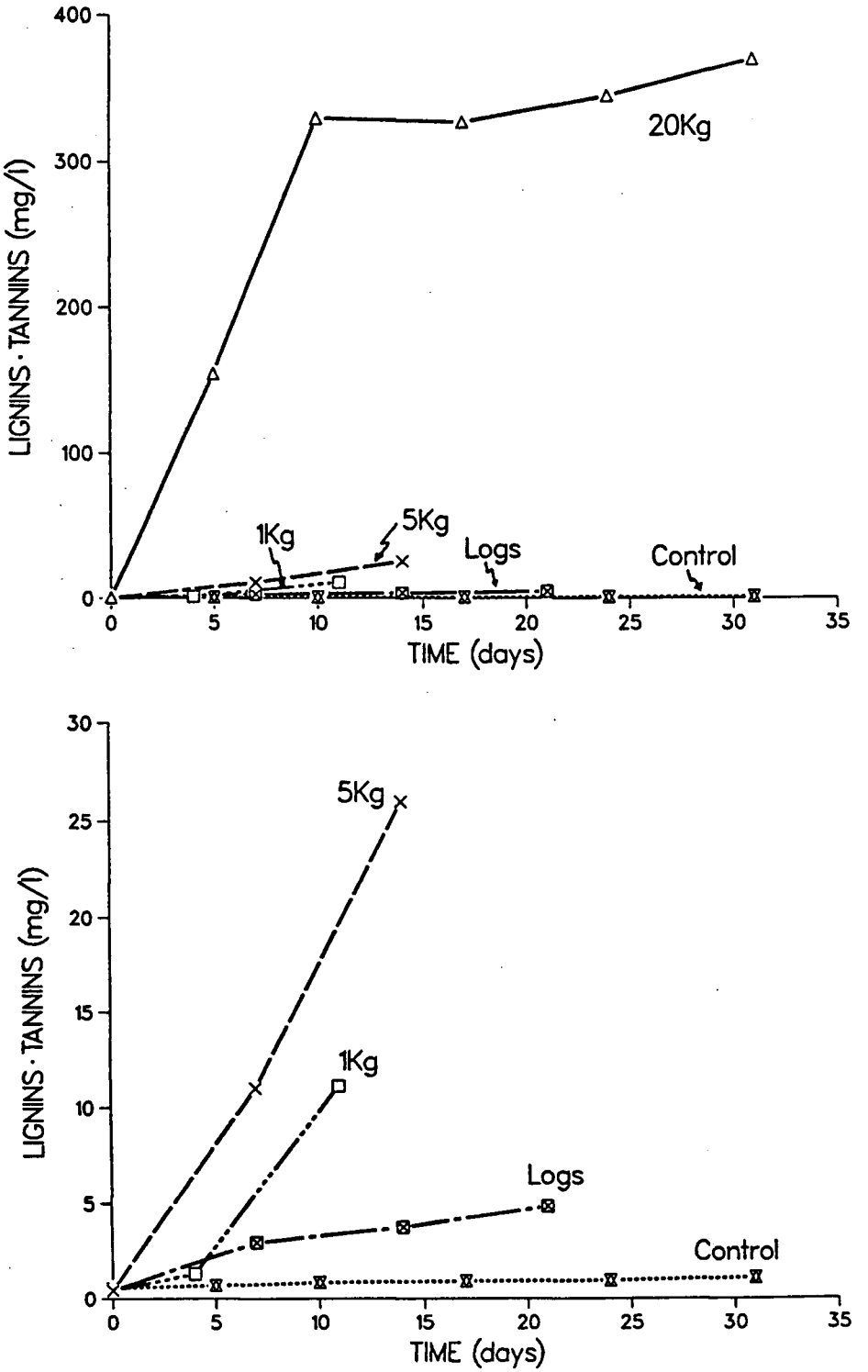

Figure 4. Lignin and tannin concentrations (mg/l) in 1984 enclosure experiments under for all treatments (top) and all treatments excluding 20 kg bark load (bottom).

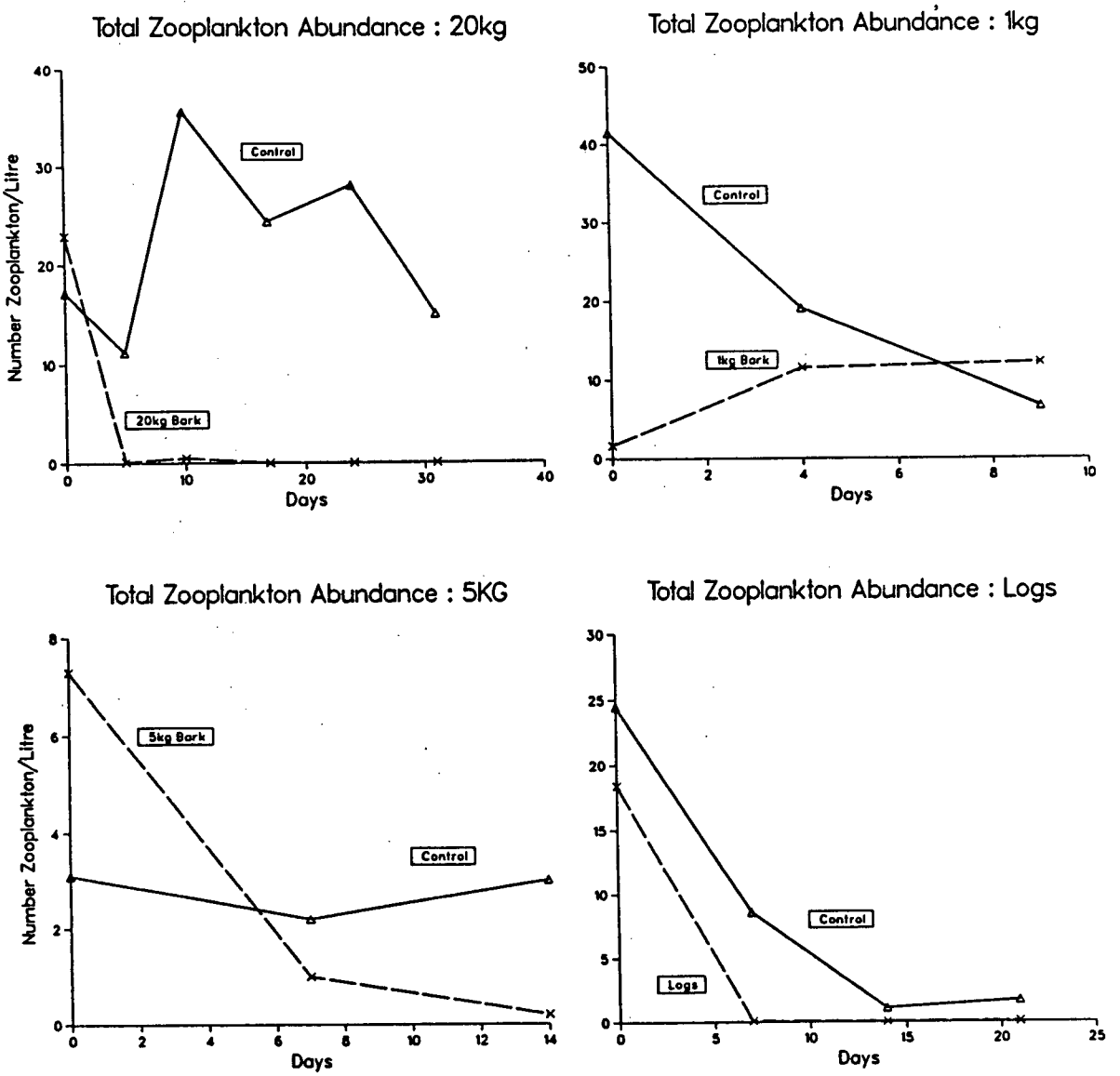

Figure 5. Total zooplankton concentrations (no./l) over time for control enclosures and enclosures treated with 20 kg, 5 kg, and 1 kg of bark, and a log raft in 1984.

2.3.2.1 Water Quality

Temperatures in the enclosures stayed within 1 degree C of lake temperatures adjacent to the enclosures. Water temperature during the June, 1985 experiments ranged from 4.1 to 12.0 deg. C; in the July, 1985 experiments water temperature ranged from 13.0 to 17.5 deg. C.

Dissolved oxygen concentration was reduced in log treated enclosures, compared to the lake and controls, but to a far greater degree in July than in June (Table 2). Oxygen depletion increased significantly with time and with the number of logs per enclosure. Pine log treatments reduced dissolved oxygen concentrations to slightly lower levels than spruce, particularly in July.

The oxygen saturation percentages (measured dissolved oxygen concentration divided by oxygen saturation concentration (Y.S.I. instrument book) for the appropriate water temperature) are presented in Figure 6. Oxygen saturation percentages remained near 100% in control and 1 log enclosures, but declined to 80% and <40% in 5 log treatments for June and July, respectively.

Lignin and tannin concentrations of enclosure water increased significantly with time and with the number of logs per treatment (Figure 7) and there was no apparent differences

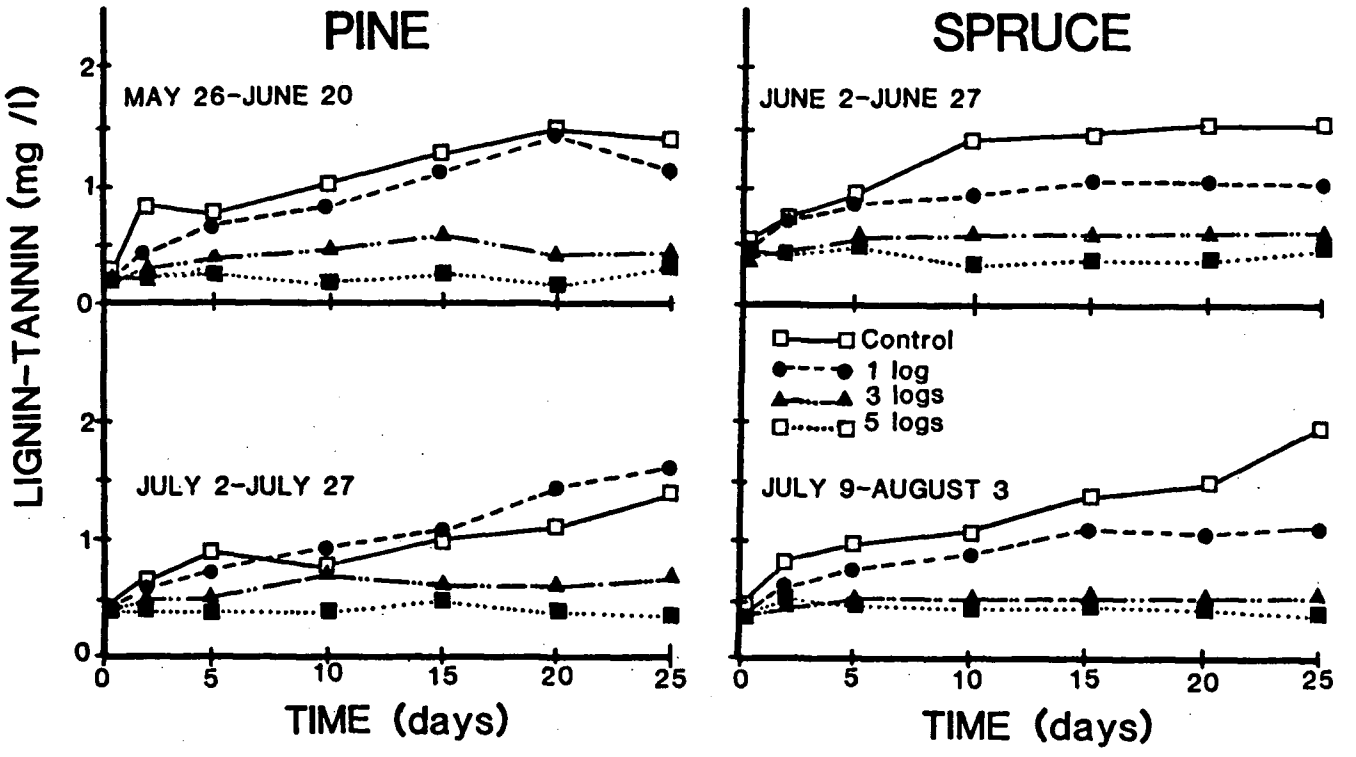

Figure 6. Oxygen saturation (%) in 1985 enclosure experiments for log treatments and controls (see legend).

Table 2. Dissolved oxygen concentrations (mg/l) in experimental enclosures for log treatments, lake, and controls (treatment; C = control, 1 = 1 log, 3 = 3 logs, 5 = 5 logs

		NUMBER OF DAYS					
Treatment	Duplicate	0	5	10	15	20	25
lake	1	10.7	11.3	11.0	10.4	10.2	11.1
pine C	1	11.6	11.1	11.4	10.7	10.4	9.8
pine 1	1	12.4	10.8	11.2	10.1	9.5	9.6
pine 3	1	11.1	10.0	9.3	8.8	8.6	9.5
pine 5	1	11.5	9.7	8.4	8.2	8.0	9.1
lake	2	10.1	9.4	9.4	9.7	9.3	8.9
pine C	2	10.3	9.4	9.3	9.6	9.0	9.0
pine 1	2	10.4	9.2	8.5	8.9	7.8	8.0
pine 3	2	10.3	8.2	6.8	5.8	5.1	3.0
pine 5	2	10.3	7.8	6.2	5.5	4.2	2.5
lake	1	10.6	11.4	10.6	11.0	11.3	11.2
spruce C	1	10.8	11.4	11.3	11.4	11.4	10.6
spruce 1	1	10.6	11.2	10.8	11.2	10.5	9.6
spruce 3	1	11.1	10.7	9.9	10.1	9.8	9.2
spruce 5	1	11.2	10.6	9.4	9.8	9.6	9.0
lake	2	9.7	9.8	10.1	9.3	9.5	9.1
spruce C	2	9.6	9.8	9.8	9.2	9.5	9.1
spruce 1	2	9.6	9.7	9.4	8.7	8.5	8.2
spruce 3	2	9.6	9.1	8.0	6.7	7.3	6.0
spruce 5	2	9.7	8.8	7.6	5.6	5.3	3.4

between June and July experiments or pine and spruce log treatments. The increases in L-T concentration were not large; the greatest change was from 0.45 to 2.00 mg/l in 25 days for the 5 log spruce treatment. Control enclosures and lake water had similar and L-T concentration (<0.50 mg/l) over the 25 day experiments.

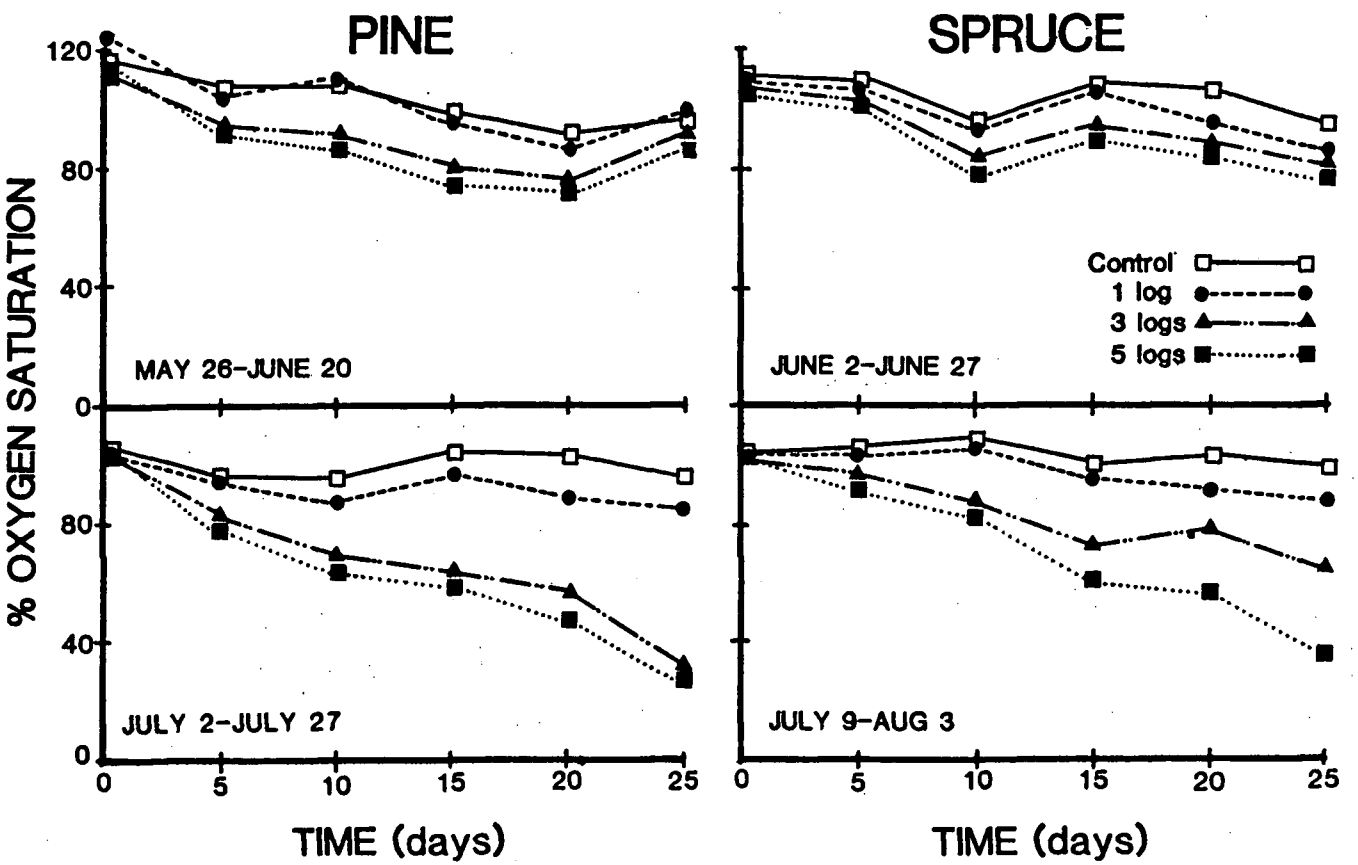

Figure 7. Lignin-tannin concentrations (mg/l) in experimental enclosures (1985) for log treatments and controls (see legend).

2.3.2.2 Zooplankton

Juvenile copepods were numerically dominant in both the lake and the enclosures, followed by Diaptomus spp., Diacyclops thomasi, Daphnia spp., and Bosmina coregoni. This zooplankton assemblage is typical of Babine Lake (Rankin 1977, Levy et al. 1984).

That initial densities of zooplankton in the enclosures were approximately 30% lower, on average, than in the lake (Figures 8 to 13) does not detract from the validity of the experiments. Lake zooplankton densities fluctuate both within and between years over a range which includes levels measured in the enclosures. The overall trends observed for the lake zooplankton (Figures 8 and 9) were the same as observed in the June and July control enclosures, which indicates the similarity between enclosures and the natural lake.

Perhaps most important, the enclosures were similar to one another. Similarity in species composition between enclosures was satisfactory (Table 3); percent similarity (Renkonen's Index; Wolda 1981) between enclosures ranged from 77% to 97%. Similarity between control enclosures stocked at different times ranged from 55% to 92%.

Zooplankton abundance decreased in log enclosures relative to control enclosures during all experiments (Figures 10 to 13;

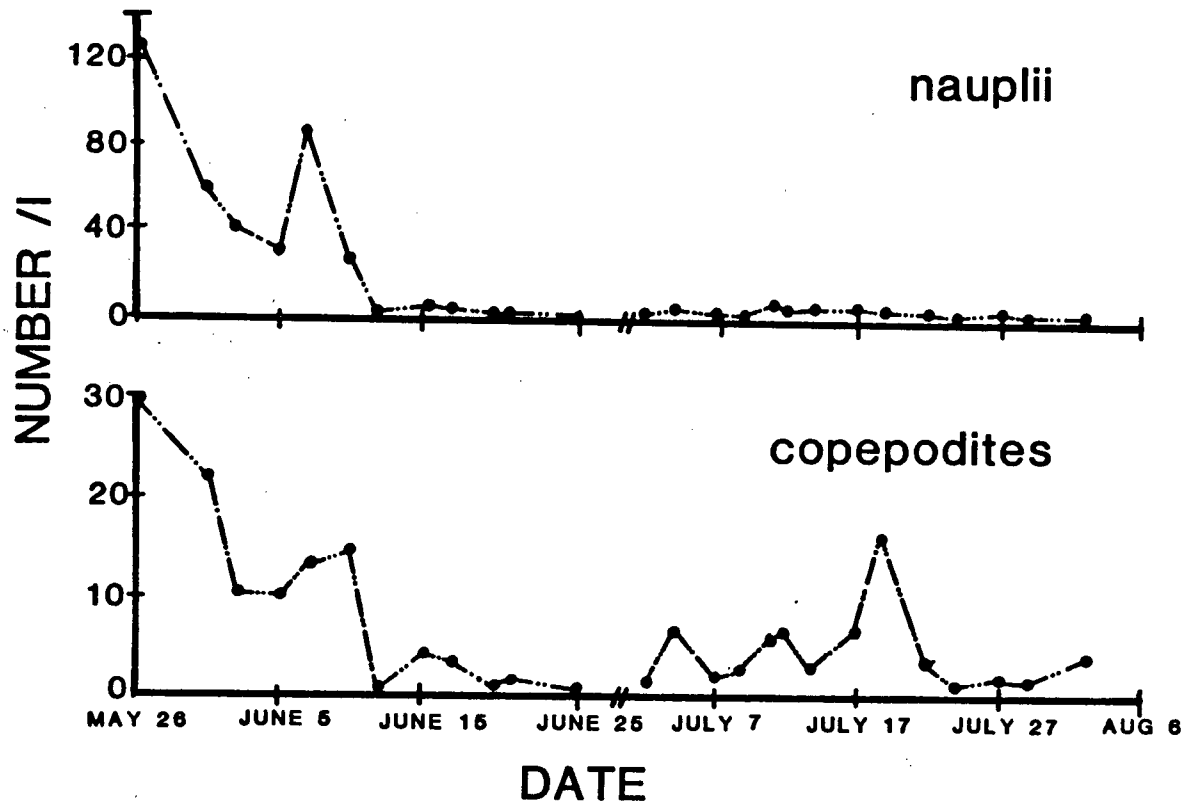

Figure 8. Nauplii and copepodite densities (no./l) in the lake (z = 3 m) from May 26-August 3, 1984.

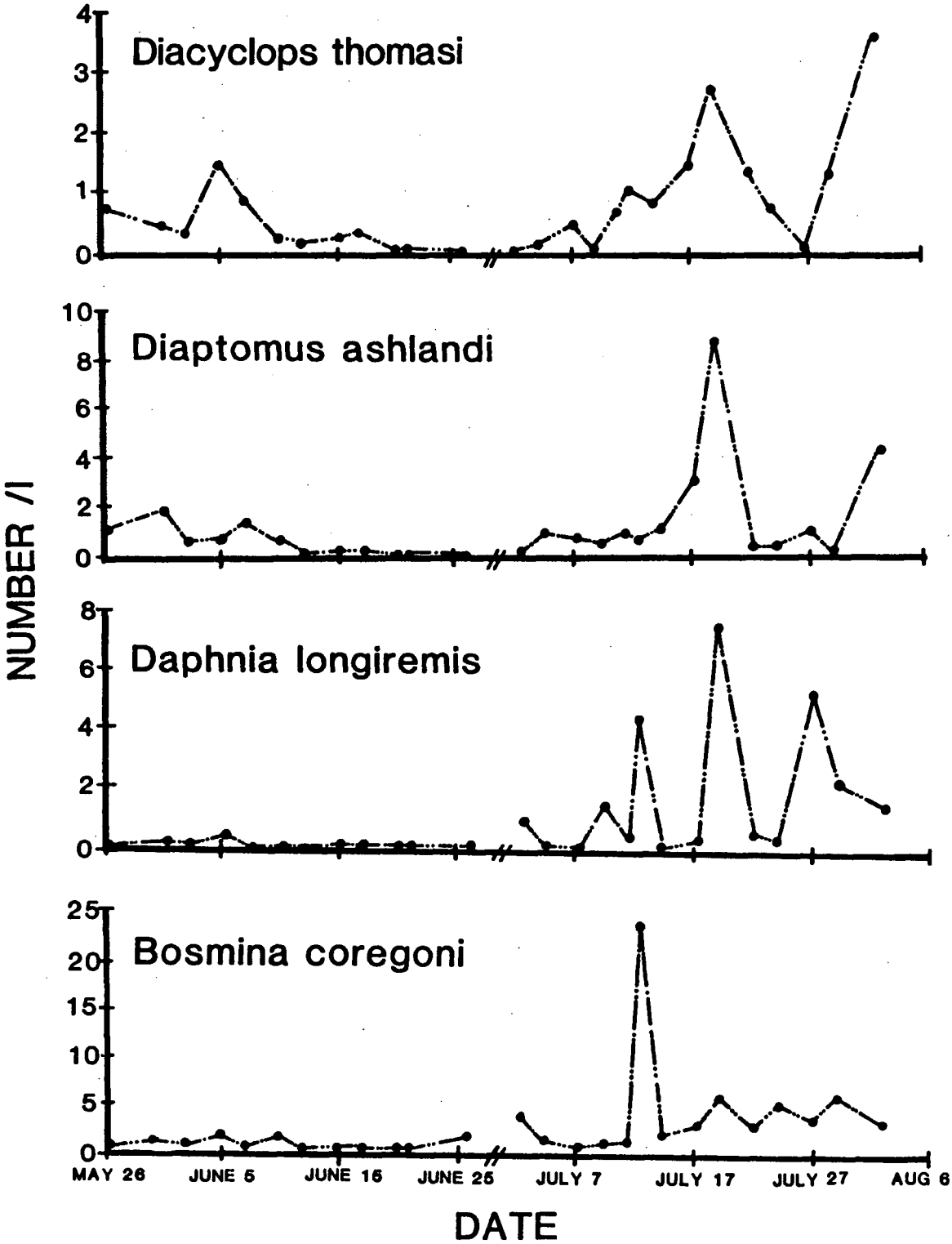

Figure 9. Densities (no./l) of Diacyclops thomasi, Diaptomus ashlandi, Daphnia longiremis, and Bosmina coregoni in the lake (z = 3 m) from May 26-August 3, 1984.

Table 3. Percentage similarity (Renkonen's index) matrix of initial zooplankton populations in 1985 enclosure experiments.

Tree-duplicate		3 log	5 log	control	lake
pine-1	1 log	95	97	87	92
	3 log	-	96	88	88
	5 log	-	-	87	92
	control	-	-	-	82
pine-2	1 log	89	88	83	70
	3 log	-	93	86	61
	5 log	-	-	90	57
	control	-	-	-	52
spruce-1	1 log	94	93	90	91
	3 log	-	96	86	85
	5 log	-	-	84	85
	control	-	-	-	81
spruce-2	1 log	89	90	81	70
	3 log	-	96	77	72
	5 log	-	-	78	69
	control	-	-	-	77

Table 4). The pine and spruce log treatments did not significantly differ in their effect on zooplankton abundance ($p=0.130$). However, log treatments reduced zooplankton density in relation to the number of logs and this effect was significant ($P=0.041$). Comparison among the means for log treatments revealed that the five log treatment was significantly different from the control ($p=0.007$) and that the three log treatment approaches statistical significance ($p=0.053$).

There were differences in the way that zooplankton species

Source of var.	Sum squares	df	Mean square	F	Sig. of F
Within cells	22.04868	64	0.34451		
Tree species	0.81042	1	0.81042	2.35237	0.130
Log number	3.01298	3	1.00433	2.91523	0.041
Zoop. species	38.71218	7	5.53031	16.05266	0.000
Tree*Log	0.41736	3	0.13912	0.40382	0.751
Tree*Zoop	2.21280	7	0.31611	0.91758	0.499
Log*Zoop	2.24914	21	0.10710	0.31088	0.998
Tree*Log*Zoop	1.31333	21	0.06254	0.18153	1.000

Table 4. Three way MANOVA (with repeated measures) of zooplankton abundance in log treated enclosure experiments.

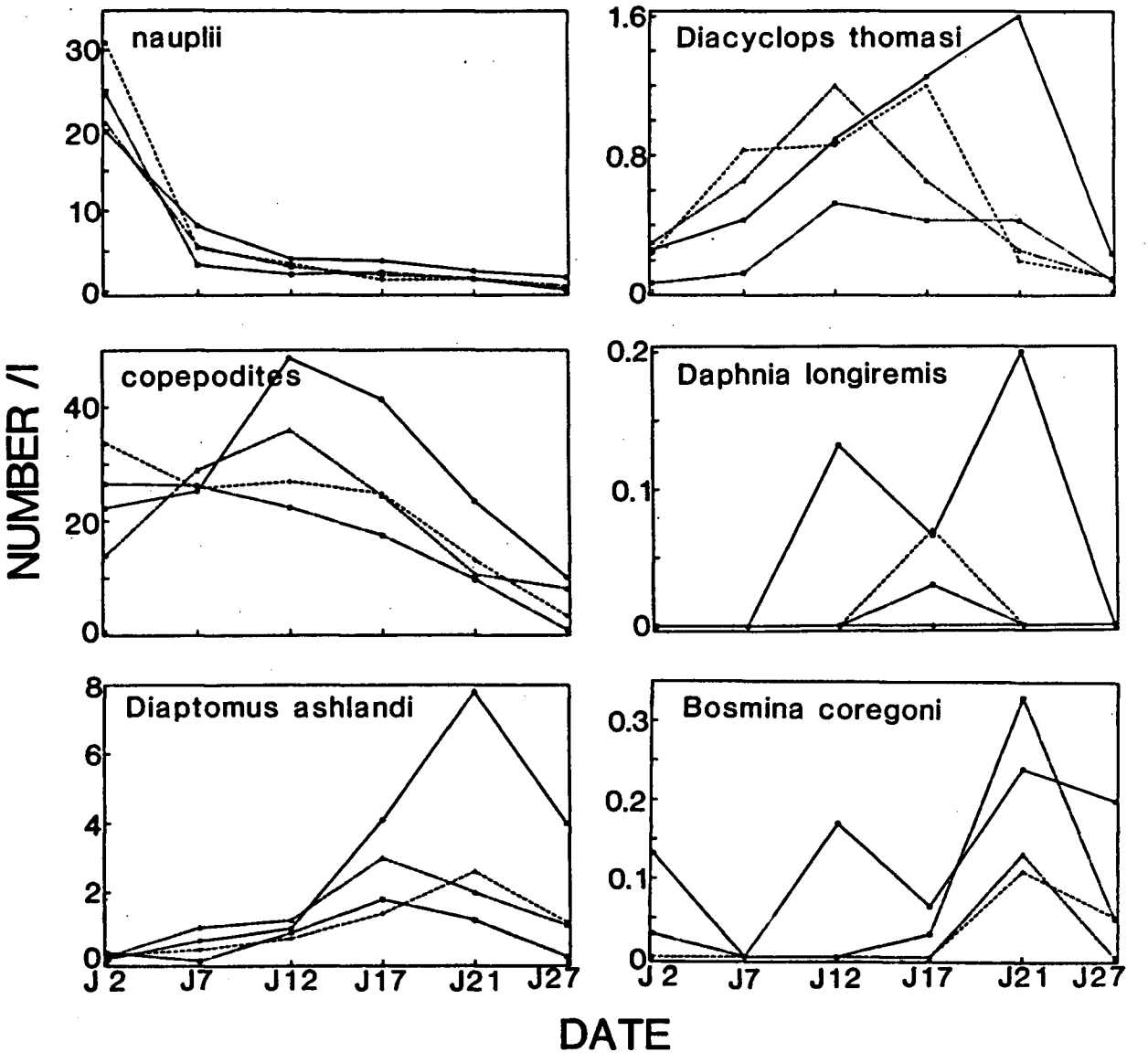

Figure 10. Zooplankton densities for common species in spruce log treated and control enclosures (June 2-June 27, 1985) (see legend).

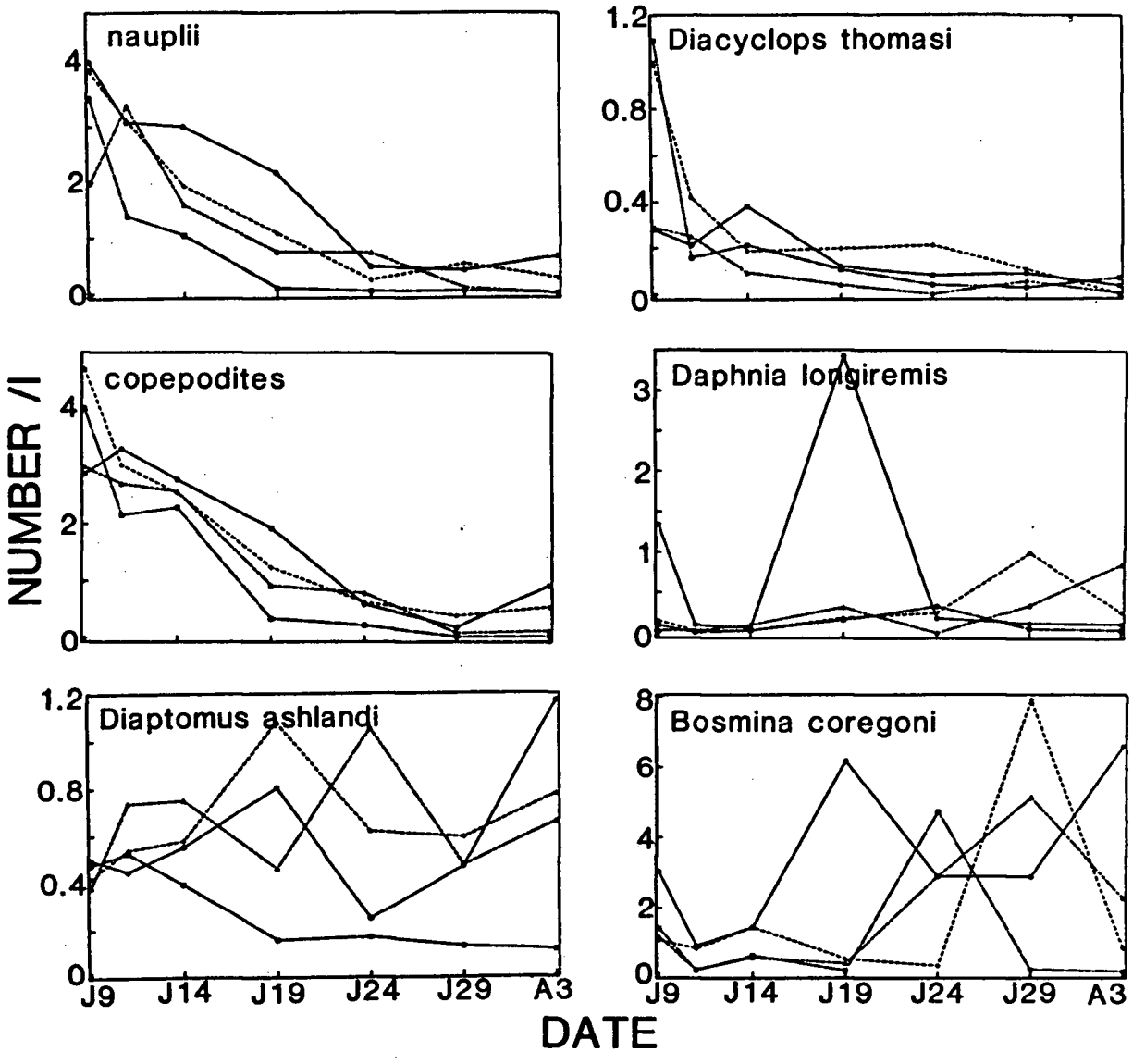

Figure 11. Zooplankton densities for common species in spruce log treated and control enclosures (July 9-August 3, 1985) (see legend).

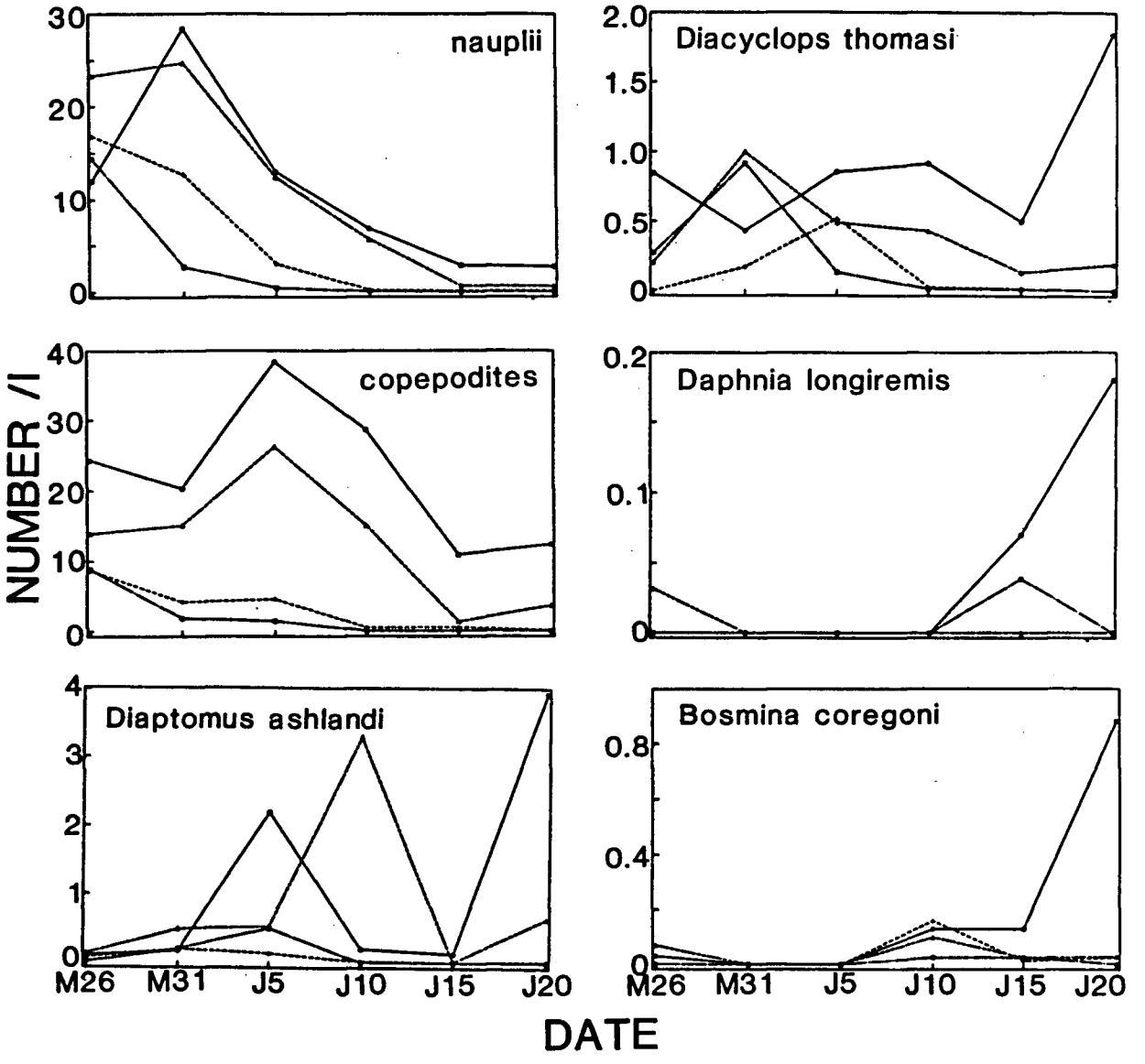

Figure 12. Zooplankton densities for common species in pine log treated and control enclosures (May 26-June 20, 1985) (see legend).

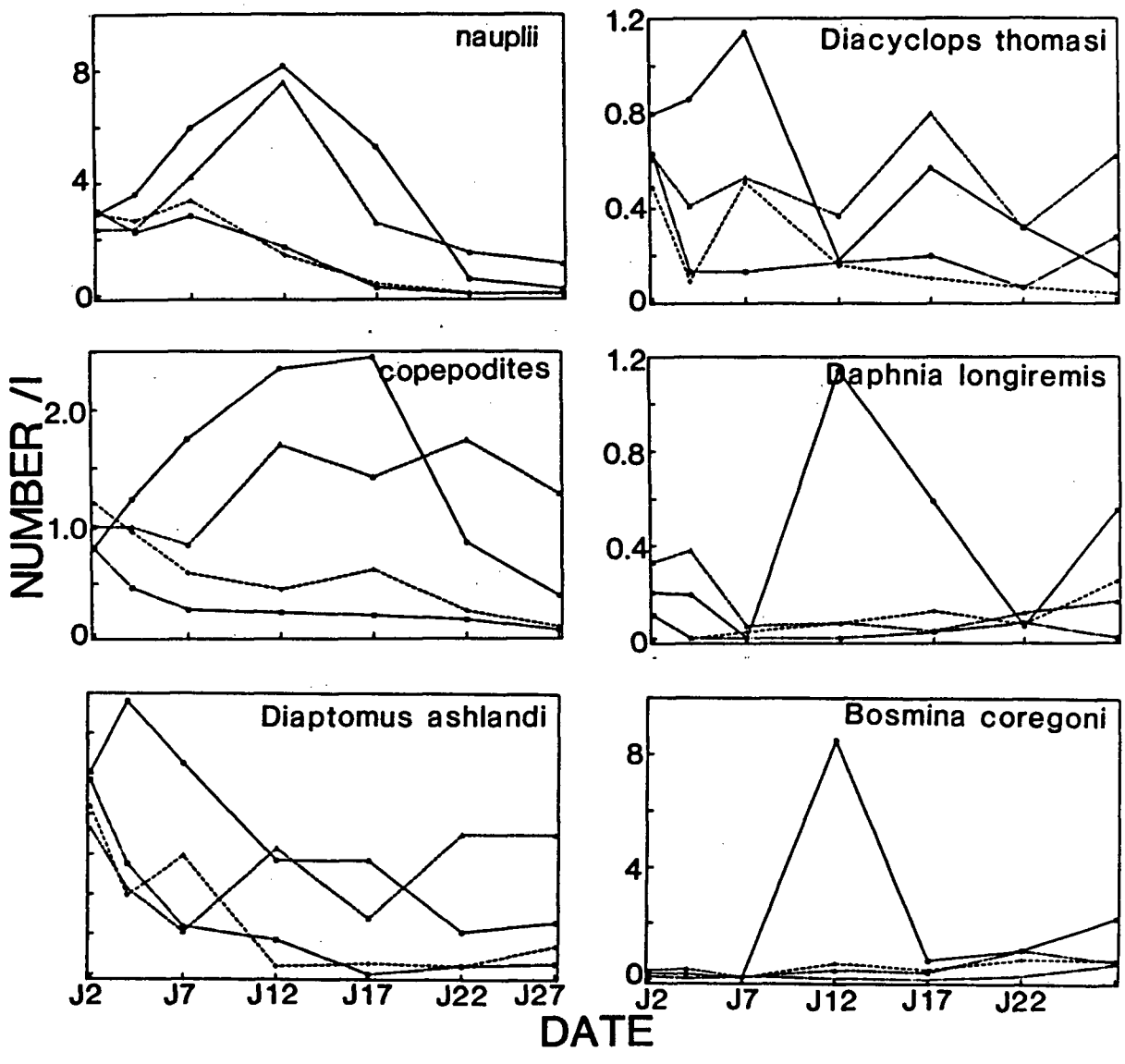

Figure 13. Zooplankton densities for common species in pine log treated and control enclosures (July 2-July 27, 1985) (see legend).

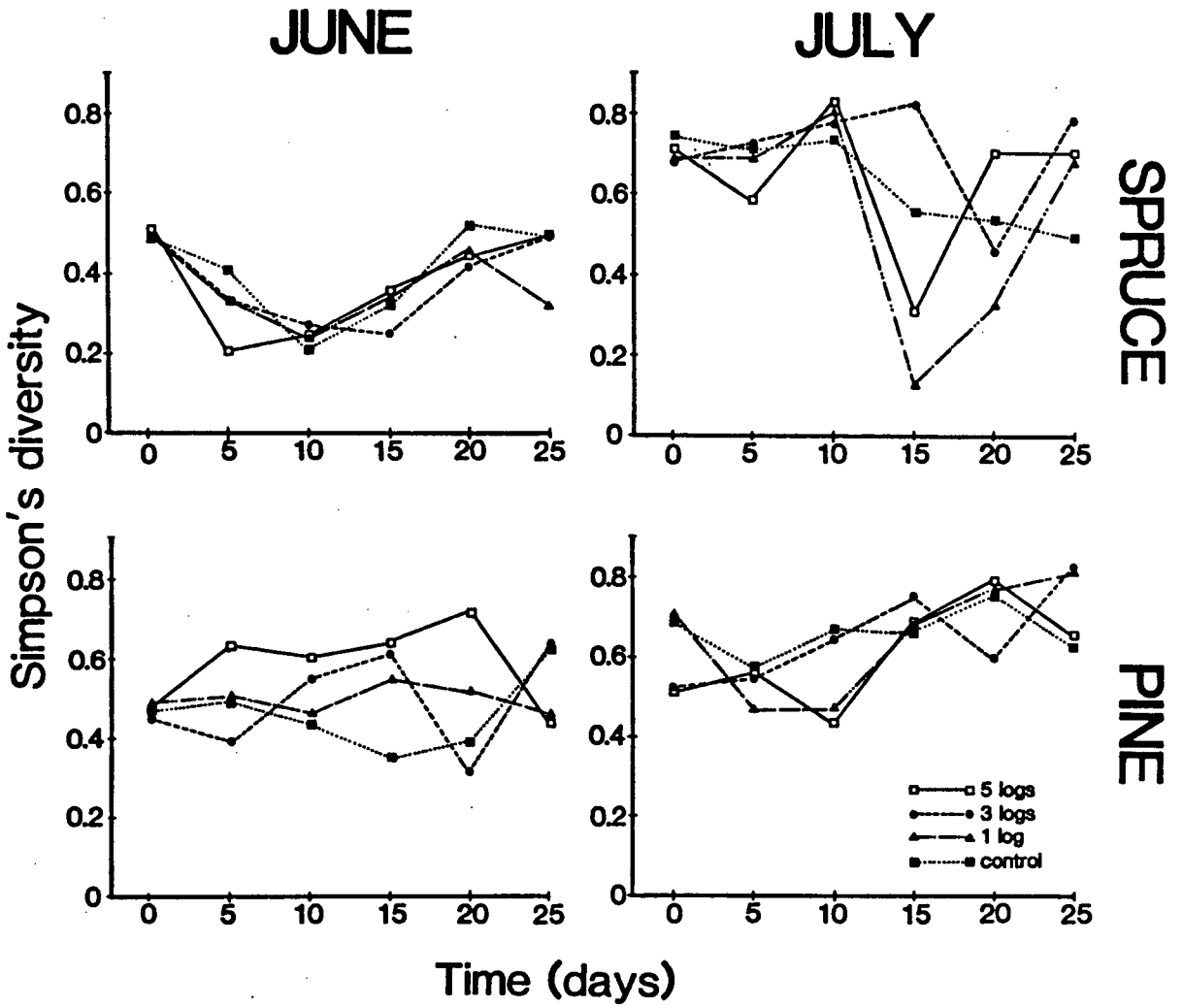

Figure 14. Diversity indices (Simpson's D) for log treated and control enclosures under all treatments (see legend).

and life stages responded to log treatments ($P<0.001$). Copepod juvenile life stages (nauplii and copepodites) were more deleteriously affected than adult zooplankton and this effect was statistically significant ($P<0.001$). The effect of log treatment on the adult zooplankton was not statistically significant. There were no significant interactions between any of the main effects (Table 4).

Simpson's diversity indices for log treated enclosures showed no consistent pattern, compared to controls, for zooplankton populations over the course of the 25 day experiments (Figure 14). With a maximum H of 0.90, diversity values ranged between 0.12 and 0.82.

2.4 DISCUSSION

Based on the water quality and zooplankton abundance results obtained from bark additions to enclosures during the 1984 field season, I was unsatisfied with the applicability of bark treatment results to the field situation where logs are the treatment factor. Firstly, the water quality conditions created by bark additions were unrealistic relative to field measurements, particularly for the 20 kg and 5 kg bark treatments. L-T levels are < 2 mg/l in the Morrison Arm log storage area during June 1985 (Levy et al. 1985b) whereas the lowest bark treatment L-T concentration was 11 mg/l after two weeks. I decided to use log additions for treatments, based on the promising results (< 5 mg/l) of the eight log treatment. Secondly, in 1984, initial zooplankton populations were not sufficiently similar between treatments and controls to draw conclusions about treatment effects. As a result, a zooplankton stocking method for enclosures was adopted for the 1985 field season. However, the qualitative trends determined for bark treatments (1984) are consistent with those measured more quantitatively in log treatments (1985).

2.4.1 Water Quality

Log treatments significantly reduced dissolved oxygen concentrations over the course of the enclosure experiments with oxygen depletion increasing with time and number of logs. This

result agrees with other water quality measurements made by Wentzell (in prep.) (eg: C.O.D. and T.O.C). It is clear from the literature (Graham 1970, Sproule and Sharpe 1970, Schaumburg 1973) that under static conditions, leachate concentration increases with both time and the ratio of wood/water volume resulting in associated water quality changes, as described for the enclosure experiments by Wentzell (in prep.). There were no clear differences between spruce and pine log treatments in enclosure water quality.

There were significant differences in the dissolved oxygen results for June and July experiments (Wentzell, in prep.). During June, for spruce and pine log treatments, dissolved oxygen levels were reduced but remained above 75% oxygen saturation, which would be adequate for fish survival. However, during the July experiments, oxygen concentration dropped below 40% oxygen saturation (lethal levels for fish; Davis 1975) for the 3 and 5 log pine treatments and the 5 log spruce treatment. It is not clear why there is such a marked difference in oxygen levels in the June and July experiments. Factors which may differ between the two time periods and contribute to differences in dissolved oxygen are temperature, leachate production and bacterial biomass, respiration and activity. Mean temperatures of 9.4 and 15.3 deg. C for the June and July experiments, respectively, may result in different leaching rates. However, there was no difference in lignin and tannin concentration between the two months, which suggests that the

oxygen differences are not a result of temperature and leachate production. Wentzell (in prep.) supports this conclusion with C.O.D. and T.O.C. data which do not significantly differ between the two months. Also, lignins and tannins are large complex compounds which are not easily broken down (Schaumburg 1973) so it is very unlikely that their concentration in July is low due to degradation. There is still a possibility that the water soluble extracts from logs cut at the two different times were responsible for differences in oxygen depletion. The chemistry of wood leachates is extremely complex and there may be seasonal changes in the chemical constituents and their concentrations which were undetectable by the various measurements made by Wentzell (in prep.). For example, wood sugars may be rapidly degradable by microbes with a high "turnover", resulting in little contribution to T.O.D. and C.O.D. despite high microbial activity (Wentzell, in prep.).

The most likely explanation for the differences in dissolved oxygen between June and July experiments is oxygen depletion as a result of temporally varying chemical and biological oxygen demands. Work by Wentzell (in prep.) on the C.O.D. of log leachate found no difference between the two months. Biological oxygen demand, although not directly measured, is related to bacterial number and activity which was determined over the course of all enclosure experiments (Wentzell in prep.). Based on the oxygen measurements, I would expect bacterial activity rate and/or numbers to have been

higher in July. This was not found to be the case; bacterial number and activity rates in July were similar to those in June (Power and Wentzell 1985, Wentzell in prep.), however, bacterial biomass was significantly lower in July. Hall et al. (in prep) suggest that the smaller bacterial biomass in July had a better ability to take up substrate due to a higher metabolic activity per organism, resulting in lower dissolved oxygen concentration. This hypothesis is supported by their bacterial uptake kinetic data.

The log treatment enclosure experiments demonstrate that water quality is extremely sensitive to log loading densities. The resulting water quality changes can produce conditions which are lethal for fish and many invertebrates. A major problem in most laboratory experiments which examine wood leachates and water quality is that loading densities are orders of magnitude higher than in field situations and produce unrealistic water quality levels. For example, Graham (1970) and Atkinson (1971) submerged logs in 50 l aquaria, producing waters with a C.O.D. as high as 600 mg/l. The maximum field measurement of C.O.D. in the Houston Forest Products log storage site was 50 mg/l (Levy et al. 1985b). The maximum C.O.D. for the enclosure experiments was 70 mg/l in the 5 log treatments (Power and Wentzell, 1985). In the enclosure experiments, lignin and tannin concentrations are similar to concentrations measured in the log storage site (Levy et al. 1985b). Agreement between experimental enclosure and field results is striking; static conditions in the log boom

result in water quality similar to that measured in enclosure experiments. The log loading density for the Morrison Arm log storage site, as calculated in Chapter 3, is 0.33 cubic meters/cubic meter water, which is higher by an order of magnitude than the highest log loading density (0.0335 cubic meters wood/cubic meter water) used in enclosure experiments. Allowing for water circulation in the log storage area, the use of enclosures proved to be an effective way to experimentally simulate log storage conditions and examine their effects on water quality. It follows that the biotic component of the enclosures will be under physical and chemical conditions similar to those which occur in the field as will be examined in the General Discussion.

2.4.2 Zooplankton

Laboratory tests have shown that log leachates are acutely toxic to several aquatic organisms such as salmon eggs and fry (Servizi *et al.* 1971, Pease 1974, Peters *et al.* 1976) and caddisfly larvae and mayfly nymphs (Peters *et al.* 1976), but toxicity to zooplankton has never been examined. The enclosure experiments were conducted at low, realistic log loadings with the possibility of measuring responses at the community level. Log leachates reduced zooplankton density in all treatments over time, with 5 log experiment populations being most severely affected. This pattern is linked to water quality, as zooplankton respond to changes such as reduced dissolved oxygen

and increases in leachate concentration. Sprague (1970) emphasizes that environmental conditions (in this case dissolved oxygen) may greatly modify toxicity and discusses several examples from the literature. Wentzell (in prep.) monitored chlorophyll a in enclosures and found that oxygen depletion and leachates generated by logs did not adversely affect algal standing crops. The algae consumed by zooplankton would have to be monitored to test the hypothesis that zooplankton numbers were reduced through food limitation.

No significant difference in zooplankton response to pine and spruce log treatments was found. This is reasonable given that there were no measured differences in water quality between treatments. Differences in toxicity between other tree species have been shown by several researchers (Atkinson 1971, Buchanan et al. 1976) but for higher leachate concentrations than occur in most log handling facilities. These studies obtained toxic leachates through extraction processes which involve high loading densities and grinding of wood chips in water. Whole log experiments produce more dilute leachates which are not usually measurably toxic to salmonids (Atkinson 1971, Schaumburg 1973, Pease 1974).

In enclosure experiments, only reductions in abundance of juvenile calanoid and cyclopoid copepods (nauplii and copepodites) were statistically significant. Early life history stages tend to be the most susceptible to toxic substances for

both fish and invertebrates (APHA *et al.* 1985, McKim 1985). Potentially, for young sockeye salmon feeding in the log storage area at Morrison Arm, there may be a reduction in local food levels since fry feed mainly on copepodites (Levy *et al.* 1984). Negative effects of wood leachate on zooplankton were determined at realistic leachate concentrations.

At the community level, differences in species diversity between log treated and control enclosures was predicted, based on the premise that community structure would change as a result of log storage. According to Washington (1984), stresses applied to a community as a result of pollution should be reflected in changes to the community structure, of which species diversity may be an indicator. Species diversity has been used in a few other studies to evaluate the impact of a toxicant upon a zooplankton community. For example, Kaushik *et al.* 1985, documented reductions in species diversity of zooplankton in response to application of permethrin (an insecticide) to enclosures.

The changes in species diversity which occurred in zooplankton communities in log enclosures showed no consistent pattern, which could be due to a number of reasons. If species diversity is a measure of community structure, than this result suggests a lack of effect of log leachate at the zooplankton community level; however, there are several alternative hypotheses. Diversity indices may not be a good or sensitive

measure of community structure. Also, responses by each zooplankton taxa may have been equal, resulting in no net change in relative abundance, upon which diversity is based. There is limited scope for determining significant changes in diversity because species richness in Babine Lake is low and small changes may not be detected by a diversity index.

Similar results at the community level using similarity indices were obtained for zooplankton in selenium enclosure experiments (Salki et al. 1985) where no acute or chronic effects were measured, despite indications from laboratory bioassays that selenium was toxic to zooplankton.

3. FIELD STUDY

3.1 INTRODUCTION

Utilization of the littoral zone by sockeye fry during their first few weeks of life has been widely observed (McDonald 1969, Levy et al. 1984) and this would seem to be the zone most severely affected by log storage on Babine Lake. Impacts such as bark accumulation, sediment compaction, reduction in benthic prey organisms and reduced water quality are described for log storage areas (Pease 1974, Conlan and Ellis 1979, Toews and Brownlee 1981). However, there are few studies which experimentally examine the direct effects of log storage on fish (Sedell and Duval 1985). This is partly due to the logistical problems associated with carrying out research in log storage areas, where, for example, submerged logs and debris make many fish capture methods impractical.

Given that sockeye fry utilize littoral habitat (McDonald 1969) for their first few weeks of their lake residence, there is concern that log handling activities during that period may detrimentally affect these juvenile salmon. Sockeye fry leaving Morrison River enter Babine Lake only 1 km from the Houston Forest Products dump site (Figure 15). Morrison Arm is essentially a migration corridor, along which fry must travel from the Morrison River to the main body of Babine Lake. Levy et al. (1985b) have examined relative utilization of the littoral

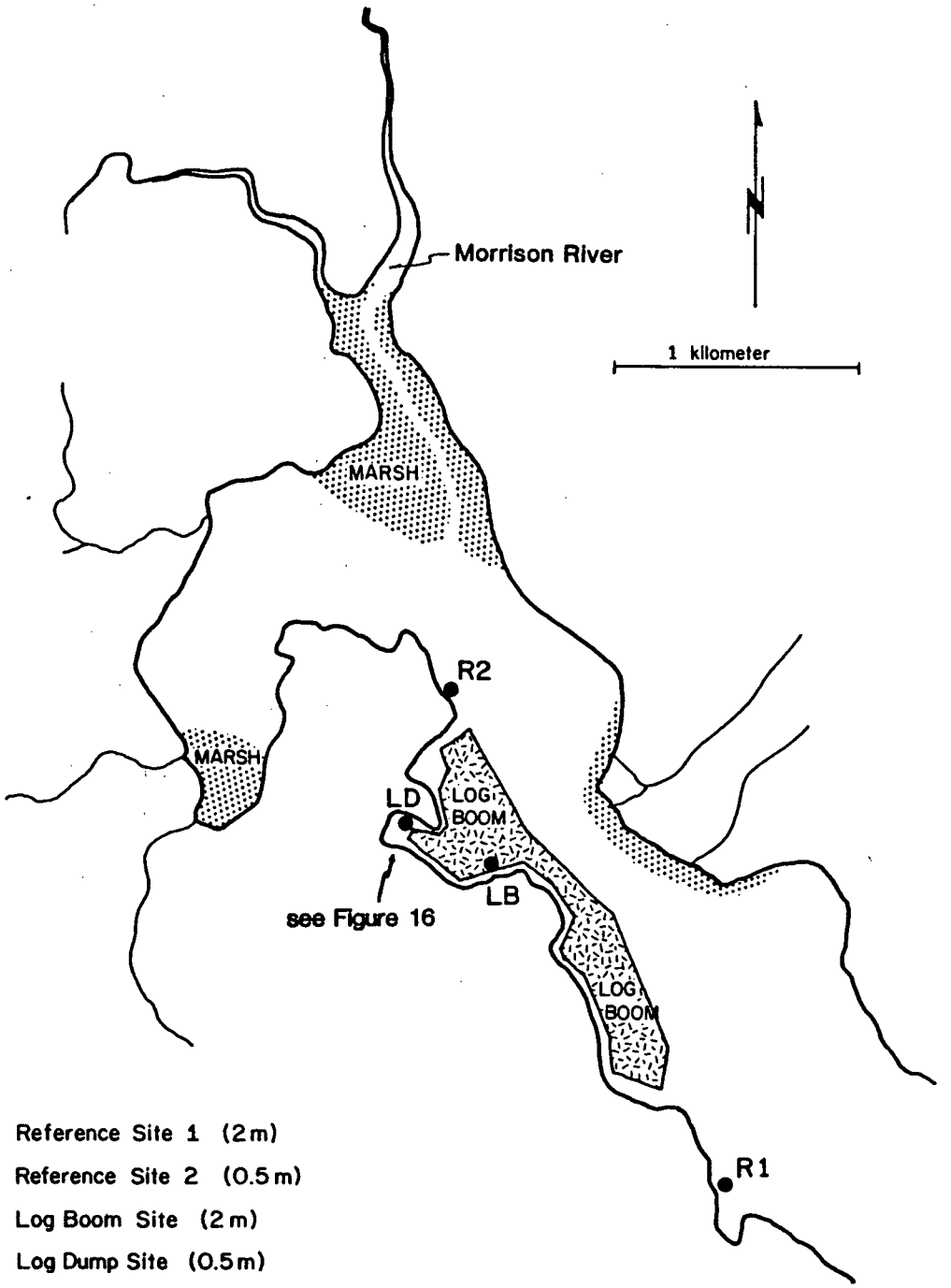

Figure 15. Map of Morrison Arm, Babine Lake, B.C. Locations of fry feeding experiments are marked and described in the legend. As indicated, see Figure 16 for a map of the log dump bay.

zone along Morrison Arm and determined that the highest numbers of sockeye fry were onshore at the head of the arm. They likely shift into the pelagic zone as they migrate towards the main basin of Babine Lake. Deterioration of water quality or habitat caused by log storage may inhibit or change migration of inshore sockeye fry within the arm. If log booms affect the zooplankton community in areas where sockeye fry feed, the diet of fry could be altered. Dietary change may detrimentally affect their growth and survival during their first few weeks of life. To look at these concerns, this field study examines (1) water residence time and water quality in log boom and control sites, (2) differences in food supply and diet for sockeye fry held in enclosures in log storage and control sites.

3.1.1 Water residence and water quality

The forest industry had acquired approximately 950 coastal leases and reserves by 1983 (Edgell and Ross 1983), 64% of which were used for log storage alone. These leases have the common characteristic of being sheltered from wind, waves and currents. Houston Forest Products' log storage site at Morrison Arm, Babine Lake is no exception; this site is in the most protected bay (Figure 15) within the region meeting the criteria for a log dump site. In addition, the presence of log booms decreases lake surface area per unit volume water, consequently further reducing the potential for mixing of the water column by wind action.

Obviously, lethal levels of oxygen (< 4 mg/l for salmonids) are limiting to fish (Davis 1975). Critical oxygen levels can also be defined as those which cause sublethal effects such as loss of equilibrium and respiration stress (Davis 1975). For young sockeye salmon Brett (1964) determined that reduced levels of oxygen detrimentally affected swimming performance which may reduce the ability of fry to capture prey. Invertebrates may also respond negatively to low oxygen levels, although they usually have higher tolerance levels than fish (Davis 1975).

Restricted flushing has been linked to deterioration of water quality (Pease 1974), particularly depleted oxygen conditions, which are caused by high B.O.D. (biological oxygen demand) and C.O.D. (chemical oxygen demand) as demonstrated in many laboratory studies (Graham 1970, Sproule and Sharpe 1970, Atkinson 1971, Toews and Brownlee 1981).

Water quality has been closely monitored in log storage sites (Levy et al. 1984, 1985a, 1985b) on Babine Lake and in the present study during 1985 within the Morrison Arm log handling site. During the 1984 field season, depleted oxygen conditions were not observed in the Morrison Arm log handling area (Levy et al. 1985a). Several studies (Schaumburg 1973, Pease 1974, Duval and Slaney 1980) conclude that water flow, in the majority of cases, is enough to prevent either accumulation of toxic leachates or any reduction in oxygen which could adversely

affect fish. Duval and Slaney (1980) could find no record of this kind of severe oxygen depletion in British Columbia.

Based on results of the 1984 field season and laboratory work, it was clear that leachates from tree species harvested in the Babine Lake watershed could exert a significant oxygen demand. Since oxygen depletion did not occur in the Morrison Arm log handling area in 1984, I hypothesized that water movement within the boom was enough to dilute the chemical effects of log storage. Water movement and residence time can be examined by introducing a dye into the water and tracking its movement over time (Kisiel et al. 1964). Therefore, I examined water residence time and water quality in the log storage area of Morrison Arm, to determine the relationship between water exchange and oxygen conditions.

3.1.2 Food supply and diet of sockeye fry

The quality and quantity of food available to juvenile fishes is accepted as being an important factor in their growth and survival (Braum 1967), as has been demonstrated in several studies (Hjort 1914, LeBrasseur 1969, Eggers 1978). A critical period for fish larvae seems to occur when they shift from yolk sac to external feeding; at this time a food shortage may make them particularly vulnerable to predation and adverse environmental conditions. A large proportion (50%) of the sockeye collected in Morrison Arm had remnants of a yolk sac

(pers. obs.).

For juvenile sockeye salmon, ration level and growth rate are linked (Brett et al. 1969, Brett and Shelbourn 1975). In the Babine-Nilkitkwa lake system, the mean growth rate of pelagic sockeye salmon fry increases assymptotically with the mean zooplankton biomass (Johnson 1961) and I would expect the same relationship in younger, littoral sockeye.

Fry to smolt mortality processes for Babine Lake sockeye have been studied by West (1983), who found high mortality rates among fish of smaller body size at emergence. It appears that these fry take longer to grow out of the size window in which mortality is the most intense. Sockeye fry-to-smolt mortality rates have been shown to decrease with increased length of lake residency (Foerester 1938).

In the present study, differences in food supply and diet of sockeye fry in log storage areas vs. undisturbed lake habitat are examined. If log storage detrimentally affects sockeye salmon feeding, I predicted that a change in food supply (quantity and/or quality) and diet would be evident. A decrease in the amount of food available or ingested by juvenile salmon held in enclosures in log storage areas could be accepted as evidence of a direct negative impact of log handling activities.

3.2 METHODS

3.2.1 Water residence and water quality

A trial dye (rhodamine-B) study was conducted on May 28, 1985 in the open water near the log dump ramp. The stock solution concentration of rhodamine-B used in experiments was chosen to be 1 g/l. At this initial concentration, dilutions of up to 15,000 times were still visible.

A control experiment (June 3, 1986) was conducted in open water within the log boom site to observe water movement in water without log bundles. The rate of movement was observed from shore.

Within the log booms located closest to the log dump site, a sampling grid was laid out to cover an area 100 m by 100 m. Ten litres of rhodamine-B dye (1 g/l) were introduced (Figure 16) in an instantaneous dose to track water movement. The objectives were to sample over time the location and dye concentration of (1) the centre of the dye cloud (2) the perimeter of the dye cloud and (3) the depth of the centre of the dye cloud.

Samples were taken by hand at the surface or by a Par diaphragm pump and hose, at depths below the surface. Sampling began immediately following the dye introduction and continued

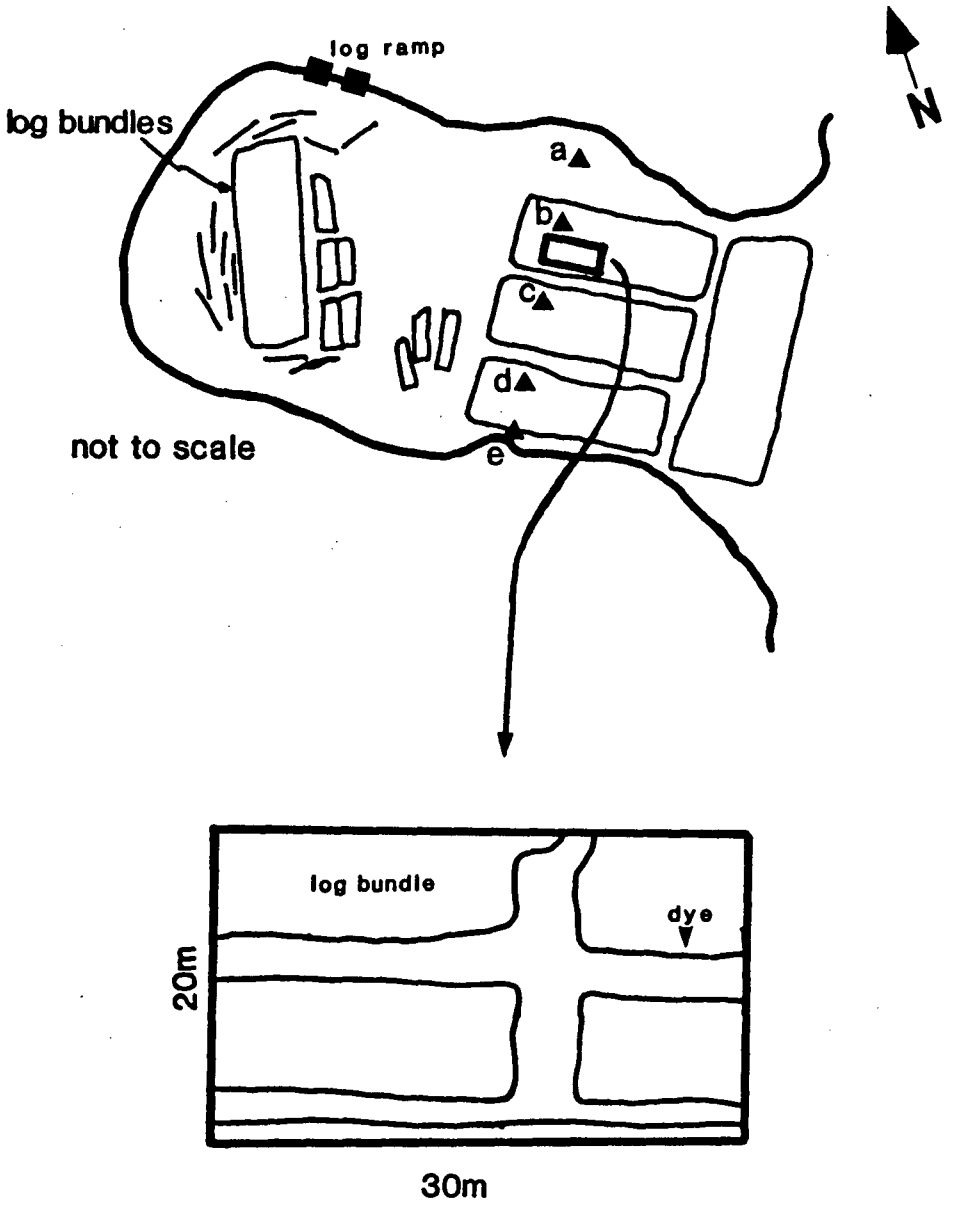

Figure 16. Sketch map of the H.F.P. log dump bay and storage area showing booms and log ramp (see Figure 15 for location in Morrison Arm). Water quality and in situ bioassay sites are marked with a triangle and letter (top). The dye study site is illustrated showing log bundle arrangement and release point for dye (below).

at 5 minute intervals for 0.5 h, then 10 minute intervals for 1.5 h at the centre and perimeter of the dye cloud. Water samples at depths of 0, 0.25, 0.50, 1.0, 2.0, 3.0, 4.0 and 5.0 m were taken at 0.5 h intervals for 3.5 h following introduction of the dye. The movement of the cloud was tracked on the grid map at 5 minute intervals. Due to the slow movement of the dye cloud, the grid size was reduced to 30 m by 20 m. The location and log bundle layout of the grid in the log storage site is marked in Figure 16.

Water quality (temperature, oxygen) measurements were taken at sites along a transect (Figure 16) and at a control site. Sites A and E are in "open" water within the log storage inner bay. Sites B, C, and D are within the log boom and the control site is in 5 m of open water near the 2.0 m reference site used in the fry feeding experiment (Figure 15).

In situ bioassays were conducted at water quality sites (Figure 16). Ten sockeye fry held at each site in flow through enclosures were monitored for respiratory stress and mortality on June 3, 1985.

Water samples were analyzed for dye concentration by visual comparison with known concentrations of rhodamine-B in a method adapted from Standard Methods (APHA et al. 1985). Two 50 ml matched Helige Aquatester tubes were filled to the 50 ml mark, one with the sample, the other with appropriate standard. By

looking vertically downward through the tubes toward a lighted white surface, standard comparisons were made. Dilutions by 10% increments of 1 g/l rhodamine-b down to 1/15000 (+ 10%) were discernible by this method.

The surface area of the boom and the head of Morrison Arm were determined using software ("lake morphometry") on an Apple computer with a graphics tablet. The volume of water below the log boom was also determined.

3.2.2 Food supply and diet of sockeye fry

Feeding studies were conducted *in situ* at Morrison Arm, Babine Lake, British Columbia in late May and early June, 1985. The following sites were selected to compare diets of sockeye fry in log handling areas (treatment) and undisturbed littoral habitat (reference):

(1) log boom treatment site (2 m)
(2) undisturbed reference site (2 m)
(3) log dump treatment site (0.5 m)
(4) undisturbed reference site (0.5 m)

The locations of these four sites are shown in Figure 15. The dissolved oxygen concentrations at all sites were above 6.0 mg/l. This study design does not provide replication of treatments; a greater number of sites in each area are required

to statistically test for differences between treatments (Hurlbert 1984).

The log boom and log dump sites represent habitats exposed to two kinds of log handling activities which sockeye fry might encounter. The shallow log dump site is used regularly over the winter and has higher rates of bark deposition than the log boom area which is relatively undisturbed once covered with log bundles (Levy et al. 1985b). Reference sites in waters of appropriate depth were selected in areas of similar undisturbed substrate and exposure to the two treatment sites.

During the feeding experiments, fry were held in flow-through enclosures (FTEs) which permitted ample water circulation and zooplankton replenishment. Sampling indicated that there was no significant difference in zooplankton abundance or species composition in samples taken within or outside the enclosures. Each FTE consisted of green vexar plastic mesh (3 mm mesh size) stretched and fastened over all sides of a 0.5 X 0.5 X 0.5 m aluminium frame box (Figure 17). One side had a removable sliding door to allow access into the otherwise completely enclosed box. On the top of the FTEs were four aluminium loops from which the cages could be suspended in the water column. Six FTEs were constructed, two of which were modified for zooplankton sampling. The addition of a small hole and mesh cover flap allowed insertion of the zooplankton pump hose into the top of the FTE.

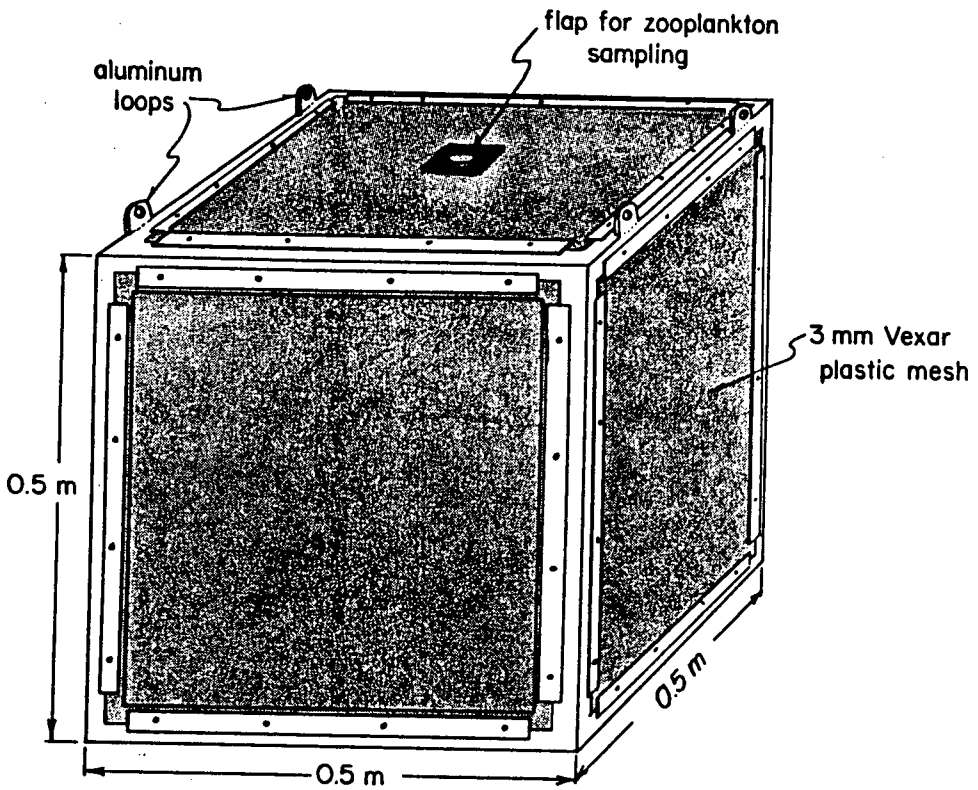

Figure 17. Design of flow through enclosures used in fry feeding experiments.

Sockeye fry used at the log boom and 2 m reference sites were obtained by beach seining from Morrison Arm, Babine Lake. The fry used at the log dump and 0.5 m reference sites were collected at Fulton River spawning channel, due to the low abundance of fry in Morrison Arm in early June. The average length (mean + standard deviation) of fry used in the experiments was 2.8 + 0.1 cm. There was no significant difference in length of fry used for each set of experiments.

The log boom experiments were conducted on May 22-23 and May 28-29, 1985 and the log dump experiment was conducted on June 4-5, 1985. It was not possible to travel to or stay at Morrison Arm on any other dates; this restricted the number of experiments which were conducted. The afternoon before the start of an experiment, three FTEs were installed at each site (two for fish, one for zooplankton sampling). For the log boom and 2 m sites, each FTE was independently suspended at 0.25 m depth from a boom and float respectively. For log dump and 0.5 m reference sites, the FTEs were placed on the sediment at a depth which just completely immersed the top of the cage.

A total of 200 fry were randomly divided into groups of 50 and each group was introduced into a FTE the afternoon before the start of an experiment. It was observed that young sockeye fry clear their stomach contents overnight, so stomach samples taken the following day contain prey captured only in the

enclosures. In trial feeding experiments it was determined that there was no significant difference in net food intake per fry when fry were held at densities of 10 or 50 fry per enclosure.

Sampling began before dawn and continued through until the morning of the following day. The sampling regime consisted of both fry and zooplankton samples being taken at each site at regular (usually 4 h intervals) throughout the experiment. Zooplankton samples were taken with a Par diaphragm pump filtered through a 100 micrometer mesh net and preserved in a sucrose-5% formalin mixture (Haney and Hall 1975). Duplicate two minute samples were taken from fishless FTE each sample time. Fish were sampled by dipnetting 5 fry from each enclosure at each sample time, and these were preserved in 10% formalin. Travel between the treatment and reference sites was timed to keep sample collection times as similar as possible for the two sites.

Zooplankton samples were examined <u>in toto</u> or by splitting into 1/5 subsamples. Rare species were counted from the total sample. Samples were ennumerated and identified under a stereo dissecting scope using Edmondson (1959) and Smith and Fernando (1978) for taxonomic classification.

Fork lengths were recorded for each fish and stomach contents were analyzed under a stereo dissecting scope. Stomachs were dissected out of preserved fish specimens by making

incisions at the esophagus and at the junction of the pyloric sphincter with the intestine. Absence or presence of the yolk sac was noted. The stomach fullness was estimated on a scale of 0 (empty) to 10 (full) (Hyslop 1980). The stomach contents were spread out in a petri dish in a drop of water and food items were identified, sorted and ennumerated. Visual estimates (+5%) of the relative volume of each prey type were made. Overall assessments of the state of stomach contents were made (eg: fresh, digested). Fresh zooplankton were intact and loose in the gut, whereas digested zooplankton were almost unidentifiable. Prey were categorized as follows:

Diacyclops

Diaptomus

Daphnia

Bosmina

nauplii/copepodites (calanoids and cyclopoids)

chironomid larvae

insect pupae

insect adults

unidentifiable (digested matter)

The numerical relative proportions of each food item in the guts were calculated for each sample time. Proportion data were transformed by square root arc sin, before means were calculated, to meet the assumption of normal distribution of the proportion data (Schefler 1980). Unidentified food items were not included. Prey which were rarely taken (adult Diaptomus

spp., Daphnia, and insects were grouped into a category called other. The diversity indices can serve as a measure of feeding habits for comparisons between fish (Hyslop 1980). The diversity of prey acquired by sockeye fry was determined for each sample time using Simpsons's Diversity Index as recommended by Washington (1984).

SIMPSON'S DIVERSITY INDEX (D)

$$D = 1 - \sum_{i=0}^{s} (p_i)^2$$

where s = no. of species

p_i = proportion of total no. of individuals of i^{th} species

3.3 RESULTS

3.3.1 Water residence and water quality

Two dye release trials in open water at the head of the log storage area indicated that the dye cloud moved at rates ranging from 15 to 30 metres per hour, when unrestricted by log bundles. Surface waters were rippled by a light breeze during both trials. Following an hour of observation, the dye clouds in open water were diluted beyond detection limits.

In contrast, the perimeter of the surface of the dye cloud released within the log boom initially moved at a rate of 20

metres/h, and then was virtually stagnant, remaining for at least 8 h. The perimeter did not expand more than 10 m in any direction for the rest of the day (Figure 18).

The centre of the dye cloud was diluted as it spread out. Two minutes after introduction it was diluted 20 times and after 7 minutes, 200 times. The perimeter dilution was 1000 times after 7 minutes, and increased to 15000 times, after 3.5 h. At this time, the "centre" of the dye cloud was diluted 3000 times, relative to starting conditions and was largely restricted to the top 0.5 m of the water within the log boom area (Figure 19). This phenomenon, in combination with the small lateral movement of the dye, resulted in the dye cloud being contained within an area approximately 15 X 10 m (between 4 log bundles) to a depth of 0.5 m. This distribution implied reduced water movement, laterally or vertically, within the log boom area. The dye persisted until nightfall, but was undetectable the following morning. However, the thick bacterial growth which covered the log bundles was heavily stained, but only within the upper 0.25 m of water.

Extreme oxygen depletion (<2.0 mg/l) and elevated temperatures (17.0-18.5 deg. C) occurred in the upper 0.5 m of water in the log boom on June 3, 1986, while "open" areas had only slightly depressed oxygen levels in surface waters (Table 5). Control site water was cooler (<14 deg. C) and oxygen levels were above 9.0 mg/l at every depth.

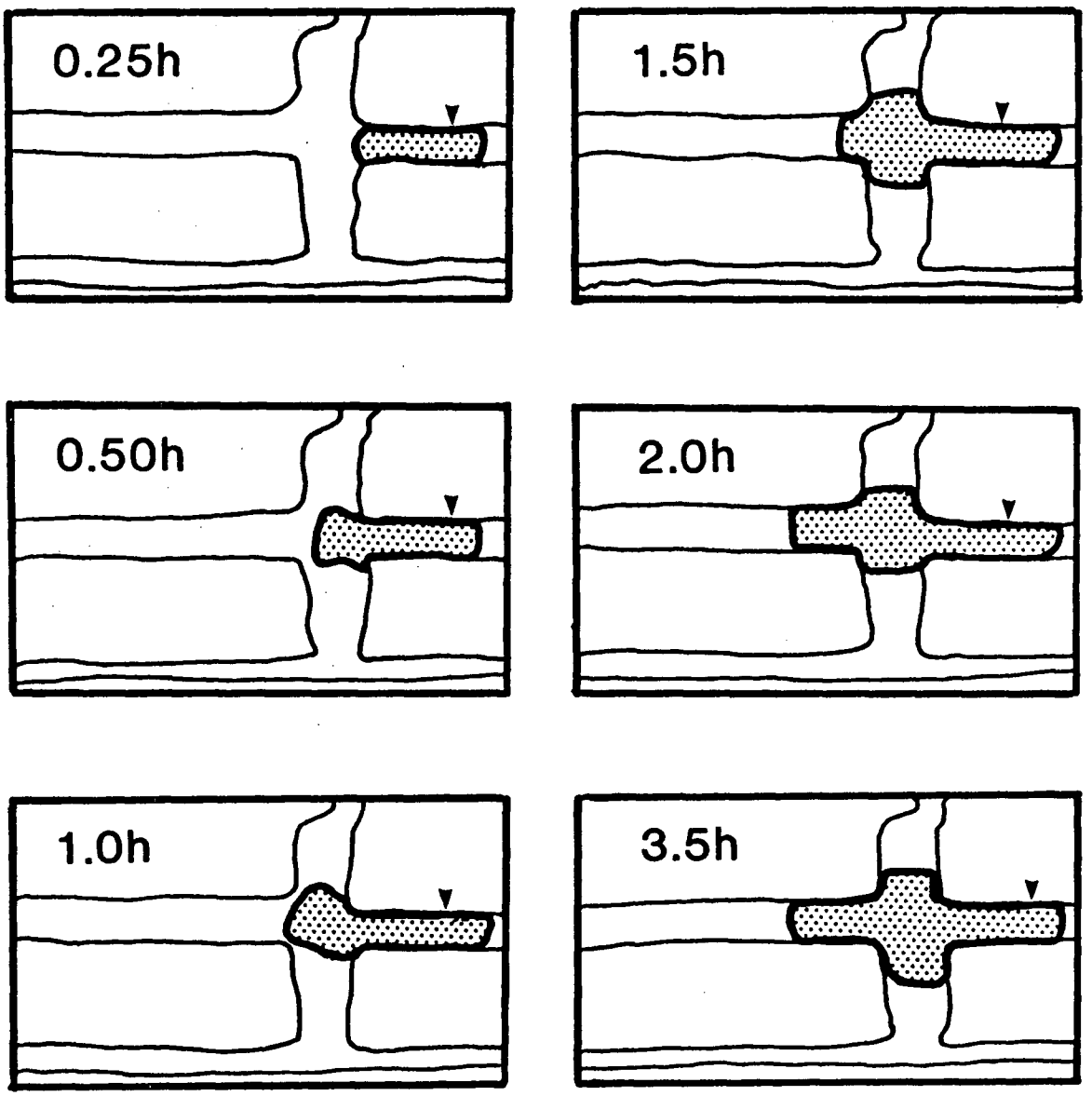

Figure 18. Dye cloud movement in surface waters of a 30 X 20 m grid within log bundles over a 3.5h period on June 3, 1985 (dye release location indicated by arrow).

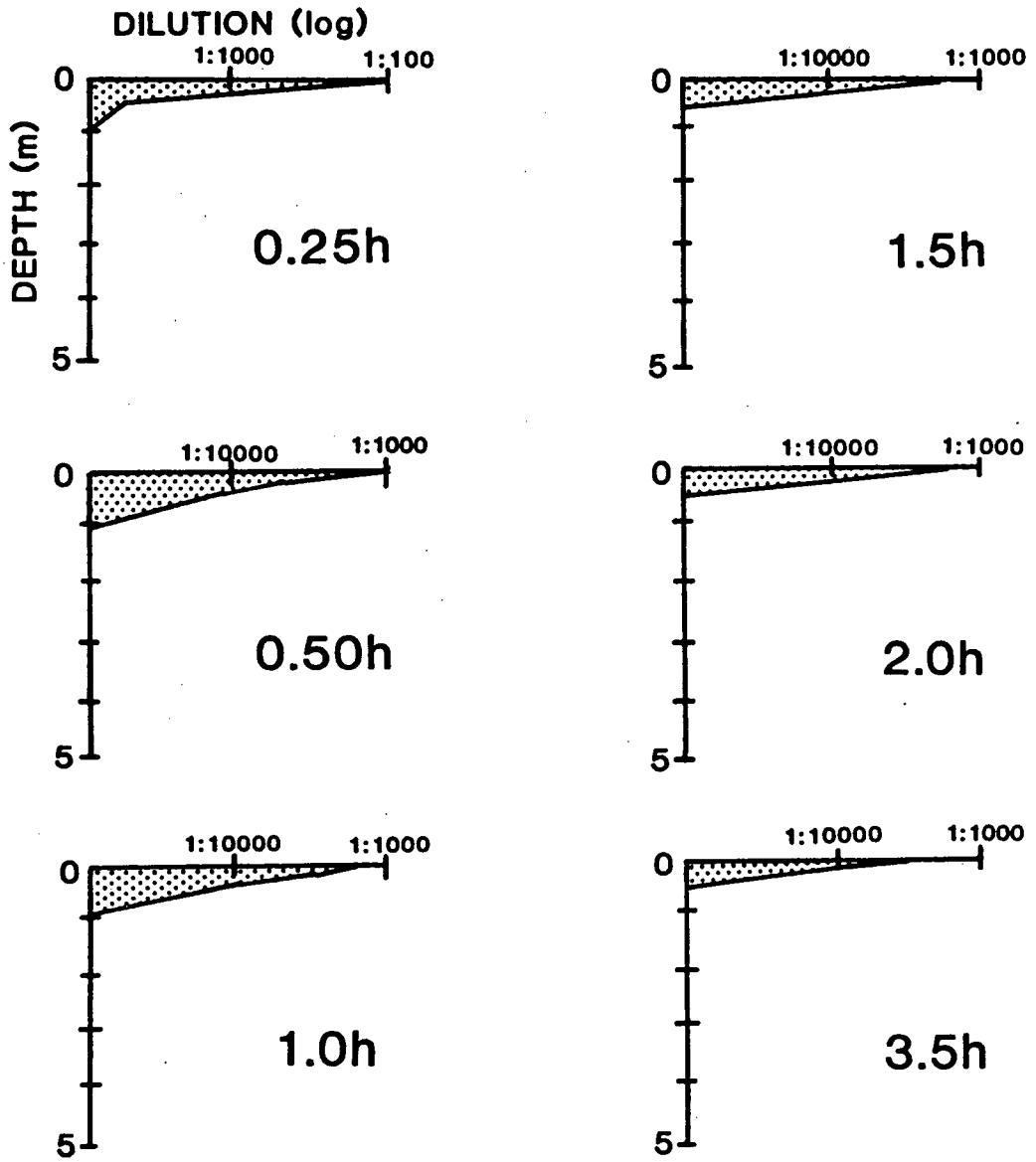

Figure 19. Depth distribution and dilution at center of dye cloud over a 3.5h period on June 3, 1985.

Depth	Site A		Site B		Site C		Site D		Site E		Reference	
(m)	T	DO	T	DO	T	DO	T	DO	T	DO	T	DO
0	16.0	7.0	18.0	1.2	18.5	0.8	18.0	1.0	17.0	7.8	14.0	9.2
0.25	15.0	7.0	18.0	1.5	18.0	1.0	18.0	0.9	16.0	7.7	13.0	9.2
0.50	15.0	7.2	17.5	2.0	16.0	1.5	16.0	1.5	16.0	7.7	12.0	9.4
1.0	15.0	9.1	15.0	4.9	15.0	3.8	16.0	2.5	16.0	8.9	10.0	9.9
2.0	12.0	8.7	11.5	7.2	11.5	8.0	11.0	8.0	12.0	8.2	10.0	9.9
3.0	12.0	8.6	11.5	9.1	11.0	7.9	11.0	8.2	-	-	9.0	9.8
4.0	-	-	9.0	9.1	9.5	8.8	9.5	8.5	-	-	9.0	9.6
5.0	-	-	9.0	8.8	9.0	8.2	9.0	9.7	-	-	9.5	9.2

Table 5. Water quality (temperature (deg. C) and dissolved oxygen (mg/l)) in Morrison Arm on June 3, 1985. See Figure 16 for locations of sites.(T= temperature, DO= dissolved oxygen)

The surface area of the log boom comprises 15% of the area of the head of Morrison Arm (Figure 15) is described by the ratio of boom:arm which is 0.15. Given that there were 165,000 cubic meters of logs in the Morrison Arm storage area (P. Ogawa, pers. comm.), and assuming that 50% of each bundle was submerged, the instantaneous wood:water density can be calculated. The volume of water below the log boom to a depth of 2 m (thermocline) was 248,000 cubic meters, therefore the wood:water ratio under static conditions was 0.33 cubic meters wood/cubic meter water.

All sockeye fry held at in situ lethal bioassay test sites within the log boom (sites B, C, and D) were dead within 15 minutes. They exhibited signs of extreme respiratory stress such as gill flaring, disoriented swimming and surface swimming. Of the ten fry held at site A, two died and the survivors displayed infrequent gill flaring. All fry in site E and the reference site survived the 24 h test period.

3.3.2 Food supply and diet of sockeye fry

3.3.2.1 Zooplankton abundance

Over the 24 h experiments there were clear differences in zooplankton abundance between reference and boom sites (Figures 20 and 21). Copepodites and nauplii numerically dominated the zooplankton community at all sites in all experiments.

Diacyclops thomasi was the second most abundant zooplankton, followed in decreasing abundance by Bosmina coregoni , chironomid larvae, Daphnia spp., and Diaptomus spp.

In the log boom experiments (May 22-23 and May 28-29), there were differences in local zooplankton abundance between the log boom and reference sites. Nauplii were 50% to 200% more abundant at reference sites than at the log boom sites on both dates (Figure 20), with peaks between 1100-1200 h. Copepodites, the most important food item for young sockeye fry, were more abundant at the reference site on May 22-23, but there was no difference on May 28-29 between reference and boom sites (Figure 20). Diacyclops thomasi was present in similar densities at both reference and boom sites on both dates, except for higher densities at the reference site near 1200 h, particularly on May 28-29 (Figure 20). The same trend was evident for Bosmina (Figure 20); this species occurred at similar local abundancies at reference and boom sites, on both dates. Chironomid larvae were present at very low densities (<1 indiv/l) almost exclusively at the reference site in the morning and evening on both experiment dates (Figure 20). Data for Daphnia and Diaptomus are not reported here because of high sampling variances associated with low densities of these species, and their absence in stomach samples.

In the log ramp experiment (June 4-5), there were significant differences in zooplankton abundance between the

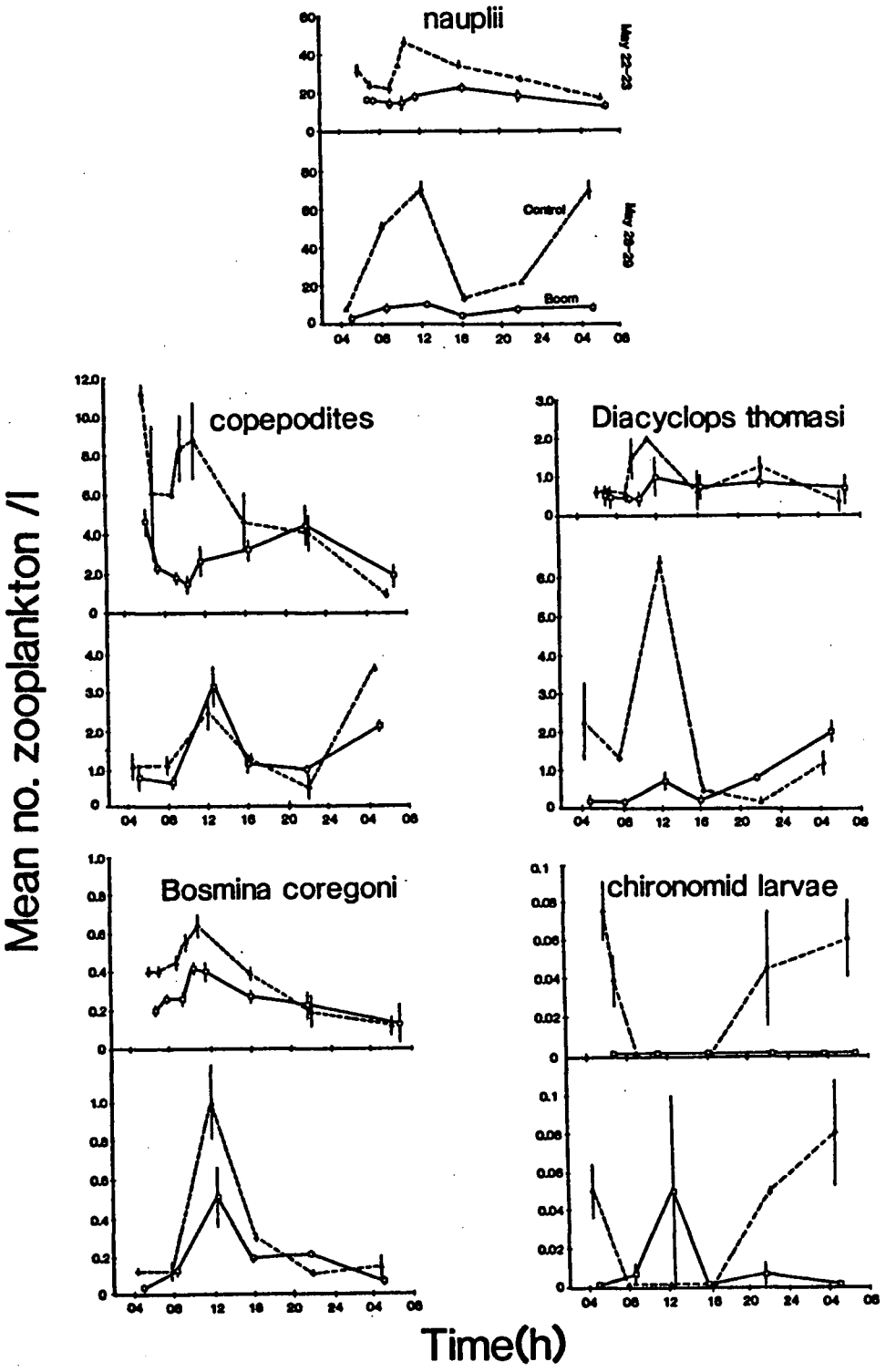

Figure 20. Zooplankton abundance (no/l ± S.E. for subsamples) in flow through enclosures at boom and control sites during sockeye feeding experiments May 22-23 and May 28-29, 1985 (dashed lines = control sites, solid line = boom site; see nauplii graph (top) for layout of May 22-23 and May 28-29).

reference and log ramp sites. The densities of nauplii and copepodites (Figure 21) were lower at the reference site than the ramp site during the morning and evening. However, nauplii and copepodite abundance at the reference site clearly increases above that of the ramp site during the middle of the day. Diacyclops thomasi and Bosmina coregoni (Figure 21) did not differ between reference and log ramp sites. Chironomid larvae (Figure 21) showed a large peak in abundance during the 0830 sample at the reference site with very low abundance during all other sample periods at both sites.

3.3.2.2 Stomach Contents

Generally, the major prey items present in the fry stomachs were copepodites, with respect to both number and volume. Bosmina coregoni were taken at a lower frequency, and nauplii, adult copepods Daphnia and insects were taken rarely, with the exception of chironomid larvae. Chironomid larvae were taken in large numbers at sites where they were available.

Digestive status of gut contents over the course of 24 h experiments showed a similar trend in each case. In general, the morning and afternoon gut samples were composed of "fresh" zooplankton; evening gut samples were relatively well digested. Sockeye salmon fry in the 2.0 to 3.0 cm size class completely clear their stomachs overnight.

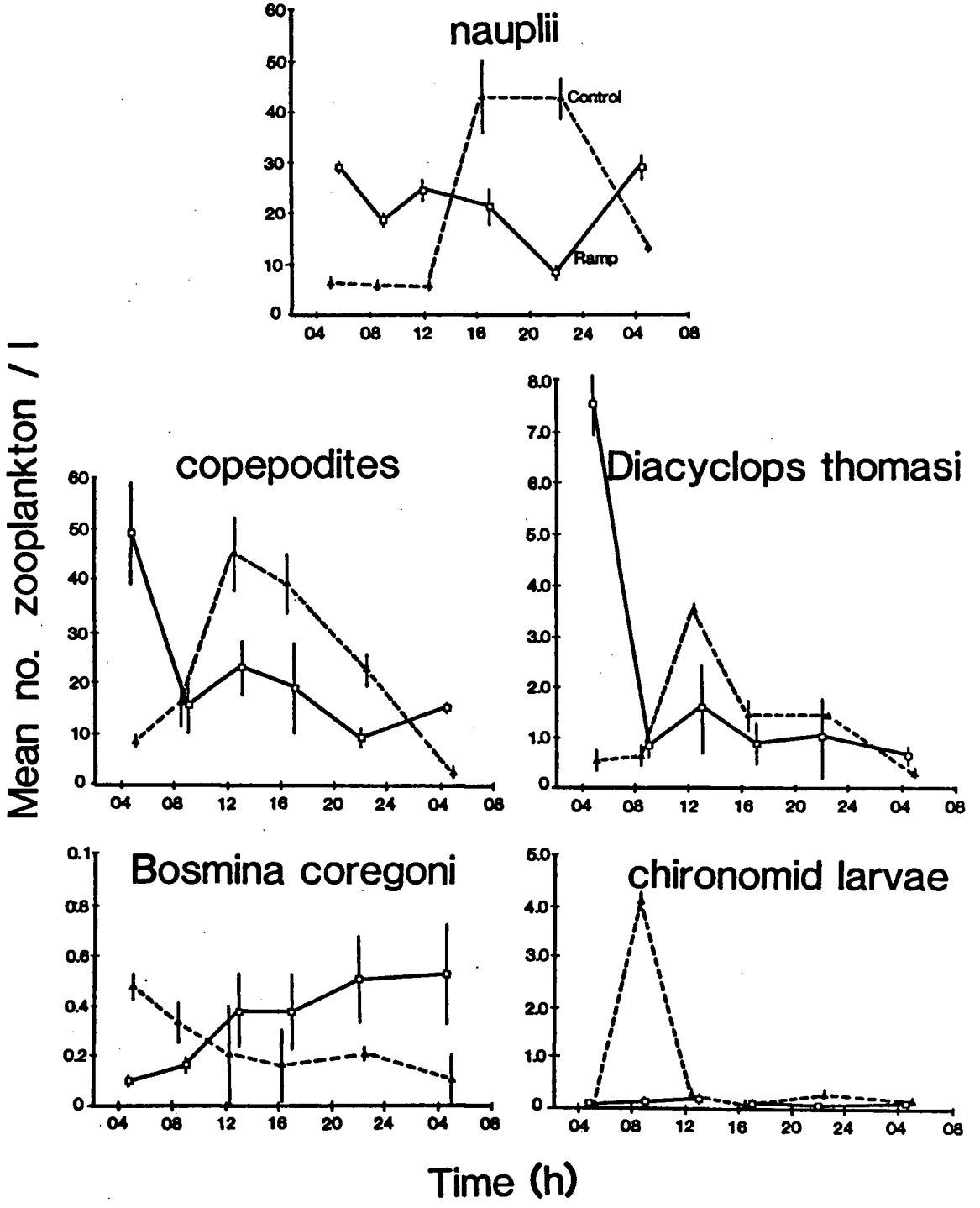

Figure 21. Zooplankton abundance (no/l ± S.E. for subsamples) in flow through enclosures at ramp and control sites during sockeye feeding experiments June 4-5, 1985 (dashed lines = control sites, solid lines = ramp site).

Information on ingested food quantity was obtained by (1) estimating stomach fullness and (2) calculating mean number of each prey item per stomach. For fry in FTEs in both the log boom and 2.0 m reference sites (May 22-23 and May 28-29), stomach fullness and mean prey/stomach (Figure 22) show similar changes over the 24 h experiments, although hourly patterns are not the same for the two dates. Food intake begins in the early morning with stomach fullness peaking between 1200 and 1600 h. Stomach fullness and mean prey/stomach then decrease until the following morning.

Fry in FTEs in the log dump and 0.5 m reference sites (June 4-5) do not exhibit similar trends in stomach fullness or in mean prey abundance per stomach (Figure 22) over the 24 h period. Fry in FTEs at the 0.5 m reference site exhibit a mid-day peak in stomach fullness and prey number, while fry held at the log ramp have relatively low values overall. Fry held in the 0.5 m reference site ingest more food than those at the log ramp site.

In both the log boom and 2.0 m reference sites (Figure 23), on May 22-23, copepodites were the dominant stomach content item throughout most of the 24 h period with Bosmina secondarily important. The same trends were seen in the May 28-29 experiments. The stomach contents of 2.0 m reference site fry generally had a greater number of prey species than those of log boom site fry (Figure 22). The higher species richness of prey

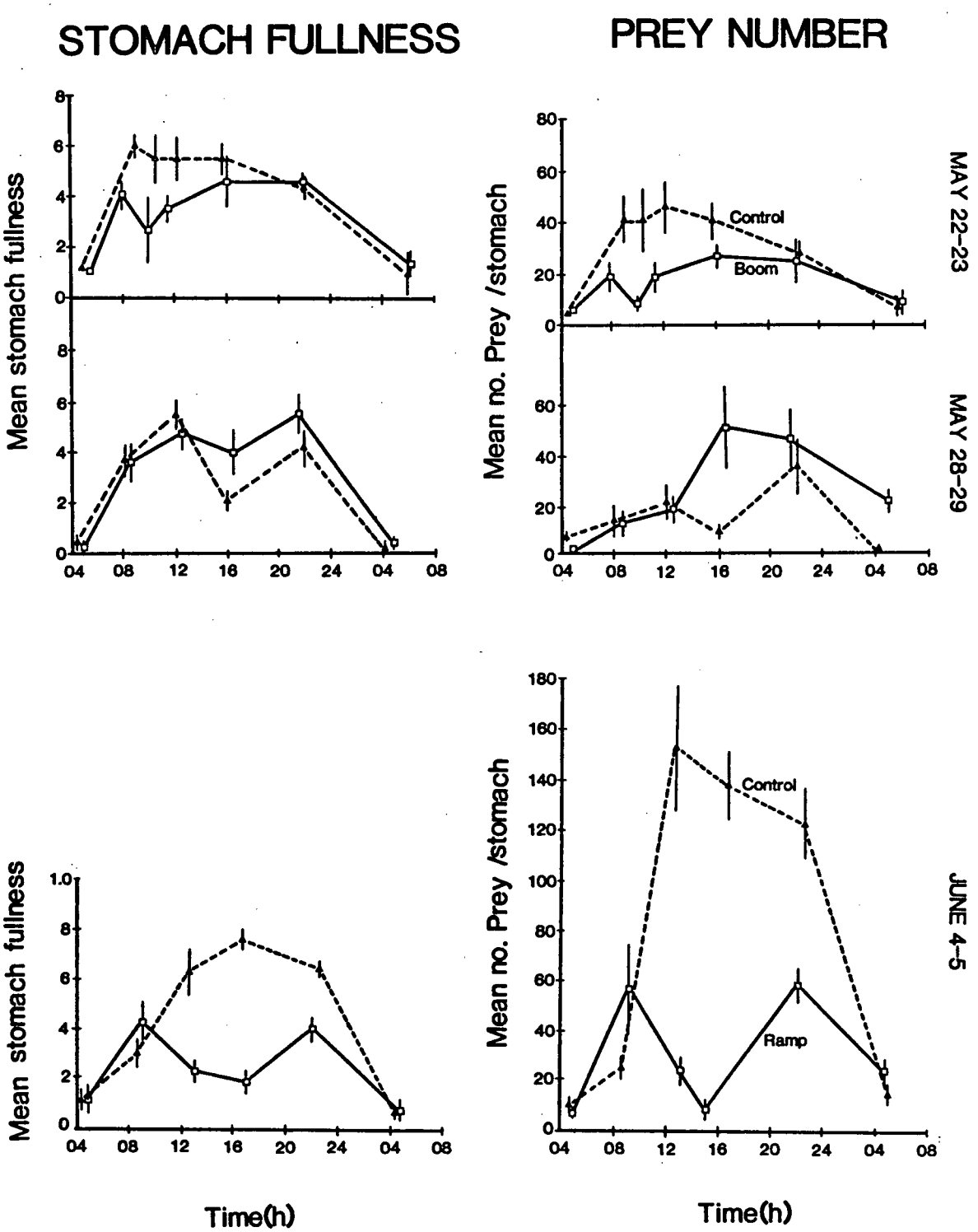

Figure 22. Stomach fullness (± S.E.) and number of prey (±S.E.) in stomachs of sockeye fry during feeding experiments at boom and control sites (May 22-23 and May 28-29, 1985) and at ramp and control sites (June 4-5, 1985) (dashed lines = control sites, solid line = treatment site)

items acquired by reference site fry is reflected in higher diversity. For both May 22-23 and May 28-29, species diversity was higher in the stomach contents of reference site fry than in boom site fry (Figure 24).

There were large differences in stomach contents between log dump and 0.5 m reference (Figure 23) fry. Although copepodites were the most abundant food item over the 24 h period, chironomid larvae became the main food item during the early morning for fry at the 0.5 m reference site. Chironomid larvae were virtually absent from the stomach samples of fry held at the log ramp site (Figure 23). Bosmina were taken at low frequencies throughout the 24 h period at both sites. There was no difference in diversity of food items acquired by fry held at ramp and reference sites.

To test for prey selectivity by sockeye fry in log handling vs . reference areas, several indices were applied to the data. Both Pinkas 's index of relative importance (1971) and Strauss' linear selection index (1979) failed to represent the feeding patterns of the sockeye salmon. The sporadic appearance of chironomid larvae and their corresponding consumption are represented as a preference for chironomid larvae over copepodites. However, the absence of chironomids in both zooplankton and gut samples at other times results in their representation as a randomly selected food item, which is not a biologically reasonable conclusion. Therefore, these two

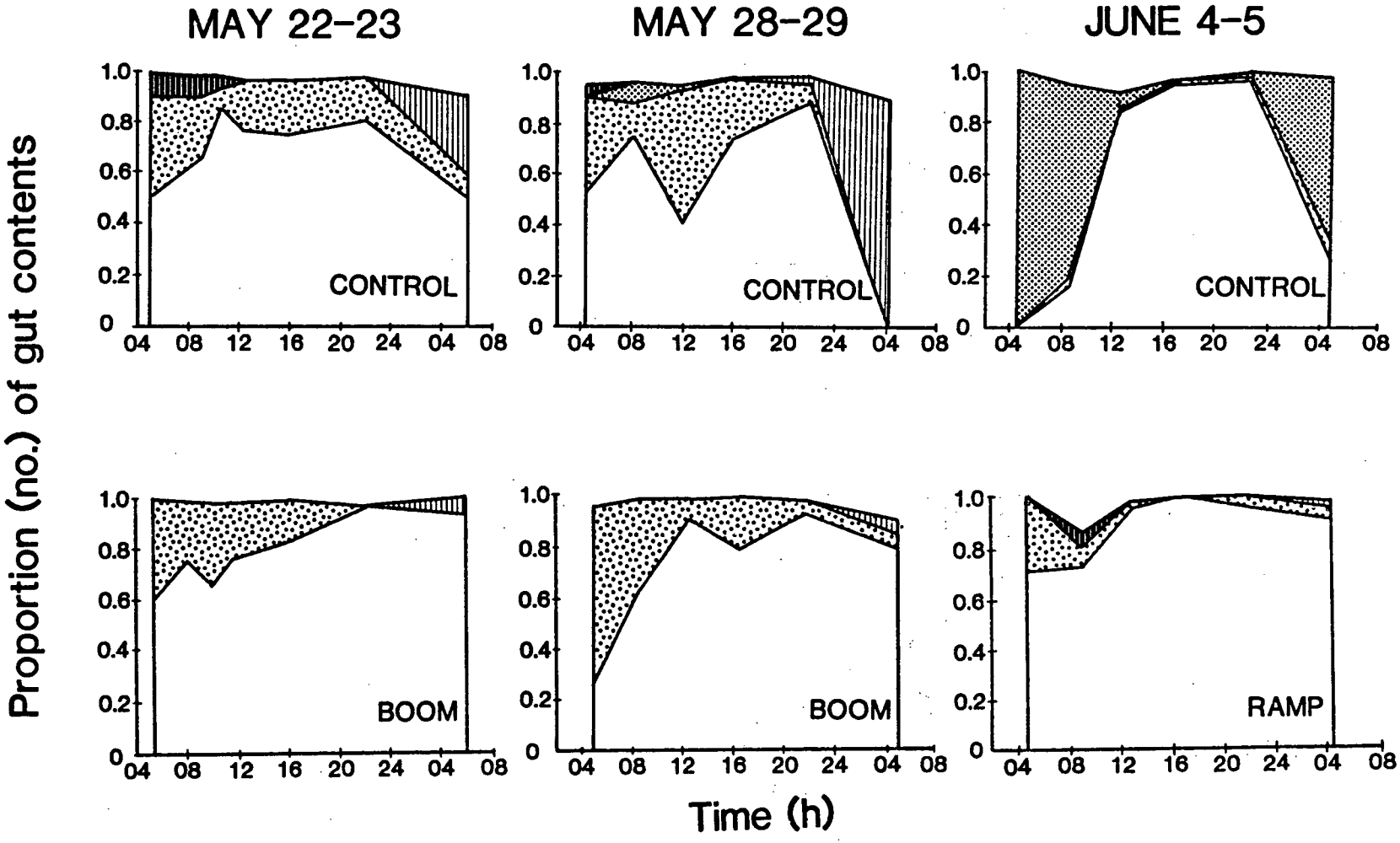

Figure 23. Gut contents (proportion (transformed by square root arcsin) of number of each prey type) for sockeye fry held at boom and reference sites (May 22-23 and May 28-29, 1985) and ramp and reference sites (June 4-5) (white = copepodites, sparse dots = Bosmina, light stripes = other, dark stripes = Diacyclops, and denser dots = chironomid larvae).

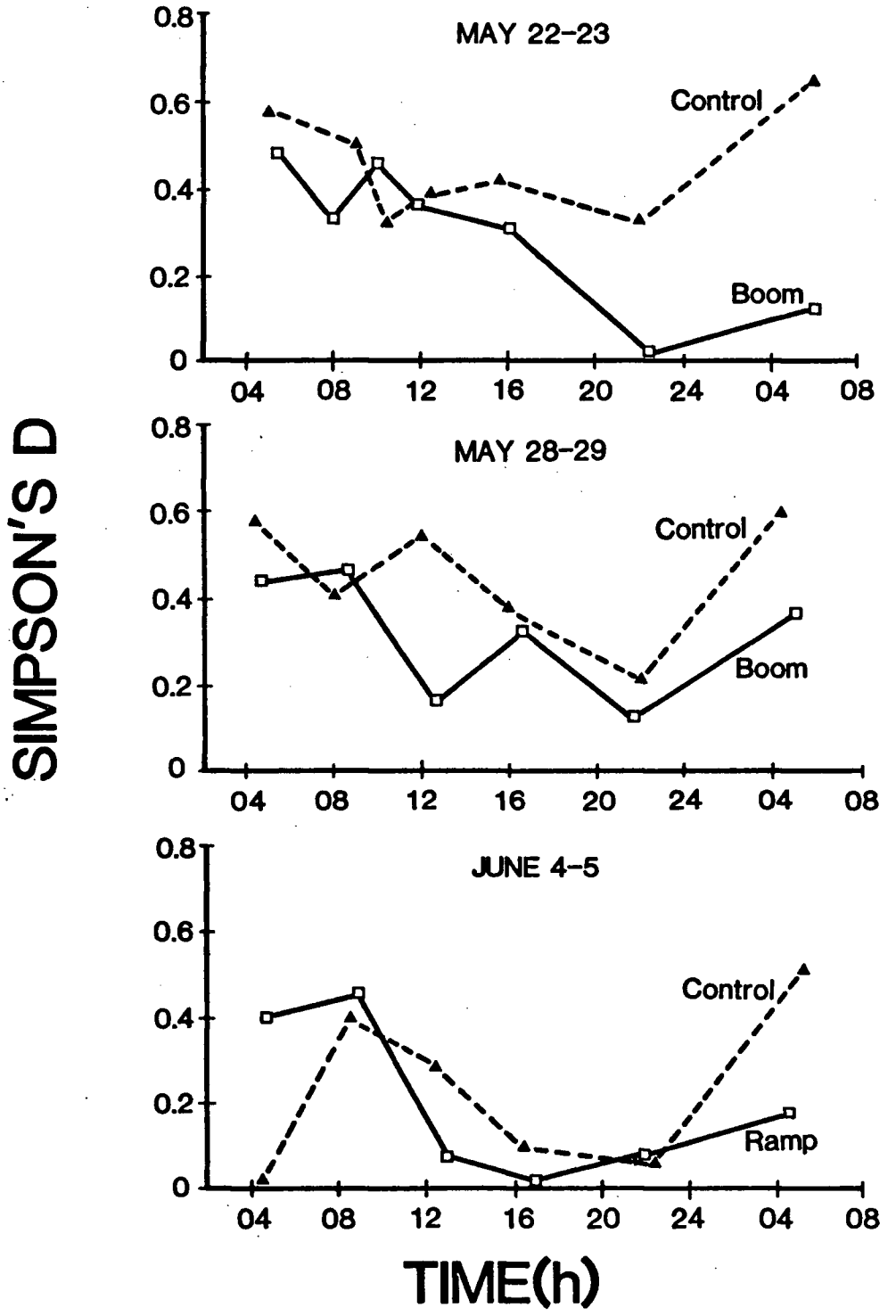

Figure 24. Diversity indices (Simpson's D) for gut contents of sockeye fry held at boom, ramp and control sites.

selectivity indices were discarded as methods to describe the feeding of sockeye fry over the course of 24 h experiments.

3.4 DISCUSSION

The results of the dye experiment suggest extreme stratification in response to density differences in the surface water within the log storage site. The stagnant condition of this water contributes to oxygen depletion due to high B.O.D. and C.O.D. Oxygen levels were lowest (<2 mg/l) in the top 0.25 m of the water column which corresponds to the most poorly mixed water layer, as demonstrated in the dye experiments. Dissolved oxygen monitoring in the Houston Forest Products site during May and June, 1986 (Bustard 1986) determined that oxygen levels again were depressed (3-5 mg/l), but not to the same extent as in 1985. The oxygen depression remained until the log booms were removed from the site. Long term temperature data (Bustard 1986) indicate that conditions were not unusual during 1985 and 1986, which suggests that this phenomenon is not anomalous and may occur on a yearly basis in late May/early June. The downstream migration of sockeye fry from Morrison River into Morrison Arm is coincident with the oxygen depletion in the log storage area.

It has clearly been shown by Levy *et al.* (1985b) that fry avoid the log storage area during this time. Avoidance of oxygen depleted water has been shown for other fishes (Davis 1975) and may be a result of increased random movement until preferred

oxygen conditions are found. Increased predation risk and/or energy expenditure as a result of avoidance behaviours might increase the mortality rate of fry passing through the log storage area. This hypothesis could be tested using a mark-recapture experiment in which fry are marked as they leave Morrison River, and then recaptured after the log storage area. The other side of Morrison Arm could be used as a control, assuming that young fry do not cross the arm. It would not be possible to attribute mortality to individual factors, but differential mortality would suggest higher risks in passing through the log storage area. Certainly, the results of the *in situ* bioassay experiments indicate that fry entering the surface waters of the log storage area during late May and early June would have died. Schools of apparently disoriented sockeye fry were observed within the booms during early June, 1985 (pers. obs.), but no bodies were found.

Prey consumed by sockeye fry in FTEs were similar to diet items previously reported for fry in Morrison Arm (Levy *et al.* 1984). Also, the zooplankton samples from the reference FTEs have a similar composition to lake samples which indicates that the food available to fry inside the FTEs is representative of the lake. However, it is possible that this is a sampling artifact since the zooplankton sample volumes were greater than FTE volumes. Consequently, zooplankton could have been drawn into enclosures resulting in a sample which may not be representative of what is available to fry. Assuming that this

effects will be equal at treatment and reference sites, I will attribute differences in fry diet between reference and treatment sites to the effects of log storage, through (1) food availability and (2) sockeye fry feeding behaviour under log storage water quality conditions in the log storage area.

The amount and digestive status of the gut contents reflects a unimodal feeding pattern, with feeding occurring from early morning to the afternoon of each day. This pattern was also described by McCart (1967) for inshore fry feeding in Babine Lake during early June. The number of empty stomachs was inversely related to the mean stomach fullness for all sample times.

Sockeye fry held in the log boom and 2.0 m reference sites (May 22-23 and May 28-29) showed no major dietary differences that were consistent for both sample dates. There were only slight differences in the food available to the fry at those two sites, although food was generally more abundant at reference sites. There was a tendency towards higher prey diversity acquired by the reference site fry, as evidenced on both experiment dates by a more complex prey species composition in gut samples. It is difficult to predict the effects of this difference on the growth or survival of sockeye fry. However, it is generally accepted that reduced food diversity is a deleterious consequence of pollution and disturbance (Washington 1984).

Sockeye fry in log dump and 0.5 m reference sites (June 4-5) displayed major dietary differences. Fry held in cages in the log dump site acquired significantly fewer prey items resulting in low stomach fullness and the species composition was markedly different. The abundance of copepodites, the major food item of sockeye fry in May and June was slightly higher in the reference site than in log dump sites for the majority of the day and this trend is reflected in the consumption of this prey. Significantly greater numbers of copepodites were consumed in the reference site compared to the log ramp site. Other food items such as Diacyclops and Bosmina do not exhibit this same pattern in either abundance or consumption; these prey items are relatively low in abundance and did not significantly differ in consumption between sites. The control site fry consumed large numbers of chironomid larvae which become available to them during the morning and evening while this preferred prey item was not available to fry in the log dump.

These results suggest that sockeye fry which feed in the highly disturbed inshore area of the log ramp will experience lower food abundance and a different diet composition than those feeding in a pristine inshore environment. Sockeye fry feed primarily on water column prey, so the presence of chironomid larvae in zooplankton and stomach samples was an unexpected result. However, these chironomid larvae appear to undergo small vertical migrations in the morning and evening, which puts them

into the "available" food supply for sockeye fry in shallow waters at those times. In gut analyses during 1984 and 1985, Levy et al. (1985a,1985b) also found chironomid larvae in stomach samples from sockeye fry.

It appears that the sockeye fry in May/June acquire prey items in proportion to their abundance in the environment. However, Rankin (1977) in a laboratory study of prey selectivity by young Babine Lake sockeye fry found that Diacyclops and Diaptomus adults were selected over copepodites. He suggested that low abundance of adults in Babine Lake may result in increased predation on the smaller copepodites and nauplii, which might be the case in Morrison Arm during May and June. In the late summer and early fall, Babine Lake sockeye depend mainly upon adult zooplankton (McDonald 1973, Rankin 1977) as has been observed for sockeye fry in Lake Washington (Doble and Eggers 1978). This feeding shift from juvenile to adult zooplankton may reflect the growth of available food items and/or an improved ability of fry to capture larger food items as they grow and move offshore.

The absence of chironomid larvae is related to disturbance from log handling activities. Severe impacts of log storage on all major groups of benthic invertebrates have been well documented for the Houston Forest Products dump site (Levy et al. 1985b, Yesaki and Levy 1986). In addition to physical disturbance from bark deposition and boom boats (Conlan and

Ellis 1979) the abundance of benthic invertebrates may be related to the chemical changes to the environment which accompany log storage. Levy et al. (1985b) demonstrate that the microbial growth under the log boom has a high rate of oxygen consumption and is associated with hydrogen sulphide production. Measurements of sediment oxygen-reduction potential in the Nanaimo River estuary show the presence of an anoxic layer over the benthos at log storage sites where logs had been removed (McGreer et al. 1984).

Sockeye fry seem to preferentially feed on chironomid larvae over copepodites when both are available; whether for taste or energetic reasons. It seems reasonable that chironomid larvae would be relatively easy to capture and result in a large energy gain relative to the foraging effort if they are abundant. Chironomid larvae are probably not a staple food item, as supported by stomach content analysis of littoral sockeye fry collected throughout Babine Lake (Levy et al. 1984), and appear to be opportunistically consumed. It is the presence of chironomid larvae which accounts for the shift in diet composition observed for fry held in reference vs. log ramp sites. Levy et al. (1982) also documented a difference in diet composition between a log handling site and a control site in the Fraser River estuary, British Columbia, which was related to food availability at those sites. The proportion of insects (adult, pupae, and larvae) consumed by released chinook fry (Oncorhynchus tshawytscha) was greater in the undisturbed marsh

than in the log storage area. In fact, insect pupae or larvae were not taken as food by chinook in the log storage area. The availability of insects as a food item seems to be positively related to the presence of marsh plants which are detrimentally affected by estuarine log storage, particularly over tidal flats (Levy et al. 1982).

I am aware of only one other experimental study that examines fish feeding in log storage areas. A similar project on feeding by chum salmon (Oncorhynchus keta) was conducted in the well flushed Nanaimo estuary, British Columbia using experimental enclosures in log storage and undisturbed sites (McGreer et al. 1983). No large differences in either prey abundance or diet composition were found between sites, which corroborates the results of the boom experiments in the present study. Not surprisingly, food quality and quantity appears to be related to the intensity and frequency of use log handling sites receive. Juvenile salmon feeding in sites which are relatively well flushed and lightly used may not be measurably affected by log storage. However, sites which have restricted mixing (Pease 1974) and heavy use (eg. log ramp area) may reduce water quality and abundance of food to levels which can affect movement and feeding of juvenile salmon fry.

The reduced abundance of zooplankton, particularly copepodites, in the inshore surface waters of the log ramp site may be related to oxygen depletion. Levy et al. (1985b) conclude

that zooplankton are insensitive to water quality conditions within the log handling site. However, their sampling regime did not include surface waters (< 1.0 m), where water quality was poor enough to expect a negative response by zooplankton. Therefore, their conclusions were only valid for zooplankton at depths greater than 1.0 m, where oxygen conditions are better than those in surface waters.

As demonstrated in the enclosure experiments (Chapter 2), zooplankton do respond to the changes in water quality which accompany log storage. Lignin-tannin and oxygen concentrations in the enclosure experiments under three and five log experiments are similar to those observed in the log storage site during May and June in both 1985 and 1986 (Levy et al. 1985b, Bustard 1986). This suggests that the static enclosure experiments are representative of the field situation.

From the results of the log ramp experiment, it seems that sockeye fry obtain reduced amounts of food under such conditions, probably as a result of the slightly depressed oxygen conditions as well as lower food availability. Differences in oxygen between reference and treatment sites were not controlled for, and therefore, must be assumed to be part of the boom or ramp treatments. The combination of these factors during the critical early weeks of a fry's existence may reduce its survival and growth (Braum 1967).

It is not possible to quantitatively compare the log boom and log ramp feeding experiment results because the sites are located in waters of different depths and, accordingly, zooplankton populations and water quality differ. This is a fundamental problem in ecological research (Cairns and Pratt, 1986) and not easily resolved. Also, in this study, several sites within the log storage and reference areas should have been used, providing replication to see if observed patterns are reproducible. Another problem, common to environmental impact studies, is that effects must be inferred from spatial pattern alone when the impact has already occurred (Green 1979). I must assume that observed differences between reference and treatment sites would not have existed if the log storage facility had not been installed.

The data suggest that the effects of log handling activities on food supply are greater at the log ramp site than at the log boom site. This is reasonable, given that this site is used actively and more intensively than the log storage site. More importantly, the ramp site is located within the bay, enclosed by log booms and subjected to greater physical and chemical influence, so water quality, zooplankton and benthos at that site are more severely affected. These changes translate into a change and reduction in food supply available to sockeye fry and the flow through enclosure experiments demonstrate that food intake by fry is reduced. However, it may be that the water

quality changes which occur in the log storage area are more important. Fry actively avoid the log storage area probably due to severe oxygen depletion and, therefore, may never encounter areas of altered food supply. The effects described here are localized and site specific, and I can only hypothesize that sockeye fry exposed to these effects might be exposed to higher mortality rates. The sockeye habitat affected with respect to water quality and food supply is a substantial proportion of the lake shoreline available to post larval sockeye.

4. BIOASSAY EXPERIMENTS

4.1 INTRODUCTION

The study of deleterious effects of chemicals on aquatic organisms (aquatic toxicology) may include measurement of mortality, growth, reproduction and other parameters affected at the sublethal level (Rand and Petrocelli 1985). Evaluation of the toxicity of compounds is most often accomplished through bioassay tests which quantify the response of organisms to the toxicant. From bioassay results, "safe" levels for compounds can be determined (Rand 1980) which gives decision makers basic information to use for regulating and controlling toxic substances.

There are standard procedures which can be used to obtain widely comparable data on toxicity. In general, acute and chronic bioassays examine short term (usually lethal) and long term (usually chronic or sublethal) effects. Standard aquatic bioassay organisms include a wide range of invertebrates and fish (APHA et al. 1985), but most commonly rainbow trout (Salmo gairdneri) and Daphnia are used. These organisms are easy to culture and maintain, relatively inexpensive and have a long history of use in the literature (APHA et al. 1980). For these reasons, I chose rainbow trout and Daphnia as test organisms for bark leachate bioassays, in addition to sockeye salmon fry, all of which are resident in the environment being affected by log

storage.

It is well documented that bark and wood leachates are toxic (Tabata 1964, Atkinson 1971, Servizi et al. 1971, Schaumburg 1973, Pease 1974, Buchanan et al. 1976, and Peters et al. 1976). However, much of this toxicity work is inconsistent, as well as fragmented and scattered by differences in methodology, tree species and leachate concentrations. Therefore, it was necessary to determine toxicity for bark leachates which would be applicable to the Babine Lake system. Also, several authors (Pease 1974, Conlan 1975) have shown that the toxicity of wood leachates is higher in freshwater than in seawater because lignin compounds precipitate out in salt water, reducing toxicity. As a result, there is a greater potential for toxic effects to be significant in freshwater (Sedell and Duval 1985).

The main purpose of these acute bioassays was to explore the range over which acute toxicity exists. Short-term tests are acknowledged (Rand 1980) as being the first step in the study of the toxic effects of a chemical compound. Acute tests can provide values for comparison of toxicant lethality between test organisms or toxicants. A 96h-LC-50 can be defined as the quantity of toxic substance in the test solution that produces 50% mortality in test organisms which are exposed for 96 hours. However, acute tests can not be used to predict a "safe" toxicant concentration which would be unlikely to harm the

ecosystem (Buikema et al. 1982). When a chemical compound is not lethally toxic it does not follow that it has no adverse effects. Chronic toxicity tests permit assessment of adverse effects on several life stages of test organisms.

It became evident from the literature and the present study that acutely toxic bark leachate concentrations were higher than those usually measured in the field. Therefore, a study of the sublethal effects of bark leachate was undertaken. As far as I can determine, nothing has been published concerning the sublethal effects of bark or wood leachate.

The objective of this study is to determine the toxicity of bark leachate at (1) the lethal level, to have a standardized measure which can be compared to literature values and (2) the sublethal level, to examine the effects of bark leachate concentrations similar to those measured in the field.

Bark leachates (pine and spruce) produced by static leaching for different lengths of time are tested for lethal toxicity using Daphnia, rainbow trout, and sockeye salmon fry. Sublethal bark leachate concentrations are tested in long term bioassays for their effect on mortality, reproduction, molting and growth of Daphnia neonates.

I predict that the L-T concentration of bark leachates will increase with the length of time the bark is allowed to leach

under static conditions; in conjunction, the toxicity of leachates to Daphnia and fish will also increase at both lethal and sublethal levels.

4.2 METHODS

4.2.1 Daphnia bioassays

4.2.1.1 Test organisms

The Daphnia culture and facilities of the Environmental Engineering Lab, University of British Columbia were used from February to April, 1985 for all bioassay tests. A brood stock of Daphnia pulex was set up in a 4.0 l glass beaker in a constant environment chamber where the temperature was maintained at 19.5+ 0.5 deg. C with a light regime of 14 h light and 10 h dark. The brood stock was maintained at low densities by siphoning off half the culture every two weeks and adding fresh dilution water.

Daphnia were cultured in a medium described by Horvath and Russo (unpublished) which is prepared by adding specific amounts of reagent grade chemicals (Table 6) to distilled water. Particular care was taken to determine a high survival rate in this medium. A synthetic medium was chosen because its composition is known and reproducible.

Table 6. Chemical reagents and quantities used in preparation of Daphnia medium (after Horvath and Russo, unpublished).

Reagent	Amount added (mg/l)
$NaHCO_3$	96
$CaSO_4 \times 2H_2O$	60
$MgSO_4$	60
KCl	4

Daphnia were fed every other day with a prepared mixture of algae (Chlorella,) trout chow and yeast. To obtain the bioassay stock, a single female was taken from the brood culture and placed in a 4 oz round glass jar with 100 ml of dilution water. Neonates were collected as produced and pipetted into their own jars. In this manner, healthy breeding females were cultured to produce enough genetically uniform neonates for bioassay tests.

4.2.1.2 Bark leachate solutions

Dry pine and spruce bark collected at a log dump ramp during the summer of 1984 was stored in sealed dark plastic bags and kept at 4 deg. C before use in bioassays. Leachates were passively extracted from bark in a static system with bark at a density of 2.50 g per litre water held at the temperature each

bioassay was conducted at. Bark was added to *Daphnia* dilution water at given densities and allowed to leach for 1, 2 and 5 day periods, after which time the bark was removed and the bioassays started. As test solutions were replaced on a daily basis, enough leachate for the entire experiment was made at once, with the majority of it stored at 4 deg. C for later use throughout the bioassay. the lignin and tannin concentration was determined for test solutions by the method described in Chapter 2. Concentrations of bioassay test solutions were diluted as percent by volume on a volume to volume basis (eg. 10% dilution equals 1 part leachate to 9 parts dilution water).

4.2.1.3 Short term bioassays

Bioassays (96 h static) were conducted in duplicate using 100 ml volumes of serial logarithmic dilutions of the bark leachates. Additional jars of the highest sample concentration were set up for measuring water quality (temperature, dissolved oxygen, pH and conductivity) over the course of the experiment. Neonates (< 24 h old) were randomly transferred to test jars using a pipette to reach concentrations of 5 neonates/100 ml. *Daphnia* were not fed during short term bioassays (APHA *et al.* 1985). Immobilization, the criterion for death, was determined by complete lack of movement even after the test jar is rotated. The percent mortality in each jar was measured at 1, 2, 8, 24, 48, 72 and 96 h. Dead *Daphnia* were removed and test solutions were changed every 24 hours after determination of mortality.

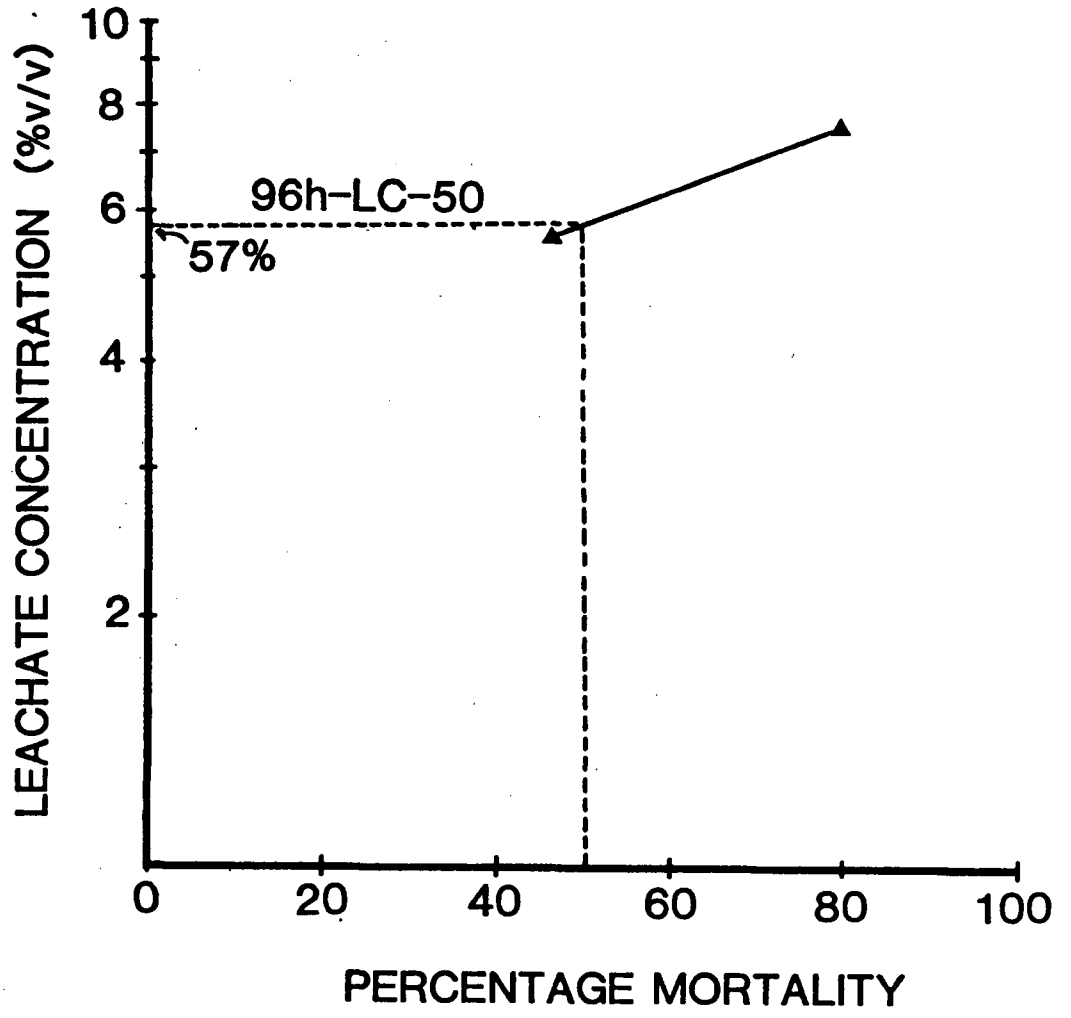

Figure 25. Example of graphical interpolation for calculation of 96h-LC-50. This graph is for Daphnia in 5 day spruce bark leachate.

Estimation of 96h-LC-50 was made by graphic interpolation (APHA et al. 1985, Atwater et al. 1983, Parrish 1983). According to standard methods (APHA et al. 1985), a LC-50 is an interpolated value based on percentages of organisms dying at two or more concentrations which produce greater and lesser than 50% mortality. These data are plotted on semilog paper with the log of concentrations vs. percentage mortality (see eg., Figure 25). A straight line is drawn between successive concentrations and the point where the line crosses the 50% mortality point is the estimated LC-50 value.

4.2.1.4 Long term bioassays

Daphnia were used to assess sublethal toxicity of bark leachates because they grow to reproductive age in <10 days and several clutches can be produced in a 30 day test period. Long term bioassays for Daphnia were conducted using serial logarithmic dilutions of 2.5 g bark/l water solutions of spruce and pine bark, leached for 24 h. Water quality was monitored every other day; oxygen levels were maintained above 6.0 mg/l. Twenty neonates (<24 h old) were used for each test dilution and maintained in 20 separate 100 ml jars. Daphnia were fed every 1 to 3 days during the 30 day experiments. Test organisms were monitored for mortality, molting and neonate production every other day, at which time test solutions were changed. Neonates produced during the experiments were counted and discarded.

4.2.2 Fish Bioassays

4.2.2.1 Test organisms

Two fish species were used for short term bioassay: sockeye salmon (Oncorhynchus nerka) and rainbow trout (Salmo gairdneri). Sockeye salmon (mean fork length = 2.9 cm) were obtained from Fulton River spawning channel, Babine Lake, B.C. and maintained in river water in holding tanks at the Department of Fisheries and Oceans laboratory located there. Fry were fed daily with Oregon Moist Pellets. Sockeye salmon produced in the Fulton River spawning channel had a high incidence of infectious hepatic necrosis (I.H.N.) disease during the 1984 field season (egg-fry survival rate of 7%, Stu Barnetson pers comm); however, during the 1985 field season the egg-fry survival rate (61%) was above average. Fry samples analyzed by Garth Traxler (Pacific Biological Station) indicated that the infection incidence of this disease was 41% in the lab-held sockeye used in bioassay experiments during 1985.

Rainbow trout fry (mean fork length = 2.6 cm) were obtained in March, 1985 from Sun Valley trout farms, a commercial supplier in the Fraser Valley, British Columbia. These fry were maintained in holding tanks with dechlorinated water at the University of British Columbia, B.C. and fed Oregon moist pellets on a daily basis.

4.2.2.2 Short term bioassays

Bark leachates for fish bioassays were produced by the same methods used for Daphnia bioassays, except that Fulton River (sockeye salmon) and dechlorinated water (rainbow trout) were used and samples were leached at 10+2 deg. C. Experiments were run with natural photoperiod (approx. 14 h:10 h of light:dark).

Bioassays were conducted in glass aquaria with 20 fish in 20 l of test solution, which is well below the maximum suggested loading density of 1g/3 l/day for static bioassays (APHA et al. 1985). All aquaria were aerated with compressed air via glass pipettes to maintain dissolved oxygen concentrations above 8 mg/l. Temperature regulation was accomplished by situating bioassay aquaria in trays with flowing water. Water quality was monitored (temperature, dissolved oxygen, pH and conductivity) daily in all aquaria.

Serial logarithmic dilutions of bark leachates in duplicate were used for 96h-LC-50 bioassays as described in Standard Methods (APHA et al. 1985). Mortality was measured at 1, 2, 8, 24, 48, 72 and 96 h and dead fish were immediately removed from aquaria and fork length and weight were recorded. Observations on the behaviour of test fry were also noted. 96h-LC-50 values were determined by straight line interpolation (Figure 25).

4.3 RESULTS

4.3.1 Bark leachate

Lignin-tannin concentrations produced by bark in waters used for bioassays varied only slightly between Daphnia and fish experiments (Figure 26). Bark leaching seemed to be equivalent in Daphnia medium, Fulton River water and dechlorinated water from U.B.C., although occurring at temperatures ranging from 9.0 to 19.5 deg. C.

Using L-T concentrations as an indicator, spruce bark produced much higher concentrations of coloured materials than pine bark. After two days, leachate concentration leveled out in both tree species, remaining the same at five days as for two.

4.3.2 Daphnia bioassays

4.3.2.1 Short term tests

Pine bark leachates were not toxic to Daphnia neonates in 96h lethal bioassay tests (Table 7). In 1, 2 and 5 day leaching experiments for pine bark there was >90% survival at all leachate concentrations and controls. For spruce bark, the 1, 2 and 5 day leachates caused significant mortality in neonates exposed in the 96h lethal bioassay tests. 96h-LC-50 values (Table 7) indicate that acute toxicity to neonates declined with

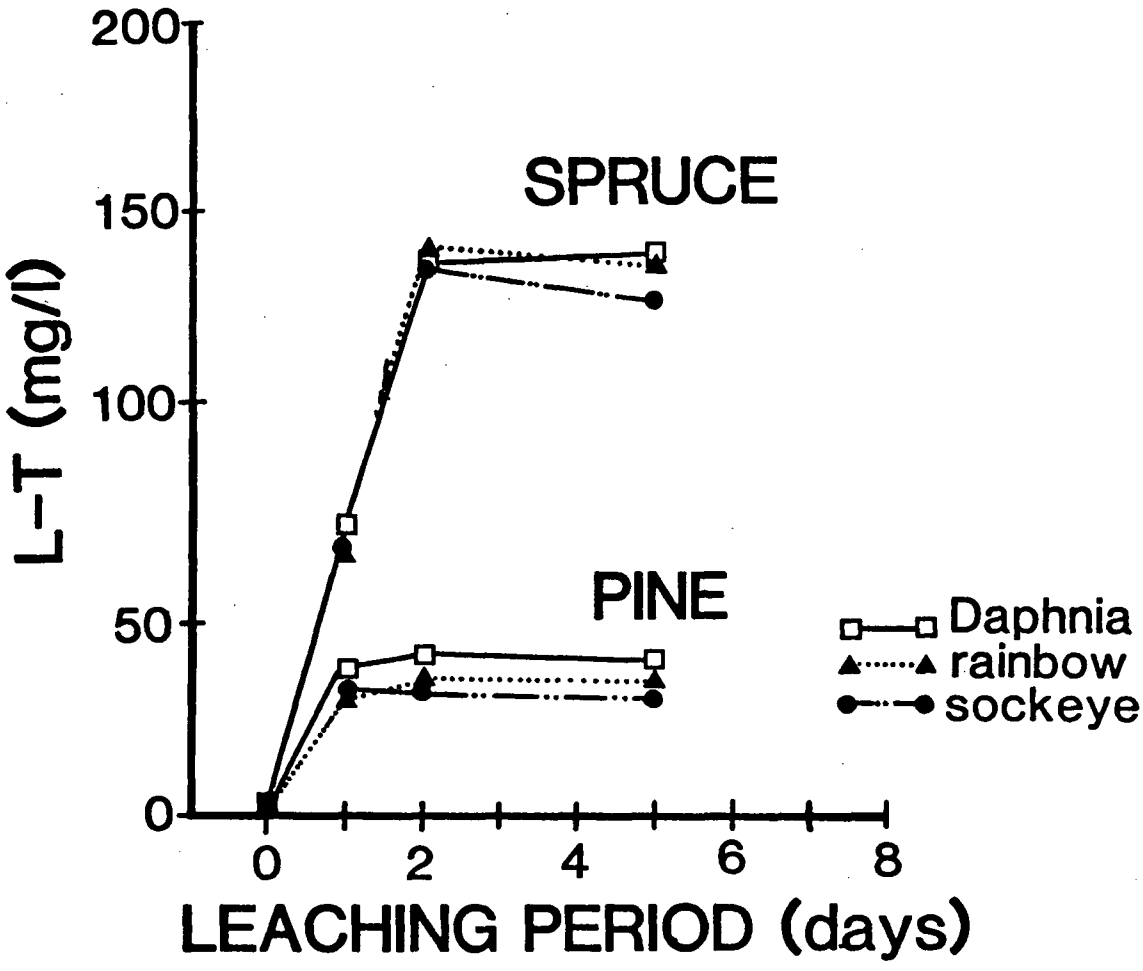

Figure 26. Lignin-tannin concentrations (mg/l) produced over time under static conditions (2.5 g bark/l water) for Daphnia and fish bioassays.

leaching time.

Table 7. 96h-LC-50 values [L-T concentration (%v/v)] for Daphnia neonates using pine and spruce bark leachate (2.5 mg/l) under uniform bioassay conditions (mg/l tannic acid as measured L-T analysis (APHA et al. 1985).

Leaching time (days)	Pine	Spruce mg/l	Spruce (%v/v)
1	not lethal	41	(58 %)
2	not lethal	67	(50 %)
5	not lethal	73	(57 %)

4.3.2.2 Long term tests

Sublethal bioassays with Daphnia neonates raised over a 30 day period demonstrate that spruce bark leachate is more toxic than that of pine bark. Long-term survival was highest for neonates raised in pine bark leachates (Figure 27), particularly at higher concentrations. Neonates in 100% spruce bark leachate were all dead (n = 20) after 7 days, whereas several neonates in 100% pine bark leachate survived until day 27 of the experiment. Oxygen levels and other water quality parameters were well within acceptable levels in all treatments relative to controls.

The proportion of neonates producing clutches in bark leachate solutions was higher for pine bark than spruce bark (Table 8). The proportion of reproductive Daphnia decreased with increased leachate concentration in both cases (Friedman two way

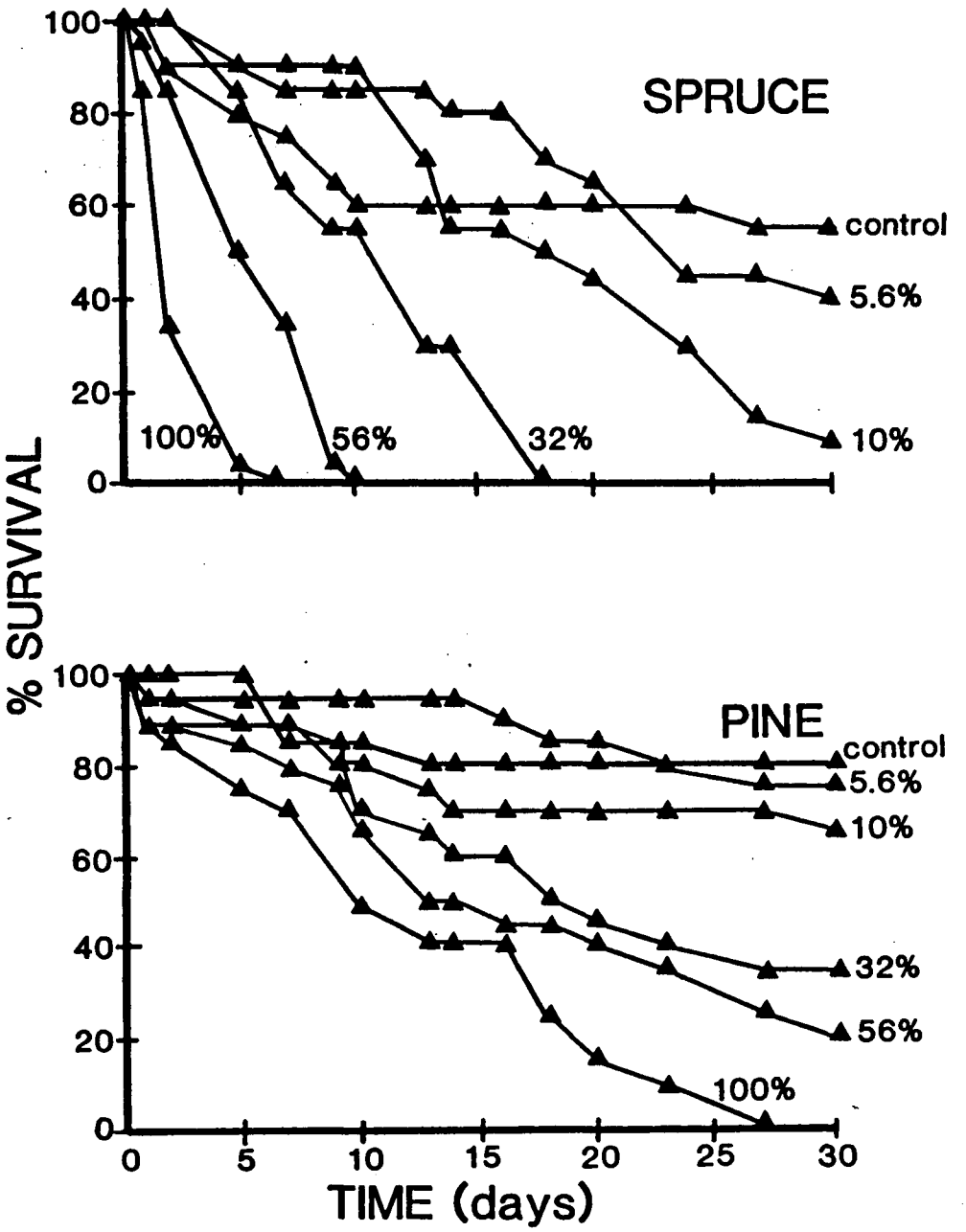

Figure 27. Survival of Daphnia neonates (n = 20) grown over 30 days in chronic bioassays for spruce and pine bark leachates. Each line represents a serial dilution of leachate, ranging from 0% to 100% concentration.

ANOVA, $p < 0.001$).

Table 8. Proportion of initial Daphnia neonate number (n = 20) that produced at least one clutch in long term bioassays for spruce and pine bark leached for one day.

Leachate (%v/v)	Spruce	Pine
100	0	0.05
56	0	0.50
32	0	0.60
10	0.45	0.70
5.6	0.80	0.80
0	0.65	0.85

The mean number of neonates produced per reproducing female was higher for Daphnia raised in pine bark leachate compared to spruce bark leachate (Figure 28). Neonate production decreased with increased leachate concentrations and there were statistically significant reductions in number of neonates produced between different concentrations of both pine and spruce bark leachate (Kruskal-Wallis one way ANOVA, $p< 0.001$).

4.3.3 Fish bioassays

Pine bark leachates were less toxic than spruce bark leachates to both rainbow trout and sockeye salmon fry (Table 9). In fact, after leaching for one and two days, pine leachate produced no mortality in bioassay fish. Only after leaching pine bark for five days was it possible to calculate a 96h-LC-50. Spruce bark leachates became more toxic with longer leaching

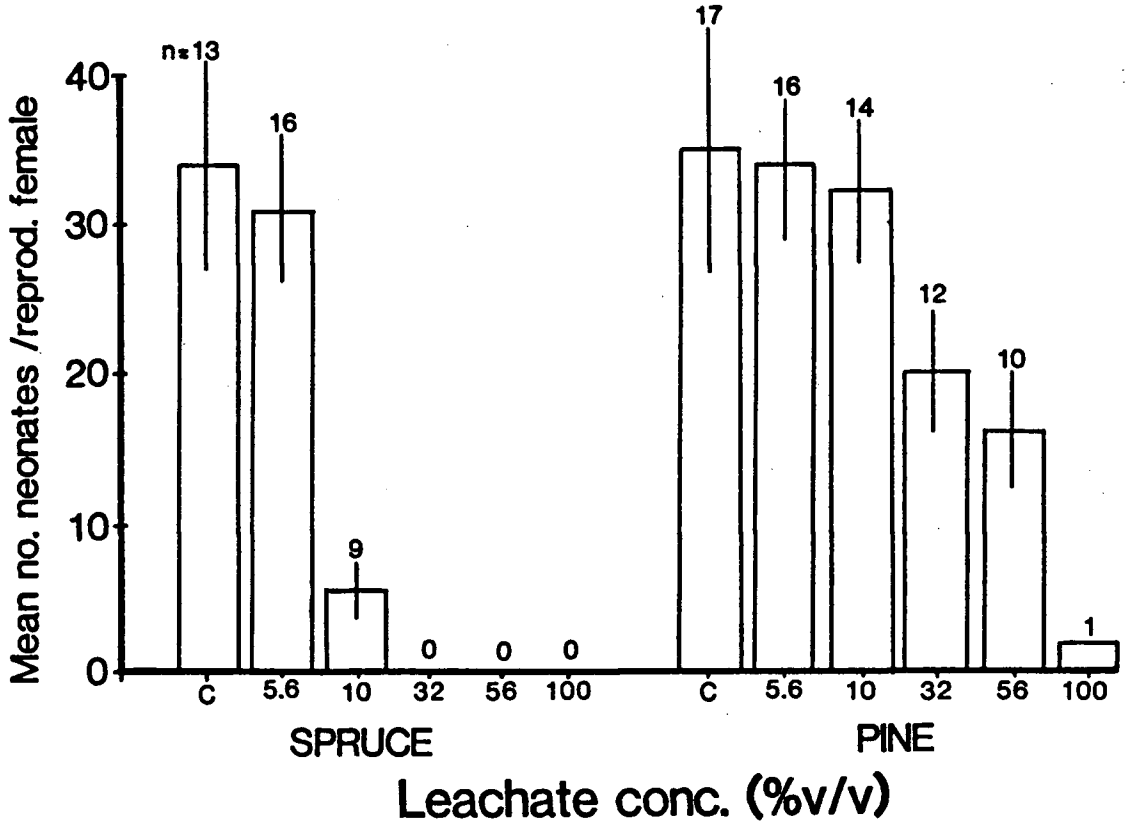

Figure 28. Mean total number (± S.E.) of neonates produced per reproducing Daphnia in sublethal bioassays with serial dilutions of spruce and pine bark (2.5 g/l) leachate (n = no. of reproducing Daphnia).

time, as demonstrated by a decrease in L-T values at LC-50.

Table 9. 96h-LC-50 values (L-T concentration mg/l (%v/v)) for rainbow trout and sockeye salmon fry using pine and spruce bark leachate.

	RAINBOW		SOCKEYE	
Leaching time(days)	Pine	Spruce	Pine	Spruce
	mg/l(%v/v)		mg/l(%v/v)	
1	not toxic	51 (73%)	not toxic	42 (54%)
2	not toxic	33 (24%)	not toxic	32 (24%)
5	15 (44%)	22 (16%)	12 (30%)	16 (12%)

Sockeye salmon fry were more susceptible than rainbow trout to the toxic effects of bark leachate. For all leachates, sockeye salmon reached 96h-LC-50 at lower concentrations of L-T than rainbow trout (Table 9). For both sockeye and rainbow fry bioassays, survival of control fish was always 95%.

4.2 DISCUSSION

The short term bioassay tests achieved the objectives of exploring the range of bark leachate toxicity and producing relative toxicity values for comparative purposes. Bark leachate is acutely toxic; it can kill 50% or more of the test organisms in a short period of time if at high enough concentrations (Rand and Petrocelli 1985). The variables which determine the bark leachate concentration are (1) bark/water ratio (2) leaching

time (3) tree species and (4) possibly other variables which I did not determine.

It has been shown in the present study and in other studies (Graham 1970, Servizi et al. 1971, Schaumburg 1973) that the loading level (wood/water) is positively related to leachate concentration. Therefore, this parameter was held constant to examine the effect of leaching time over a realistic time range; most static laboratory studies of bark leachates span leaching periods of thirty days or more (Graham 1970, Sproule and Sharpe 1970, Schaumburg 1973). There were significant differences in the L-T concentrations and often the toxicity of bark leachates produced over short time periods in my experiments.

For Daphnia, pine bark leachates were not acutely toxic, although L-T concentration did increase after one day to concentrations above 30 mg/l, where it stabilized. In spruce bark leachates, L-T concentration increased dramatically between days 0,1 and 2. This increase was not accompanied by increased toxicity. Per unit L-T, toxicity to Daphnia neonates actually decreased with increased leaching time, quite the opposite to my prediction. The toxic component to Daphnia may be volatile (Rand and Petrocelli 1985) or a colourless compound. If it exists, it was undetectable by all analytical methods used by Wentzell (in prep.).

Toxicity results for fish matched more closely to my

predictions. Again, pine bark leachate was less toxic than spruce and only acutely toxic after leaching for 5 days. For spruce both L-T concentration and toxicity increased with increased leaching time, as did toxicity to fish per unit L-T. This is opposite to the results obtained for Daphnia, despite studies which show good agreement between Daphnia and fish bioassay tests (Atwater et al. 1983).

This anomaly underlines two of the main problems of acute toxicity tests which seem to be recognized but not emphasized in aquatic toxicology research. Chemical compounds can cause different responses under a variety of test conditions (Buikema et al. 1982). For example, in the present study, due to the requirements of Daphnia and fish, they must be tested at different temperatures and in different dilution media which may affect the activity and behaviour of toxic compounds. Consequently, the measured toxicity of a compound may differ substantially between experiments (Sprague 1970, Rand and Petrocelli 1985). Also, the systematic effects of chemicals may be different for different organisms, depending on the mode of uptake and the corresponding effect. This point leads to the second problem in assessing toxicity of chemical compounds. Very little is known about the chemical structure and behaviour of many toxicants, including bark leachate. The toxicity of a chemical can be modified by toxicological interaction and small changes in constituents (Rand and Petrocelli 1985). To circumvent this problem some researchers have experimentally

extracted known compounds (eg. tropolones and lignans) from wood and examined their toxicity (Peters 1974, Peters et al. 1976). However, this information is of limited use in practical assessments of log leachate toxicity. I am not satisfied with any of the current methods for characterizing or quantifying bark leachate concentrations. These include the Pearl Benson Index, total organic carbon, C.O.D. (Graham 1970, Schaumburg 1973) and L-T concentration. L-T concentration is a good indicator for the amount of coloured materials present in leachate, but this is not necessarily related to toxicity, as demonstrated in the short term Daphnia bioassays. Unfortunately, L-T concentration has not been used by any other investigators, and it is not possible to compare my results to those obtained by others. Peters et al. (1976) measured the acute toxicity of lignan (a related compound) to coho fry and obtained 96h-LC-50 values of 60 and 64 mg lignan/l which is in the same order of magnitude as for lignins in the present study.

There were differences in the toxicity of bark leachates to rainbow trout and sockeye salmon. Sockeye salmon reached LC-50 at lower %v/v and L-T concentrations than rainbow trout. This difference is not surprising since rainbow trout are noted as being easy to culture (APHA et al. 1985), which suggests they are hardier than sockeye salmon. Also, the sockeye salmon fry in the Fulton River system had a relatively high incidence of I.H.N. disease in 1985. This disease may make them more susceptible to stresses (Garth Traxler, pers comm) which would

include exposure to toxicants. I did not have sufficient bioassay data to construct a dose response curve which may have provided me with more information on factors affecting the toxicity of bark leachates (Buikema et al. 1982) and an indication of when acute lethality ceases (Rand 1980). Sprague (1970) emphasizes that modifying environmental conditions (eg. D.O.) greatly modify toxicity and discusses several examples from the literature. For pulp and paper mill wastes, Alderdice and Brett (1957) determined that reduced D.O. concentration lowered LC 50 values, perhaps due to higher respiratory rates. In the case of wood leachates, it appears that dissolved oxygen levels may be the operative lethal agent rather than leachate, as was determined for fry in in situ bioassay experiments (Chapter 3). Pickering (1968) found that dissolved oxygen replaced zinc as the main mortality agent in bioassays with Lepomis. As a result, sublethal bioassays to test breathing rates and respiratory irregularities such as coughing have been developed (Walden et al. 1970) and would perhaps be suited to detecting toxic effects of wood leachates.

However, for the purposes of this study, I was able to demonstrate that acute lethality occurred at bark leachate concentrations higher than those measured in the field (< 2.0 mg/l L-T; Levy et al. 1985b). Therefore, it is unlikely that mortalities such as those measured in acute lab bioassays would occur in the Morrison Arm log storage area. The bark weight to water volume ratio should be measured as bark surface area to

water volume since the surface area of the bark determines the amount of extractives leached (Wentzell, in prep.). In the 1984 field season, bark treatments were used in enclosure experiments and a 0.25 g bark/l load produced greater leachate concentrations than the 1 log (0.24 g bark/l) treatment in 1985. Schaumburg (1973) also found that nearly all the colour is contributed by bark, compared to wood. Also, a higher percentage of extractives are found in the inner bark, relative to the outer bark (Wentzell, in prep.) so I would expect "loose" bark to produce leachates with a higher toxicity than "attached" bark. The toxicity of whole spruce and pine logs was examined in Chapter 2 in enclosure experiments which more realistically simulate conditions in the surface waters of a log storage area. Another way of examining the toxic effects of bark leachates at concentrations near those measured in the field is by conducting chronic bioassays. By determining the concentration of a chemical that will interfere with normal growth, development or reproduction, a more sensitive measure of toxicity than acute lethality is obtained.

Spruce bark leachate was more toxic than pine bark leachate in chronic bioassays, which supports the results of the short term bioassays. Similarly, toxicity increased with leachate concentration to the point where all test organisms in the 100 %v/v solutions died before the end of the 30 day experiment. Neonates raised in the most dilute leachate concentrations (5.6 %v/v) had survival rates very similar to controls. Survival

rates in chronic tests illustrate that leachate concentrations which did not cause mortality in short term bioassays can be chronically lethal over the lifetime of an individual.

In addition, sublethal bark leachate concentrations reduced the proportion of reproducing Daphnia and their fecundity (neonate production). that at low leachate concentrations (3.9 and 1.8 mg/l L-T for spruce and pine, respectively) Daphnia reproduction rates were very similar to those of control animals. The next step in examining the sublethal toxicity of bark leachates would be to run a long term bioassay with a logarithmic series from 10% to 1%. The concentrations would span the L-T concentrations measured in the field.

Given that stagnant water conditions do occur in the log boom (Chapter 3), it is possible that zooplankton are deleteriously affected, at least at sublethal levels. The chronic bioassay results may illustrate the mechanism for reduced zooplankton abundances observed in log treated enclosures.

5. GENERAL DISCUSSION

The effects of log storage on the aquatic ecosystem have been examined in a series of laboratory and field experiments which focus on responses by zooplankton and juvenile salmonids. Here I shall consider the consistency of the results of these experiments and their application to field situations.

The response by zooplankton to wood leachates was examined because these organisms are important to sockeye fry as a food source (Narver 1970, Rankin 1977). In enclosure experiments, field measurements, and laboratory bioassays there is evidence that zooplankton are reduced in abundance by the changes which accompany log storage. Short term bioassay results indicate that the levels of wood leachates that are lethally toxic to zooplankton are higher than those measured in Babine Lake or in enclosure experiments, assuming the response by Daphnia to leachate is similar to that of Babine Lake sockeye. Therefore, the mechanism for reduced zooplankton abundance may be linked to chronic lethality or reduced fecundity, both of which were demonstrated for Daphnia exposed to sublethal leachate concentrations. Also, examination of zooplankton abundance over the course of enclosure experiments (Figures 10 to 13) and field measurements (Figure 20) revealed a pattern common to both; large "spikes" in zooplankton density almost always occurred in control populations, whereas treated populations remained unchanged. This may be linked to a number of factors, such as

patchiness, inadequate water quality or reduced zooplankton fecundity in log affected areas. A process of verification by comparison of toxicity tests, microcosm and field effects is required according to Buikema et al. (1982), yet they make the point that this kind of work has rarely been done.

I had predicted that zooplankton diversity would decrease in response to the stress applied to the systems in the form of log treatments. Although species' abundancies declined, there was no consistent change in community diversity, so my results do not support the hypothesis that stresses applied to a community can be determined by changes in diversity indices. This could be for a number of reasons as discussed in Chapter 2. However, in the field experiments, zooplankton community diversity was consistently lower at log boom sites than at undisturbed littoral sites.

If reductions in zooplankton abundance occur under conditions of log storage in Morrison Arm, as is supported by my results, the food supply for young sockeye fry feeding in that area is potentially reduced. 24 h feeding experiments indicated that fry diet is sensitive to small changes in food abundance. The sockeye fry feeding experiments also clearly demonstrate the single peak feeding time of fry, in contrast to the feeding behaviour of fry when they become pelagic and feed in a diel pattern (Eggers 1978).

If the very earliest fry stages are the most critical (Braum 1967, Hjort 1914) and susceptible to starvation and environmental conditions, then fry entering the Morrison Arm log storage area may be deleteriously affected (reduced food intake, stress due to low oxygen levels) by the conditions within the log storage site at Morrison Arm. This scenario is often described in studies of environmental effects because it is difficult to obtain direct measurements of fish responses to environmental conditions. Oxygen concentrations within the surface waters of the log storage area dropped to lethal levels. Avoidance of the Morrison Arm log storage area by sockeye fry during periods of low oxygen was well documented (Levy et al., 1985b). However, the critical information which is required is the fate of those fry. Do they undergo higher mortality rates as a result of their contact with the log storage area?

Given that the log storage area covers about one fifth of the lake perimeter at the head of Morrison Arm, fry are effectively excluded from a large proportion of the habitat available to them upon entry to the lake from Morrison River. In the marginal habitat available within the log storage area, food levels are reduced.

Relative to the littoral area of Babine Lake, the Morrison Arm log storage area is only a minute fraction of the environment available to sockeye fry in total. However, there are strong rationales for minimizing the impact of log handling

activities on the Babine Lake system. Firstly, deleterious effects will be felt exclusively by Morrison River sockeye, a wild stock that Department of Fisheries and Oceans is anxious to maintain. The migration of sockeye fry from Morrison River is coincident with conditions which can lead to oxygen depletion and leachate accumulation. Secondly, the responsibility of minimizing or possibly mitigating environmental impacts should not be shirked, despite the apparent small scale of the problem relative to Babine Lake as a whole. There are some simple ways to reduce the severity of problems associated with log storage. Recommended practices, specific to the Morrison Arm log storage site include the following:

(1) Minimize accumulations of bark and wood debris around the dump site area, particularly where it comes in contact with water. Implement collection of loose bark and debris at regular intervals during active use of the site.

(2) Minimize the length of time the booms remain in the bay following spring break up of the ice. Before a thermocline develops, move booms to the dewatering site near Topley Landing.

(3) Consider moving booms out of the most sheltered areas first, to facilitate mixing by wind and reduction of the probability of oxygen depletion occurring.

(4) The aeration system which keeps the log storage area open in the winter could be operated briefly to destratify water in areas of low oxygen. However, there could be a problem with an immediate high B.O.D. of microbial growth on and around the log bundles.

Implementation of these practices would reduce the magnitude of effects associated with log storage in the Morrison Arm site.

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