

Ministry of Environment
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Williams Lake Laboratory Aeration Experiments

Richard N. Nordin, Ph.D. R.P. Bio
Water Quality Branch
Victoria, B.C.

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Background

Williams Lake has displayed less than ideal water quality for many years. The lake has large recreational potential if water quality were improved. Improvements to the sport fishery would also be expected with improved water quality. At present, the city of Williams Lake draws its drinking water from groundwater wells but this is of relatively poor quality and the lake might be considered as a source if water quality were improved.

Evaluation of the lake water quality has been carried out (McKean et al 1987) with the purpose of recommending a course of action to improve water quality. Aeration has been considered but until recently, the size of the lake (720 ha) has always been considered too large for this technique. New techniques recently available (oxygen generation, shore based facilities) now make the concept practicable.

Before a full scale project were even considered, the fundamental question whether aeration would achieve the intended goal of reducing phosphorus input from the sediments had to be assessed. At a meeting of the agencies involved in this question held in Williams Lake in April 1990 it was decided to test this question by using a laboratory scale experiment to simulate aeration effects in Williams Lake and to use the results as part of an evaluation of this particular approach.

Methods.

The laboratory experiments used small sediment/water column environments inside PVC tubes which were then subjected to a variety of environmental conditions. The experimental vessels were made from 67 mm ID PVC pipe, of 23 cm length with glued end caps on one end and press on caps on the other. The pipes had 16 gauge hypodermic needles inserted through the walls of the tubes about 1/4 of the way from the bottom of the tube and sealed with silicone. Approximately 250 mL of sediment was added to the tube and then 600 mL of water was carefully added by siphoning through a tygon tube. Gas supply was connected to the hypodermic needles; compressed air for tubes designed to remain at oxygen saturation, and nitrogen gas to purge oxygen from columns intended to maintain near anaerobic conditions. The number of water quality characteristics which could be monitored was limited by the volume of water in the experimental tubes.

Two experiments were run. The first was a simpler, preliminary experiment designed to provide initial results and data for the design of a second more detailed experiment.

The general sediment chemistry for Williams Lake sediments is given in Table 11 of McKean et al.(1987) and interstitial water chemistry for the sediment samples used in these experiments is given in Table 1 of this report.

The design for the first experiment is shown in Figure 1, for the second in Figure 2.

Results

EXPERIMENT 1. Three separate treatments were used: low dissolved oxygen condition (nitrogen gas purged), a set of static (no gas bubbled through) and a set of tubes aerated with compressed air. The results are compiled as Table 2. In general the three treatments achieved their expected goals - the nitrogen bubbled tubes were generally less than one mg/L dissolved oxygen, the aerated tubes at or near saturation and the static tubes intermediate (Figure 3). Redox measurements followed the same pattern with low D.O. tubes generally less than -100 mV and both static and aerated greater than +100 mV (Figure 4).

The pH was relatively stable in all treatments, near the ambient pH of the lake (8.4). The high total dissolved solids of the lake water provides a strong buffering capacity for pH.

Sulphate was interesting, showing a reduction in low D.O. columns (likely from sulphate reduction) and increasing in the aerated columns (likely from oxidation of sulphur compounds in the lake sediments) (Figure 5).

Iron increased in low D.O. columns but was relatively stable in the aerated columns (Figure 6).

Nitrogen fractions were variable in each treatment and appear to have reflected several processes occurring. From starting concentrations of less than 5 µg/L, ammonia, not unexpectedly, increased substantially in the low D.O. treatment reaching a peak of 3.45 mg/L at day 11 but decreasing over the balance of the experiment (Figure 7). Ammonia concentrations in the other treatments was generally very low. Nitrate plus nitrite starting concentration was 440 µg/L and was generally low in the low D.O. treatment. The

exception was a reported value of 15.6 mg/L (noted as predominantly nitrate) on day 20. Without more detailed data it is difficult to even speculate on what was going on in the treatment. In both the static and aerated treatments, nitrate and nitrite increased significantly, albeit irregularly (Figure 8).

The key characteristic for the experiment was phosphorus and there was no definite result which could be seen from the treatments. In the low D.O. columns, there was an immediate increase from the background phosphorus concentrations of 67 $\mu\text{g/L}$ ortho P and 77 $\mu\text{g/L}$ of total dissolved P, to almost 500 $\mu\text{g/L}$ then both decreased back to very low levels by the end of the experiment. It is assumed that the phosphorus was transferred to total P in the form of biomass, and then lost to the sediments from the water phase; however, without any analysis of the phosphorus fractions, this remains speculative. In the static tubes, there was an increase to about 200 $\mu\text{g/L}$ for both dissolved forms and in the aerated tubes the change was not large: an initial decrease and then an increase to about twice the initial concentration (Figures 9 and 10).

Conclusions from the first experiment.

There are a variety of shortcomings of a very small scale experiment such as this, in trying to extrapolate to a whole lake response. The simulation of low D.O. and high D.O. appears to be successful and a number of the water chemistry characteristics responded as might be expected. The difficulty arose in interpreting trends in the two key characteristics - nitrogen and phosphorus.

There appears to be a favourable response to aeration as seen in the phosphorus results but it is not unequivocal evidence that aeration would reduce the phosphorus input from the bottom sediments in Williams Lake.

EXPERIMENT 2.

A second more complex experiment was run to attempt to resolve some questions about (a) the potential success of aerating Williams Lake, particularly the mechanisms controlling supply of phosphorus from bottom sediments, (b) whether the input of phosphorus from sediments is controlled by physical conditions (i.e., the classical mechanism described by Mortimer (1941, 42) and Tessenow (1974) which is a physico-chemical one responding to redox and oxygen concentration) or by biological activity (i.e., the result of bacterial activity as described by Boström (1984) and other recent

studies),(c) whether decomposing algae as well as the lake sediments could be the source of phosphorus.

The experiment was set up as 8 separate treatments:

1. Aerated tubes (12 tubes, 3 for each treatment type)

- 1.1 mud + water
- 1.2 mud + water + formalin
- 1.3 mud + water + algae
- 1.4 water + algae

2. Low dissolved oxygen tubes (12 tubes)

- 2.1 mud + water
- 2.2 mud + water + formalin
- 2.3 mud + water + algae
- 2.4 water + algae

The tubes were set up as the previous set with 250 mL sediment (except treatment 4), 600 mL water, and to those tubes with algal biomass, 100 grams of filamentous green algae (from Swan Lake near Victoria). The experiment was run for 31 days with sampling done twice a week for physical characteristics (pH, redox, dissolved oxygen) and samples for chemical analysis done on roughly a weekly basis.

Results

It would appear that in this experiment as well as the previous one, the oxygen conditions appear to have met expectations. Aerated tubes were, with a couple of exceptions of equipment failure, generally kept above 8 mg/L in the columns with sediment and the formalin treated column. The two columns with algae had slightly lower D.O. concentrations, presumably because of the added B.O.D. from the decomposition of the algae (Table 3, Figure 11). The mass of algae generally decreased rapidly over the period of the experiment - from a dense surface cover to being almost absent by the end of the experiment. Likewise the redox values from the aerated columns were generally above +100 mV in the mud and water columns but between 50 - 100 in the poisoned column and in the two with added algal biomass.

In the low D.O. columns, oxygen was generally less than 1 mg/L except for the formalin (poisoned) treatment where it is assumed that C.O.D. rather than B.O.D. was

responsible for oxygen depletion . Oxygen depletion was more intense in the columns where bacterial respiration and decomposition was taking place (Figure 21). The anoxia in most of the non-poisoned tubes was fairly intense if the redox potentials (Table 3) or the strong odours are used as an indicator of the severity of the reducing conditions.

Nitrogen

In the aerated tubes the ammonia was generally low and, except for a few cases, less than 25 ug/L (Figure 14). In the low D.O. tubes, ammonia generally was low but there are occasional inconsistent results of very high concentrations of ammonia (1.9 mg/L, 7.75 mg/L and 2.6 mg/L) in different treatments (Figure 24).

Nitrate plus nitrite is relatively low in three of the four aerated tubes (poisoned, water plus algae and surprisingly the water + algae + sediment) and not surprisingly in all of the low D.O. tubes. The one treatment with relatively consistent high nitrate + nitrite is the aerated water plus sediment with concentrations up to 1.8 mg/L (Figures 15 and 25).

Phosphorus

In the second experiment, the three available fractions of phosphorus were monitored but problems with available volume for analysis meant that not every analysis could be done with every sample.

In the aerated tubes without algae the phosphorus concentration did not increase significantly above the background (beginning) levels of 67 mg/L ortho P and 77 mg/L total dissolved P - except in both tubes with algae. In those cases there was considerable elevation of all forms of phosphorus - up to 4 mg/L! (Figures 16 - 18).

In the low D.O. tubes the situation was very similar with low concentrations of P in the water + mud tubes and the water + mud + formalin. In the two tubes with algae (mud + algae + water and water + algae) there were extraordinary concentrations of phosphorus in all forms - in both cases concentrations greater than 25 mg/L total P were measured (Figure 27).

Discussion

With regard to the questions which were posed at the beginning of the description of the experiment, it would appear that some qualified answers might be deduced. First it would appear on the basis of the experiments, there does not seem to be substantial amounts of phosphorus released from the sediment even under severely reducing conditions and there seems to be no substantial release of iron so the lake does not show the conventional iron/phosphorus response to low dissolved oxygen conditions. The other potential mobilization process for phosphorus is by bacterial mineralization of sediment organic materials containing phosphorus. The results of the tubes which were poisoned with formalin, and thus with presumably negligible bacterial activity, seemed to indicate that there was little if any phosphorus originating from bacterial activity on organic sediment.

The most obvious observation from this experiment is that the tubes with added algal biomass liberate large amounts of phosphorus under both aerated and low D.O. conditions although the amounts are an order of magnitude higher in the low D.O. tubes.

It would appear that the decomposing biomass of algae is a significant source of P and would be expected to be in the lake. The experimental biomass loading rate of 100 g wet weight per tube (i.e. 100 g wet weight/21 cm² surface area for 30 days) is probably somewhat high - it is equivalent to about 20 g C/m²/d based on a wet weight to dry weight ratio of 20%, a dry weight to ash-free dry weight of 10% and an ash-free to carbon conversion of 50% (Vollenweider 1969) thus a 1% wet weight to carbon assumption. For a eutrophic lake like Williams Lake, a production rate of 5 g C/m²/d would be expected (Wetzel 1975). Thus the experimental algal input may have been about four times what might be expected in nature, so the response may be exaggerated; however, this was what was designed for the experiment so a definite response would be noted if one occurred (and it did!).

How this would bear on the question of whether or not an aeration installation would benefit Williams Lake water quality is still open to some interpretation. It does seem clear that fish habitat would be improved by aeration by greatly increasing fish habitat. If the source of P to the water column is biomass, as seems to be indicated by the experiment and also from the field data for the lake itself (McKean pers. comm.), the difference between the concentrations produced by the aerated and low D.O. conditions would seem to indicate that

aeration may cause some improvement to occur in the lake water quality. However, before extrapolation of these very small bench scale aeration results to a whole lake treatment, some additional investigation would be prudent to confirm that the origin of the oxygen depletion and phosphorus liberation is biological and chemical oxygen demand in the water column rather than sediment oxygen demand. This could be done by a detailed field sampling program using sediment traps and analysis of the C.O.D. and B.O.D. of the sedimented material and of the water column at different times of the year in comparison to the oxygen demand of the lake sediment at the same time.

Aeration is designed to deal primarily with reduction of internal phosphorus loading, and the evidence both from this experiment and from field data appears to indicate that there is little release of phosphorus from the lake sediments. Nürnberg (1987) found that there was a good correlation between estimates of internal loading derived from laboratory incubations of sediments and estimates derived from mass balance calculations. Although the methods for the lab experiments differ from those used by Nürnberg, the procedure should have produced evidence of internal loading if this were occurring in Williams Lake. Similarly, to test whether there is evidence from field samples, using a mass balance approach, the data for 1990 were used to see whether the amount of phosphorus in the lake increased in summer in amounts consistent with internal loading occurring. The results shown in Figure 30, show some increase in late summer but the amounts and the pattern are not indicative of internal loading.

Recommendations

In any consideration of improving lake water quality, one of the first considerations has to be a consideration of a phosphorus budget for the lake (a quantification of amounts of phosphorus from various sources) so that priorities for action can be set. One of the conclusions from this work has been to show that internal loading appears to be a minor component of the input of phosphorus to the lake. Thus it would appear that the goal for future efforts should be directed to the known significant inputs from the Williams Lake watershed (agriculture, erosion, lake shore septic tanks).

Although there are a variety of in-lake restorative techniques which have either been used or are being tested by limnologists around the world there are none which seem to be entirely applicable to Williams Lake. Each has been shown to be useful in particular circumstances; however, one of the difficulties in the emerging technology of lake restoration is determining the appropriate technique for a specific lake. The experiment discussed here was designed to test whether the most likely technique (hypolimnetic aeration) was feasible

consideration for improvement of water quality in Williams Lake by reduction of internal loading. The advantage that hypolimnetic aeration has over destratification aeration is the maintenance of a more or less normal temperature stratification which can be important for the protection of aquatic life, particularly salmonid fish, which might require a cold water refuge and for maintenance of a less disturbed aquatic community. Hypolimnetic aeration appears to be best suited to lakes where internal loading is a significant source of phosphorus to the lake water column and to enhance growth or survival of fish populations. In the case of Williams Lake, it would now appear from this experimental data as well as previously collected field data, that high concentrations of phosphorus in the hypolimnion appear to be the result of water column decomposition of algal biomass rather than movement of phosphorus from the sediment to the water column (internal loading). In such circumstances, hypolimnetic aeration would not appear to be the appropriate technique to use.

Overall, it would seem that the major factor influencing the high nutrient content of the lake is the external loading from the San Jose River. Attempts at reducing the loss of nutrient-bearing materials from cattle waste and soil disturbance are key in any reduction of algal growth and subsequent oxygen depletion and nutrient regeneration which occurs in Williams Lake. At a workshop held in Vancouver in February 1991 to discuss the possibilities for water quality improvement of Williams Lake, the use of settling ponds for the inflowing San Jose was also discussed. The goal of this strategy would be to trap suspended and loosely bound phosphorus before it entered the lake. This technique has been used in a variety of other situations to reduce loadings to lakes and ponds. The applicability of this approach depends on whether or not the structures to impound the water can be practically constructed. However it seems to be far more prudent in this case to direct efforts to reducing the inputs at source rather than mitigating their presence by the use of settling ponds.

For any future actions it would appear that control of external loads at the source rather than reducing internal loading processes by aeration or considering construction of sedimentation basins should be the priority.

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TABLE 1
INTERSTITIAL WATER QUALITY DATA SUMMARY
 Williams Lake sediment data July 11 1990

CHARACTERISTIC	mg/L	other units
Hardness	326	
pH	8.0	pH units
Specific conductivity	835	uS/cm
Sulphate	8.1	
METALS (diss)		
Aluminum	<0.10	
Cadmium	<0.01	
Calcium	43.0	
Chromium	<0.01	
Copper	0.04	
Iron	0.11	
Lead	<0.1	
Magnesium	53.2	
Manganese	1.11	
Molybdenum	<0.01	
Zinc	0.02	
NUTRIENTS		
N - ammonia	14.6	
N - Kjeldahl	14.8	
N - nitrate + nitrate	1.56	
N - organic	0.200	
P - ortho	1.43	
P - tot. diss.	1.45	
P - total	2.35	

Table 2. Experiment 1. Day zero concentrations are lake water analysis. Not included are total nitrogen (1030 $\mu\text{g/L}$) and Kjeldahl N (590 $\mu\text{g/L}$) Experiment was done at room temperature (20°C)

Low D.O. Conditions (tubes 1 - 4)								all units mg/m^3 ($\mu\text{g/L}$) except pH, D.O. (mg/L) and redox (mV)	
day	pH	DO	redox	SO_4	Fe-D	NH_3	NO_2/NO_3	OP	TDP
0	8.4			15100	30	<5	440	67	77
11		1.65		12900	53	3450	140	456	487
14	8.4	0.2	-293						
20		<1.2	-109	3800	397	1100	15600 (NO_2)	7	228
22									
25		0.8	-160						
27				13750	732	732	40	5	32

Static Conditions (tubes 5 - 8)									
day	pH	DO	redox	SO_4	Fe-D	NH_3	NO_2/NO_3	OP	TDP
0	8.4			15100	30	65	440	67	77
11		3.3							
14	7.7	0.9	+120						
20		1.4	+85	13300		<5	3900	337	365
22	7.6	2.75							
25		3.5	+120						
27				26800		<5	2530	204	208

Aerated Conditions (tubes 9 - 12)									
day	pH	DO	redox	SO ₄	Fe-D	NH ₃	NO ₂ /NO ₃	OP	TDP
0	8.4			15100	30	<5	440	67	77
11		7.25		25300	46	115	3350	53	71
14	8.4	7.85	+193						
20		7.48	+111	34000	15	<5	5850	82	97
22	8.5	8.35							
25		8.35	+100						
27				38500	50	<6	1615	127	136

Table 3. Experiment 2 (detailed)

Aerated Conditions

Mud + Water (Tubes 1 - 3)										
day	DO	redox	pH	NH ₃	NO ₂ /NO ₃	OP	TDP	TP	Fe	SO ₄
0			8.4	<5	440	67	77		30	15.1
4	8.6	+65								
7	8.3	+190								
10	8.3	-		10	1820	47	62			
13	8.7	-								
17	8.5	+97		6	1320	81	100			
21	9.0	-								
25	8.8	+130					102	102	10	
27	8.3									
31	8.3		8.9 (Zenon)	<5	850	72	82	91		65.9

Mud + Water + Formalin (Tubes 4 - 6)										
day	DO	redox	pH	NH ₃	NO ₂ /NO ₃	OP	TDP	TP	Fe	SO ₄
0			8.4	<5	440	67	77		30	15.1
4	8.2	+43								
7	8.3	+87								
10	8.5	-		40	140	53	63			
13	8.6	-								
17	8.6	+77		25	90	75	85			
21	9.1	-								
25	8.9	+98		19	60	86	93	96	40	
27	8.1									
31	8.2		8.6 Zenon	75	50	91	104	120		15.1

Mud + Water + Algae (Tubes 7 - 9)										
day	DO	redox	pH	NH ₃	NO ₂ /NO ₃	OP	TDP	TP	Fe	SO ₄
0										15.1
4	7.2	+113								
7	4.2	-15 prob								
10	5.0	-		11	<20	1700	1700			
13	6.9	-								
17	7.1	+83		42	<20	4100	4450			
21	7.9									
25	7.9	+82		7	20	2550	2600	2700	30	
27	7.2									
31	6.9		8.4	5	20	2550	2680	2680		59.1

Water + Algae (Tubes 10 - 12)										
day	DO	redox	pH	NH ₃	NO ₂ /NO ₃	OP	TDP	TP	Fe	SO ₄
0										15.1
4	7.2	+150								
7	2.8	+90								
10	4.8	-		12	<20	2400	2640			
13	6.7	-								
17	7.4	+82		58	<20	150	4150			
21	8.1	-								
25	8.0	+77		7	<20	1950	1980	2100	20	
27	6.9									
31	7.0		8.4	5	20	2850	3050	3050		42.4

Low D.O.

Mud + Water (Tubes 13 - 15)										
day	DO	redox	pH	NH ₃	NO ₂ /NO ₃	OP	TDP	TP	Fe	SO ₄
0			8.4	<5	440	67	77		30	15.1
4	<1	-393								
7	<1	-173								
10	<			7	<20	18	139			
13	<1									
17	<1	-228		6	<20	3	48			
21	<1									
25	<1	-215		<5	<20	34	-		87	
27	<1		7.9	1900	<20	80	160	1.11		2.6
31										

Mud + Water + Formalin (Tubes 16, 17, 18)										
day	DO	redox	pH	NH ₃	NO ₂ /NO ₃	OP	TDP	TP	Fe	SO ₄
0			8.4	<5	440	67	77		30	15.1
4	3.1	-85								
7	3.0	-97								
10	<1			30	110	110	120			
13	<1									
17	<1	-77		54	60	93	106			
21	1.4									
25	<1	-93		52	20	149	-	168	160	
27	1.3									
31			8.4	1218	20	90	97	108		14.1

Low D.O.

Mud + Water + Algae (Tubes 19 - 21)										
day	DO	redox	pH	NH ₃	NO ₂ /NO ₃	OP	TDP	TP	Fe	SO ₄
0			8.4	<5	440	67	77	30		15.1
4	<1	-378								
7	<1	-123								
10	<1			7750	40	12800	13800			
13	<1									
17	<1	-273		302	<20	17400	19700			
21	<1									
25	<1	-280		19	30	10500	16200	-	1170	
27	<1									
31	<1		6.9	-	<20	12000	26600	34500		19.8

Water + Algae (Tubes 22 - 24)										
day	DO	redox	pH	NH ₃	NO ₂ /NO ₃	OP	TDP	TP	Fe	SO ₄
0			8.4	<5	440	67	77		30	15.1
4	<	-430								
7	<1	-100								
10	<1			110	30	13500	14000			
13	<1									
17	<1	-180		28	<20	18100	25400			
21	<1									
25	<1	-190		32		9050	-	16200	1750	
27	<1									
31	<1		6.3	2600	20	482	13800	13800		111

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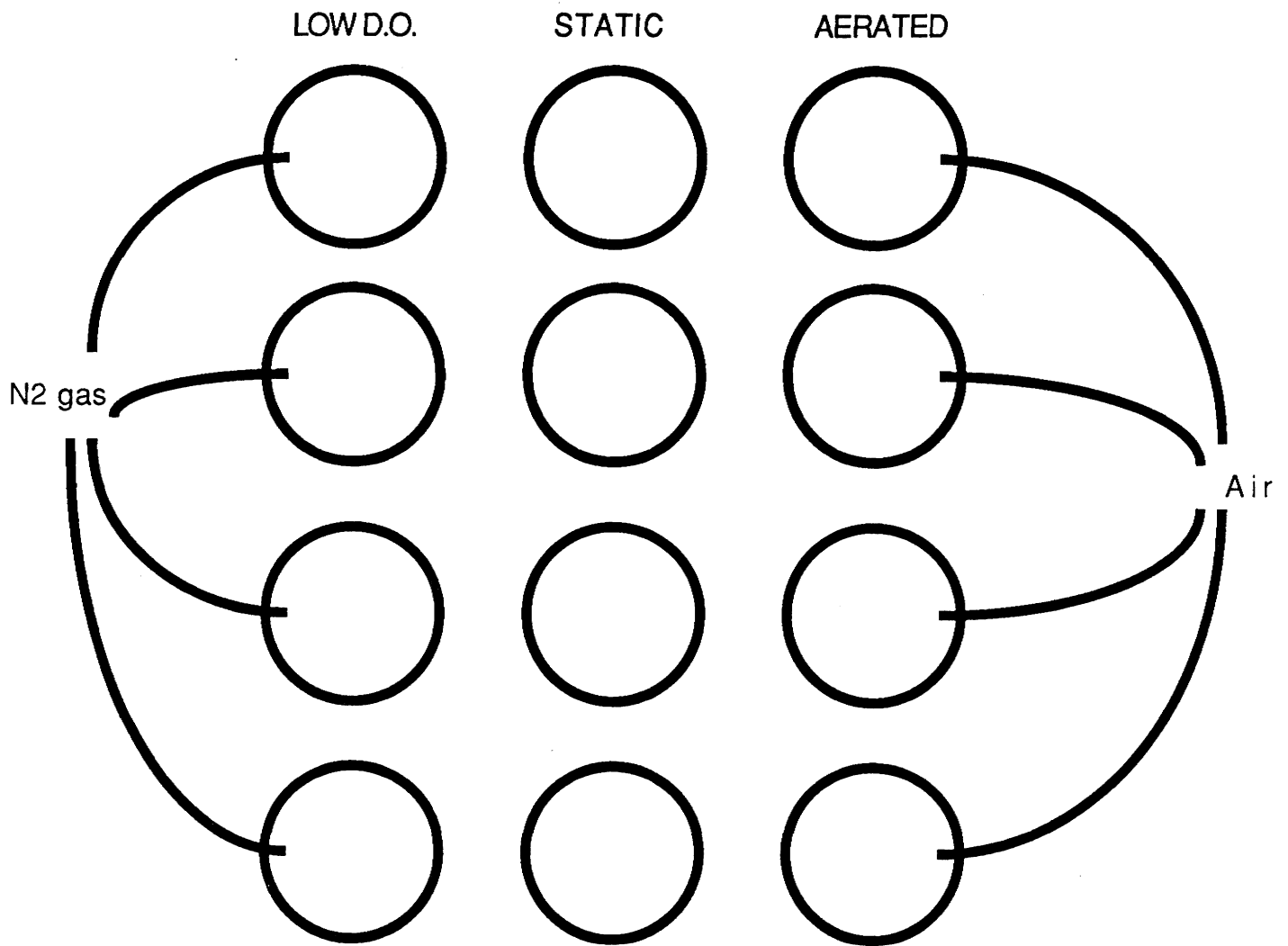


Fig 1. Physical design of experiment 1: twelve tubes and three sets of conditions.

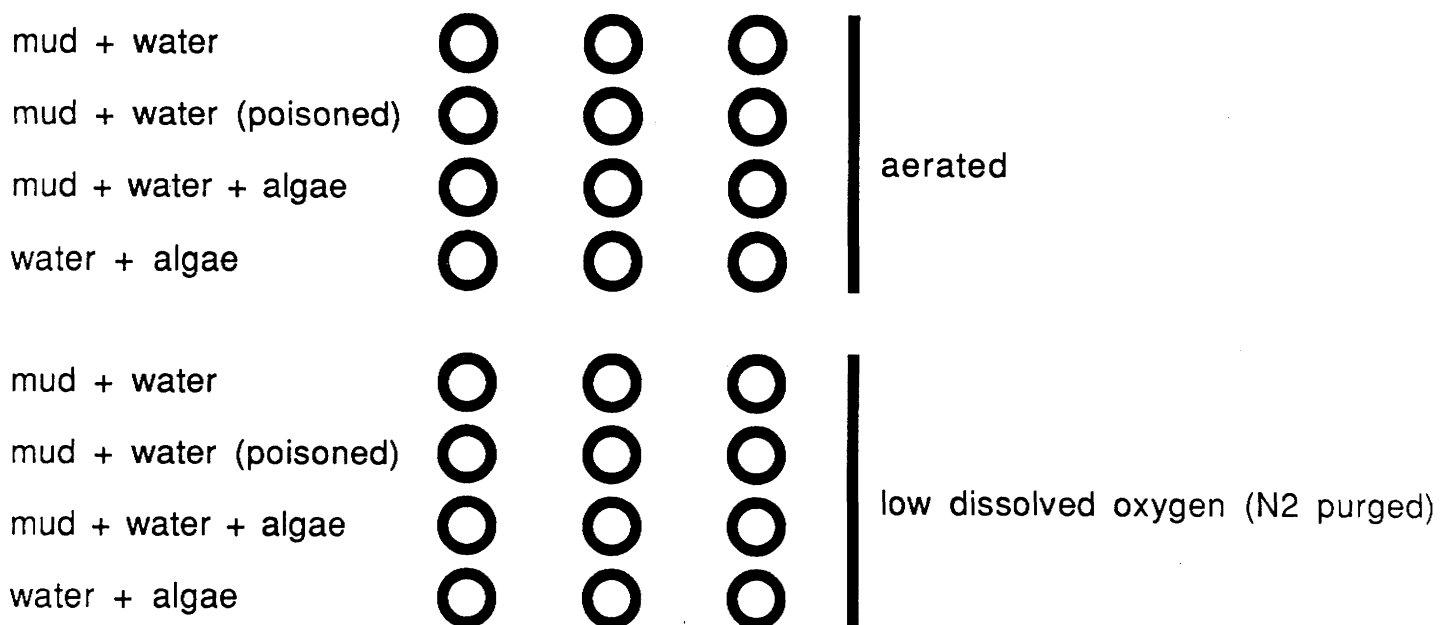


Fig 2. Physical design of experiment 2: 24 tubes and 8 different sets of conditions.

Figure 3. Experiment 1 dissolved oxygen.

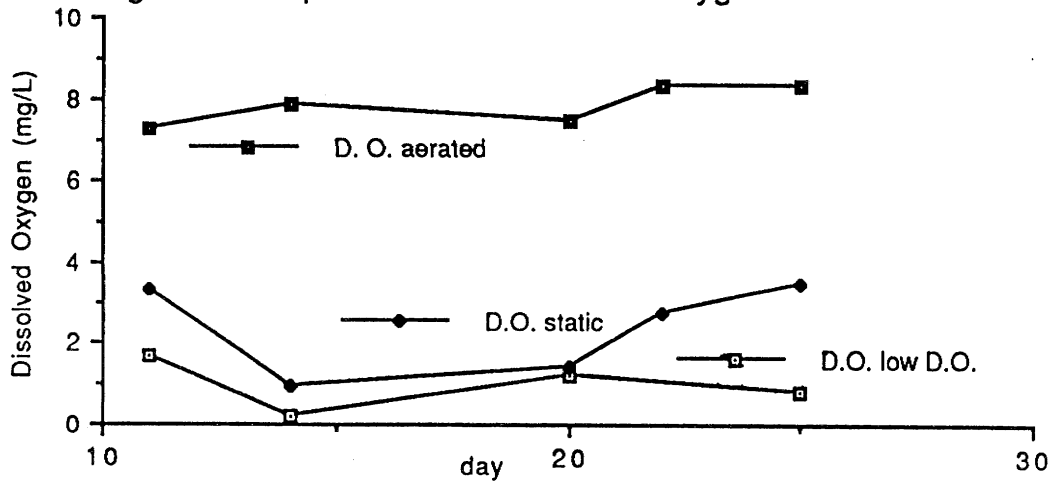


Figure 4. Experiment 1 redox.

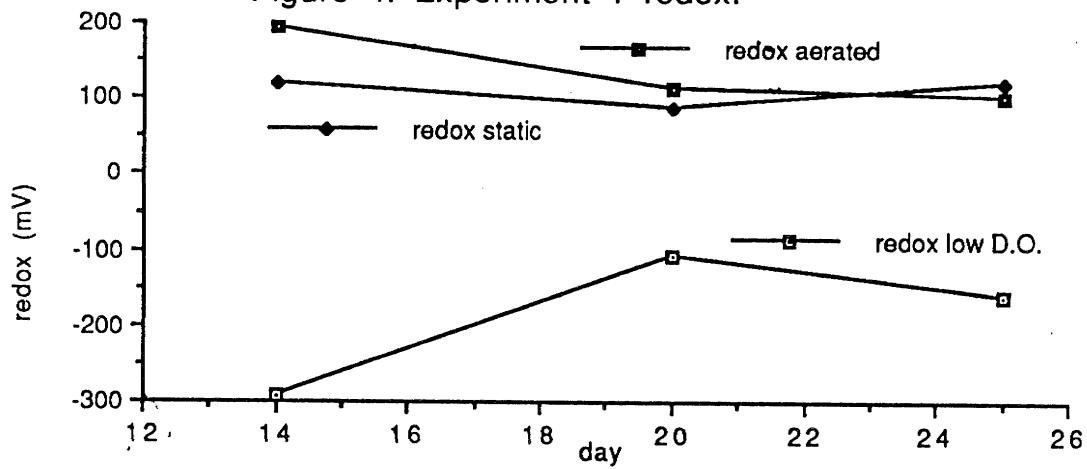


Figure 5. Experiment 1 sulphate.

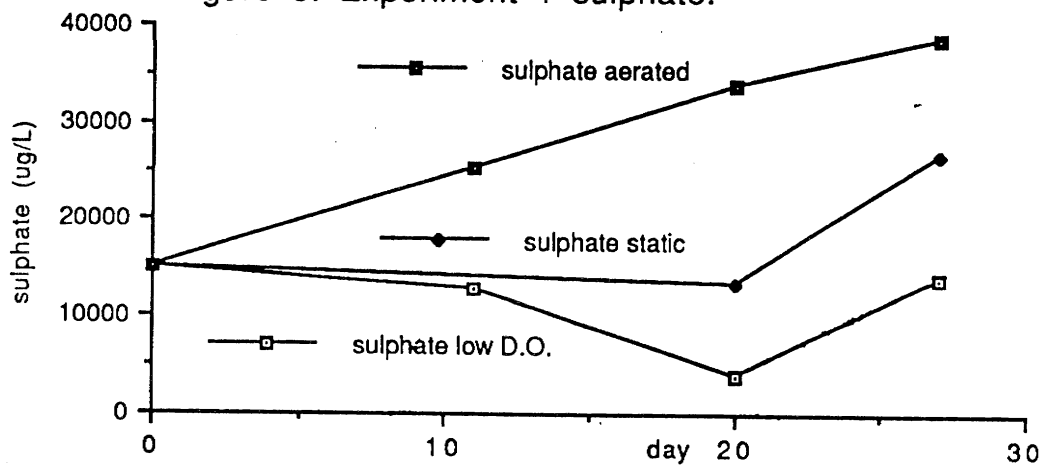


Figure 6. Experiment 1 dissolved iron.

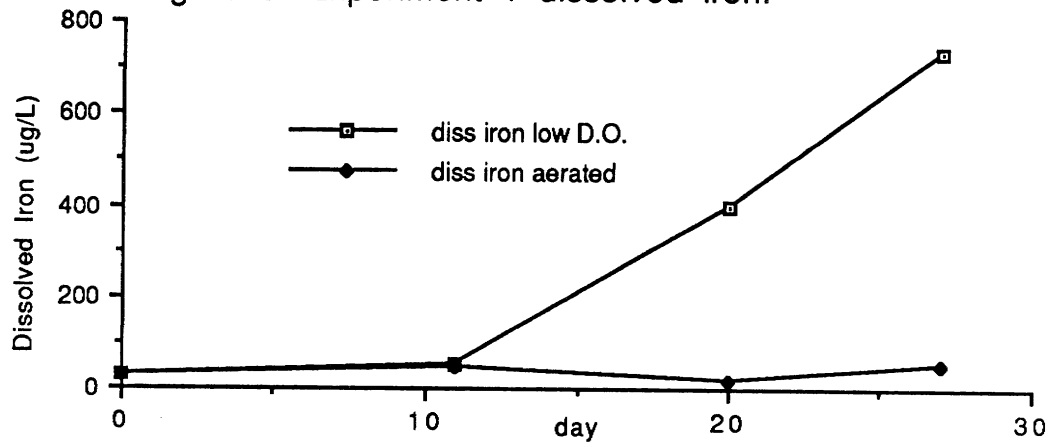


Figure 7. Experiment 1 ammonia.

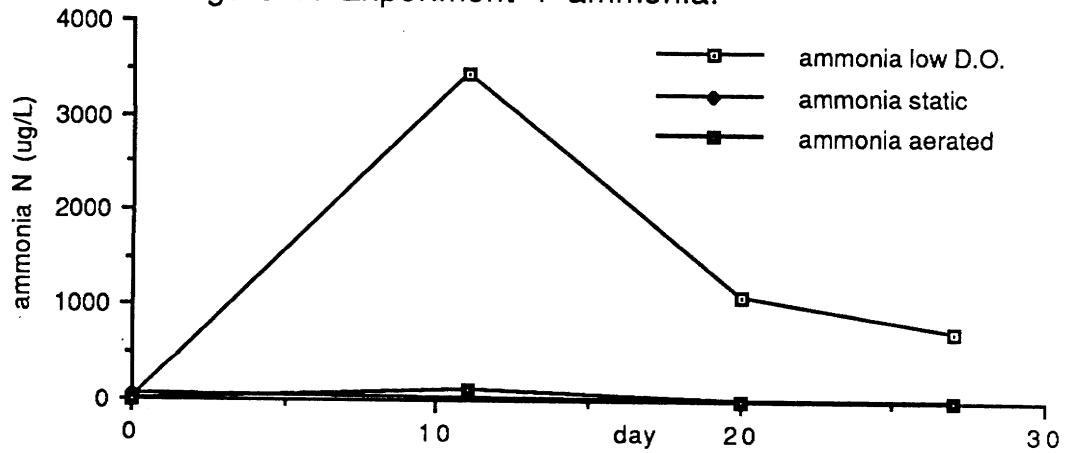


Figure 8. Experiment 1 nitrate-nitrite

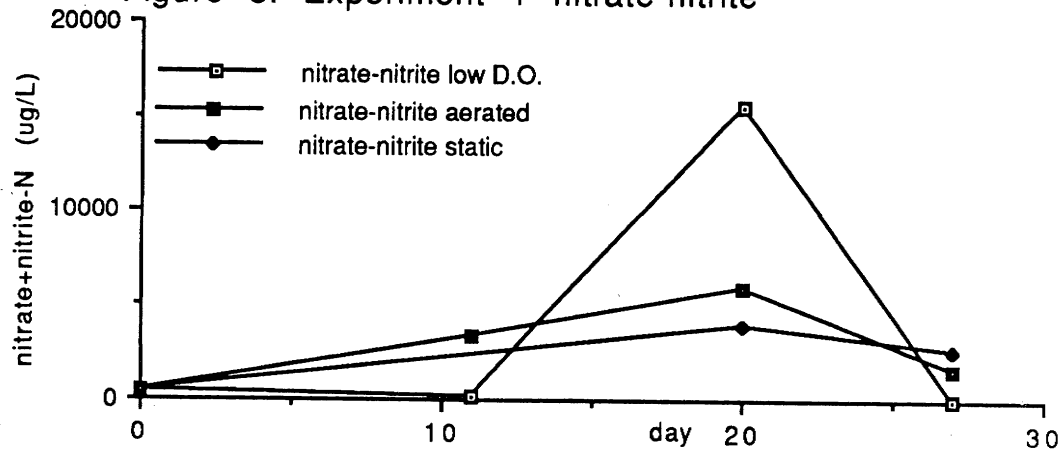


Figure 9. Experiment 1 ortho P.

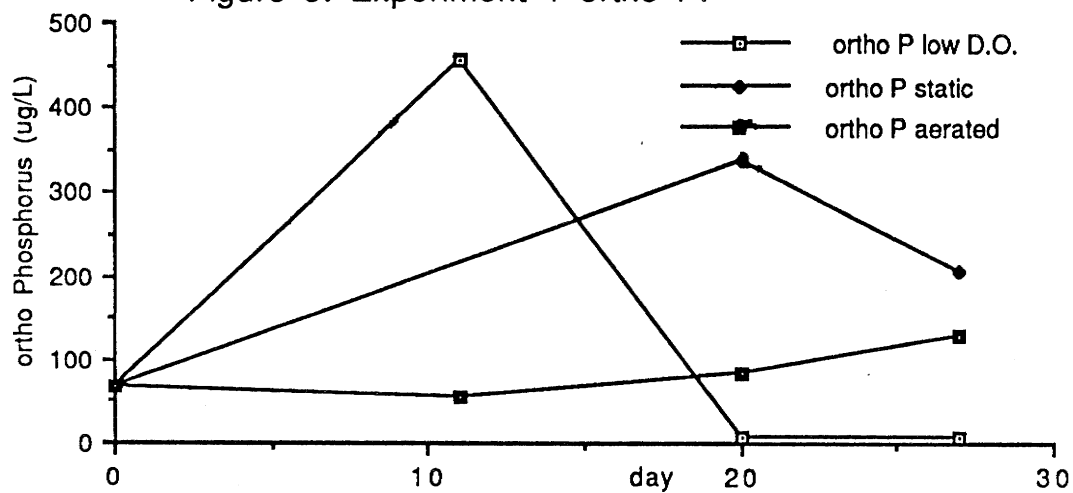


Figure 10. Experiment 1 total dissolved P

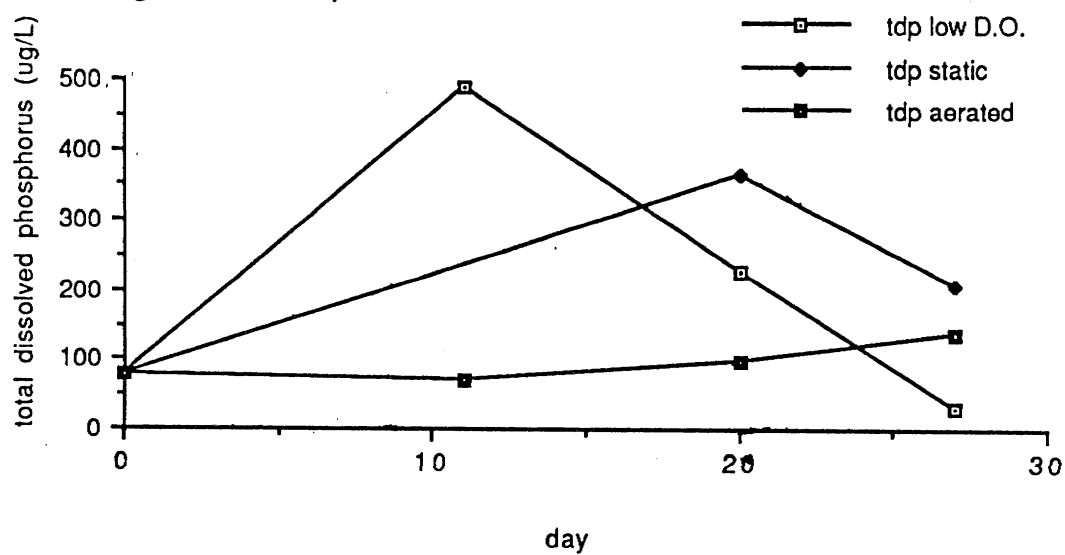


Figure 11. Experiment 2 (aerated) dissolved oxygen.

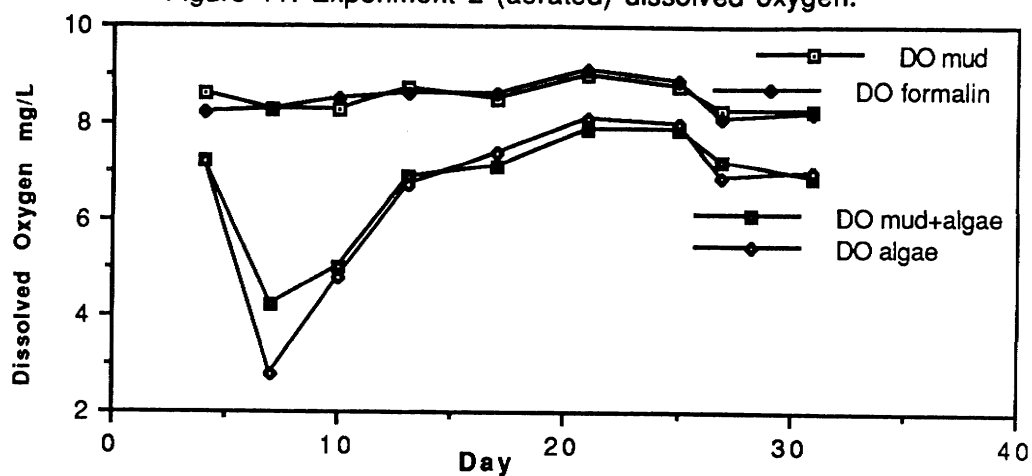


Figure 12. Experiment 2 (aerated) redox.

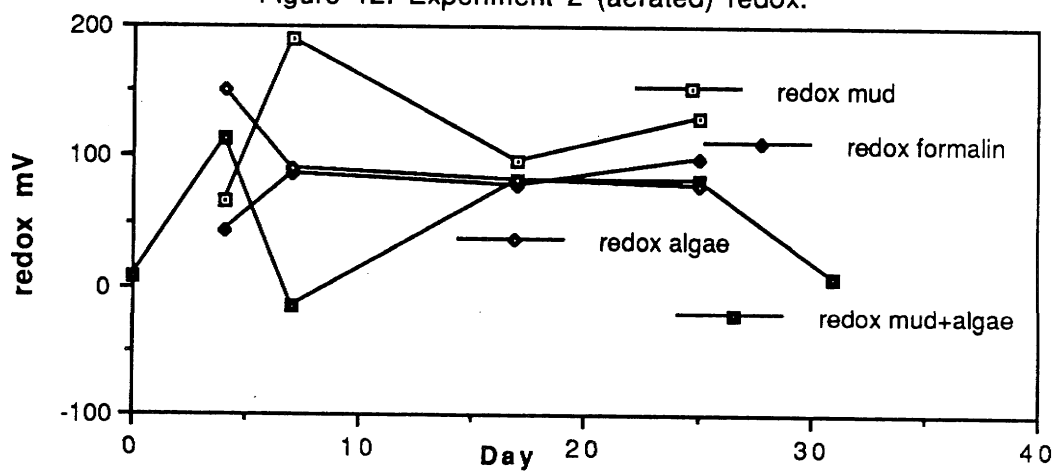


Figure 13. Experiment 2 (aerated) pH.

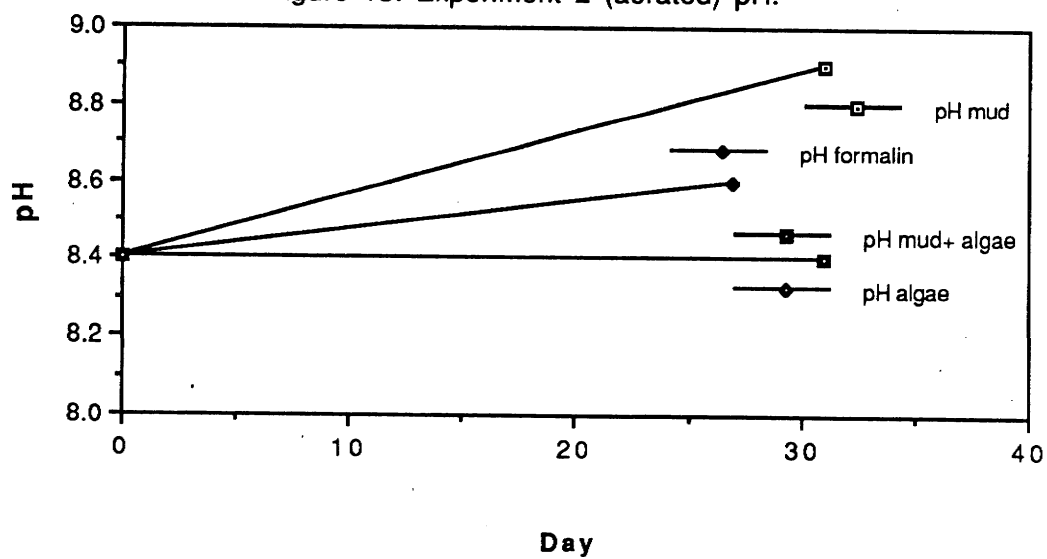


Figure 14. Experiment 2 (aerated) ammonia.

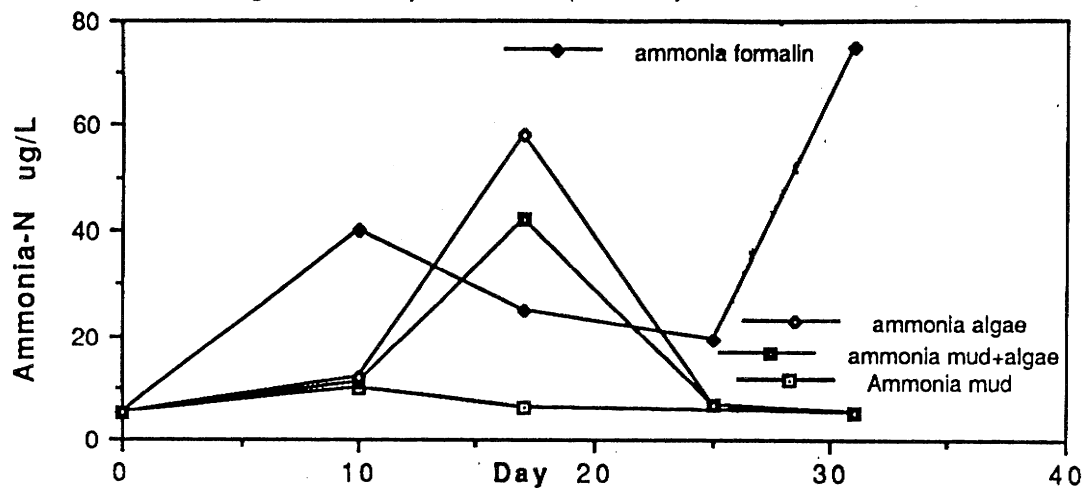


Figure 15. Experiment 2 (aerated) nitrate+nitrite

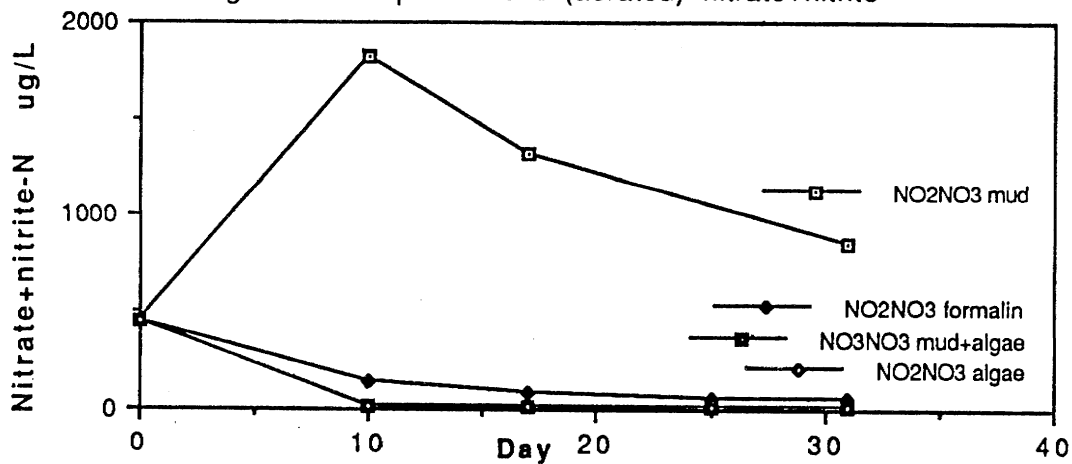


Figure 16. Experiment 2 (aerated) ortho P.

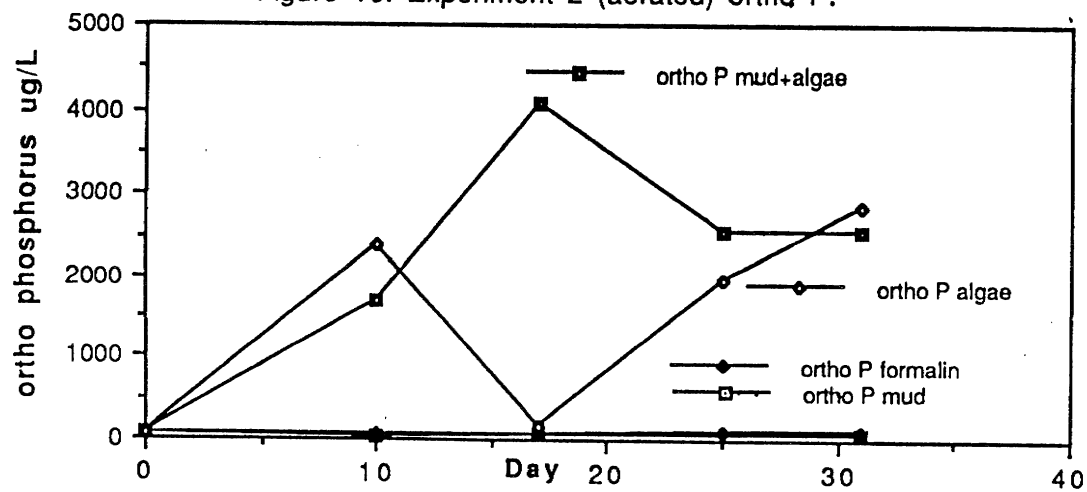


Figure 17. Experiment 2 (aerated) dissolved phosphorus.

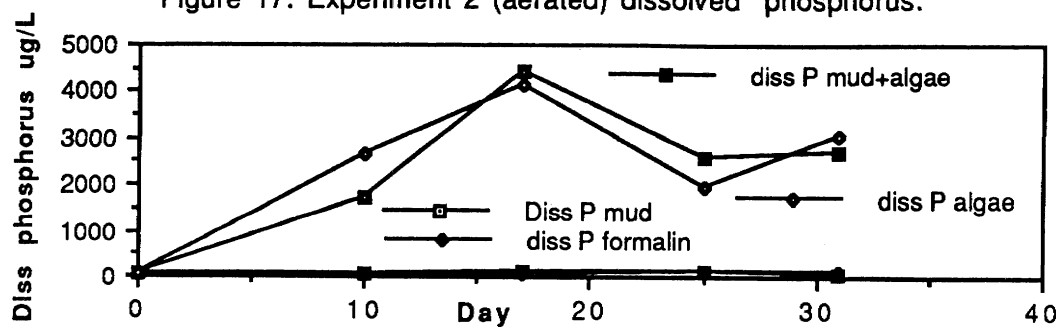


Figure 18. Experiment 2 (aerated) total phosphorus.

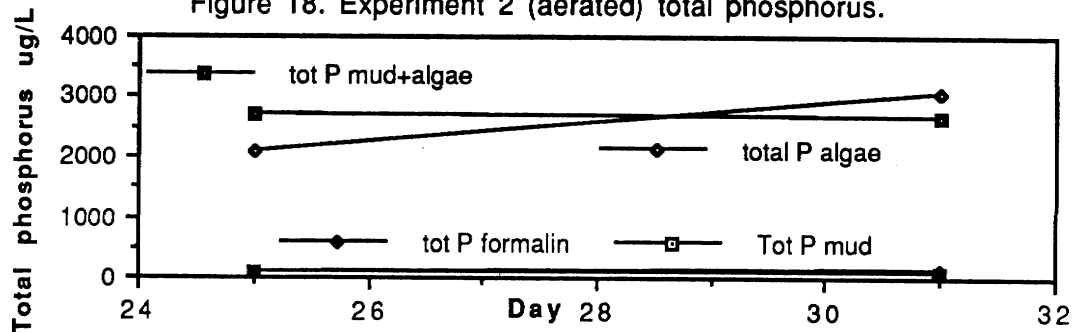


Figure 19. Experiment 2 (aerated) dissolved iron.

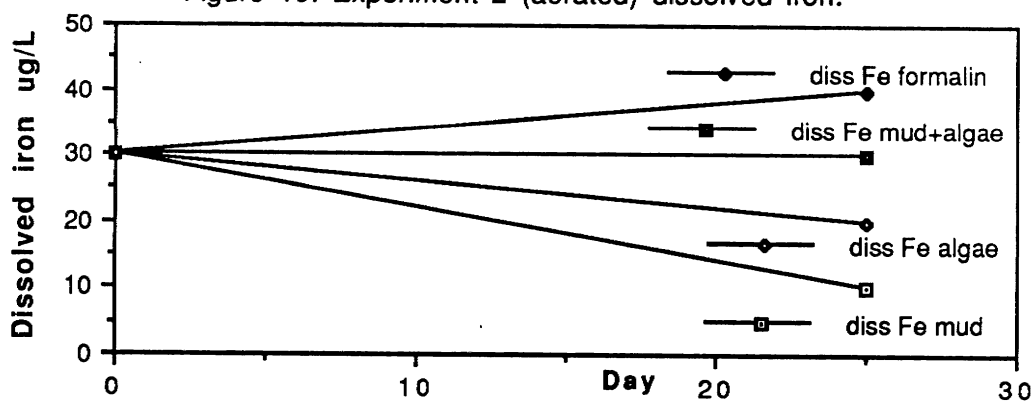


Figure 20. Experiment 2 (aerated) sulphate.

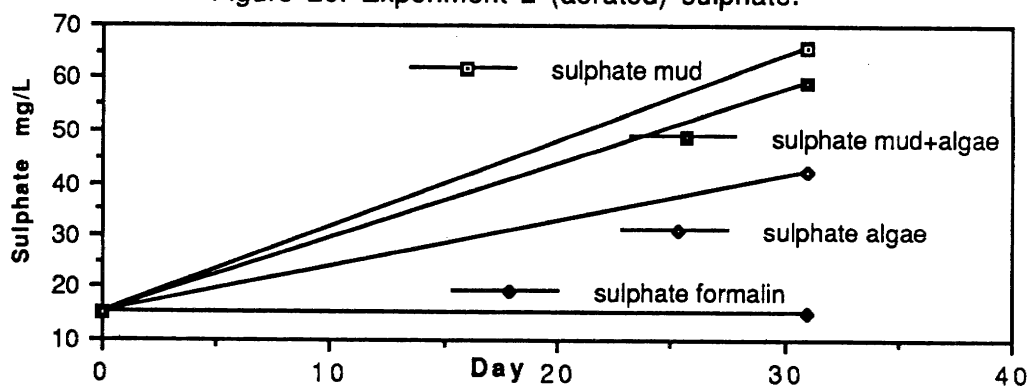


Figure 21. Experiment 2 (low D. O.) Dissolved Oxygen

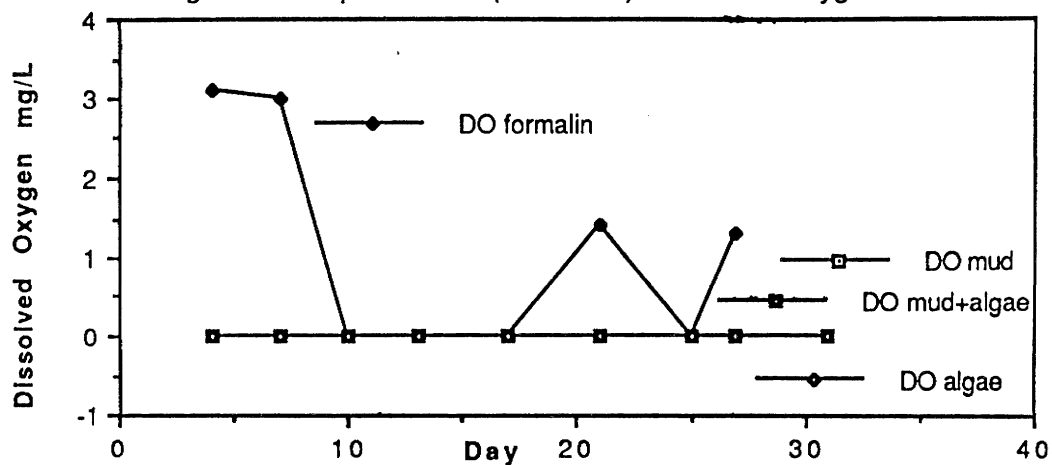


Figure 22. Experiment 2 (low D. O.) redox.

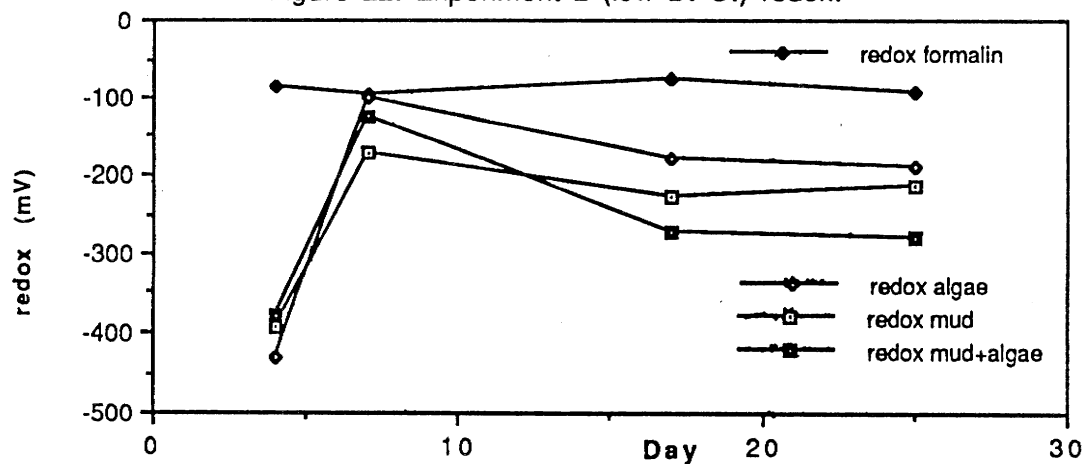


Figure 23. Experiment 2 (low D. O.) pH.

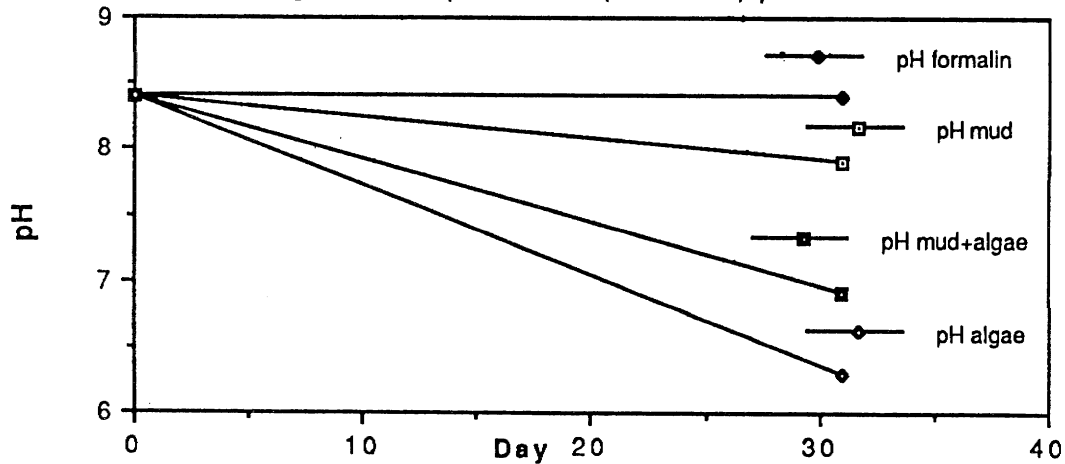


Figure 24. Experiment 2 (low D. O.) ammonia.

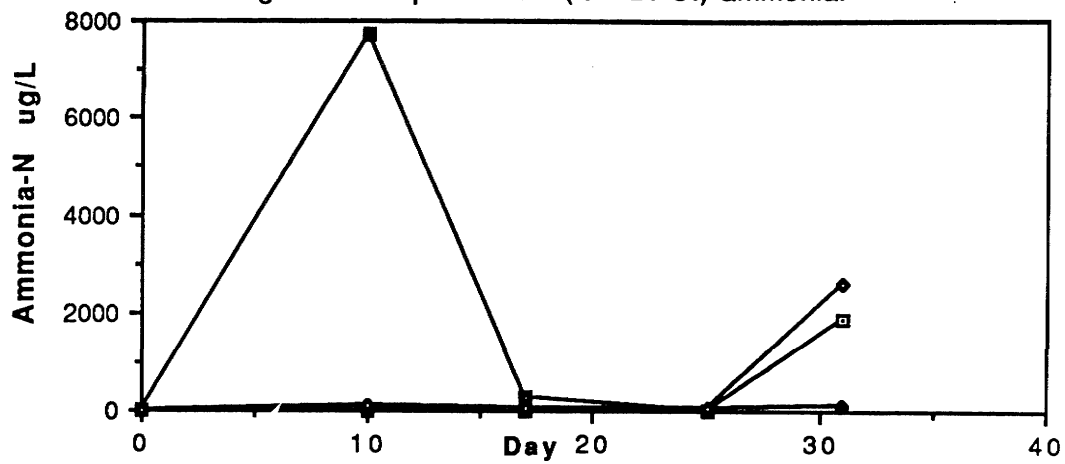


Figure 25. Experiment 2 (low D. O.) nitrate plus nitrite.

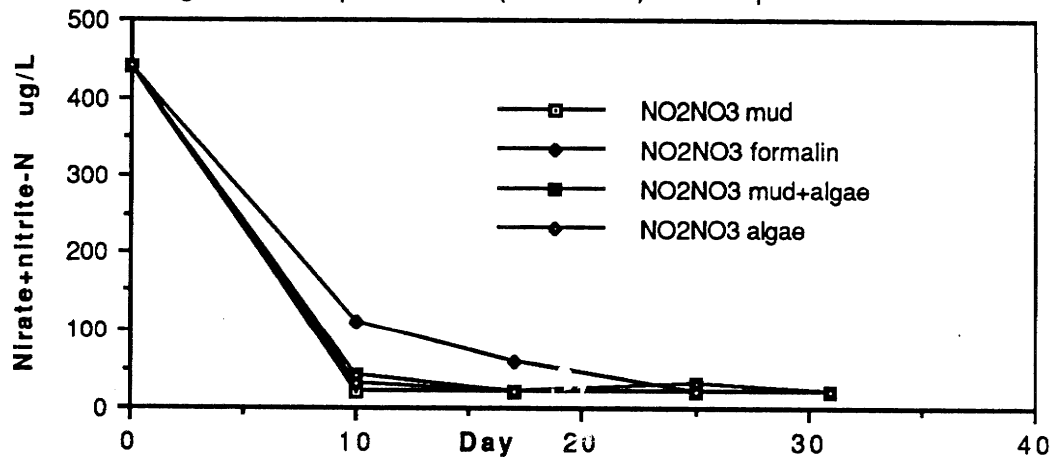


Figure 26. Experiment 2 (low D. O.) ortho phosphorus.

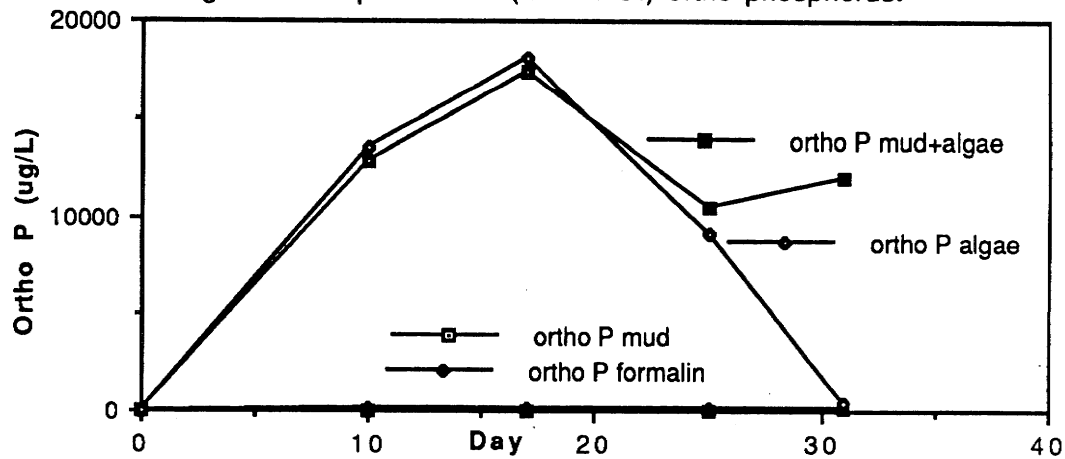


Figure 27. Experiment 2 (low D. O.) dissolved phosphorus.

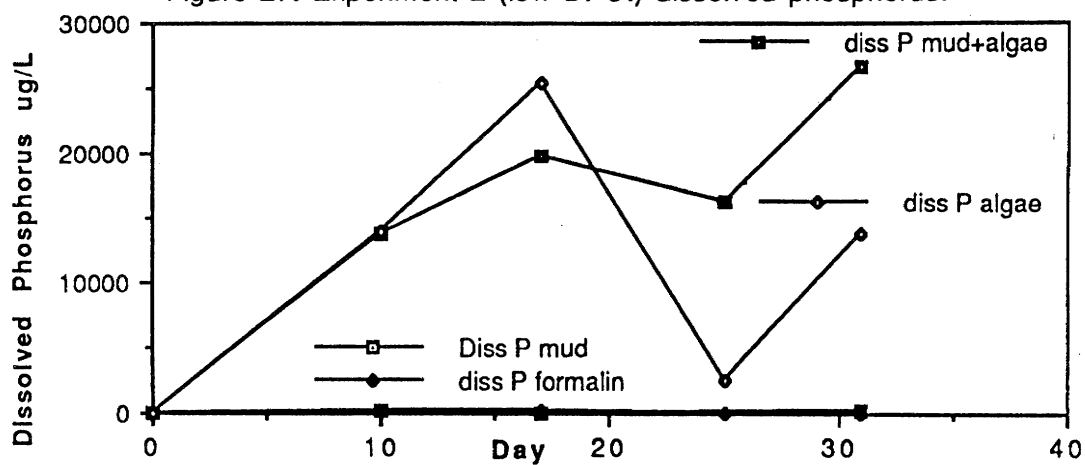


Figure 28. Experiment 2 (low D. O.) dissolved iron.

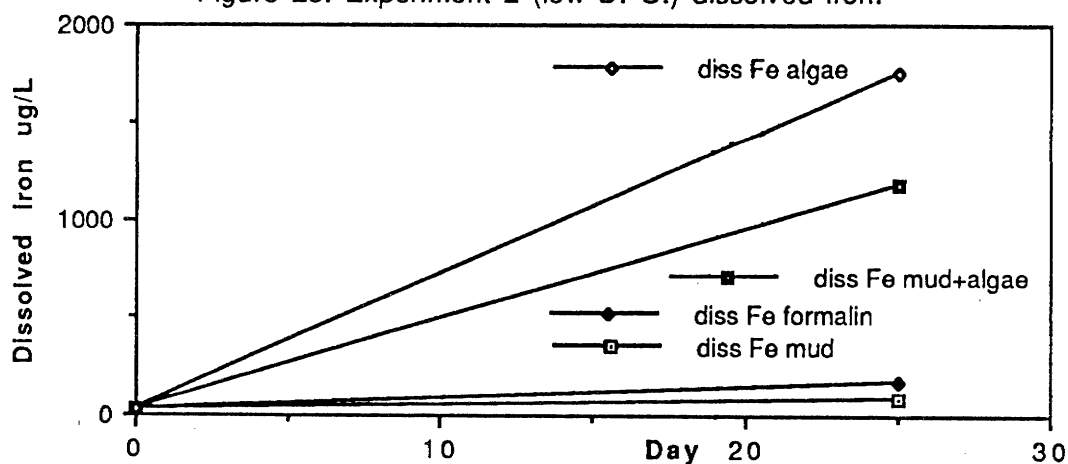


Figure 29. Experiment 2 (low D. O.) sulphate.

