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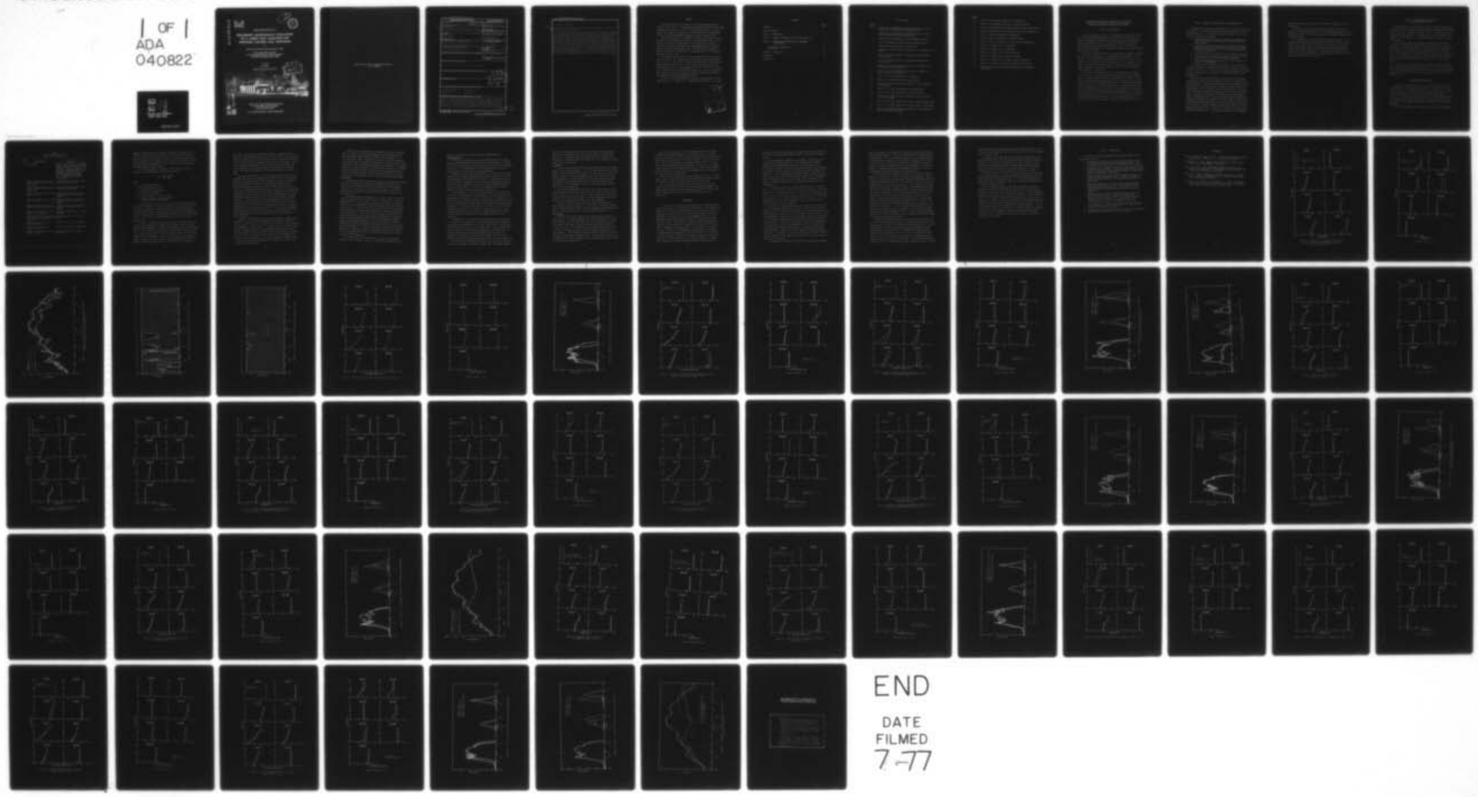

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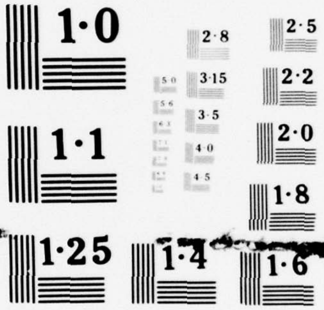

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PRELIMINARY WATER-QUALITY EVALUATION OF A LOWER POOL ELEVATION FOR PROPOSED LAFARGE LAKE, WISCONSIN

by

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May 1977

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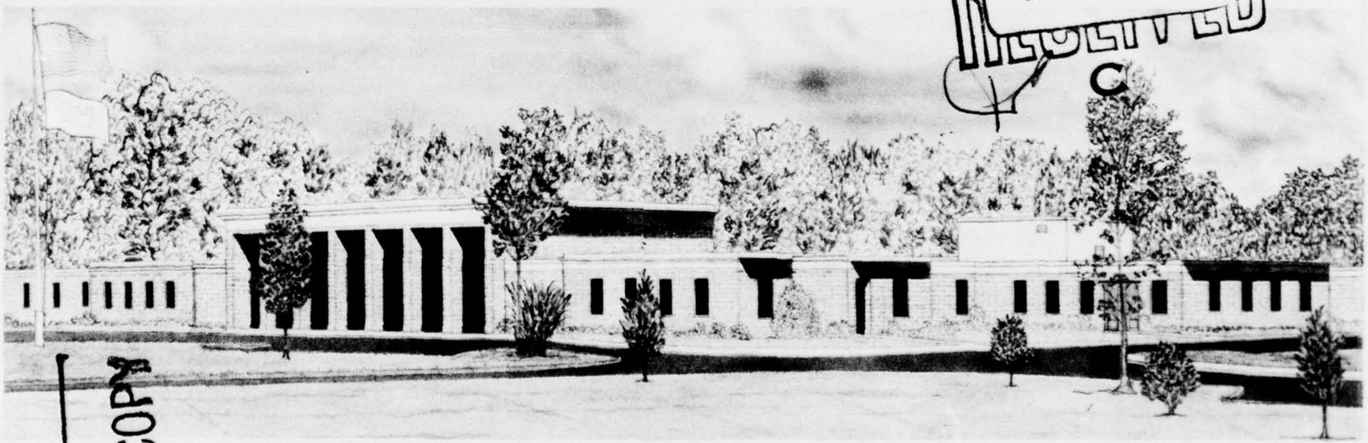

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The water quality of the proposed LaFarge Lake, Wisconsin, was investigated for a lower pool elevation (250.5 m MSL) using a version of the reservoir portion of the Water Quality for River-Reservoir Systems (WQRRS) ecological model. As in a previous study at a higher elevation (256 m MSL), the effort concentrated on temperature, dissolved oxygen, and algae. All model coefficients from the previous study were retained with the exception of Secchi disk

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depth, which was reduced from 3 to 2 m to account for probable increased turbidity at the lower pool elevation.

The simulations indicated that LaFarge Lake would be a dynamic impoundment at the lower pool elevation. Temperature stratification was weak and intermittent. Dissolved oxygen was distributed throughout the pool during periods of complete mixing and was rapidly depleted in the hypolimnion during periods of stratification. A bloom of diatoms and green algae during the spring and at least two major blooms of blue-green algae during summer and fall were predicted. The phasing of these blooms was similar to that found at the higher elevation, but the two blue-green algae blooms increased in magnitude at the lower pool level. Coliforms were found to persist longer and to be distributed throughout more of the pool at the lower elevation. The phosphorus loadings at the lower pool elevation were approximately twice as large, but since the flushing rate was increased approximately three times, the eutrophication potentials of the two pool levels, based on Vollenweider's loading curves, were similar. It was not possible to meet downstream temperature objectives for a cold-water fishery.

PREFACE

The work described in this report was performed during November-December 1976 by the U. S. Army Engineer Waterways Experiment Station (WES), Vicksburg, Mississippi, for the U. S. Army Engineer District, St. Paul (NCS). The project was authorized by Intra-Army Order for Reimbursable Services No. NCS-IA-76-86-EDH, Amendment 1, dated 9 November 1976.

This report describes a preliminary evaluation of the water quality expected in the proposed LaFarge Lake at elevation 250.5 m MSL. It is intended to provide initial guidance on the expected water quality. It does not represent a comprehensive and thorough analysis of the water quality for the proposed LaFarge Lake. Some parameters of possible interest could not be evaluated with the existing data base or within the time frame of this study.

The project was undertaken by the Environmental Effects Laboratory (EEL) at the WES. The research was conducted under the direct supervision of Mr. D. L. Robey, Chief, Ecosystem Modeling Branch, and under the general supervision of Drs. R. L. Eley, Chief, Ecosystem Research and Simulation Division, and John Harrison, Chief, EEL. Drs. D. E. Ford and K. W. Thornton served as principal investigators. Mr. Robert Engelstad, NCS, assisted in the data preparation and model simulations at the WES and provided information throughout the study.

Director of the WES during the preparation and publication of this report was COL J. L. Cannon, CE. Technical Director was Mr. F. R. Brown.

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PRELIMINARY WATER-QUALITY EVALUATION OF A LOWER POOL ELEVATION FOR PROPOSED LAFARGE LAKE, WISCONSIN

PART I: INTRODUCTION

This report presents results of a preliminary water-quality evaluation of the proposed LaFarge Lake at elevation 250.5 m MSL based upon initial mathematical modeling and associated studies. This short-term study was conducted at the U. S. Army Engineer Waterways Experiment Station (WES) for and with the assistance of the St. Paul District (NCS), Corps of Engineers (CE), beginning in November 1976 and concluding with the subject report.

Previous to the present study, a preliminary water-quality evaluation was conducted at elevation 256. The present study supplements the 256 analysis. It is recommended that WES Miscellaneous Paper Y-76-5 be consulted for the specifics at the 256 analysis. For the remainder of this report, the study at elevation 256 will be referred to as the previous study. Elevations 256 and 250.5 will be referred to as the higher and lower elevations, respectively.

The project as authorized includes a multiple-purpose reservoir above LaFarge, in Vernon County, Wisconsin, for flood control, enhancement of fish and wildlife resources, and recreation. The dam, located on the Kickapoo River, would be an earth-fill structure with an overall length of 1207 m and a maximum height of 31.4 m. The project configuration as analyzed in this study would include a conservation pool 8.8 m above the streambed at the dam. The top of the conservation pool would be at elevation 250.5 m and would impound approximately 1.26×10^7 m^3 of water having a surface area of 356 ha. This would create a lake extending 12 km upstream from the dam. At full pool elevation 265.3, 1.53×10^8 m^3 of water would be impounded, inundating approximately 1679.5 ha.

PART II: SUMMARY OF PREVIOUS STUDY FOR ELEVATION 256

A preliminary water-quality evaluation of the proposed LaFarge Lake was conducted to identify potential water-quality problem areas and to provide guidance for a more definitive study (Thornton et al., 1976). The study included the following:

- a. Algal bioassays were conducted to determine available and limiting nutrients in the Kickapoo River and Blackhawk and Redstone Lakes.
- b. A version of the Water Quality for River-Reservoir Systems (WQRRS) reservoir ecological model was applied to simulate water-quality conditions in the proposed impoundment under a variety of conditions.
- c. Nutrient loadings were computed for the proposed impoundment.
- d. The results of the studies were compared with data from Redstone, Blackhawk, and Twin Valley Lakes, Wisconsin.
- e. Temperature routings were performed for water releases downstream of the impoundment.

The algal bioassays indicated that phosphorus was the limiting nutrient in Redstone and Blackhawk Lakes, while nitrogen was limiting in the Kickapoo River. All the soluble inorganic phosphorus and soluble inorganic nitrogen were in available forms for algal uptake in the Kickapoo River.

Simulations using the WQRRS model indicated that the proposed LaFarge Lake probably would be thermally stratified from May through early September. Discharge temperatures under the recommended release schedule were adequate to support a cold-water fishery downstream to the confluence of the Kickapoo River with the West Fork. The recommended temperature schedule was disrupted only by a simulated summer storm. During this storm event, an alternative release schedule of surface withdrawal was used. Dissolved oxygen (DO) would probably be distributed throughout the reservoir until late summer. The hypolimnion would probably become anoxic for a short duration until fall overturn occurred. A bloom of diatoms and green algae would be expected during the spring with two or three major blooms of blue-green algae during the summer and fall. The simulated blooms were of magnitudes similar to those actually occurring in Redstone, Blackhawk, and Twin Valley Lakes.

Considering phosphorus precipitation and sedimentation and the selective withdrawal characteristics of the proposed LaFarge Lake, total phosphorus loadings were in the same range as Blackhawk, Redstone, and Twin Valley Lakes.

At elevation 256, the proposed LaFarge Lake was expected to be a eutrophic impoundment with water quality and clarity similar to Blackhawk and Twin Valley Lakes. Depending upon the reservoir operation, release temperatures might exceed stream standards during simulated summer storm events. There was an indication that coliform bacteria may also exceed 10,000 MPN/100 ml at certain times of the year at least in the upper portions of the reservoir.

PART III: MATHEMATICAL SIMULATIONS OF IMPOUNDMENT WATER QUALITY

The WQRRS model was originally modified for the Hydrologic Engineering Center, Davis, California, by Water Resources Engineers (WRE), Walnut Creek, California. This model has been applied to a variety of CE project studies in the past and is considered one of the more comprehensive ecological models available. An improved version of the model was used in this study.

Two of the most important assumptions and limitations of WQRRS include the one-dimensional assumption and the restriction to aerobic environments. Since the assumptions and limitations of the WQRRS model play an important role during output interpretation, it is important that they be understood. The WQRRS model, the modifications incorporated into it by the Environmental Effects Laboratory (EEL) at the WES, and its major assumptions and limitations were described in the previous study.

The 1974 data base was described in the previous study. Missing data were estimated using flow-weighting analyses or linear interpolation. Rate coefficients used in the simulations were also used in the previous study and were obtained from the literature, personal experience, and Dettmann (1974).

Water-Quality Simulations

A series of simulations were run on the 1974 data base from April through October to investigate the sensitivity of the model to various coefficients and to investigate the effects of various reservoir regulation schemes and flood events on the impoundment water quality (Table 1). The results will be discussed for the following conditions: (a) the base case, (b) sensitivity of coefficients, (c) varying light penetration, (d) alternative withdrawal schemes, and (e) flood events.

Base case

In ecological modeling, the initial calibration of the temperature

Table 1

Summary of the Simulation Runs

Run No.	Modifications	Objectives
1	Base condition	Establish base conditions. Bottom withdrawal. Secchi disk = 2.0 m. Settling velocity: ALG 1 = 0.13 m/day, ALG 2 = 0.08 m/day. Growth rate: ALG 1 = 2.2/day, ALG 2 = 1.9/day. Dettmann's half-saturation coefficients: ALG 1 = 0.002 mg/ℓ N and 0.005 mg/ℓ PO_4-P, ALG 2 = 0.018 mg/ℓ N and 0.04 mg/ℓ PO_4-P. Self-shading coefficient = 0.3 per mg/ℓ/m.
2	Changed inflowing nutrient concentrations by a factor of 0.5 times base concentrations	Determine sensitivity of algae to varying nutrient concentrations
3	Changed inflowing nutrient concentrations by a factor of 2.0 times base concentrations	Determine sensitivity of algae to varying nutrient concentrations
4	Changed the maximum Secchi disk depth from 2.0 to 1.0 m	Determine the effect of light penetration on the water quality of the impoundment
5	Changed the maximum Secchi disk depth from 2.0 to 3.0 m	Determine the effect of light penetration on the water quality of the impoundment
6	Changed fraction of solar radiation absorbed in top layer from 0.4 to 0.7	Determine the effect of light penetration on the water quality of the impoundment
7	Changed the self-shading coefficient from 0.3 to 0.6 per mg/ℓ/m	Investigate seasonal variations in light extinction
8	Modified the release schedule so that all of the outflow was released from the top gate	Determine the effect of surface withdrawal
9	Simulate a 25-year storm event starting on 10 April	Determine the effect of a spring flood
10	Simulate a 2-year storm event starting on 10 July	Determine the effect of a summer flood
11	Changed the coliform decay rate from 1.4/day to 1.2/day	Determine sensitivity of coliforms to decay rate

submodel is important since it affects and moderates biological and chemical reactions and the mixing regime. Less stratification was expected at the lower pool elevation because of the smaller volume and larger flushing rates. Approaches used to estimate the degree of stratification included consideration of flow-through rates and comparisons with other lakes of similar geometry.

Orlob (1969) described a densimetric Froude number F_D that is a function of flow-through rate and reservoir geometry.

$$F_D = \frac{LQ}{HV} \sqrt{\frac{1}{g\varepsilon}} \qquad (1)$$

where

L = reservoir length (m)

Q = flow-through rate (m^3/sec)

H = mean depth of reservoir (m)

V = reservoir volume (m^3)

g = acceleration due to gravity (m/sec^2)

ε = average normalized density gradient (taken by Orlob to be 10^{-6}/m)

If F_D is much less than $\frac{1}{\pi}$ or 0.32, then the reservoir is likely to be strongly stratified. At elevation 250.5 with a flow-through rate of 6 m^3/sec, F_D equaled 0.52, which is typical of conditions during the onset of stratification. This indicated that the proposed LaFarge Lake at elevation 250.5 would be only weakly stratified, if it was stratified at all. In contrast, at elevation 256 with the same flow rate, F_D equaled 0.16.

In Figure 1, simulated temperature profiles are compared with temperature profiles measured at Turtle Lake--a natural lake of similar mean and maximum depth, but slightly smaller surface area that is located near St. Paul, Minnesota, approximately 240 km northwest of the project site. Temperature profiles for Turtle Lake were taken from Ford (1976) and were used to determine the effects of morphology on the temperature regime. Temperature data for lakes of similar morphology in the immediate vicinity of LaFarge Lake were not available at the time of this study. The predicted temperature profiles for LaFarge Lake were similar to those measured in Turtle Lake from the middle of April (Julian Day 105) through the end of June (Julian Day 180). Thereafter, advective forces kept the reservoir isothermal through the end of October. Periods of stratification and complete vertical mixing were intermittent and highly dependent on local meteorological conditions and flow-through rates. The simulated temperature profiles were typical of a polymictic lake.

Since the hydrodynamics of a reservoir significantly influence impoundment water quality, consideration was given to the vertical placement of the inflowing waters and the withdrawal zone of release waters. Because the placement of inflow is significantly influenced by the density differences between the inflowing stream and the reservoir waters, the temperature differences of the two waters must be considered. A comparison of observed inflow temperatures, predicted reservoir surface temperatures, and predicted reservoir release temperatures from bottom withdrawal provided insight as to the inflow and withdrawal placement (Figure 2). Vertical mixing within the impoundment was sufficient to completely mix inflowing waters, even though these waters were cooler and more dense after day 200. The inflows were, therefore, distributed throughout the entire impoundment (Figure 3). The lack of any substantial temperature stratification allowed water to be withdrawn from the entire impoundment even when releases were made from the reservoir bottom (Figure 4).

Changes in DO concentrations were closely related to changes in the temperature structure of the impoundment and to changes in the phytoplankton population (Figure 5). The DO was initially isotropic throughout the impoundment. During periods of stratification, the DO profiles were typical clinograde curves. The oxygen demand in the lower layers resulted from benthic respiration and the respiration and decay of biomass that settled out of the upper layers. Whenever the impoundment mixed completely, the DO returned to an isotropic condition. During periods of algal blooms, when the impoundment was stratified, DO became supersaturated in the epilimnion.

Simulation of the phytoplankton population indicated a prolonged bloom of diatoms and green algae during the spring months with two blue-green algae blooms during the summer and fall (Figure 6). In all simulations, Algae 1 represented a composite of diatoms and green algae, while Algae 2 represented a composite of blue-green algae. While the magnitudes of the phytoplankton predictions should not be considered absolute, the temporal pattern of algae blooms is indicative of probable conditions, and the predicted concentrations are within the range of visible and nuisance blooms. These plots represent the surface layer concentrations of phytoplankton only and not a depth-weighted average.

Sensitivity

In the previous study, sensitivity analyses were conducted on Algae 1 settling and growth rates, nitrogen and phosphorus half-saturation coefficients, and Algae 2 growth rate and phosphorus half-saturation coefficient. None of these coefficients were changed. Sensitivity analyses were conducted on inflowing nutrient concentrations and coliform decay rate.

Inflowing nutrient concentrations (i.e., NH4-N, NO3-N, PO4-P) were doubled and halved to assess their effect on DO and algae. In general, during periods of stratification, doubling the nutrient concentrations resulted in an increased oxygen demand and slightly lower DO concentrations (Figure 7). During periods of complete mixing, DO concentrations either increased or decreased depending upon the magnitude and phasing of the algae blooms. Larger algal blooms resulted in supersaturation during blooms and increased oxygen demand following them. Halving the nutrient concentrations resulted in less oxygen demand and slightly higher DO concentrations during periods of stratification (Figure 8). Doubling the nutrient concentrations resulted in increases in all three algal blooms (Figure 9). The first blue-green bloom nearly doubled in magnitude. Halving the nutrient concentrations resulted in a reduction in the spring bloom and a shift from blue-green to green algae in the other two blooms (Figure 10).

The sensitivity of the coliform decay rate was investigated by changing it from 1.4/day to 1.2/day. No substantial difference in the frequency occurred when coliforms exceeded 10,000 MPN/100 ml.

Light penetration

The effect of increased and decreased light penetration on water quality was investigated by varying the Secchi disk depth, the fraction of solar radiation absorbed in the top layer, and the algae self-shading coefficient. In the WQRRS model, the Secchi disk depth is used as a relative measure for the depth of light penetration.

Changing the Secchi disk depth from 2 to 1 m and from 2 to 3 m had minimal effect on thermal stratification (Figures 11 and 12). The surface temperatures were similar for all cases. A Secchi disk of 1 m resulted in slightly stronger stratification, but stratification did not last any longer. Changing the fraction of solar radiation absorbed in the top layer from 0.4 to 0.7 had a more noticeable effect, but the change in stratification was still minimal (Figure 13).

Light penetration or Secchi disk variation had a more apparent effect on DO. During periods of stratification, DO concentrations in the hypolimnion were higher with a Secchi disk value of 1 m (Figure 14). On days 210 and 225, a slight temperature stratification restricted the surface DO from entering the lower layers and resulted in lower oxygen concentrations. Differences in DO between Secchi disks of 2 and 3 m were, in general, less than 1 mg/ℓ (Figure 15). When the fraction of solar radiation absorbed in the top layer was changed, shifts in the timing of algal blooms also accounted for the observed differences in DO (Figure 16).

The magnitude of the green-diatom bloom was less at a Secchi disk of 1 m than at 2 m (Figure 17). The first blue-green bloom increased in magnitude and lagged the base by about 10 days, while a second blue-green bloom did not develop. At a Secchi disk of 3 m, the green-diatom bloom was similar in both timing and magnitude to that found for 2 m, while the blue-green bloom occurred slightly earlier (Figure 18). Changing the fraction of solar radiation absorbed in the top layer resulted in the three blooms being delayed from 5 to 15 days (Figure 19).

Increasing the self-shading coefficient from 0.3 to 0.6 had little effect on the thermal stratification (Figure 20). The oxygen demand at the higher self-shading value was less and resulted in greater concentrations of DO throughout all depths during periods of stratification (Figure 21). The phytoplankton concentrations were all reduced with the higher self-shading coefficient (Figure 22). The phasing was slightly later for the three blooms.

Withdrawal

It was previously shown that release water was taken from throughout the pool with bottom withdrawal. Due to the weak stratification and withdrawal of release waters from throughout the pool, there was little difference between the release temperatures from bottom and surface withdrawal (Figure 23). Neither withdrawal scheme was capable of meeting the proposed Wisconsin Department of Natural Resources (DNR) target temperatures. The temperature profiles for the two withdrawal schemes were also similar (Figure 24).

Since surface withdrawal removed more water from the upper layers, organic matter remained in the lower layers for longer periods of time. DO was, therefore, lower in the deeper strata (Figure 25). On day 150, the deeper strata were nearly anoxic. Once the impoundment mixed completely, DO concentrations returned to the base condition.

The three algal blooms were similar in timing and magnitude for both withdrawal schemes (Figure 26). There were no differences between the two blue-green algae blooms since the impoundment was isothermal and nearly isotropic with respect to most water-quality variables.

Flood events

Two flood events were simulated to investigate their effects on water quality. The first simulation was of a 25-year annual flood event that occurred on 10 April (day 100). The second simulation was a two-year annual flood event; this storm event was simulated to begin on 10 July (day 191). The probability of this event occurring on 10 July is actually less than two years since the frequency was based on all flood events. A more appropriate frequency would be based only on flood events occurring during the summer months. For the months of June, July, and August, the monthly frequencies of the two-year annual event are 16, 40, and 62 years, respectively.

The 25-year spring flood event had minimal impact on the thermal stratification pattern simulated in the reservoir since it occurred during the isothermal or weakly stratified period of the year (Figure 27). The temperatures predicted on day 120 were slightly cooler than the base case. Since the impoundment was completely mixed on day 135, no differences were observed through the remainder of the season. The summer stratification and release temperatures were not significantly affected by the summer storm event because the impoundment was also isothermal at the time of the flood (Figure 28).

Both floods had minimal effect on DO (Figures 29 and 30). For the summer flood, a slight temperature stratification on days 210, 225, and 255 resulted in lower DO concentrations in the deeper strata.

The spring flood delayed the green-diatom bloom by a few days (Figure 31). The two blue-green algae blooms remained unchanged. The summer flood prolonged the first blue-green algae bloom by about eight days and delayed the second blue-green bloom by about ten days (Figure 32).

Discussion

It is important to realize that mathematical ecosystem models are in the early stages of development and verification. The original documentation of the WRE model was published by Chen and Orlob in 1972. During the succeeding four years, there have been significant improvements made to the model, but neither the WQRRS model nor any other mathematical water-quality model is capable of predicting so-called "absolute values." Preliminary evaluations indicate that predictive values for some parameters at times may be in error by more than an order of magnitude of measured values or by two weeks of the onset of an event. However, if the assumptions and limitations of the model are understood and incorporated in the interpretation of the output, the general response predicted by the model will provide valuable insight into the water quality of the proposed LaFarge Lake. Despite their limitations, existing modeling techniques properly used in conjunction with other appropriate study approaches may be considered as one of the more reliable and versatile tools for making water-quality analyses.

The one-dimensional assumption of the WQRRS is important in the interpretation of modeling results. The predicted water-quality conditions are considered most applicable for the deepest part of the pool near the dam. This area has the greatest volume, the strongest stratification profile, and the most significant impact on downstream conditions. The one-dimensional assumption, however, implies that the model cannot predict isolated water-quality conditions in the headwaters, coves, or embayment areas. These shallow, usually well-mixed areas cannot be adequately represented by considering the main pool of the impoundment.

The simulations also are considered representative of conditions three to four years after filling of the reservoir when it is in a more stable state. Transient conditions are known to occur during the first several years after reservoir filling. These transient conditions were not considered in the simulations made as part of this study. With an understanding of these basic assumptions, various predicted water-quality conditions can be discussed.

The predicted release temperatures and thermal stratification profiles are probably representative of actual values that will occur in LaFarge Lake if the project is constructed. The thermal predictions are based on physical processes that are reasonably well understood. Since the impoundment is predicted to be only weakly stratified, the thermal structure will be highly dependent on local meteorological conditions and flood events. Variations in the number, duration, and strength of the stratification cycles occurring during a year may be expected to have significant effects on water quality. The weak stratification would also make it impossible to meet downstream temperature release objectives. In contrast, at the higher elevation, the summer stratification was predicted to be stable and the downstream release objectives could be met (Figure 33).

The seasonal DO concentrations predicted for the proposed LaFarge

Lake are dependent on the thermal structure and on the phasing and magnitude of algal blooms. In 1974, no prolonged period of stratification was predicted. It is quite possible, however, that a prolonged period of stratification could occur if the inflows were small and the weather was warm and calm. Under these conditions it is probable that the DO would go to zero in the deeper strata of the impoundment. Bottom withdrawal, in comparison with surface withdrawal, would be more effective in keeping DO concentrations high in the deeper strata.

As indicated by the sensitivity analyses in the previous study, a number of factors would influence the growth of phytoplankton. Half-saturation coefficients were selected to simulate phosphorus limitation in the phytoplankton population. The Secchi disk for the base case was changed from 3 m in the previous study to 2 m to simulate the increased turbidity expected with the smaller volume of the impoundment at elevation 250.5. The larger flushing rates at the lower elevation would result in more turbulence, which would suspend more matter. This study indicated, as did the previous study, that there will probably be a bloom of diatoms and greens during the spring with at least two major blooms of blue-green algae during the summer and fall. However, the magnitude of the second blue-green blooms would be much greater at elevation 250.5 (Figure 6). In the spring and the fall of the year, the pool at both elevations would be isothermal. Any inflowing nutrients would be distributed throughout the pool. Despite the larger flushing rates at the lower pool elevation, in situ nutrient concentrations would be larger resulting in larger blooms.

Halving the inflowing nutrient concentrations had the greatest effect of all the sensitivity analyses by changing the composition of the second and third blooms from blue-green to green-diatom algae. This indicates that a reduction in inflowing nutrient concentrations could possibly have beneficial effects. Doubling the inflowing nutrient concentrations had the greatest effect on the first blue-green algae bloom. Overall, the algae were more sensitive to inflowing nutrient concentrations at the lower pool elevation. At the 256 elevation, the larger volume of the impoundment would have a greater capacity to dilute inflowing nutrients during isothermal periods; during periods of stratification, the inflowing nutrients could flow directly into the hypolimnion out of the photic zone.

At elevation 256, light penetration would have minimal effect on the phytoplankton since nutrient concentrations were generally limiting (Thornton et al., 1976). This might not be true at elevation 250.5. Increased turbidity resulting from a smaller total volume and larger algal blooms could possibly result in light limitation. Secchi disk readings less than 1 m are also probable at this lower elevation.

The smaller volume at the lower elevation would result in coliforms being less diluted, while extended isothermal periods would result in greater distribution throughout the pool. The coliforms would also persist longer since it would take longer for the higher concentrations to decrease to acceptable limits.

Total phosphorus loadings to the proposed LaFarge Lake would be doubled at the new pool elevation since the surface area would be approximately one-half of the previous surface area. The total phosphorus loading was predicted to vary from 4.8 to 19.8 $g/m^2/yr$ at the new elevation, but the overall position of LaFarge Lake on the loading curve increased only slightly. The hydraulic residence time decreased from about 100 days at elevation 256 to about 30 days at elevation 250.5. The increased flushing rate would moderate the increased loadings. As stated in the previous study, it is difficult to assess the degree of eutrophication based on Vollenweider's criteria at these higher phosphorus loadings. However, eutrophication potential would not be improved at the lower pool elevation.

PART IV: CONCLUSIONS

Conclusions concerning impoundment water quality based upon this study are as follows:

- _a_. The proposed LaFarge Lake at elevation 250.5 would be weakly or intermittently stratified during the summer months. No permanent stratification typical of dimictic lakes is expected.
- _b_. Downstream target temperatures proposed by DNR could not be met at the lower pool elevation. During the summer months of July, August, and September, downstream release waters are expected to be warmer than natural stream water.
- _c_. Dissolved oxygen would probably be distributed throughout the reservoir during isothermal periods. Prolonged periods of stratification could result in the depletion of oxygen in the lower layers.
- _d_. There would probably be a bloom of diatoms and green algae during the spring with at least two major blue-green algae blooms during the summer and fall. The magnitude of the second blue-green bloom would be greater at the lower pool elevation.
- _e_. Nutrient loadings would be approximately twice as great at the lower pool elevation. Although the flushing rate would be increased by about three times, the eutrophication potential is still expected to be at least equal to the eutrophication potential at the higher pool elevation.
- _f_. If constructed at elevation 250.5, the proposed LaFarge Lake will probably be a eutrophic impoundment with lower clarity than at the higher pool elevation.
- _g_. Coliforms would be expected to persist longer and be distributed through more of the pool at the lower elevation.

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Thornton, K. W., Ford, D. E., and Robey, D. L. 1976. Preliminary evaluation of water quality of proposed LaFarge Lake, Kickapoo River, Vernon County, Wisconsin. WES MP Y-76-5.

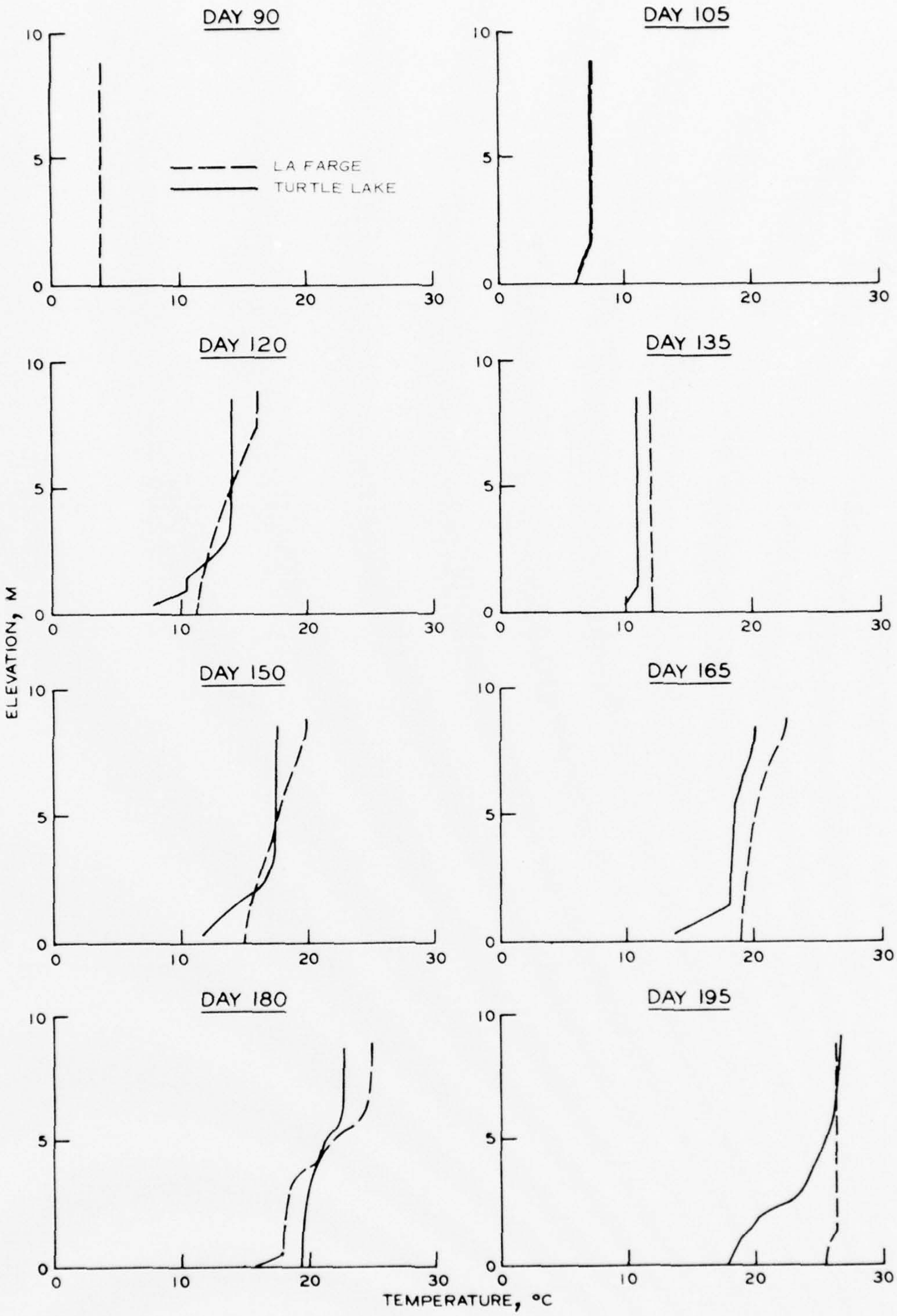


Figure 1. Comparison of temperature profiles at elevation 250.5 with Turtle Lake, a natural lake of comparable size (sheet 1 of 2)

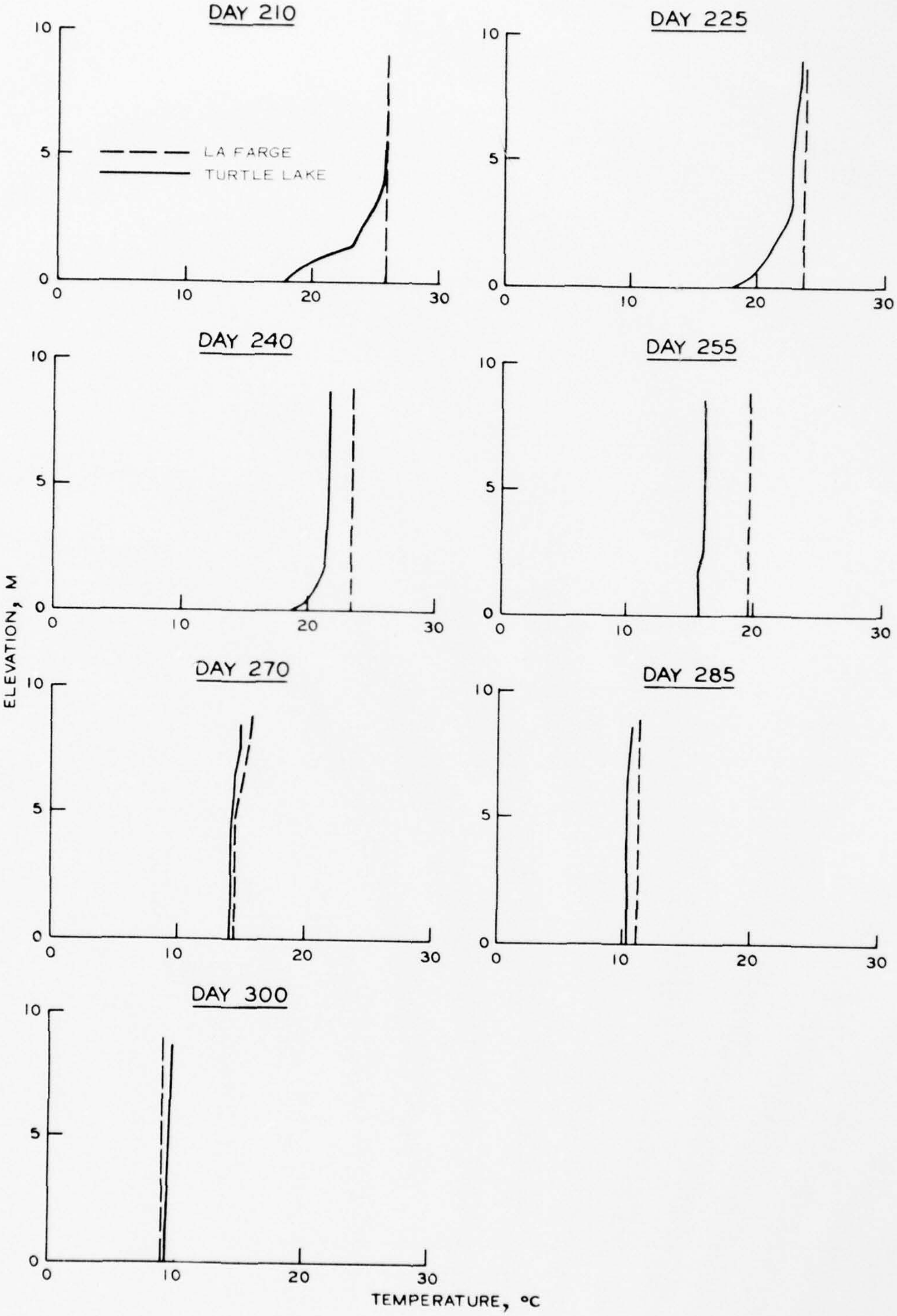


Figure 1 (sheet 2 of 2)

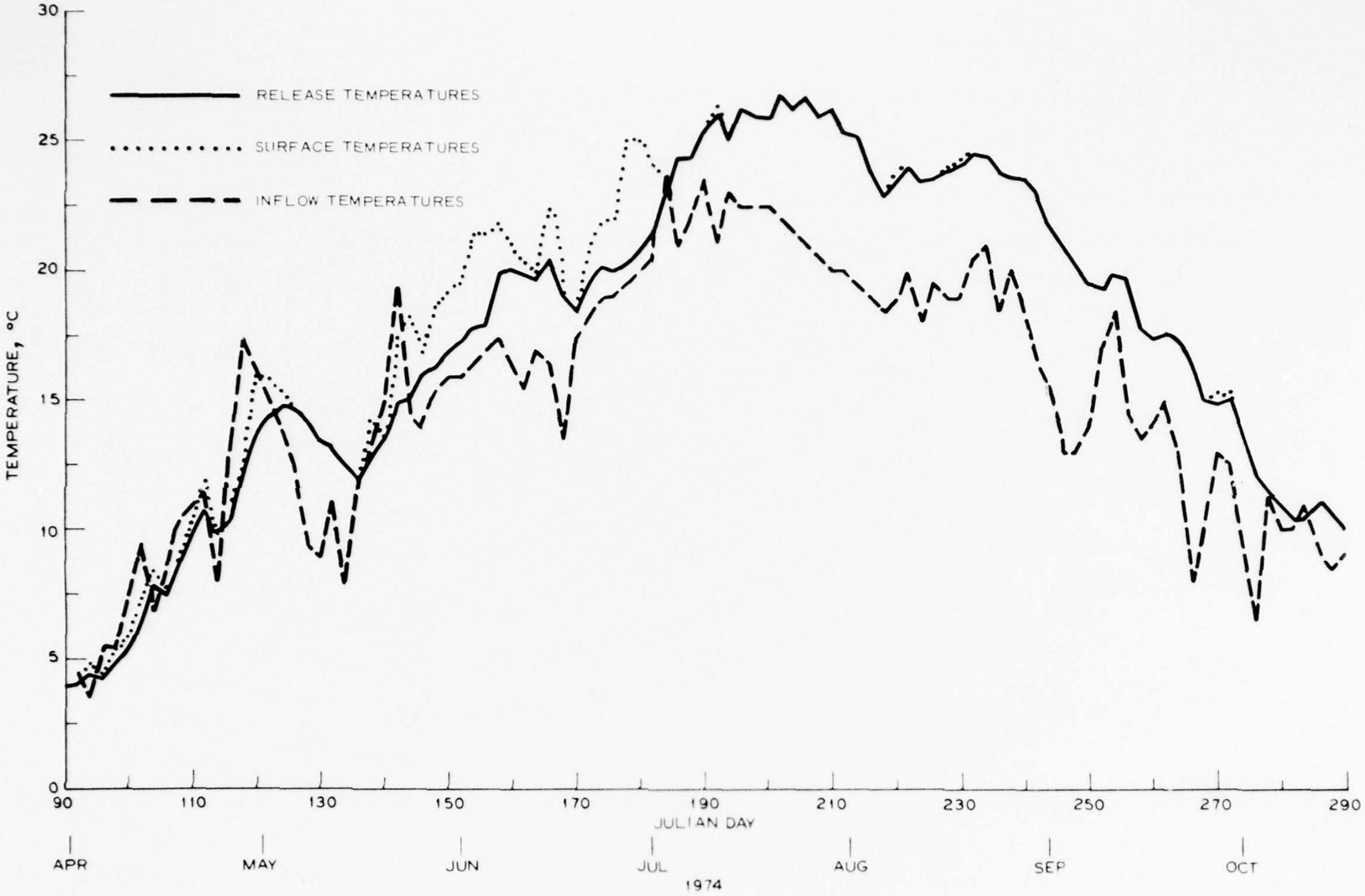


Figure 2. Comparison of observed inflow temperatures with predicted reservoir surface and release temperatures

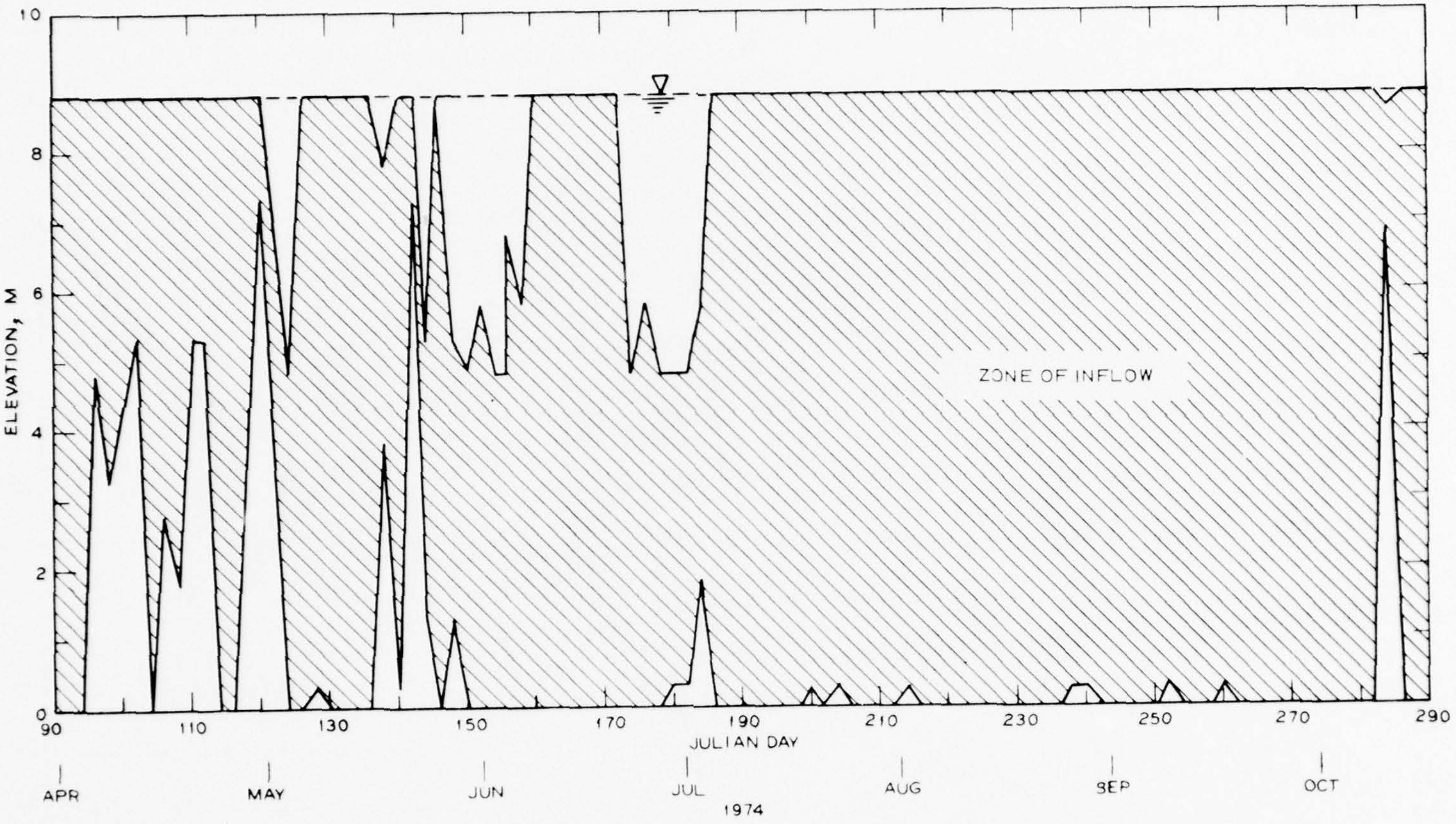


Figure 3. Distribution and placement of inflow throughout the pool

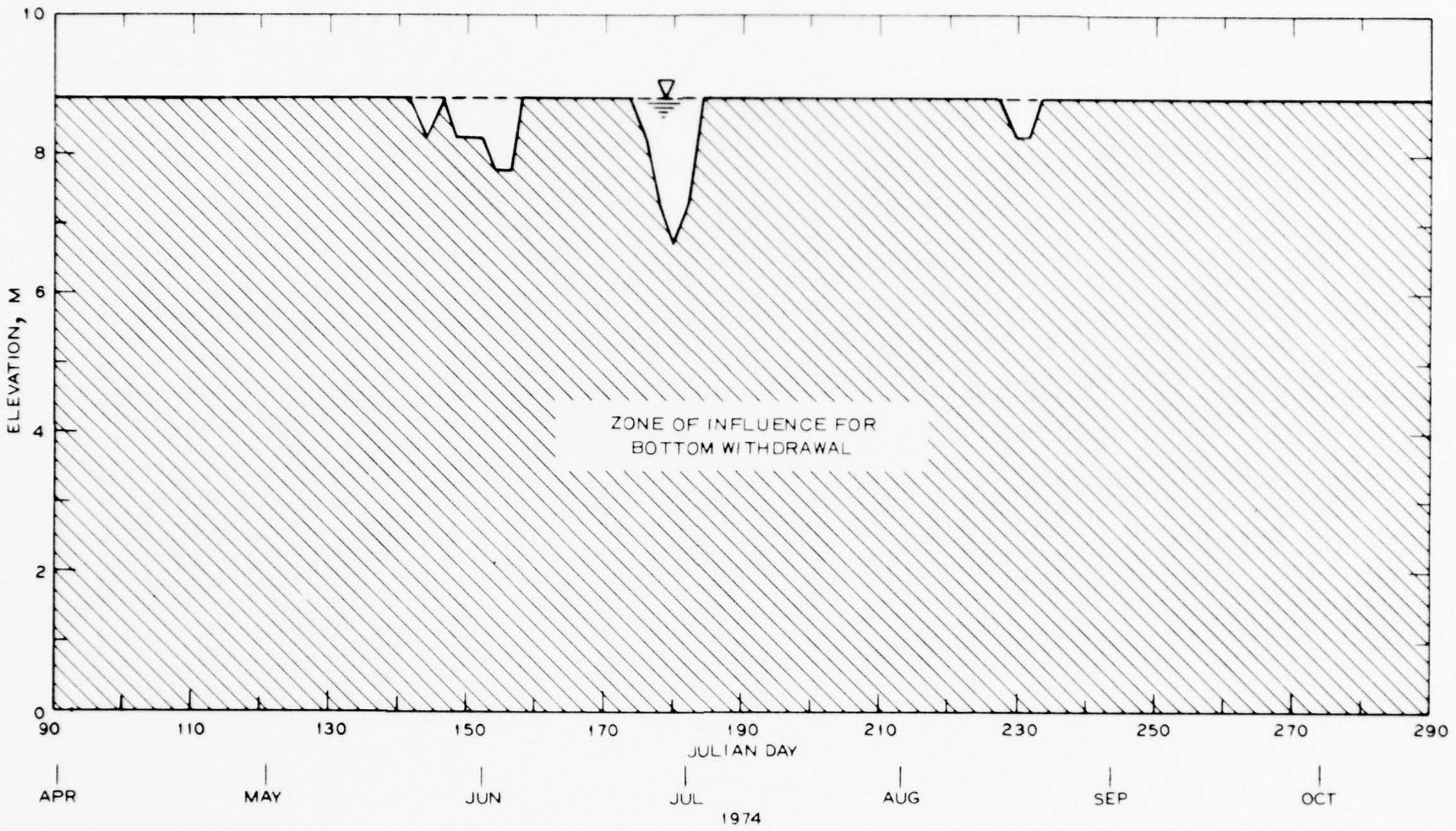


Figure 4. Zone of withdrawal of releases from throughout the pool

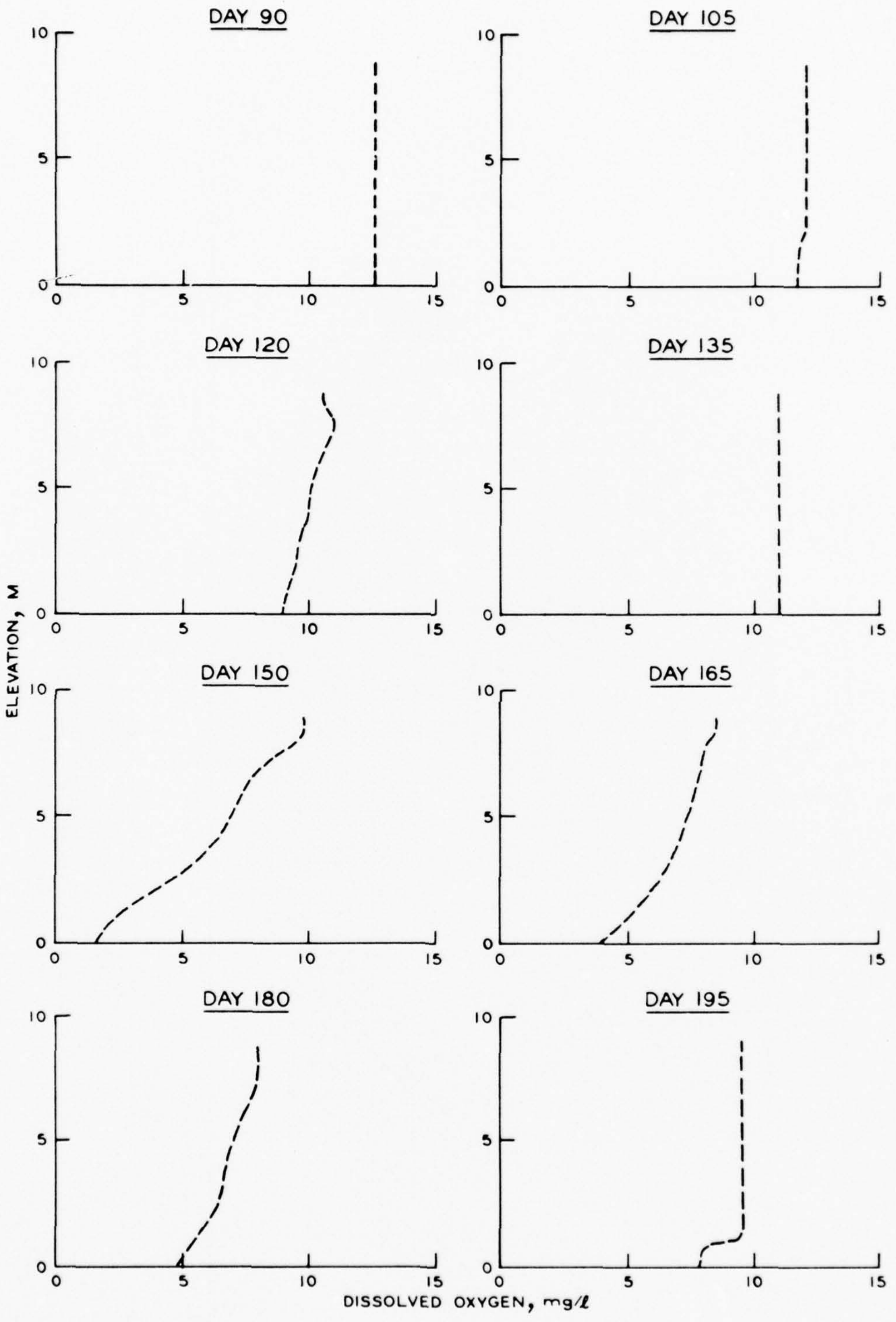


Figure 5. Dissolved oxygen profiles for the base case (sheet 1 of 2)

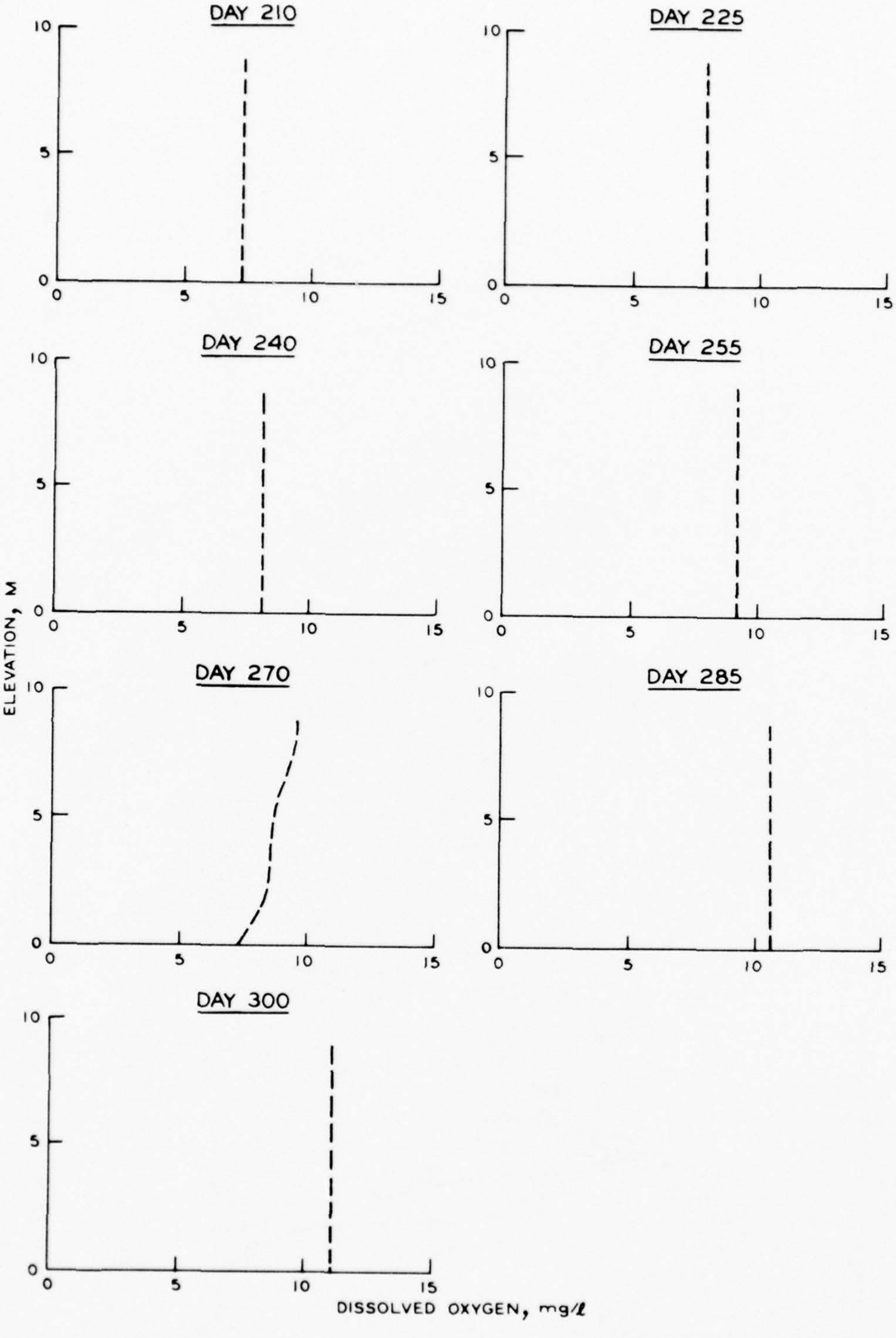


Figure 5 (sheet 2 of 2)

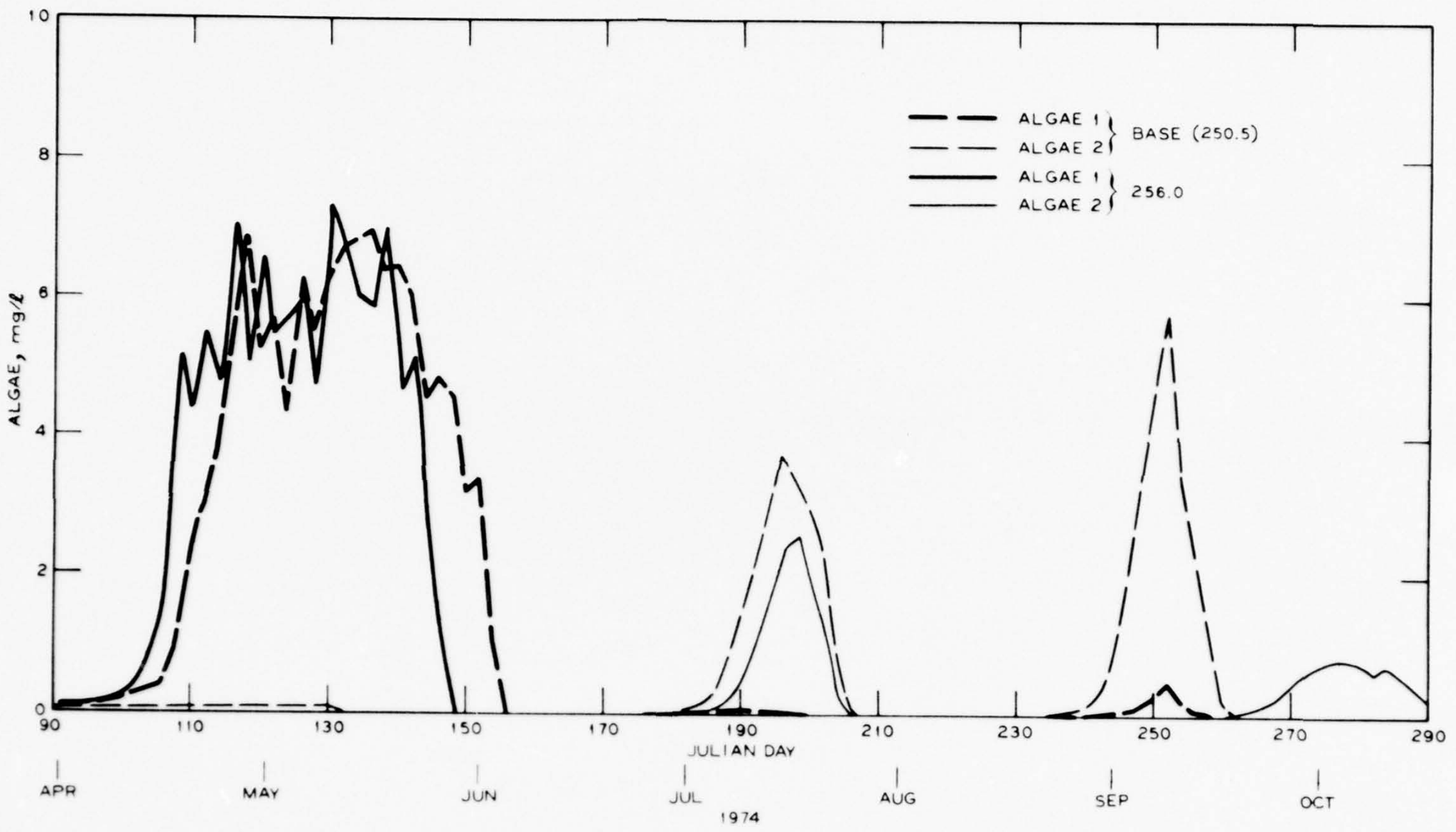


Figure 6. Comparison of surface algae concentrations at elevations 250.5 and 256

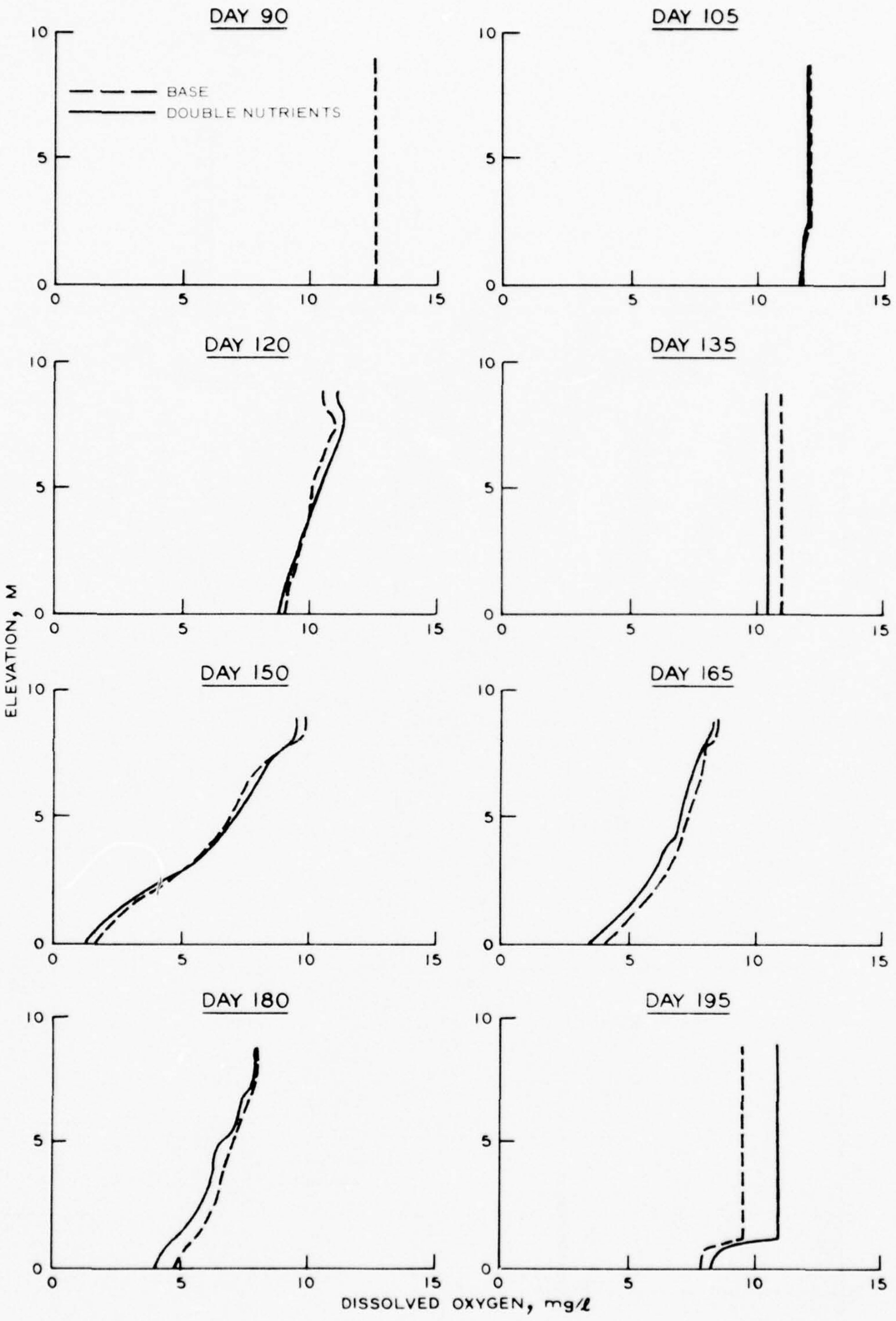


Figure 7. Effect on dissolved oxygen of doubling the inflow nutrient concentrations (sheet 1 of 2)

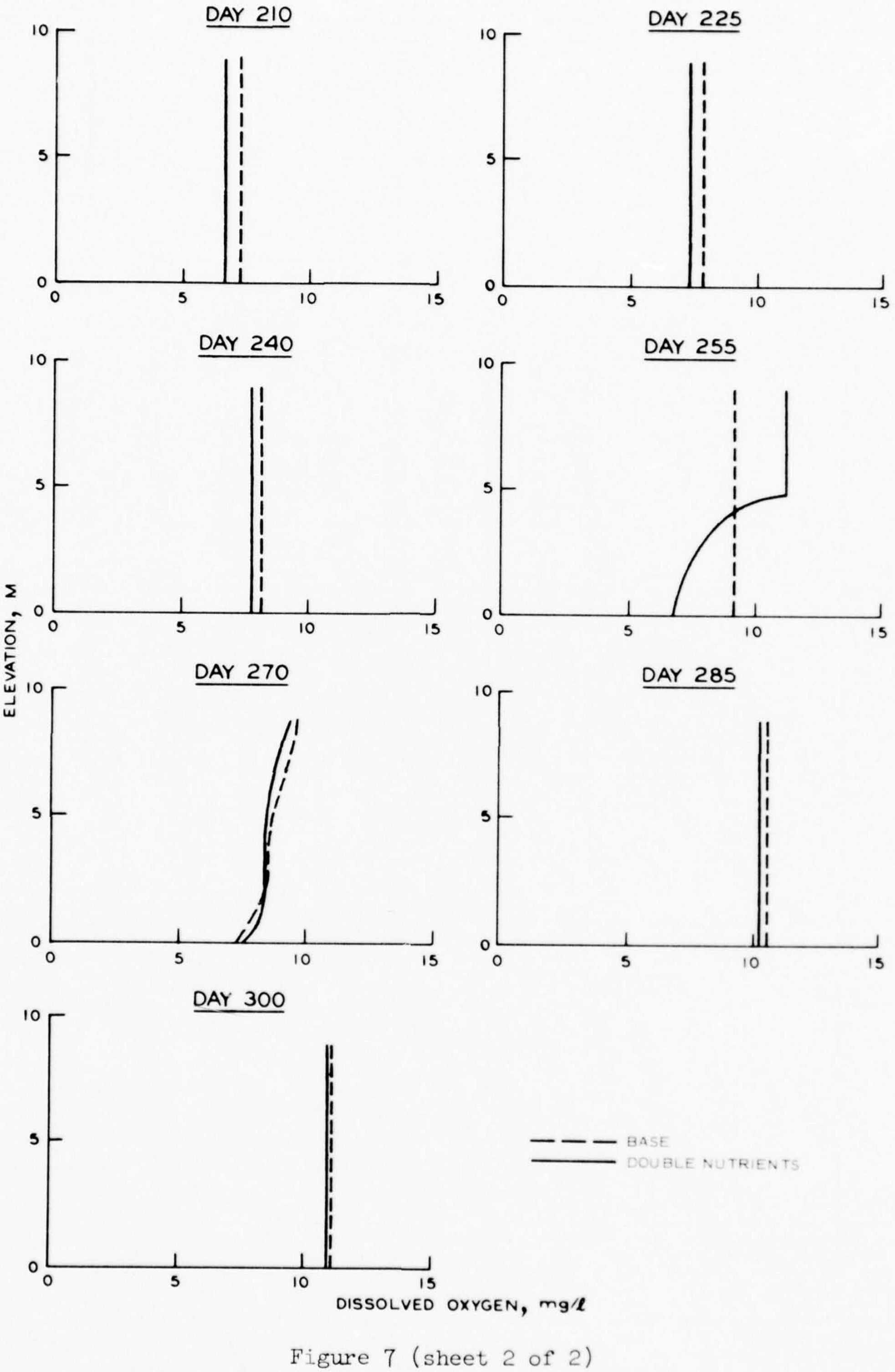


Figure 7 (sheet 2 of 2)

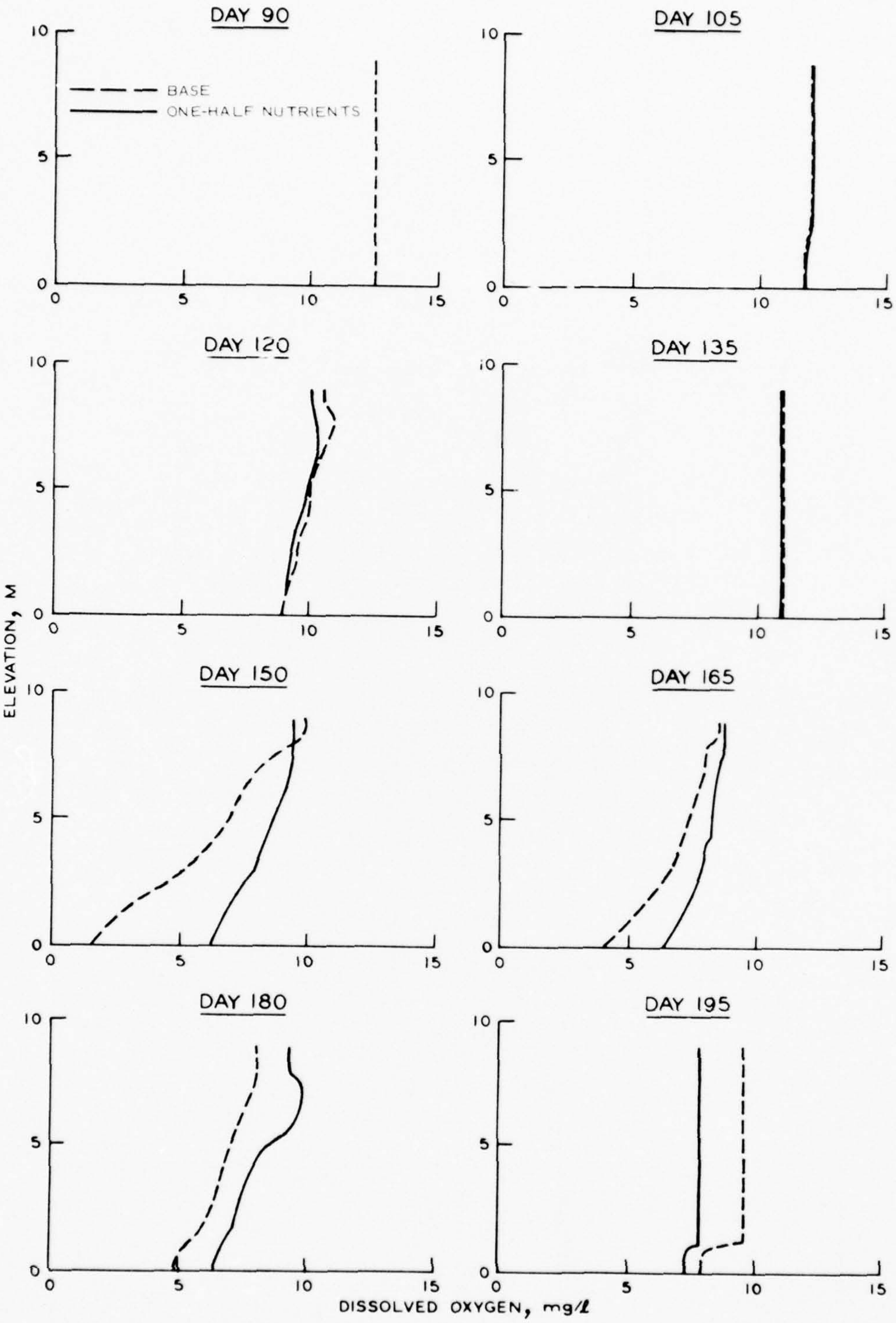
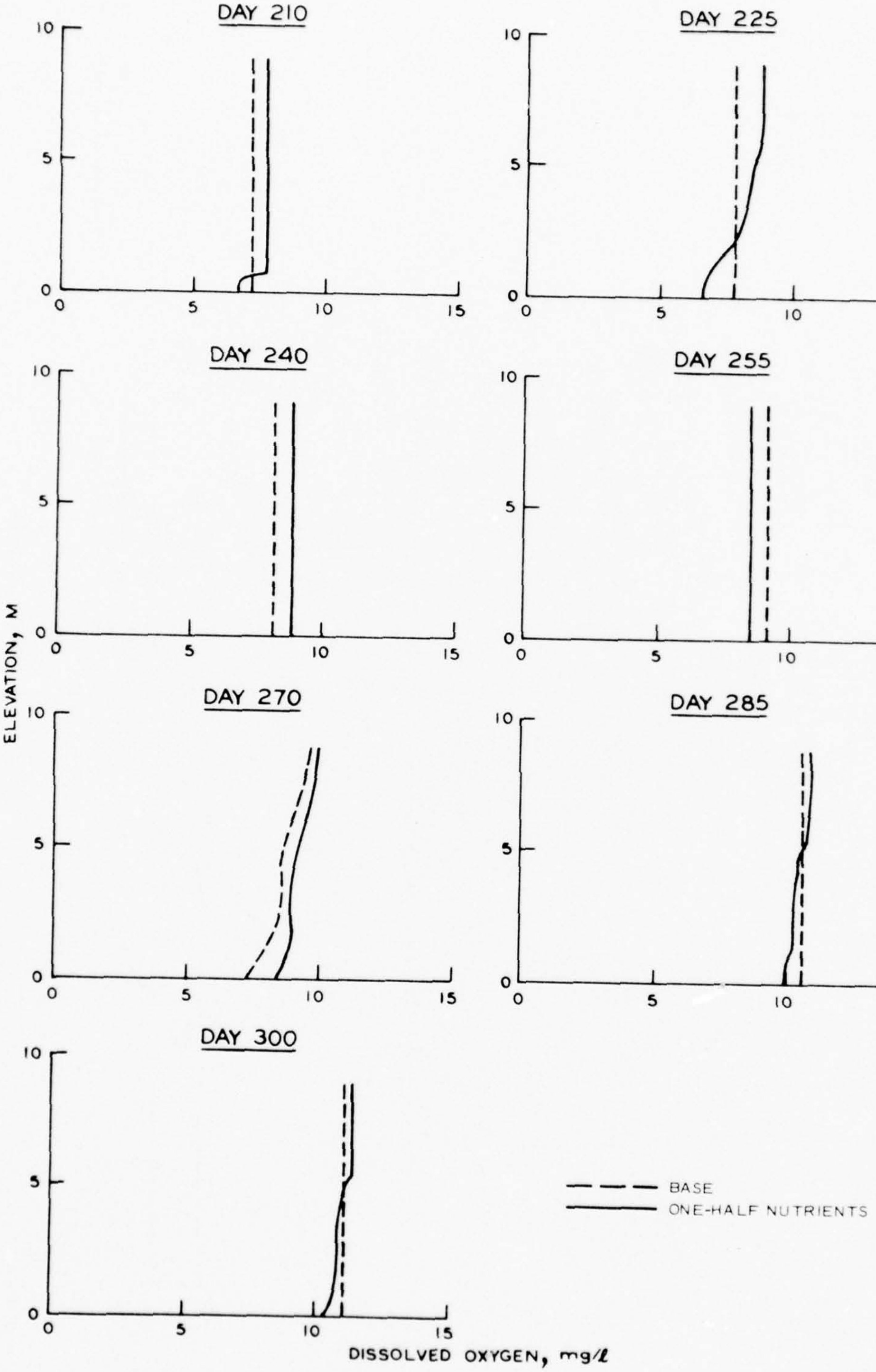


Figure 8. Effect on dissolved oxygen of halving the inflow nutrient concentrations (sheet 1 of 2)



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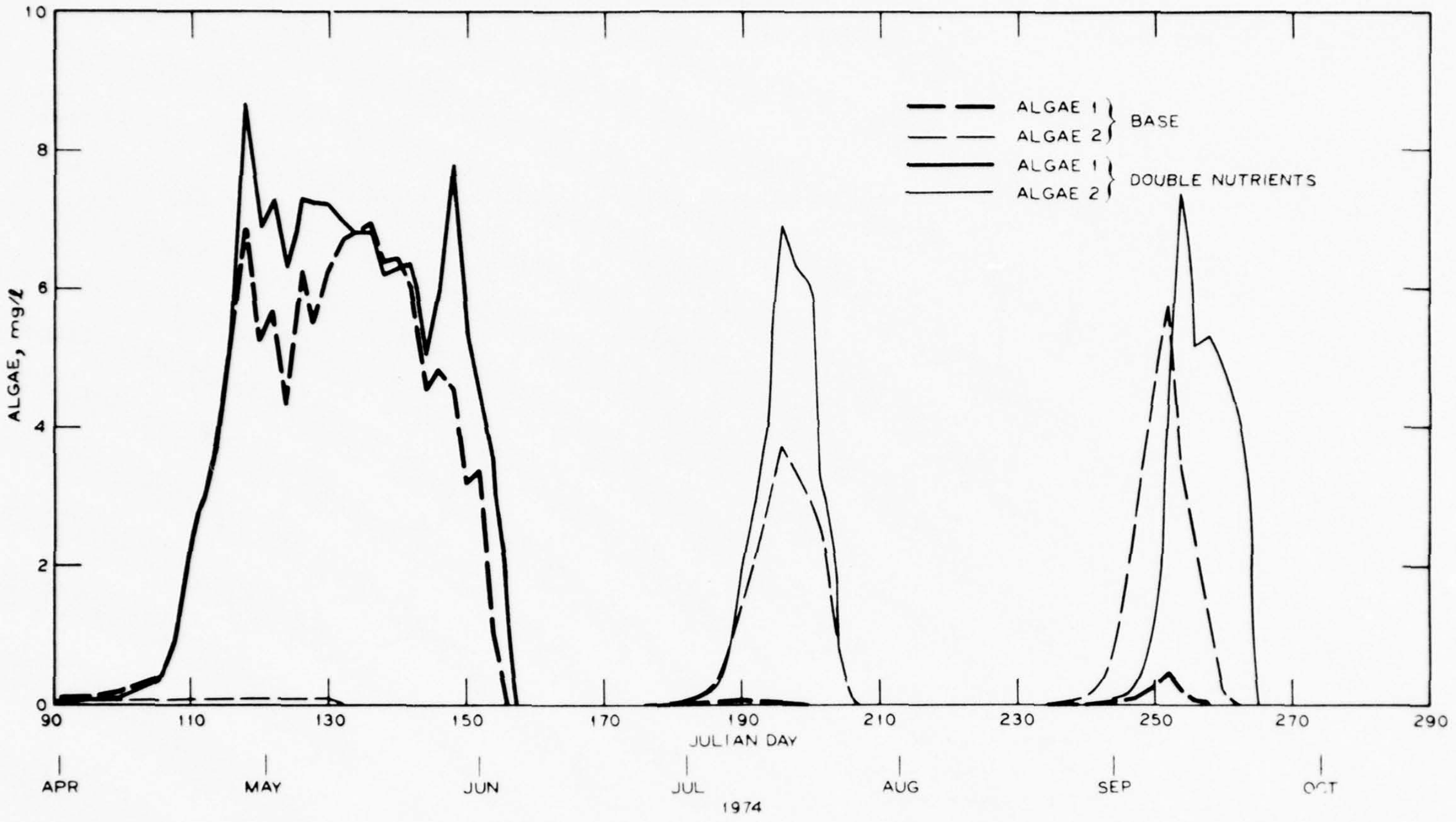


Figure 9. Effect on the surface algae concentrations of doubling the inflow nutrient concentrations

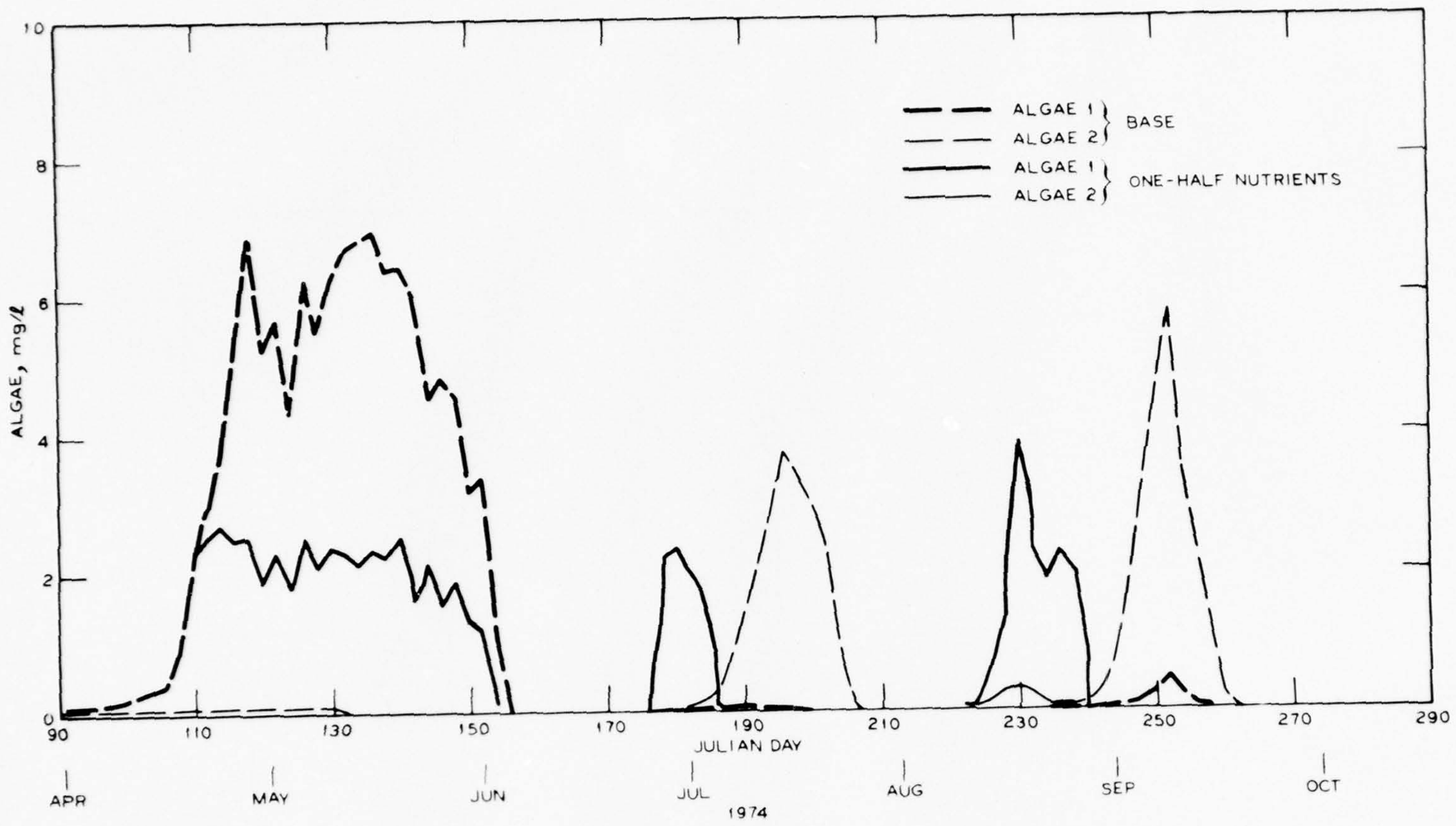


Figure 10. Effect on the surface algae concentrations of halving the inflow nutrient concentrations

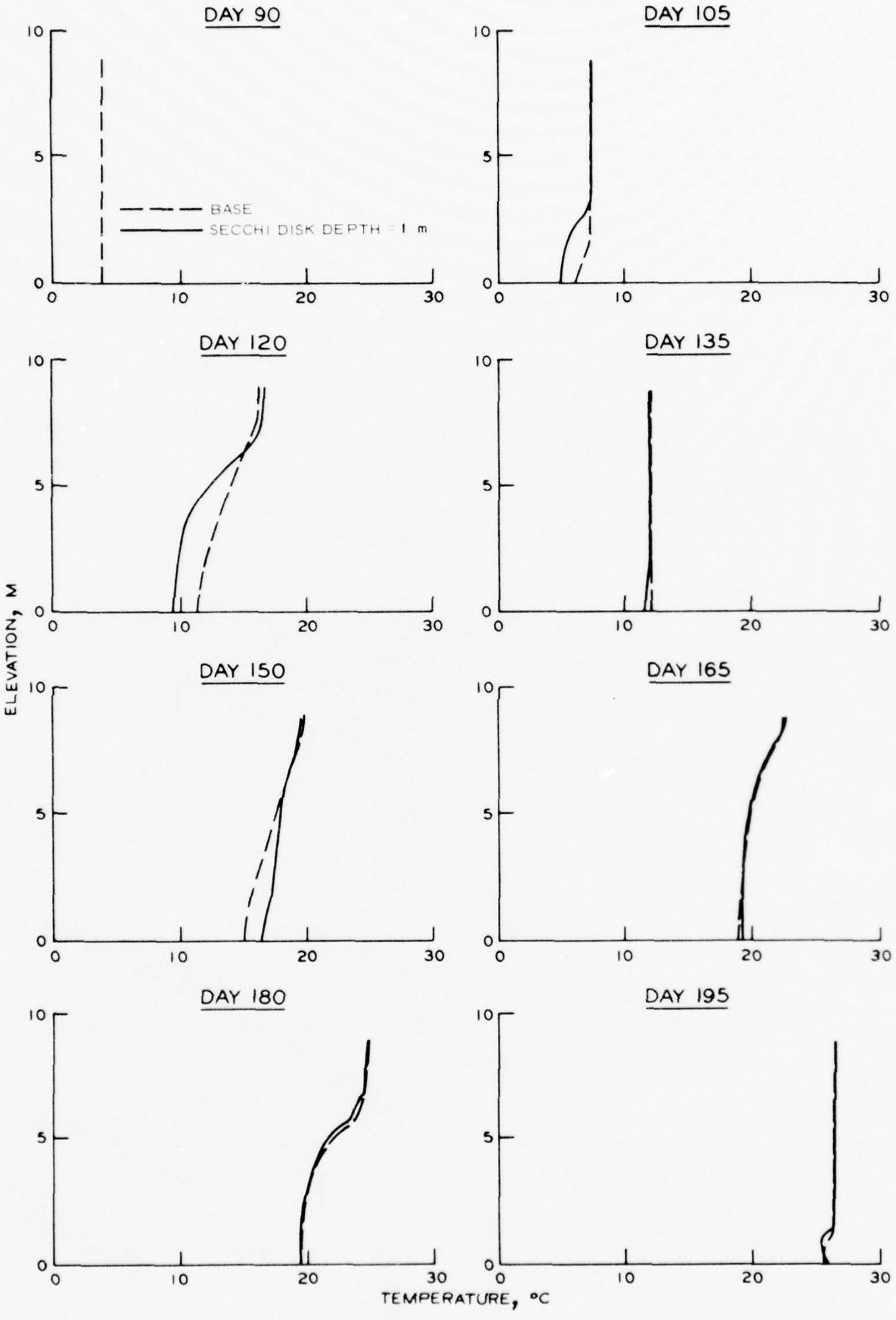


Figure 11. Effect of 1-m light penetration on temperature (sheet 1 of 2)

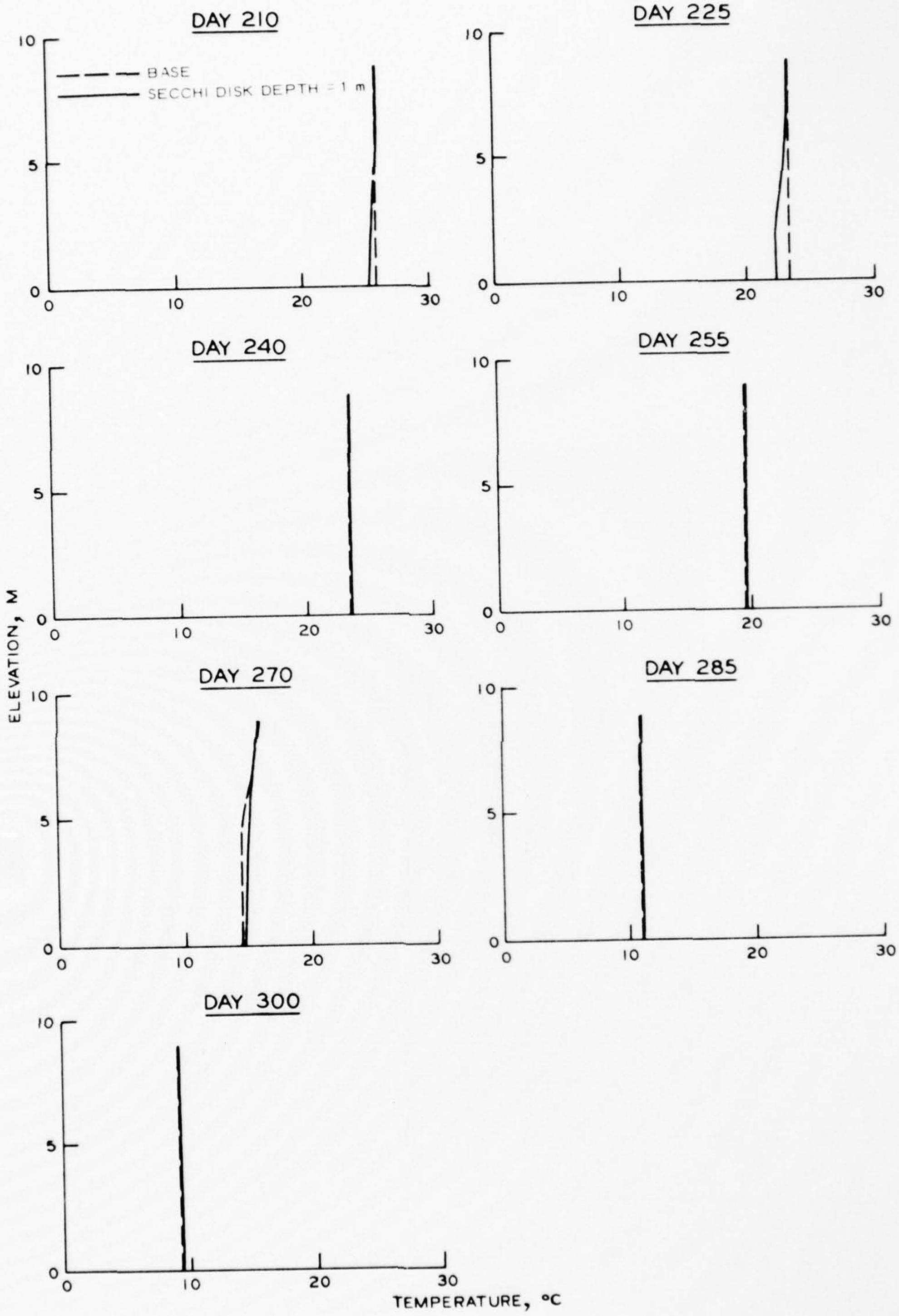


Figure 11 (sheet 2 of 2)

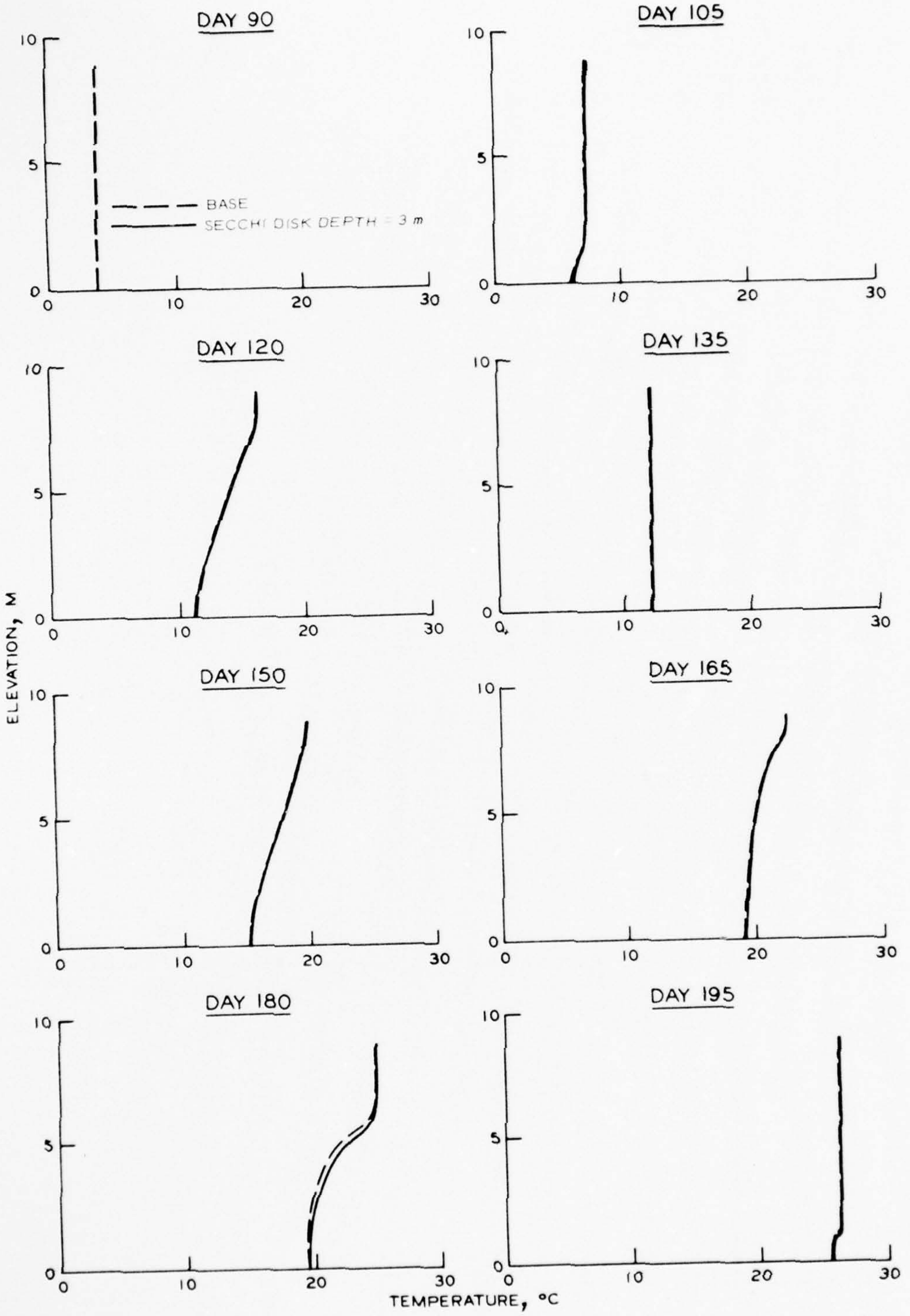


Figure 12. Effect of 3-m light penetration on temperature (sheet 1 of 2)

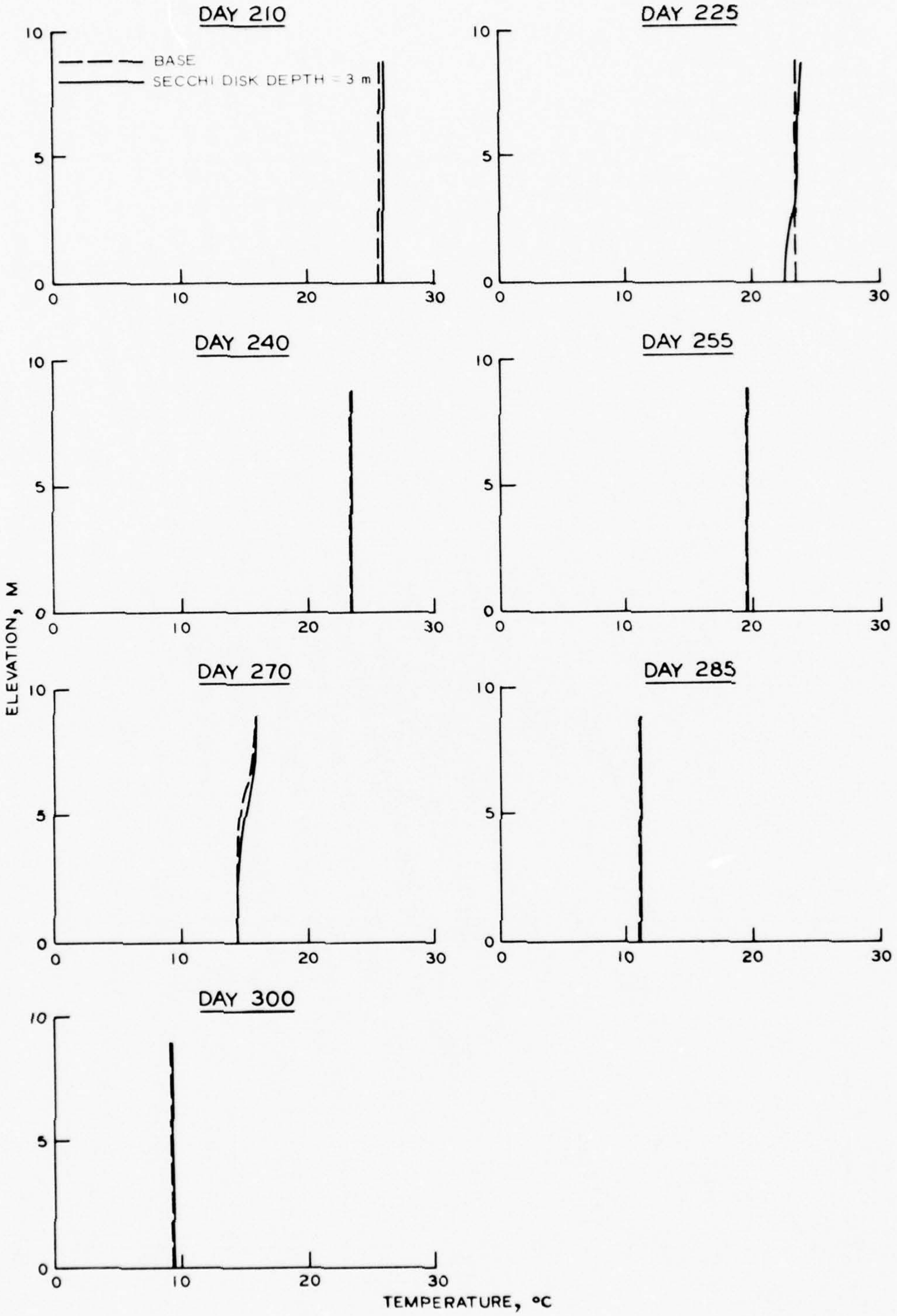


Figure 12 (sheet 2 of 2)

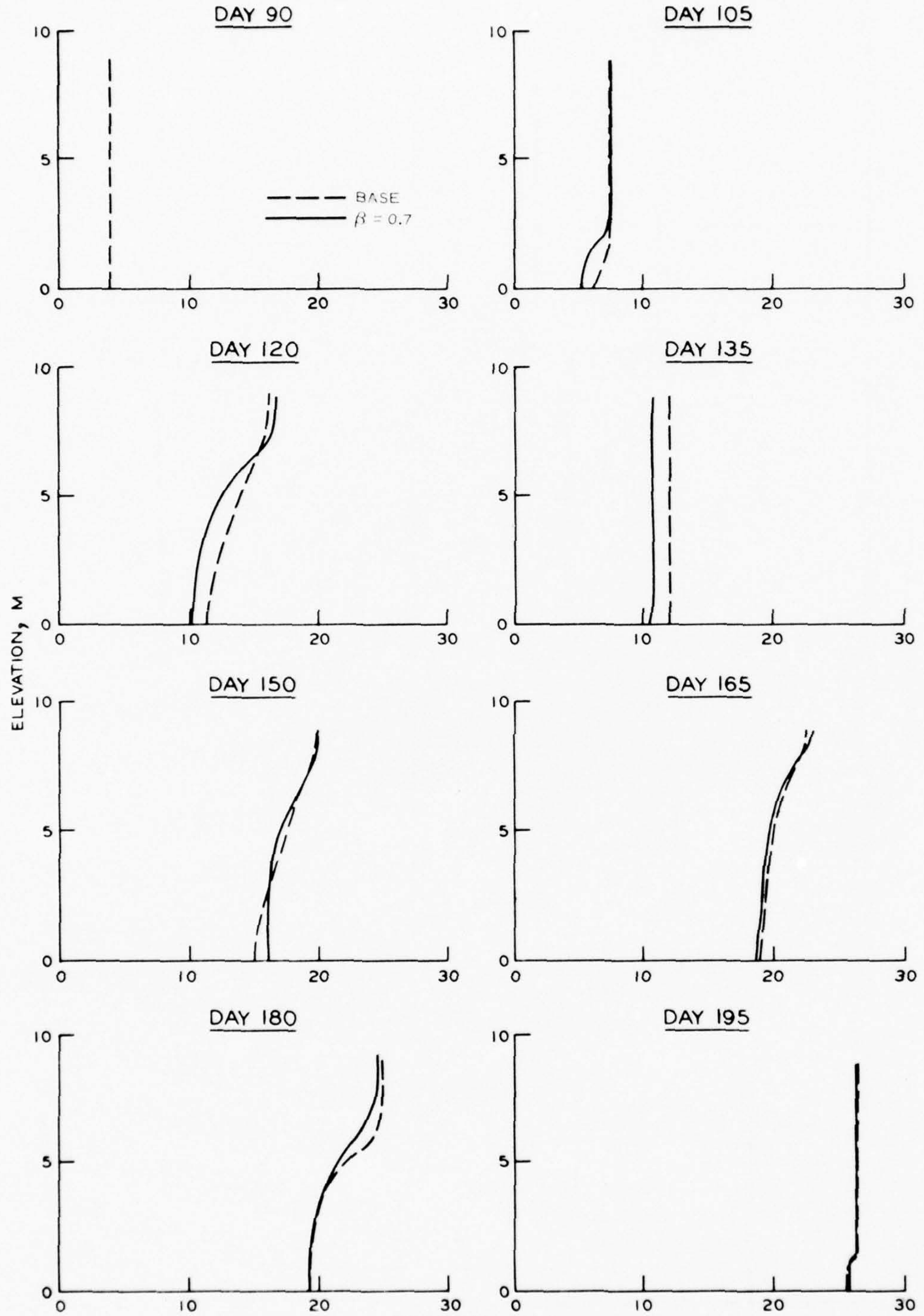


Figure 13. Effect of increased top-layer absorption of solar radiation on temperature (sheet 1 of 2)

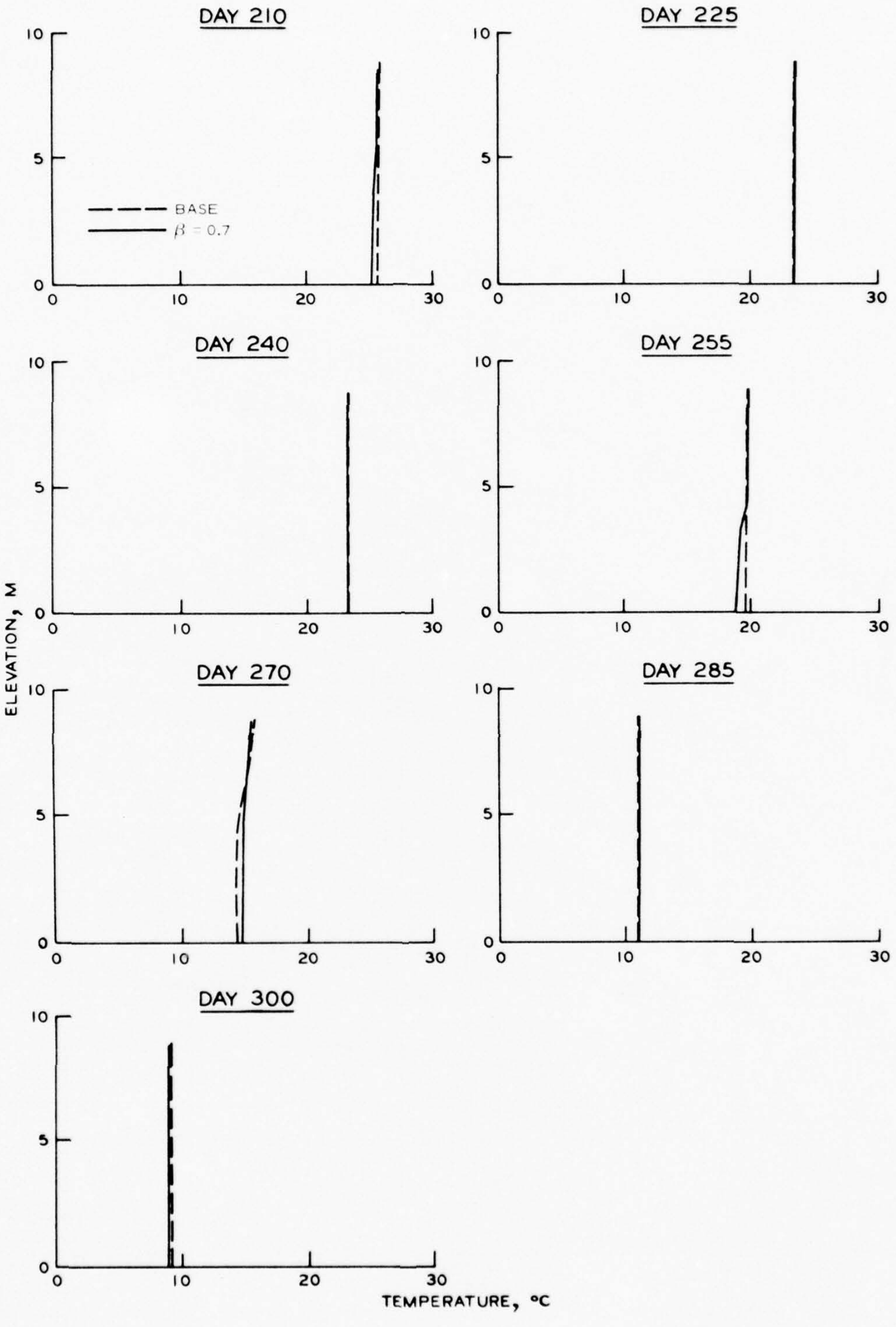


Figure 13 (sheet 2 of 2)

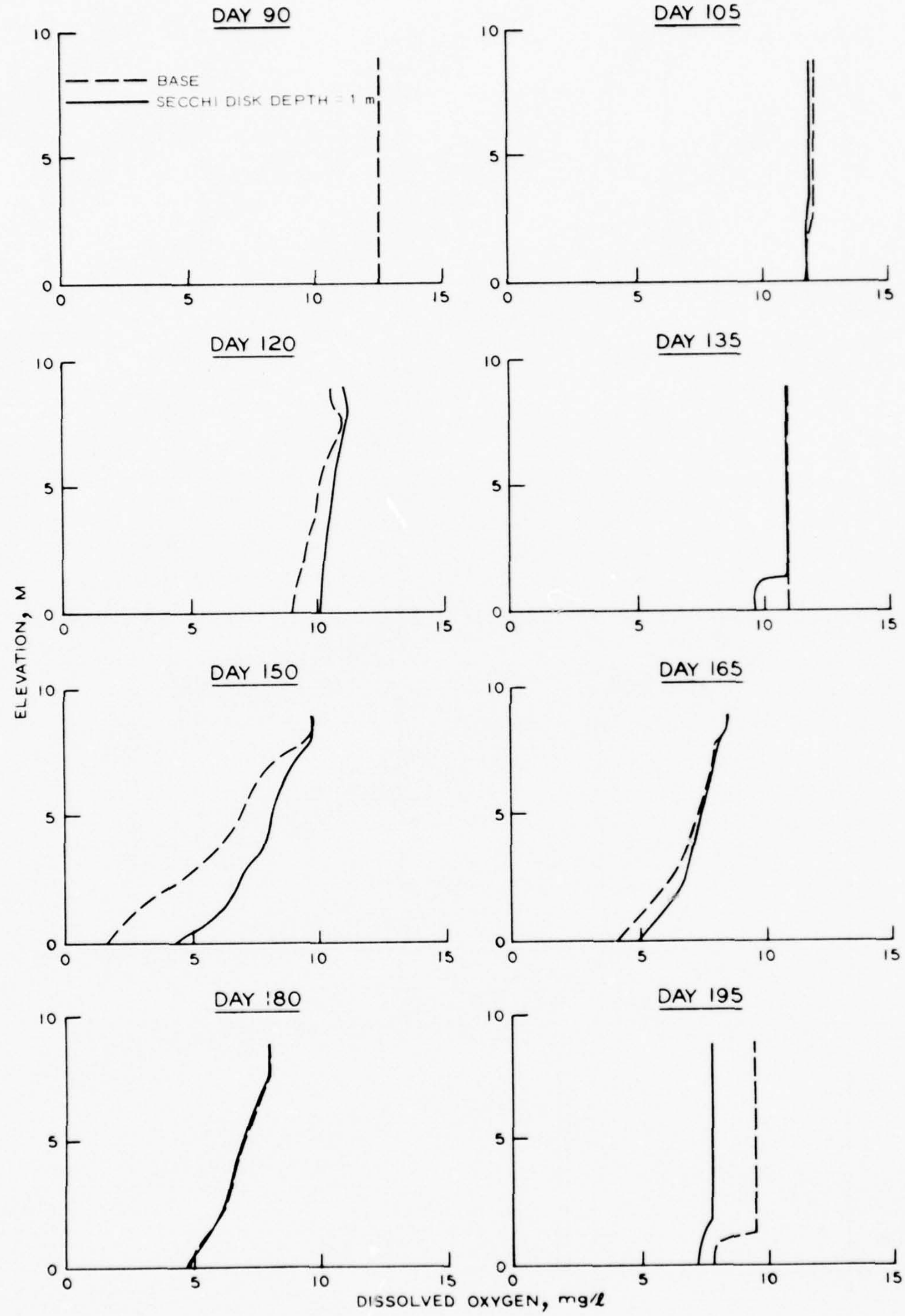


Figure 14. Effect of 1-m light penetration on dissolved oxygen (sheet 1 of 2)

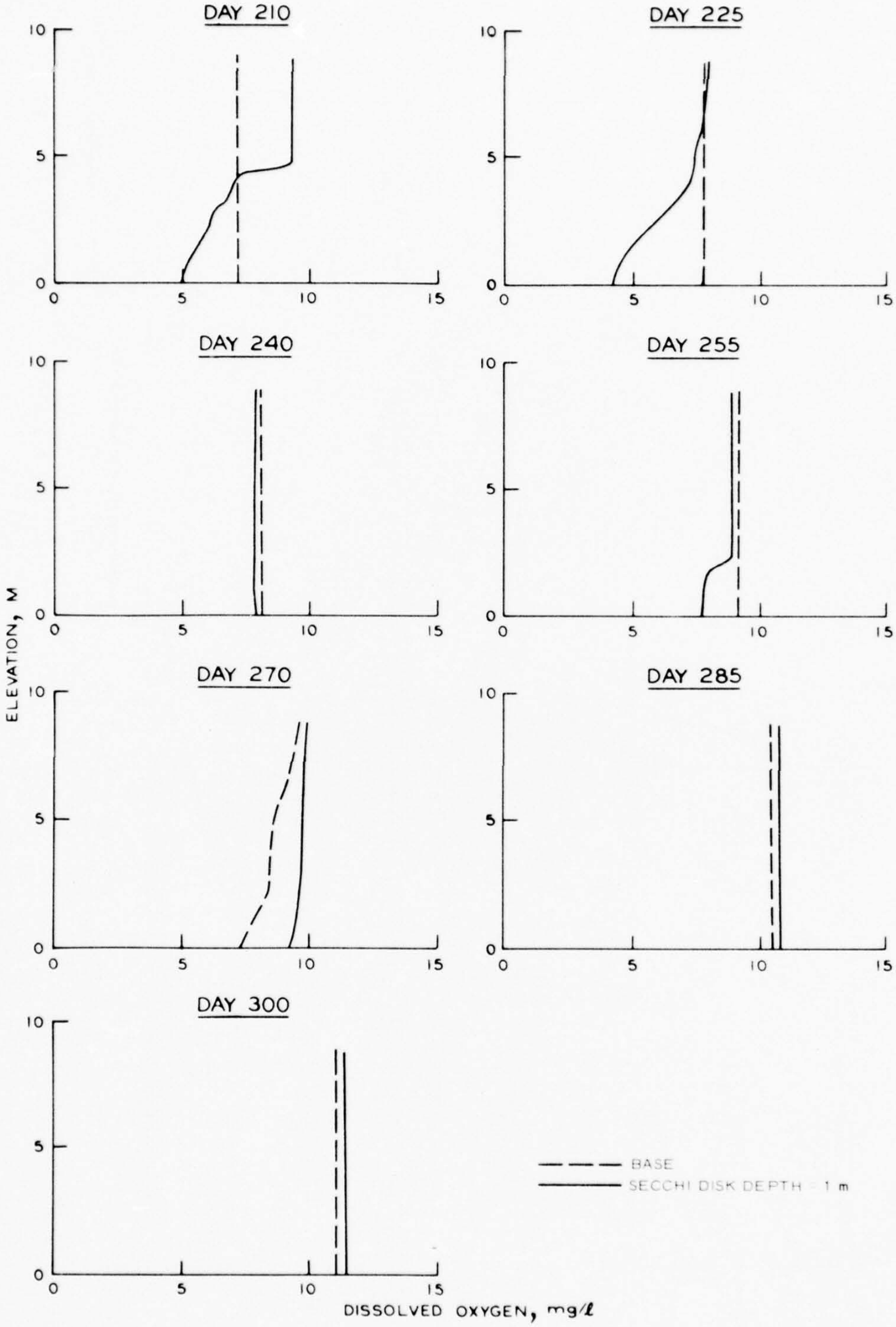


Figure 14 (sheet 2 of 2)

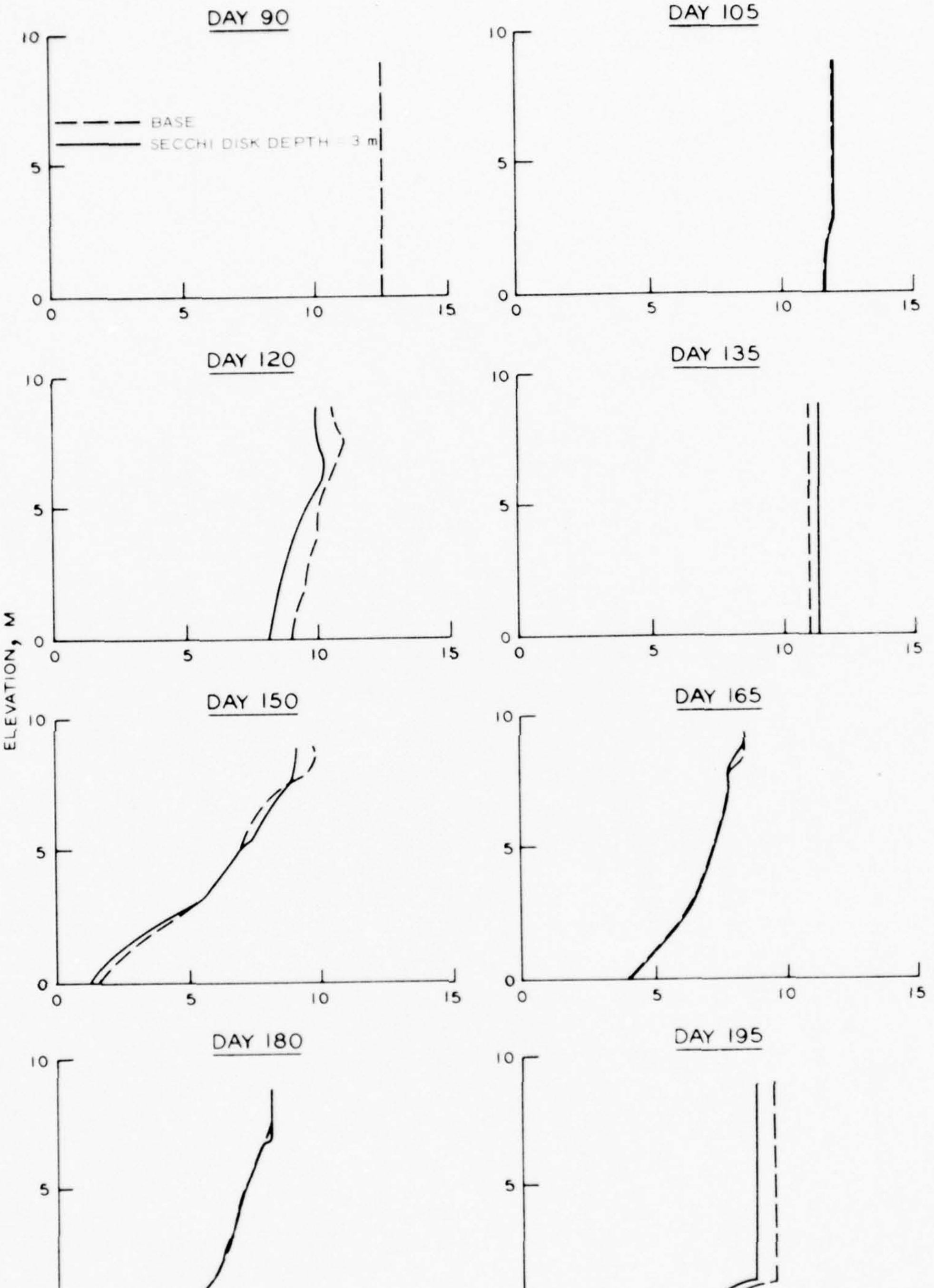
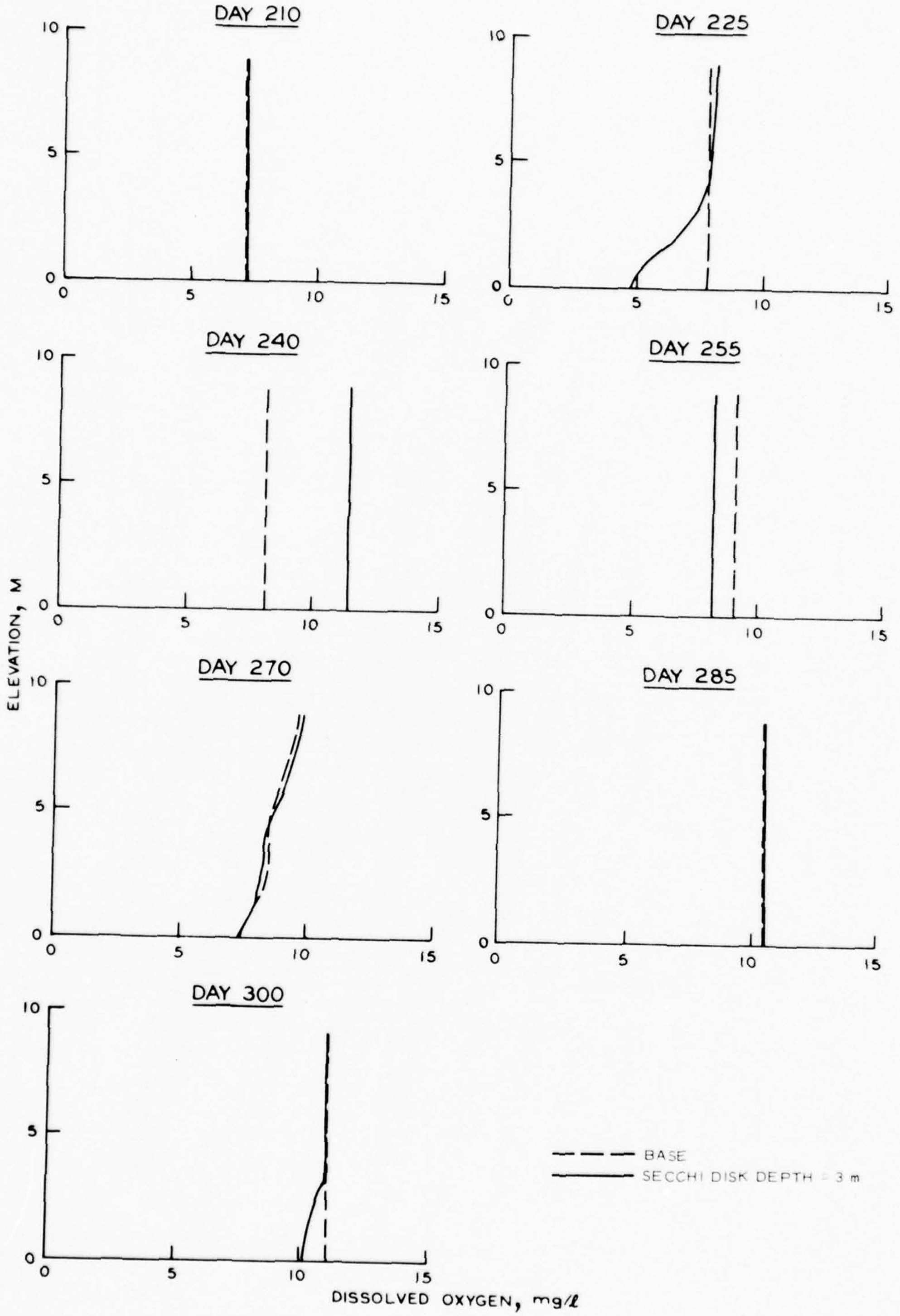


Figure 15. Effect of 3-m light penetration on dissolved oxygen (sheet 1 of 2)


DAY 210

DAY 225

DAY 240

DAY 255

DAY 270

DAY 285

DAY 300

Figure 15 (sheet 2 of 2)

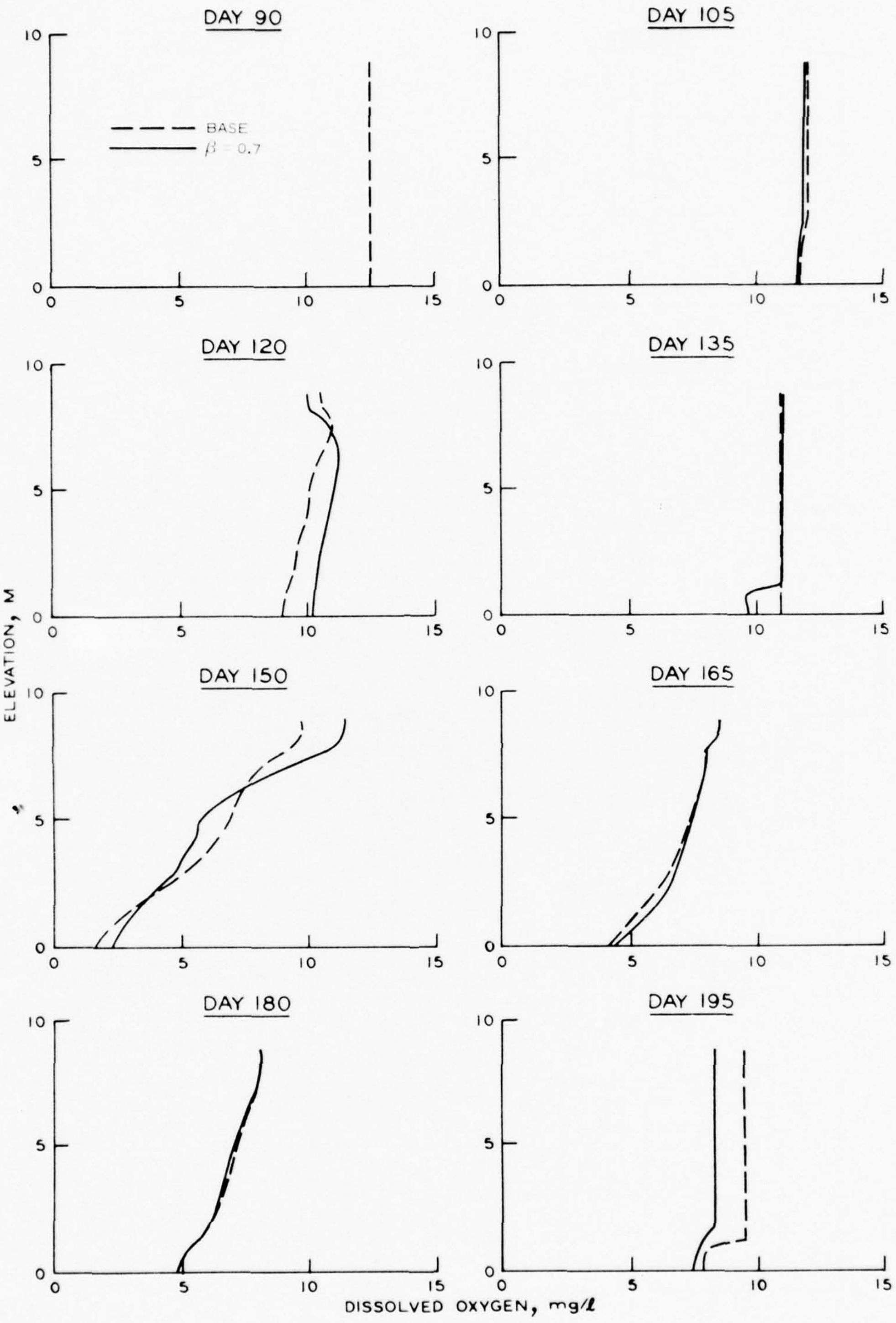


Figure 16. Effect of increased top-layer absorption of solar radiation on dissolved oxygen (sheet 1 of 2)

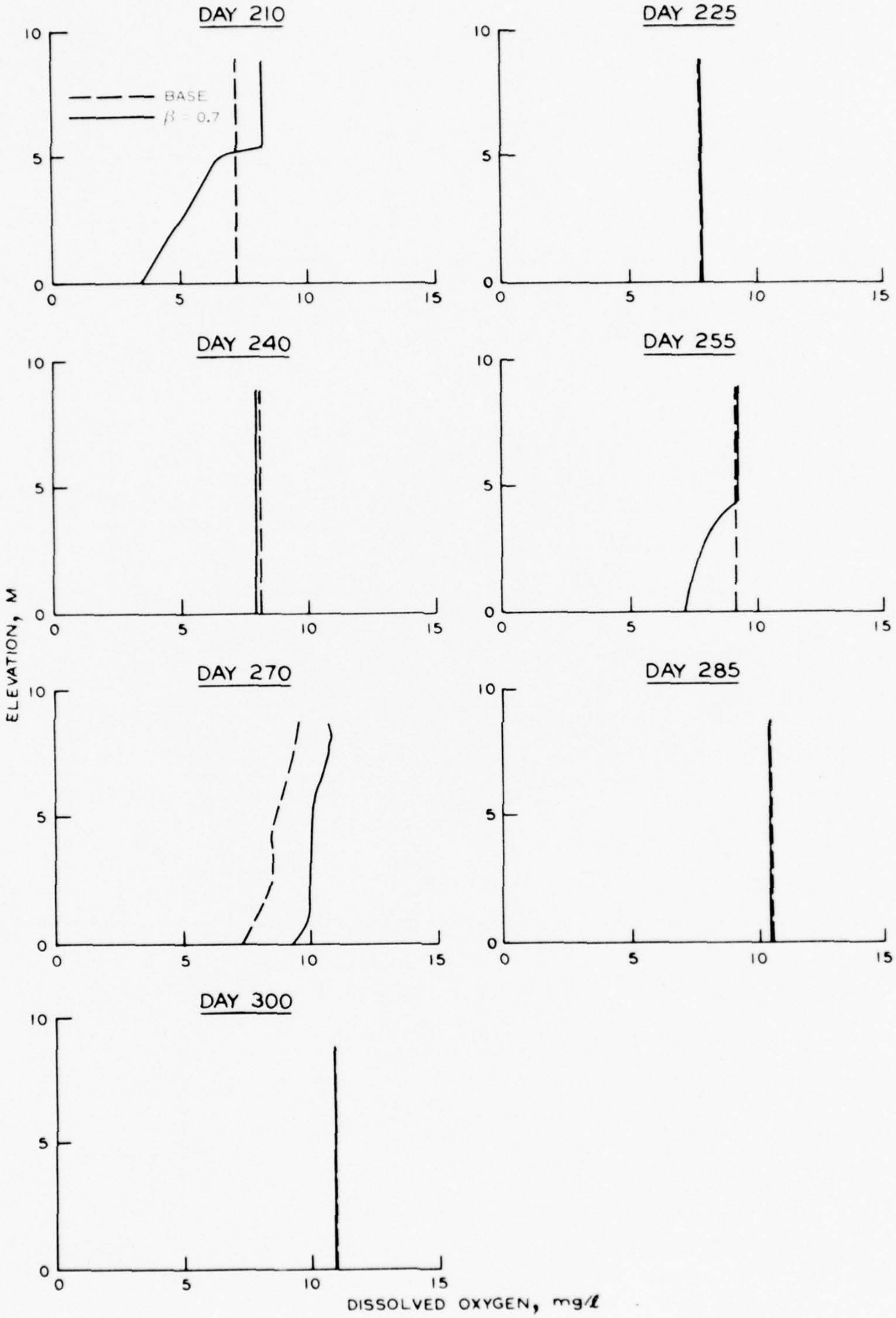


Figure 16 (sheet 2 of 2)

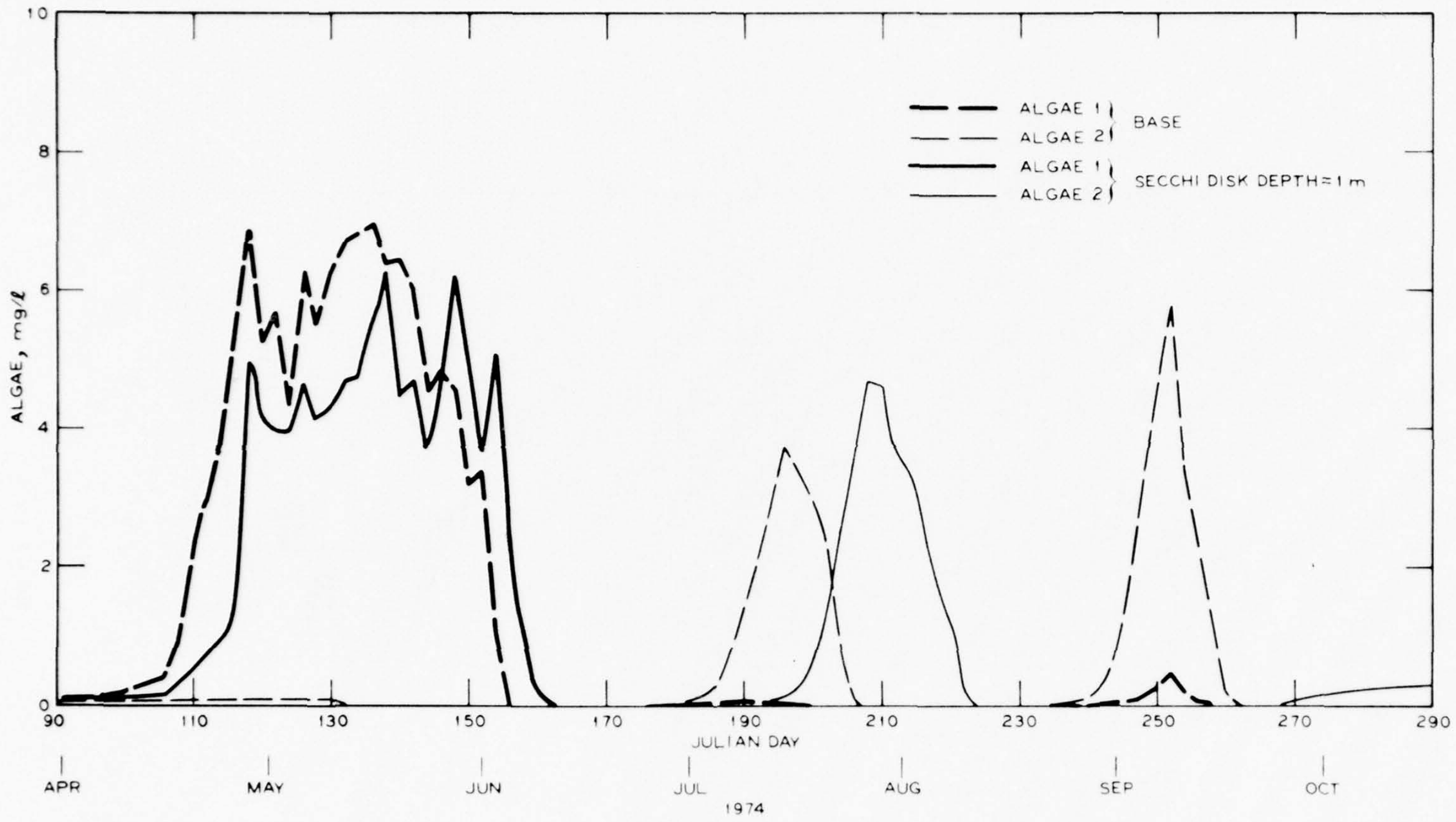


Figure 17. Effect of 1-m light penetration on surface algae concentrations

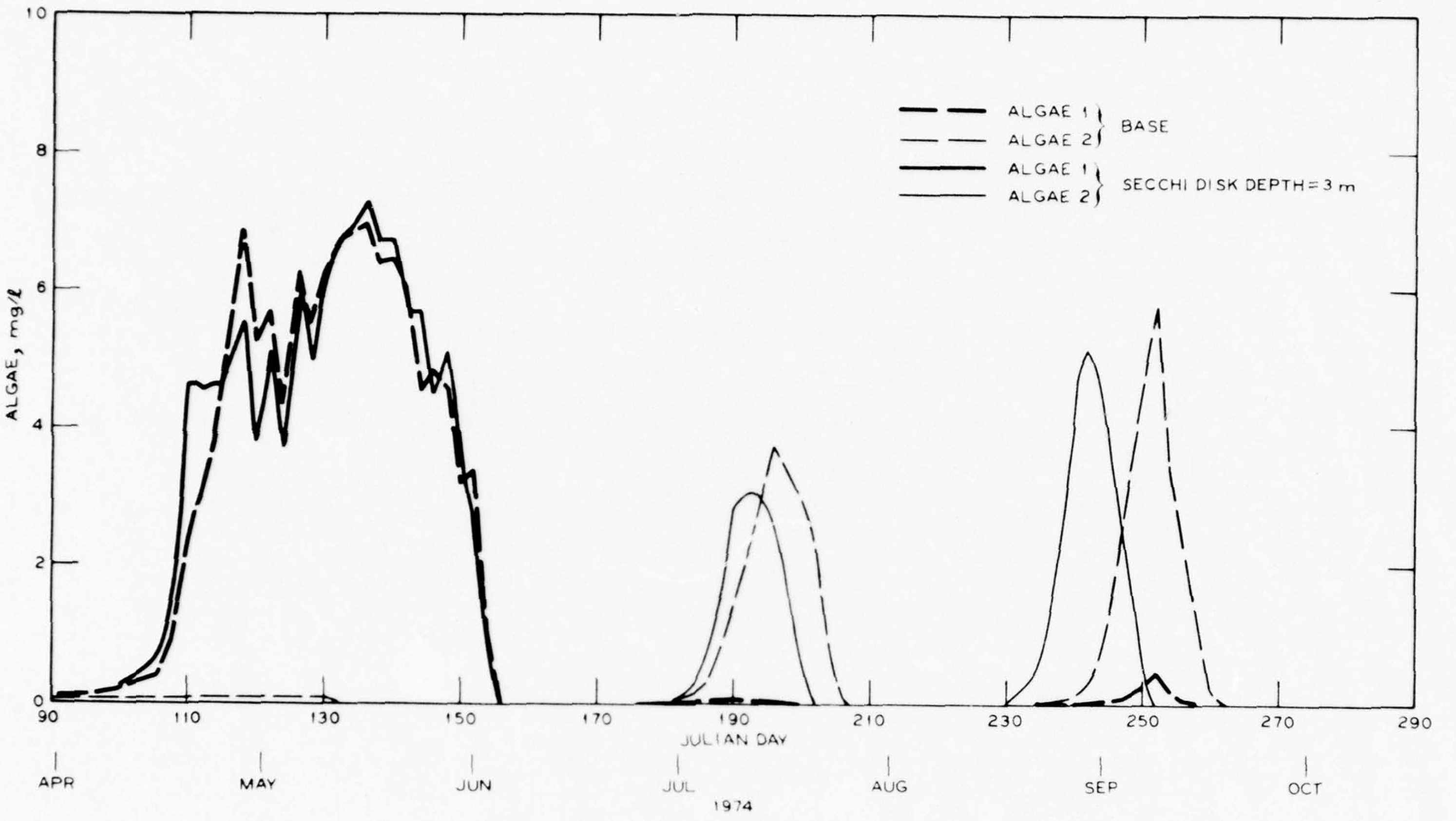


Figure 18. Effect of 3-m light penetration on surface algae concentrations

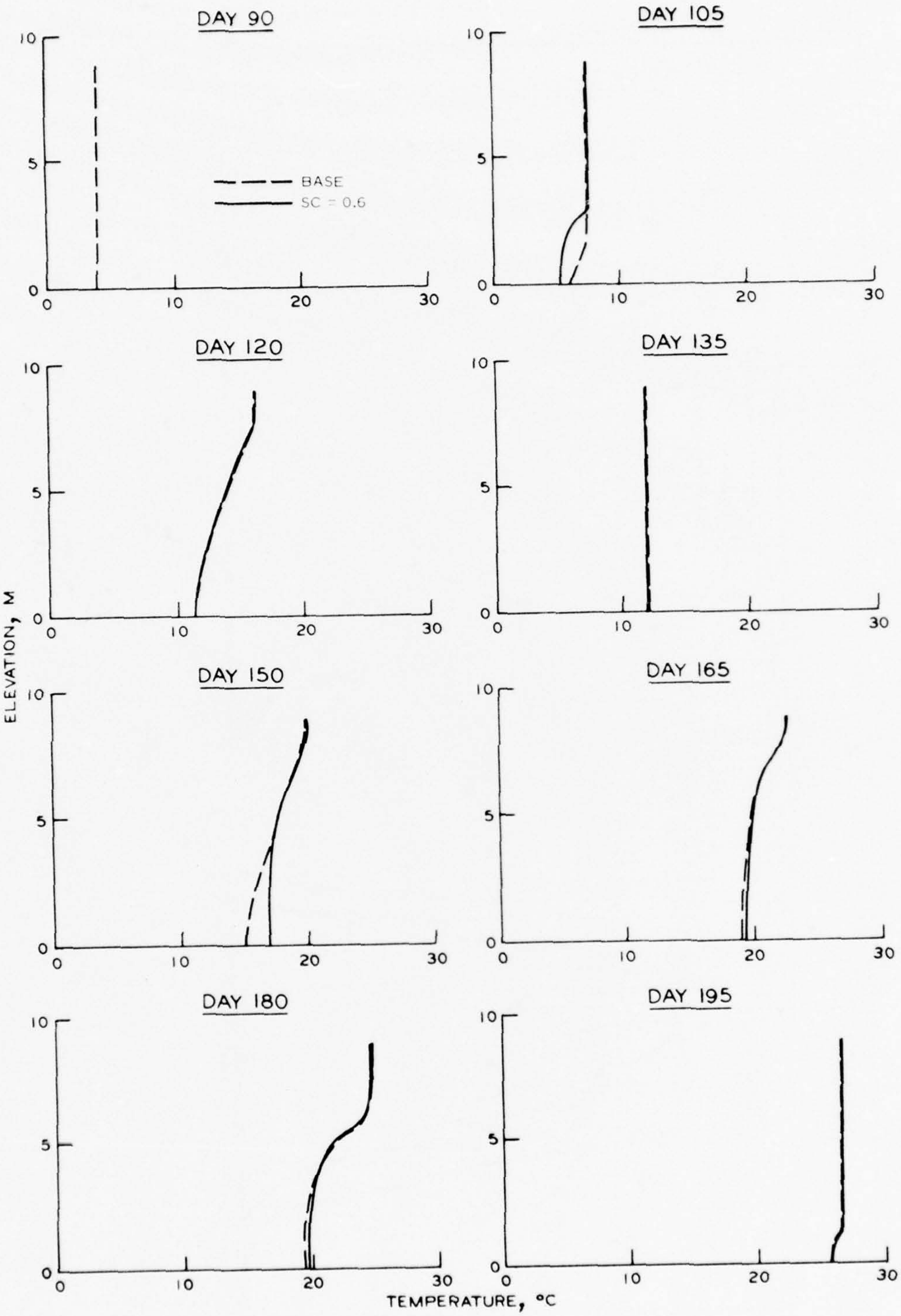


Figure 20. Effect of self-shading coefficient on temperature (sheet 1 of 2)

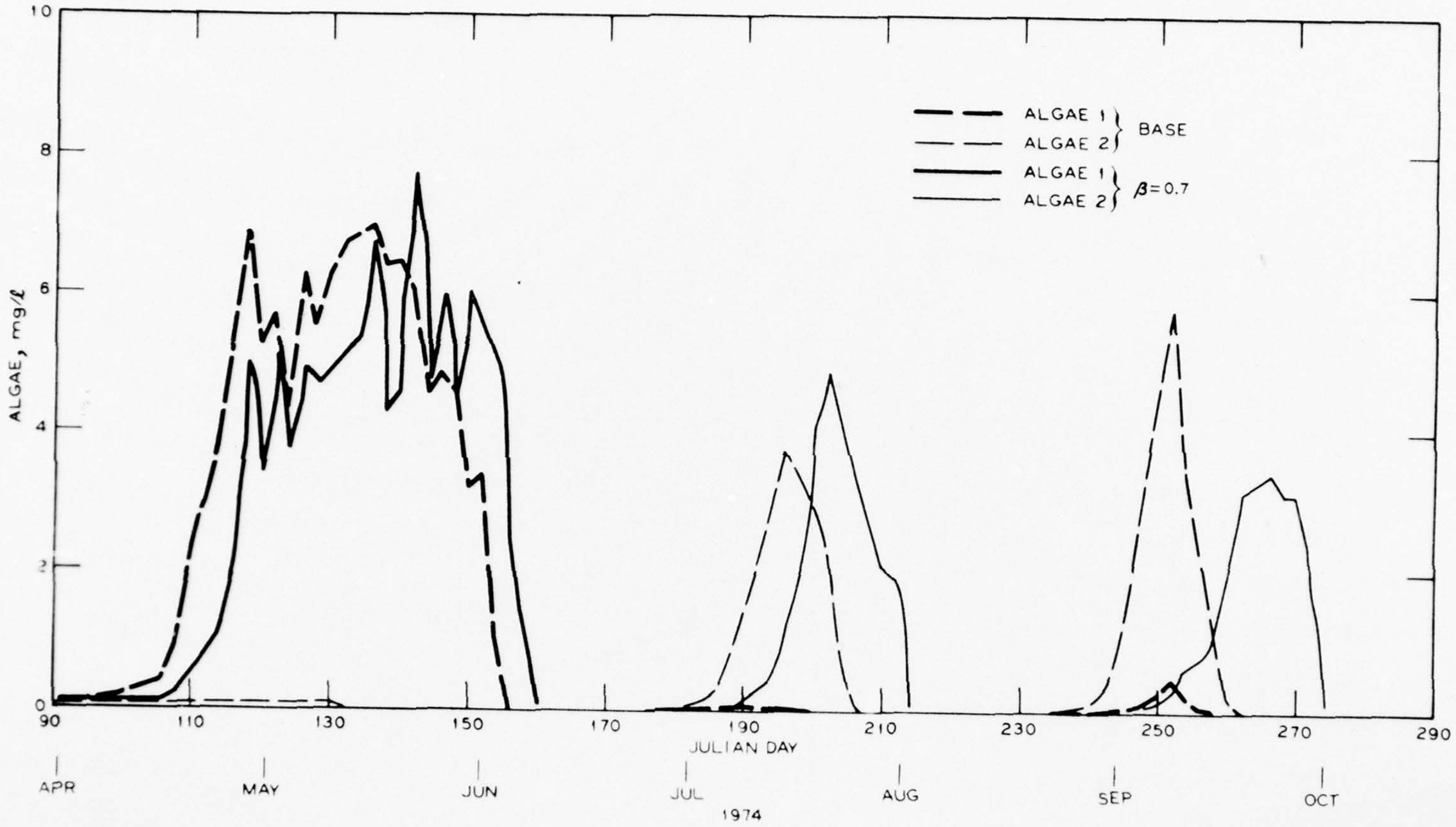


Figure 19. Effect of increased top-layer absorption of solar radiation on surface algae concentrations

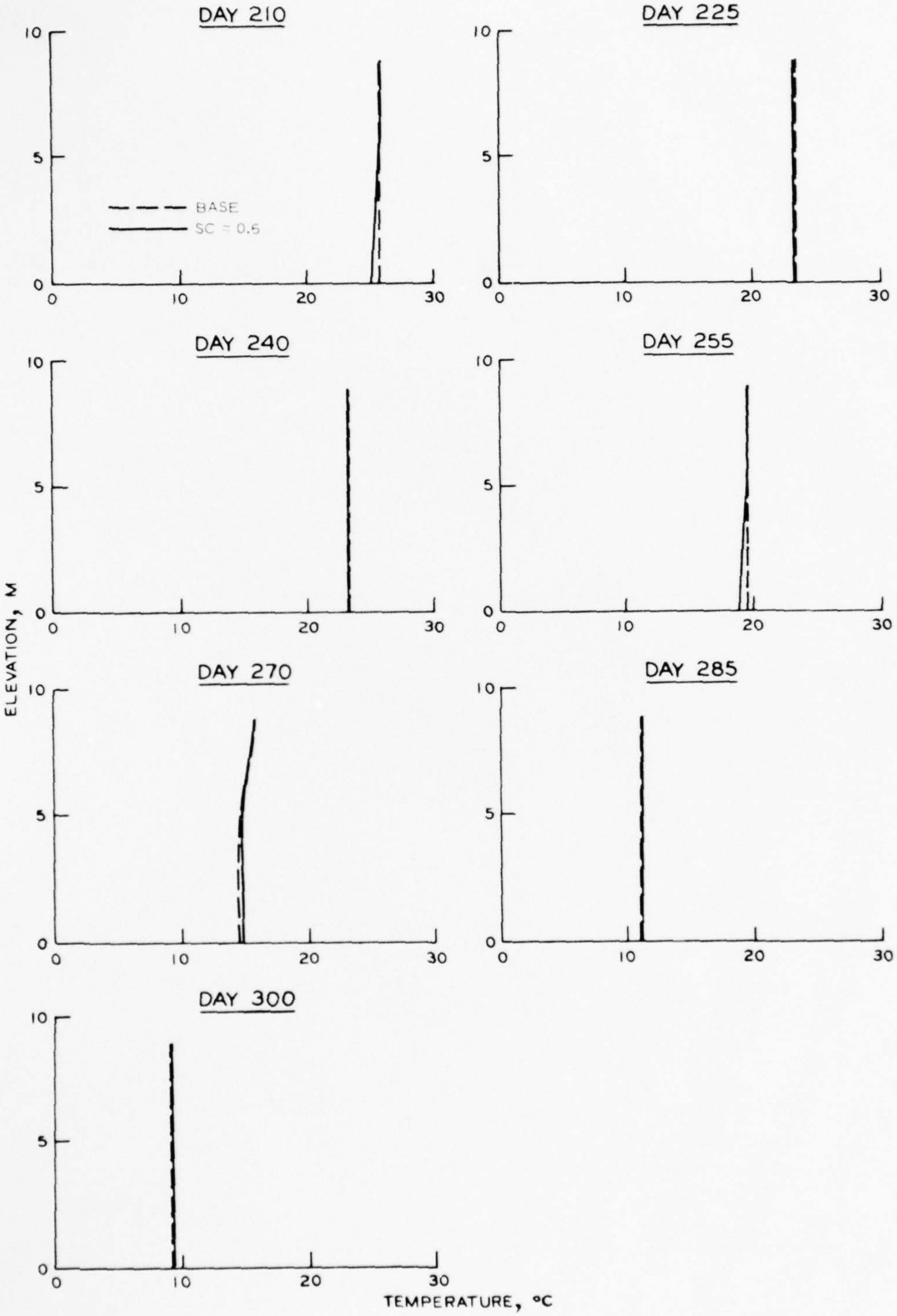


Figure 20 (sheet 2 of 2)

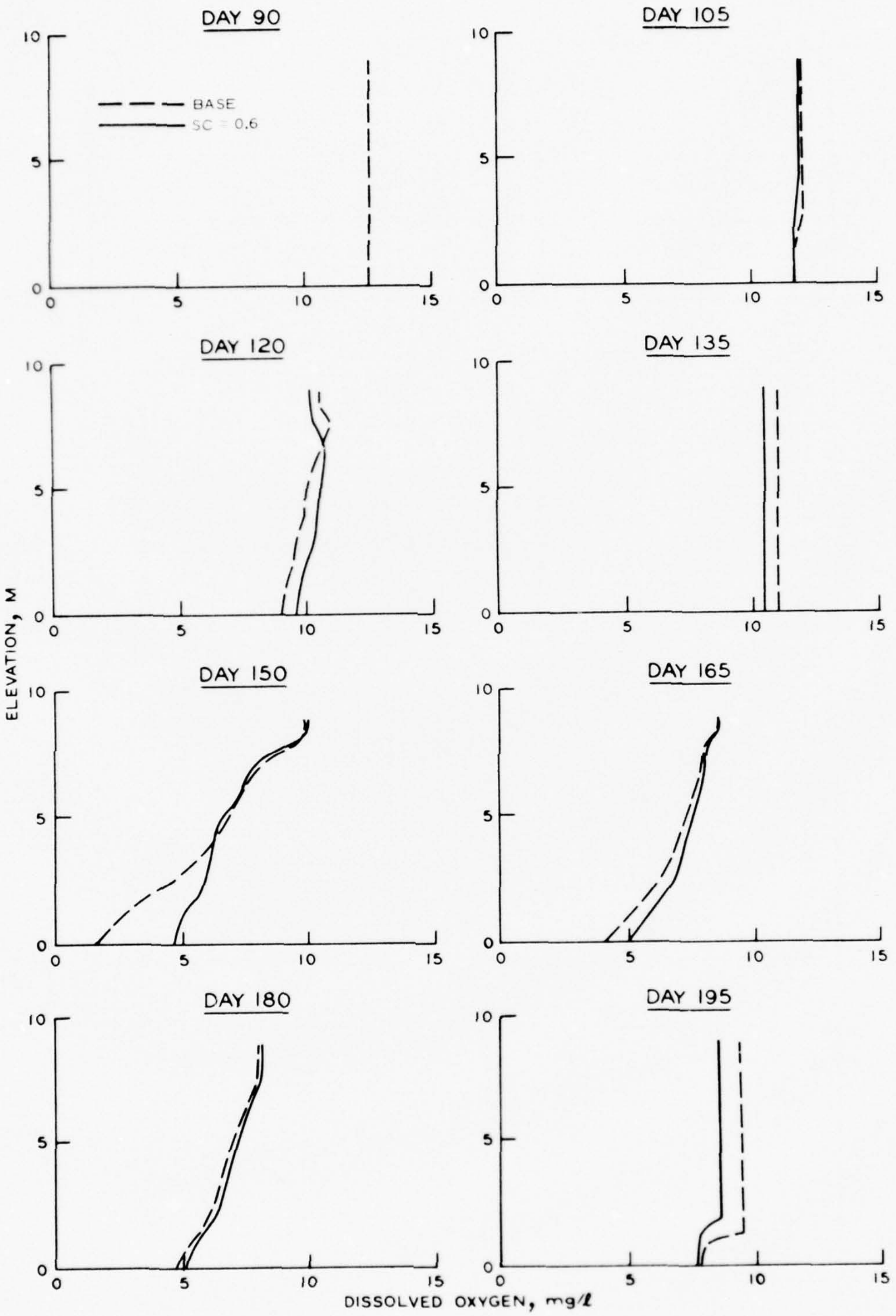


Figure 21. Effect of self-shading coefficient on dissolved oxygen (sheet 1 of 2)

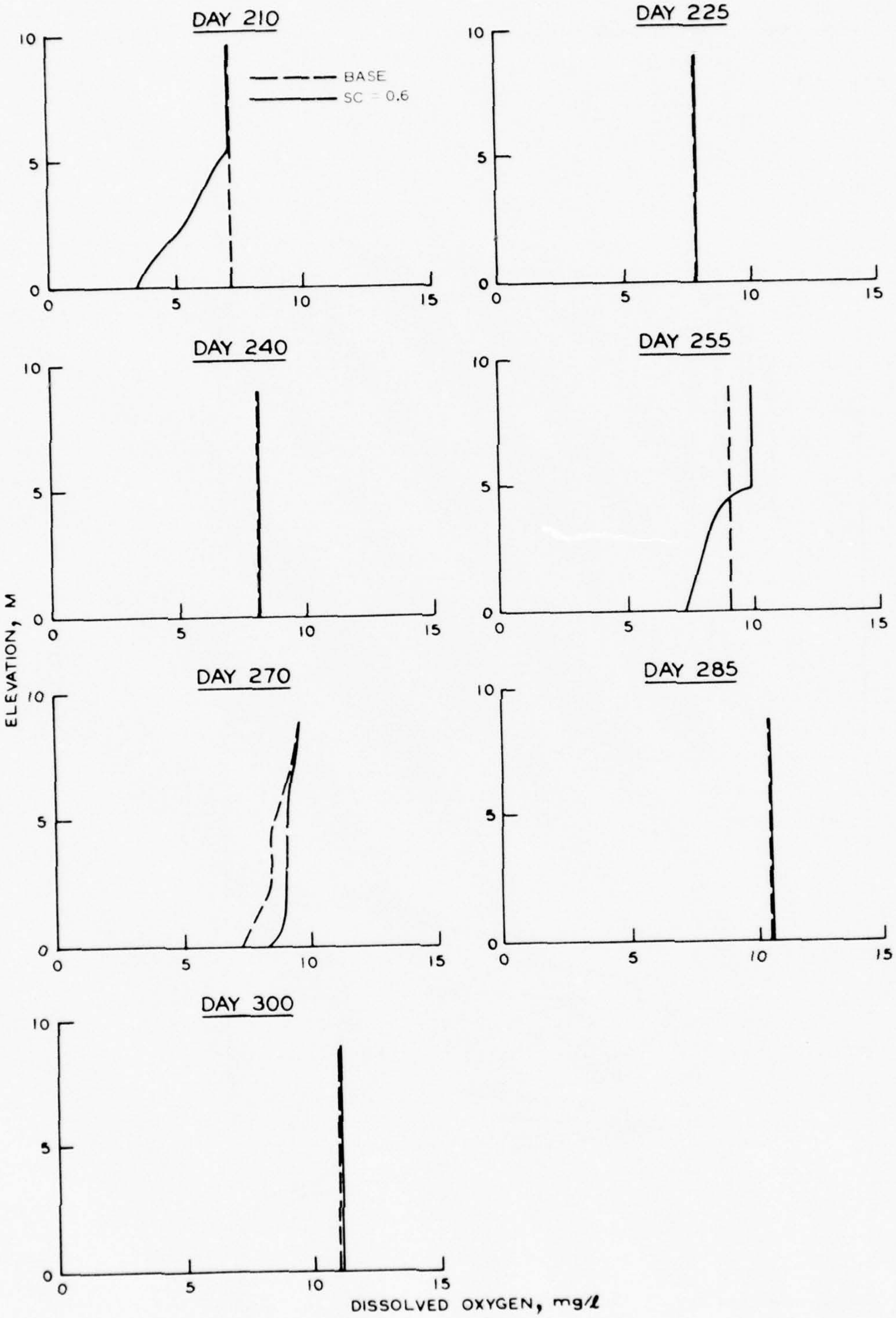


Figure 21 (sheet 2 of 2)

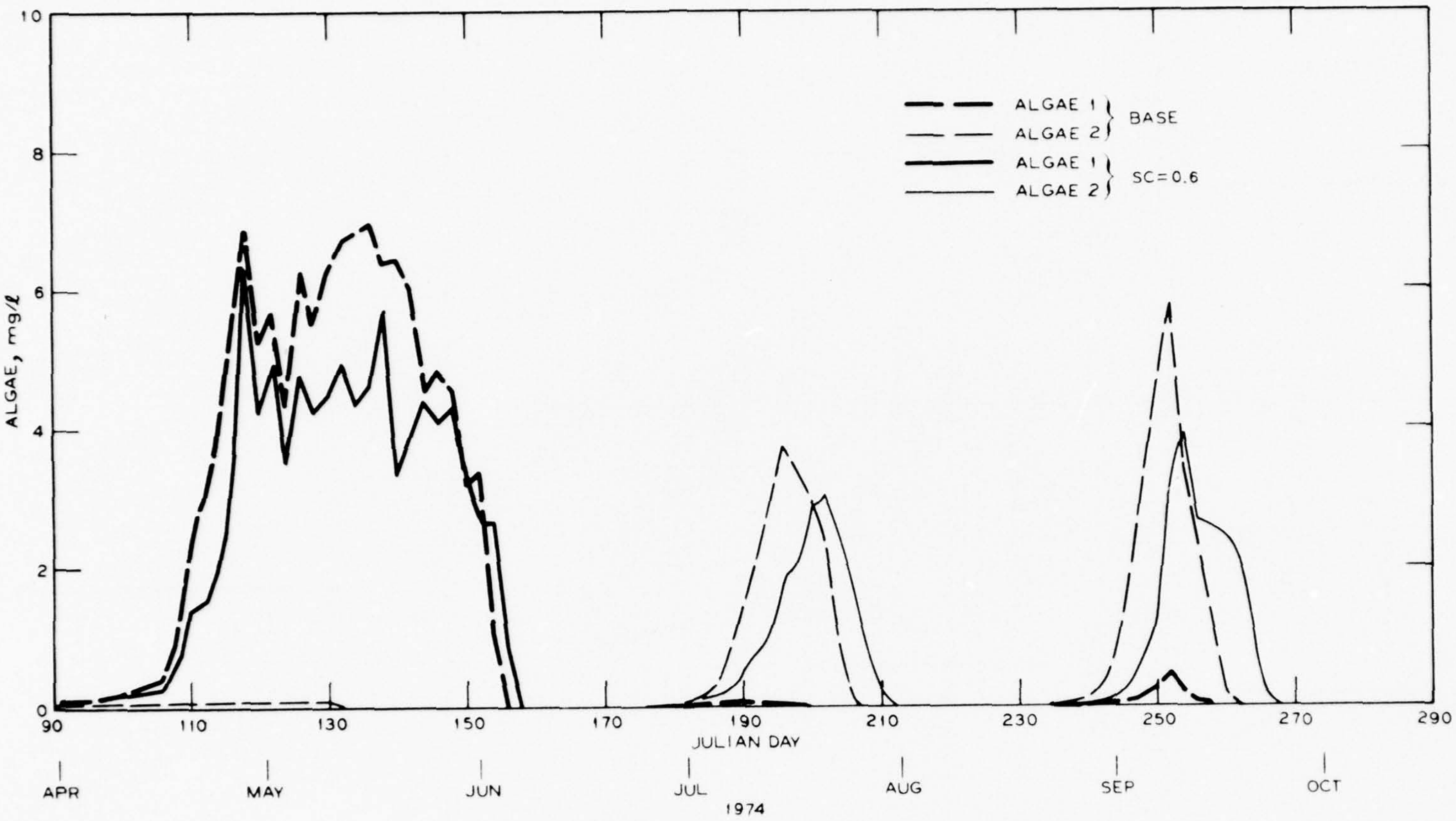


Figure 22. Effect of self-shading coefficient on surface algae concentrations

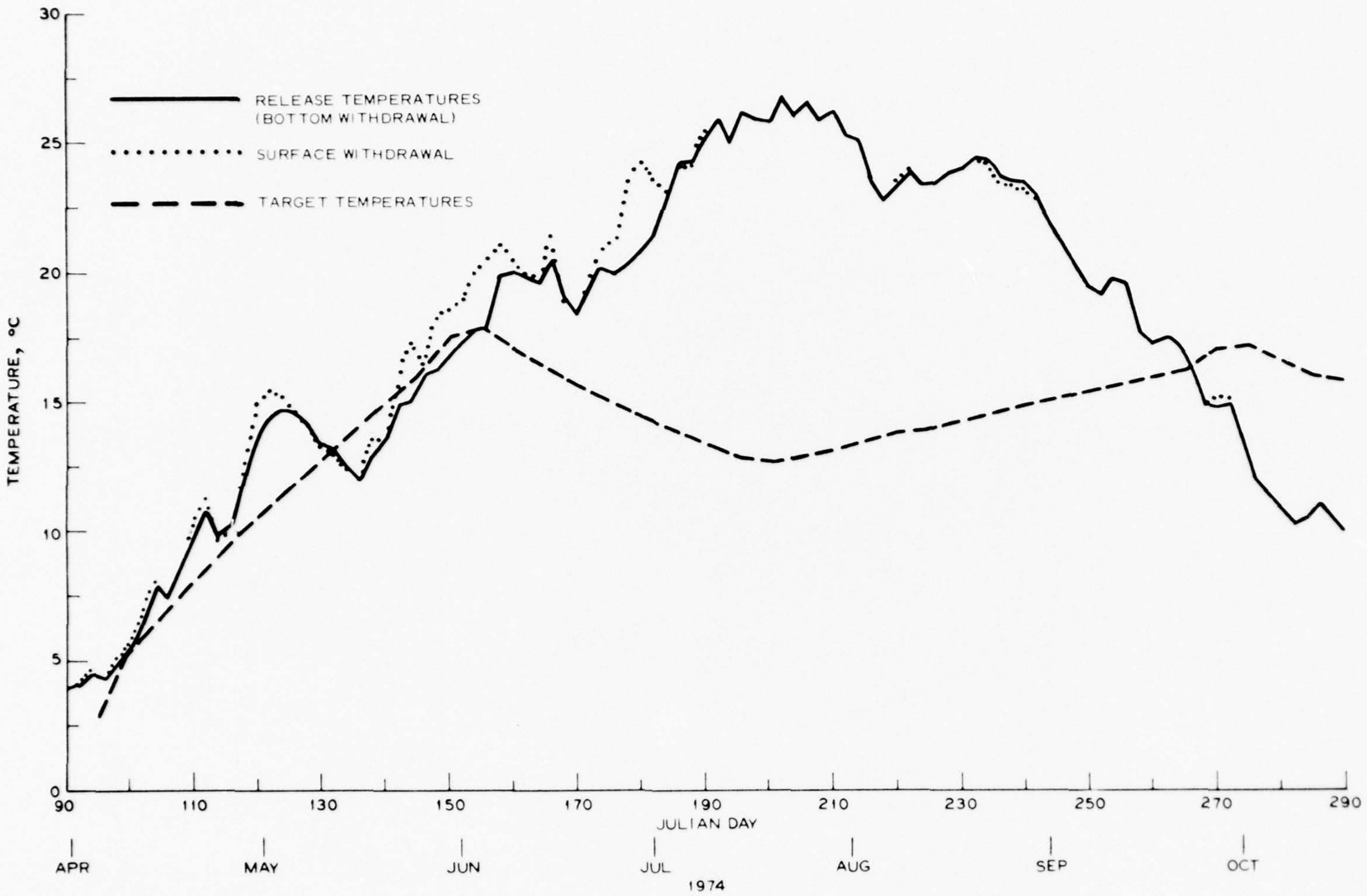


Figure 23. Effect of surface and bottom withdrawal on release temperatures

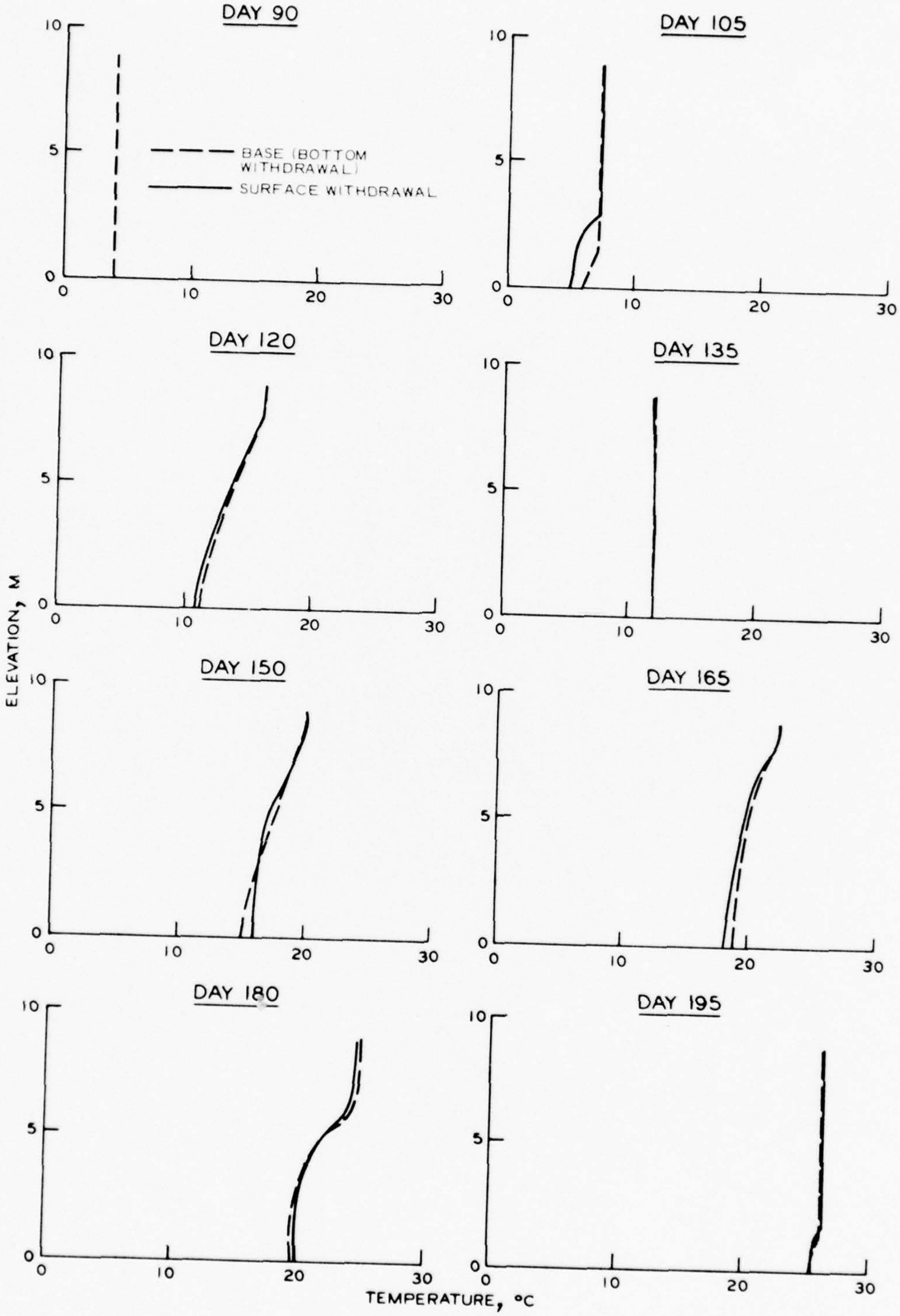
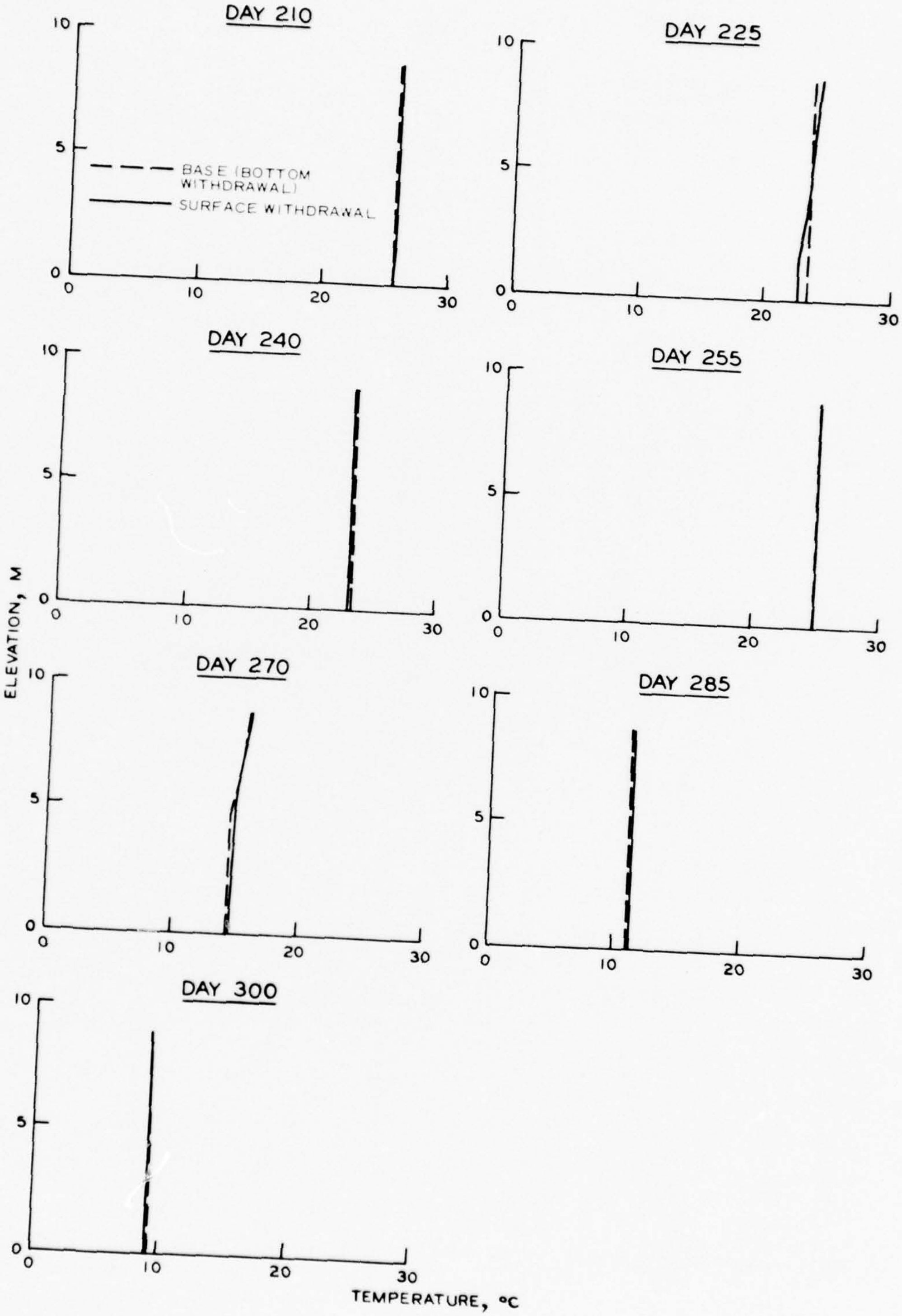


Figure 24. Effect of type of withdrawal on temperature (sheet 1 of 2)



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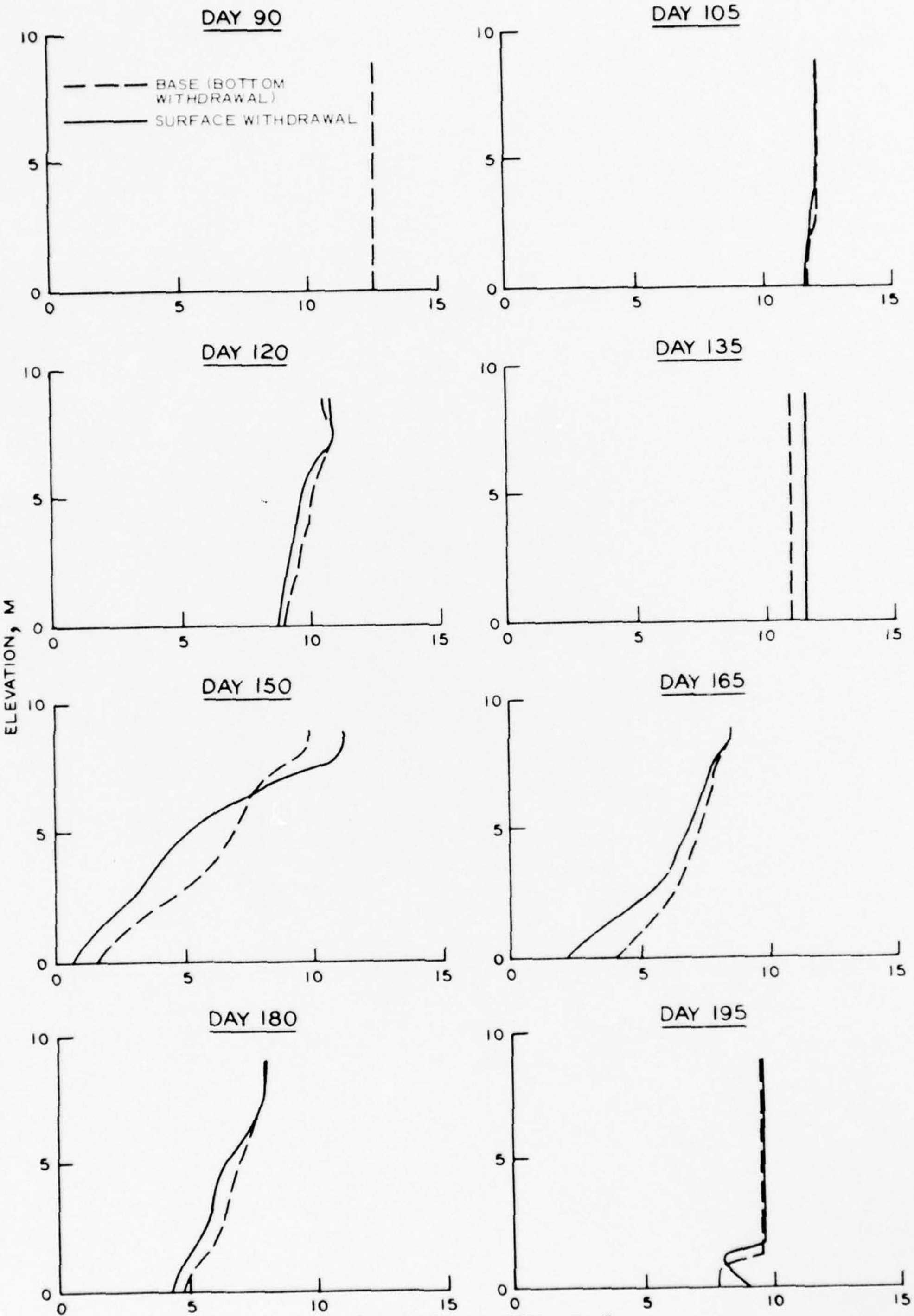

Figure 25. Effect of type of withdrawal on dissolved oxygen (sheet 1 of 2)

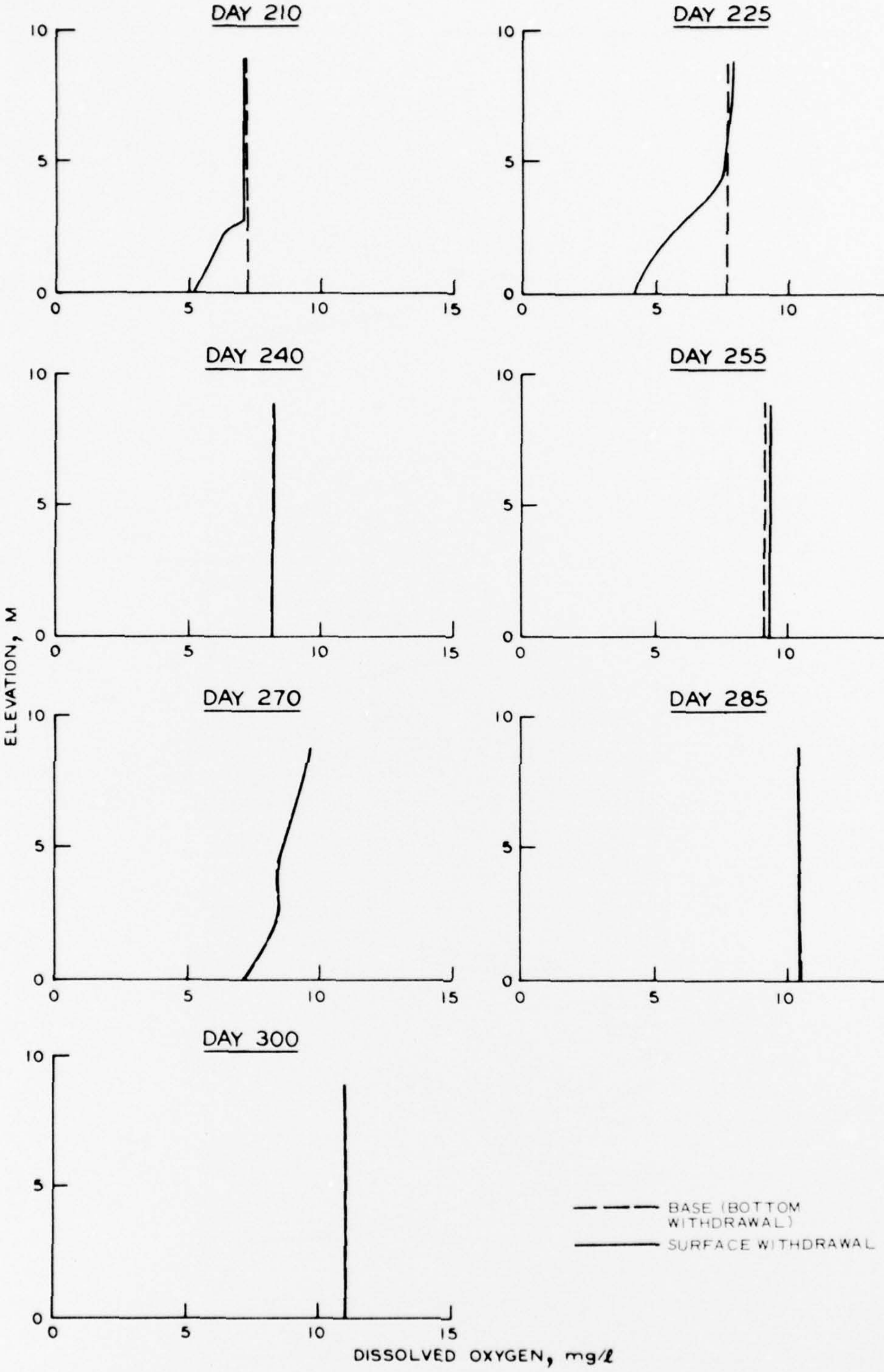

Figure 25 (sheet 2 of 2)

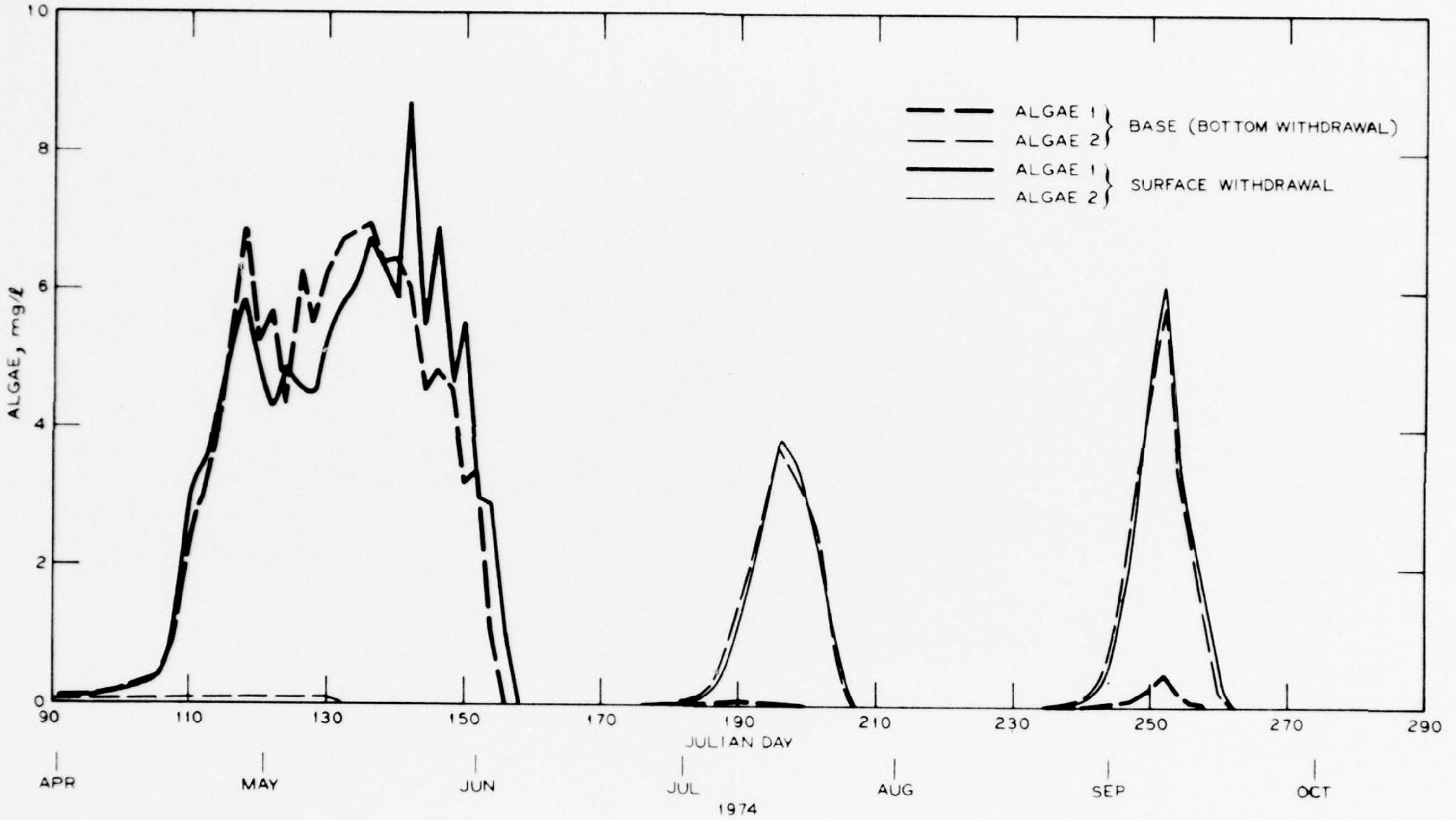

Figure 26. Effect of type of withdrawal on surface algae concentrations

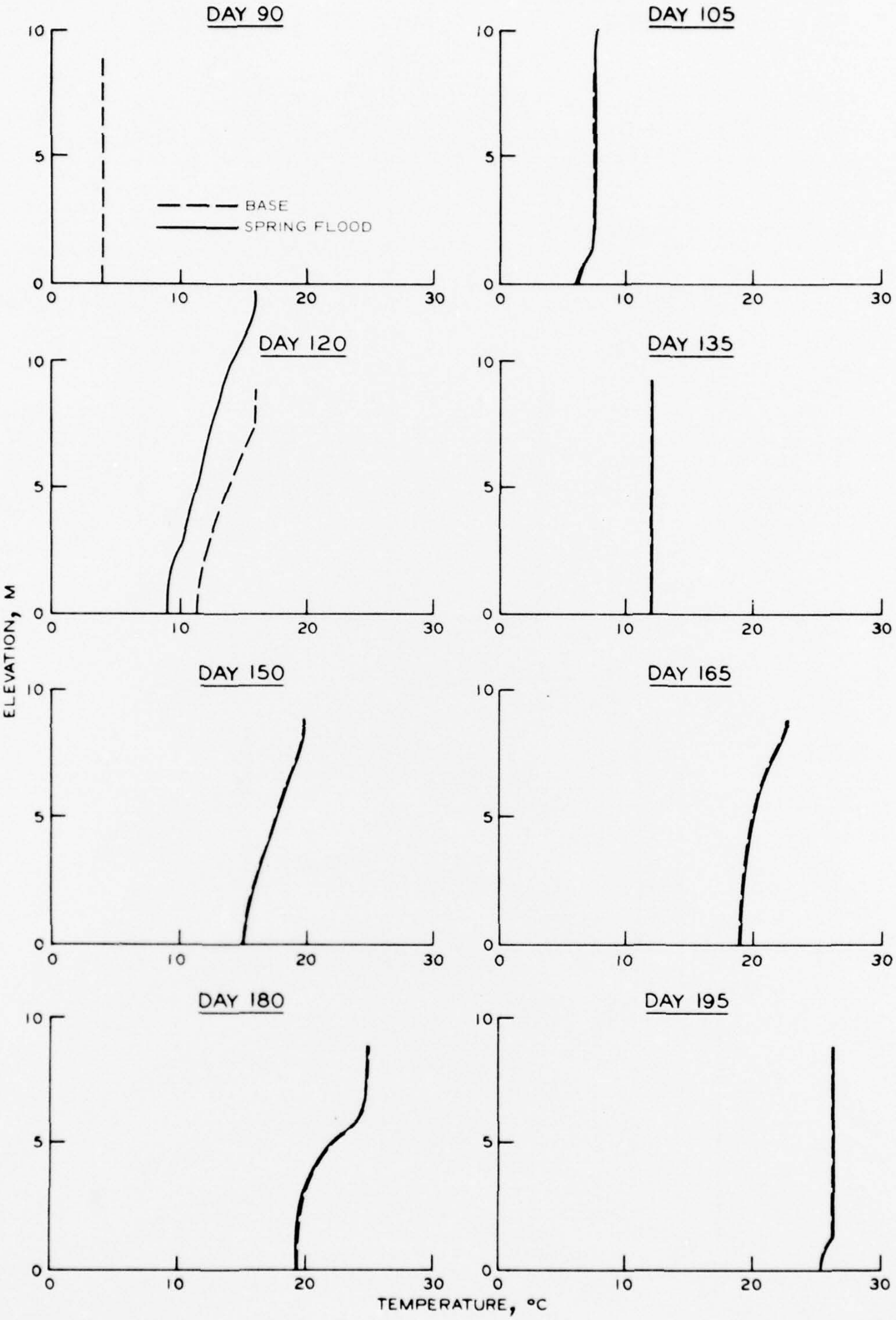

Figure 27. Effect of a spring flood on temperature (sheet 1 of 2)

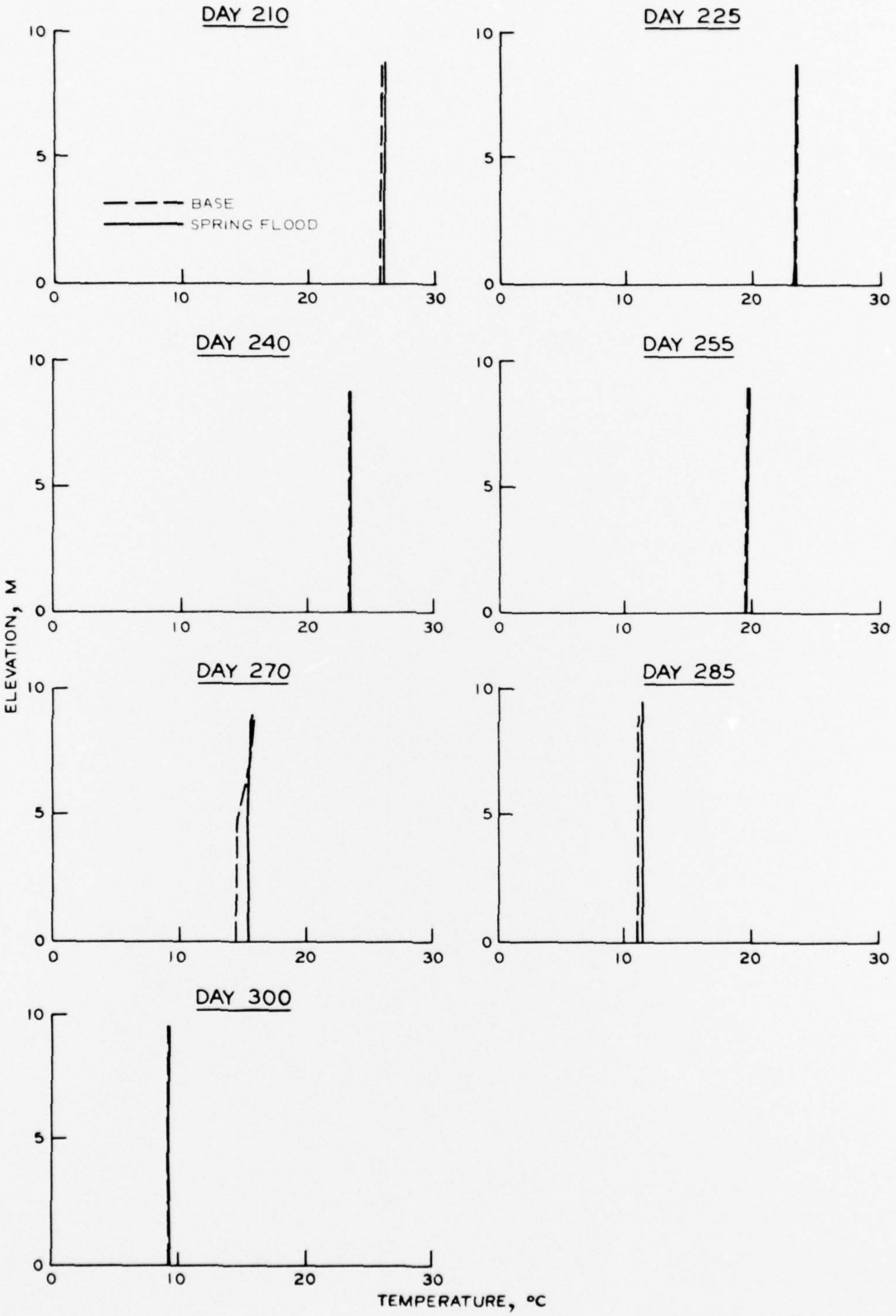

Figure 27 (sheet 2 of 2)

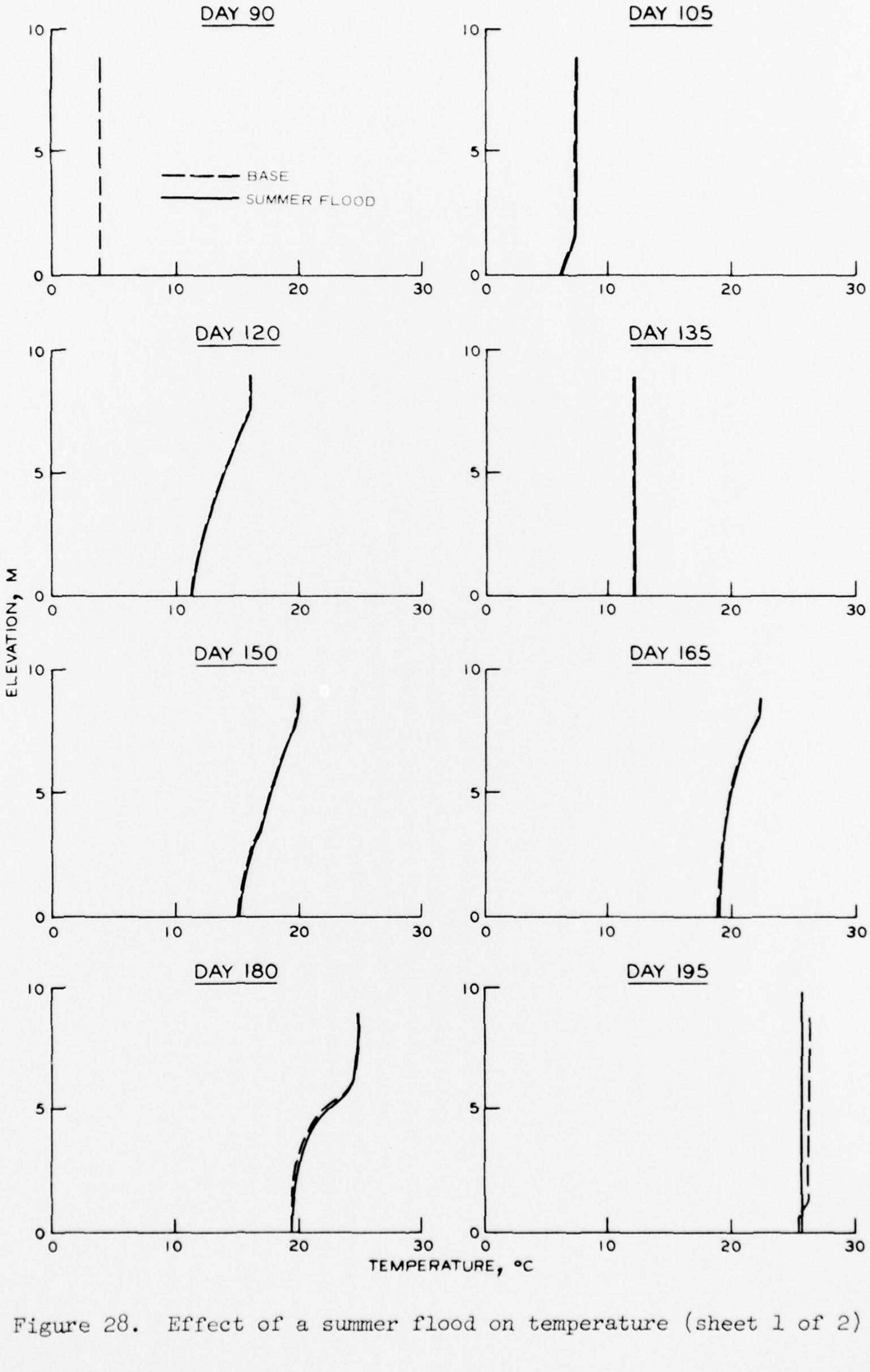

Figure 28. Effect of a summer flood on temperature (sheet 1 of 2)

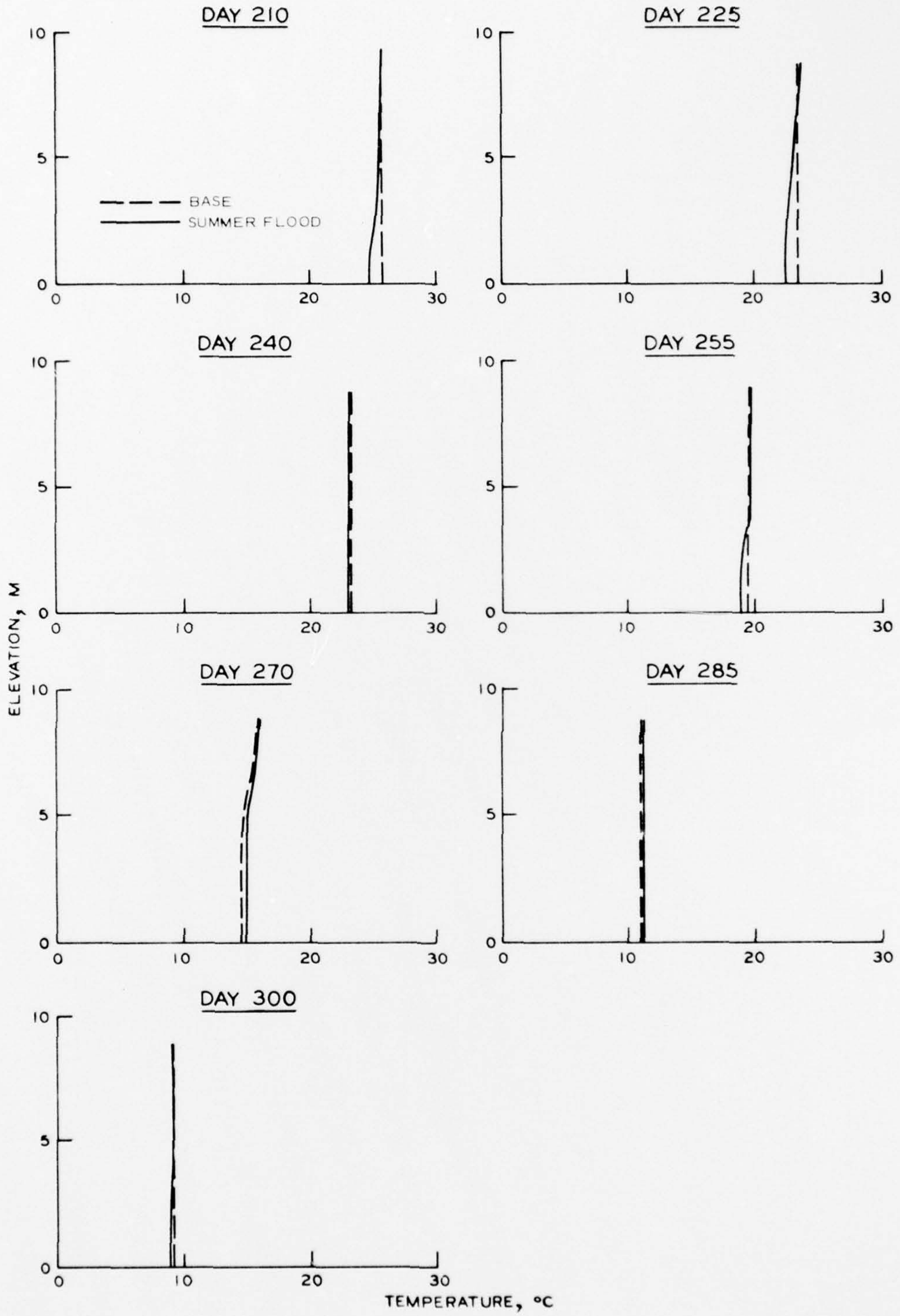

Figure 28 (sheet 2 of 2)

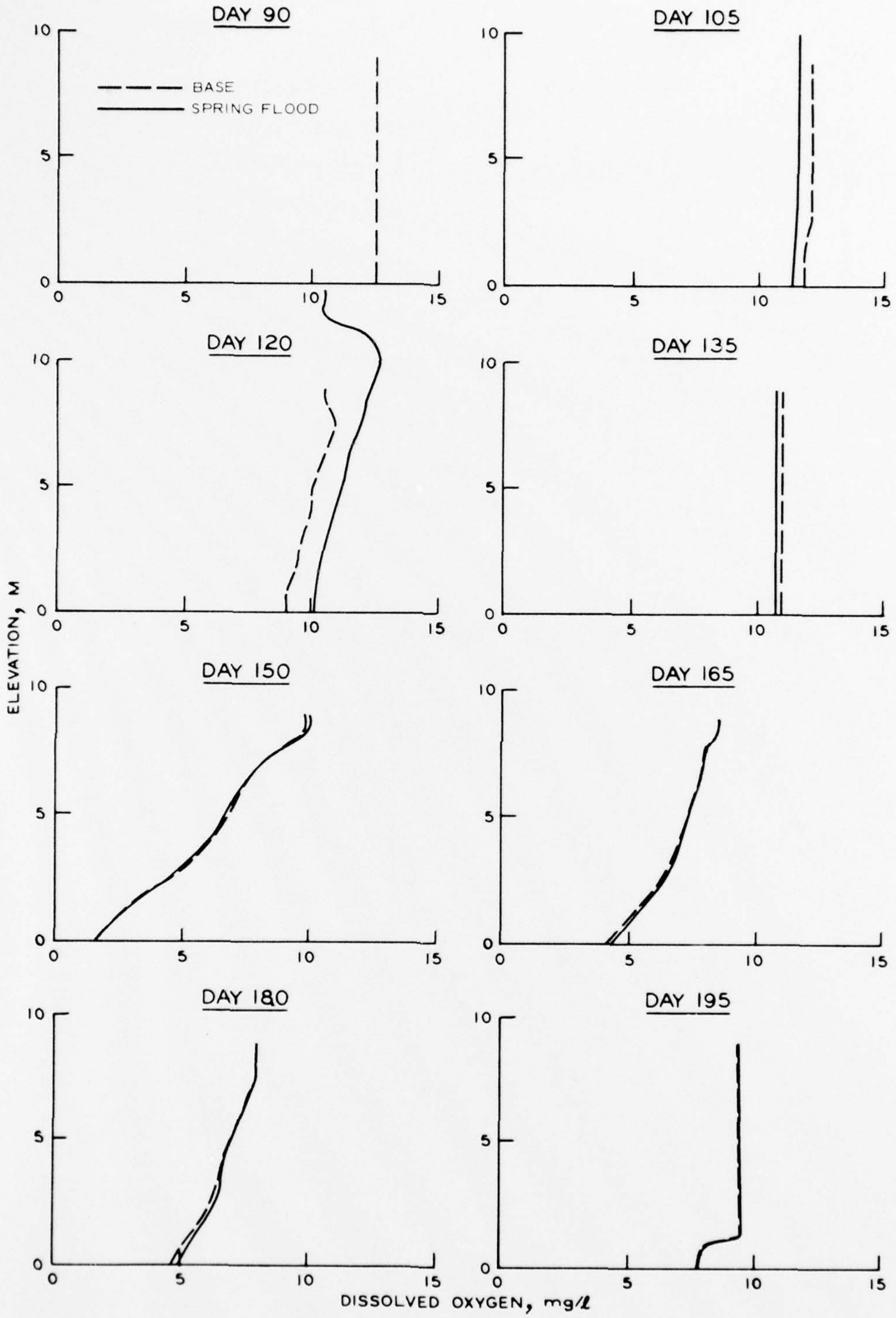

Figure 29. Effect of a spring flood on dissolved oxygen (sheet 1 of 2)

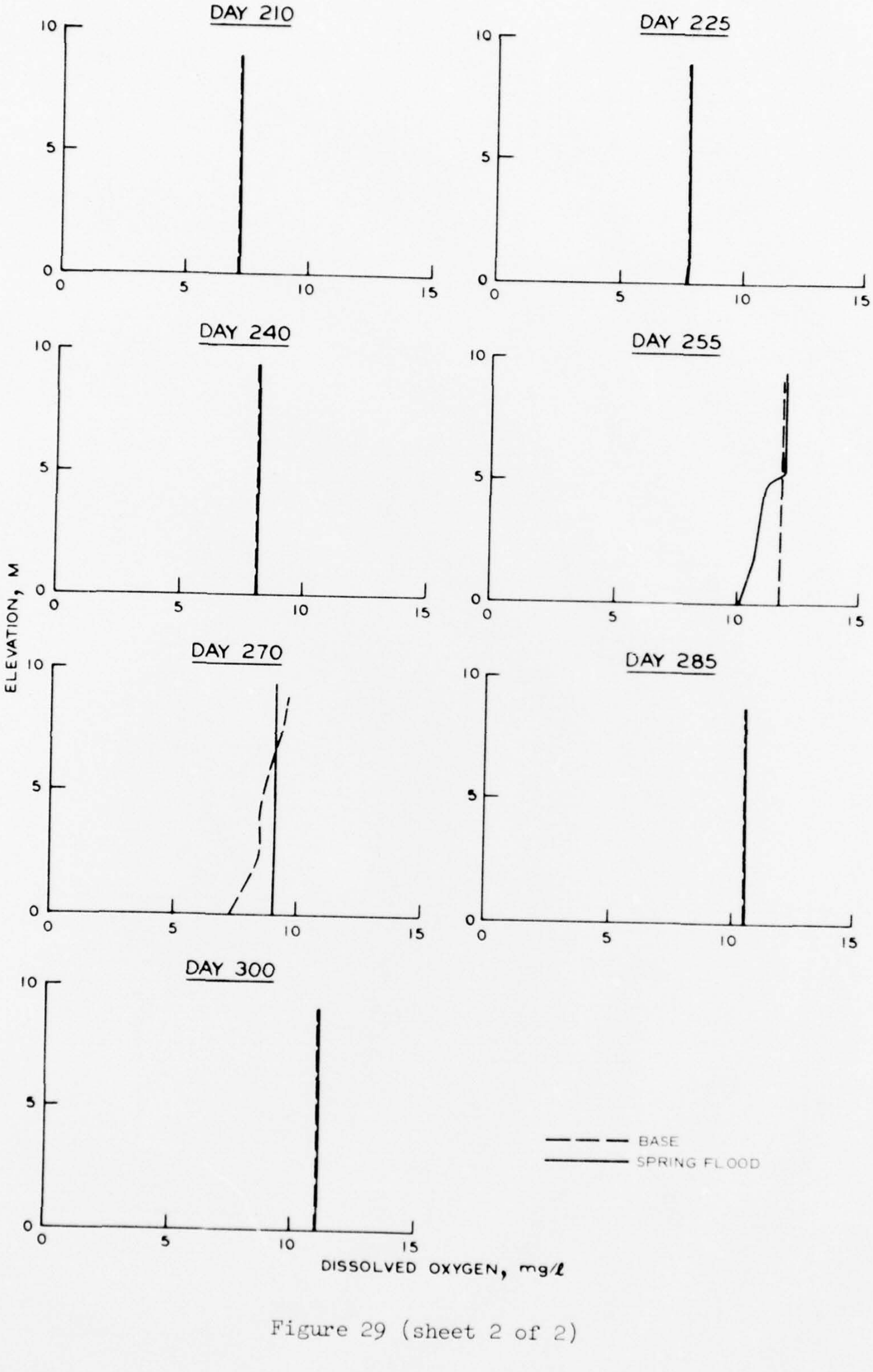

Figure 29 (sheet 2 of 2)

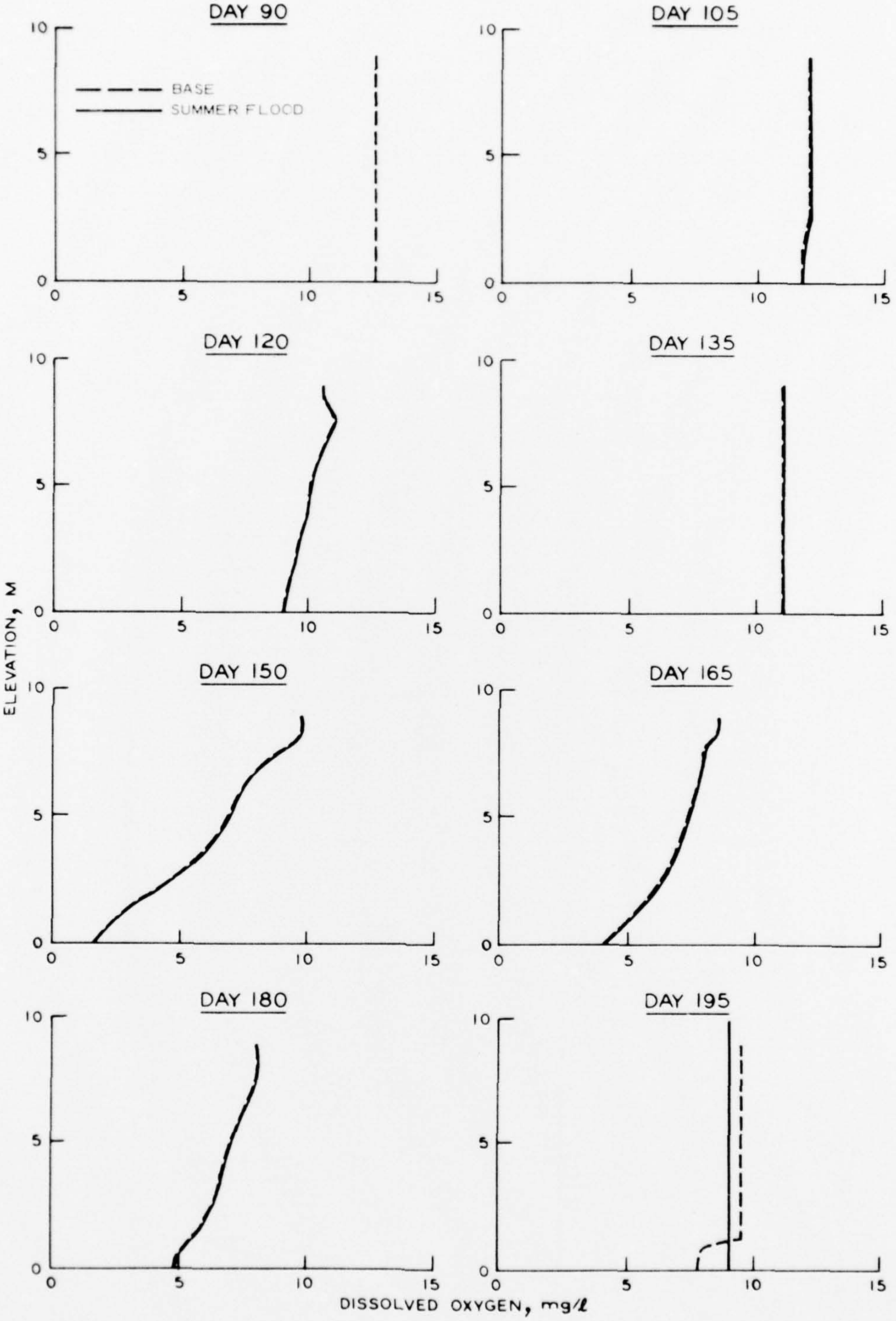

Figure 30. Effect of a summer flood on dissolved oxygen (sheet 1 of 2)

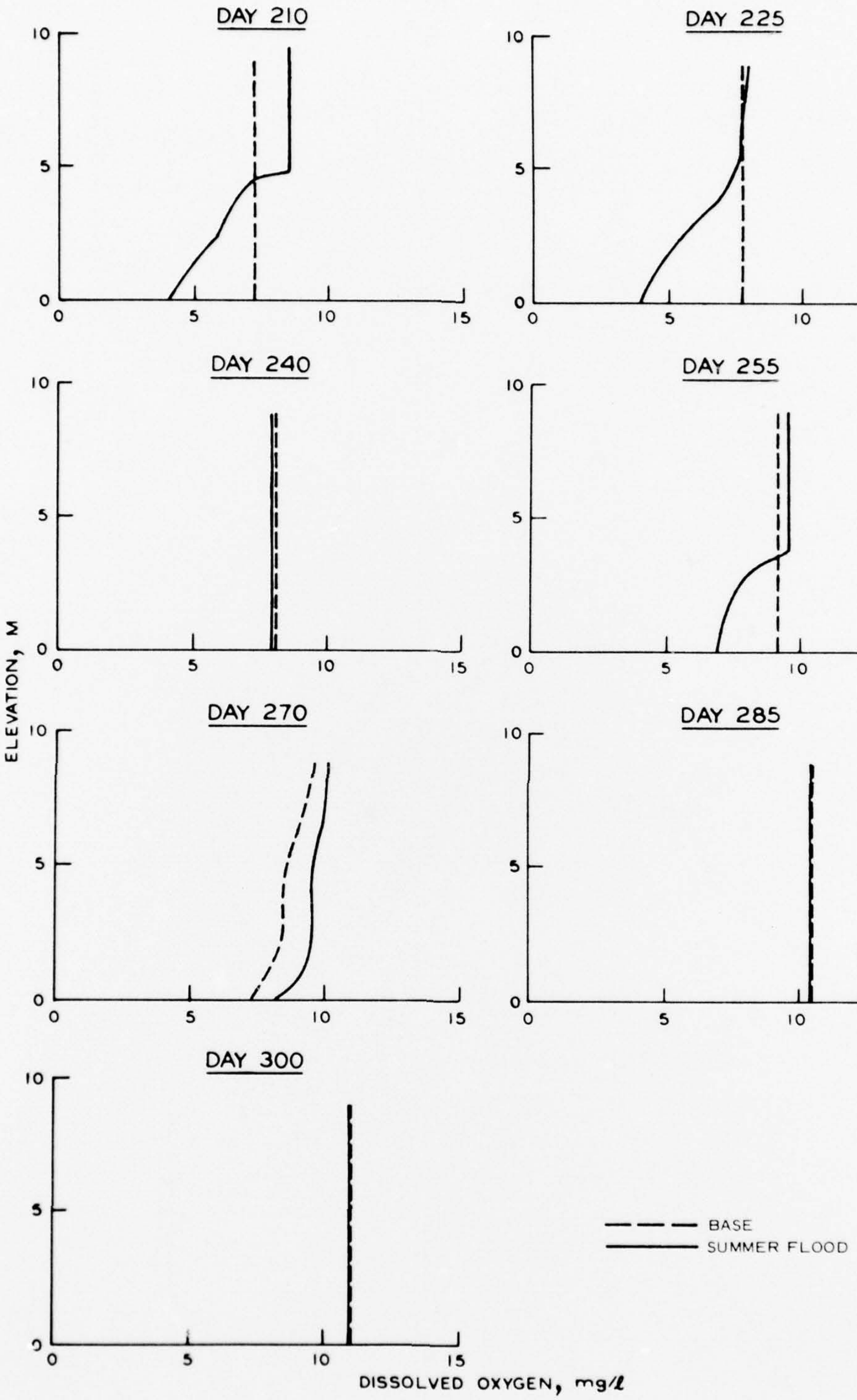

Figure 30 (sheet 2 of 2)

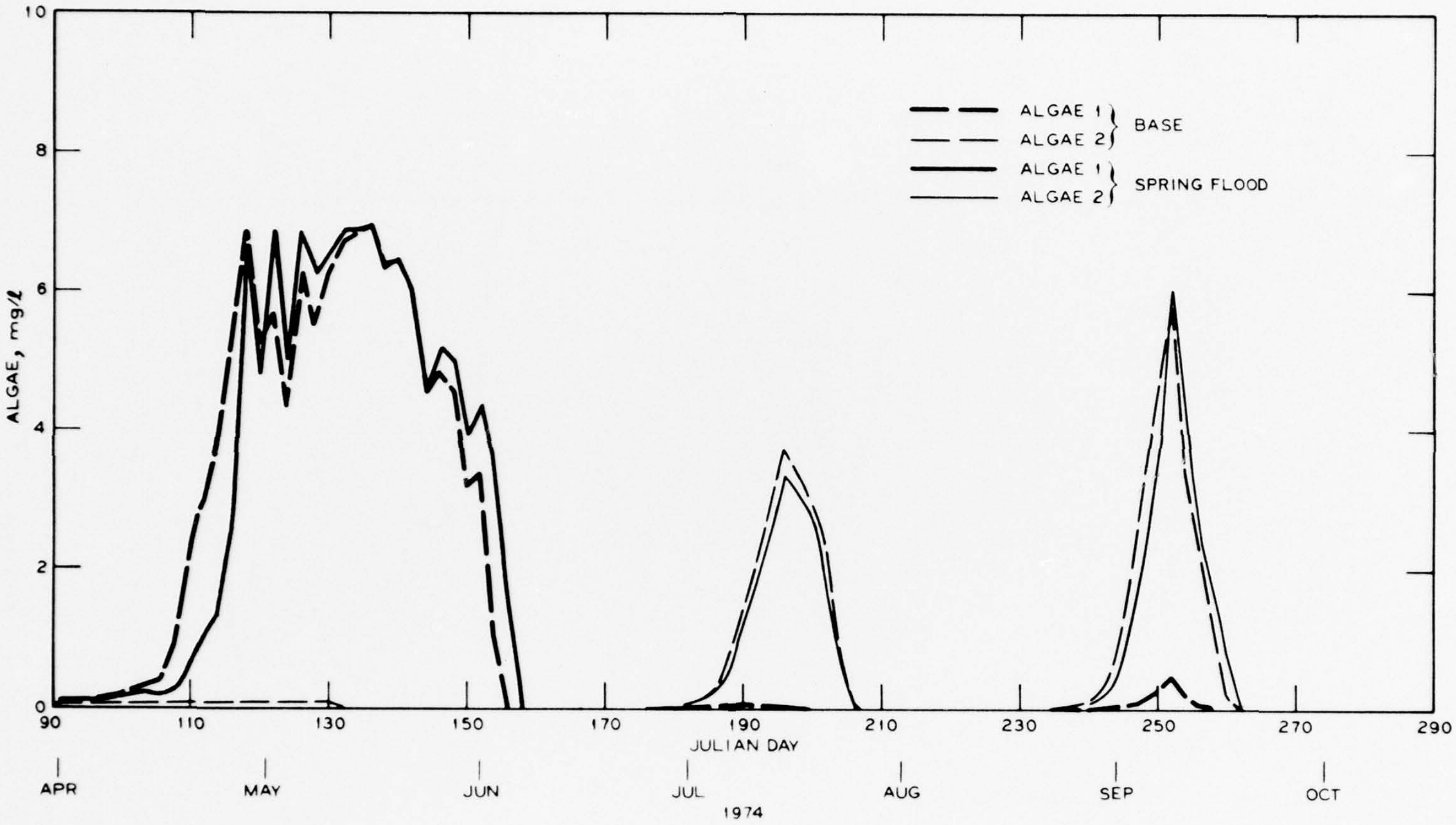

Figure 31. Effect of a spring flood on surface algae concentrations

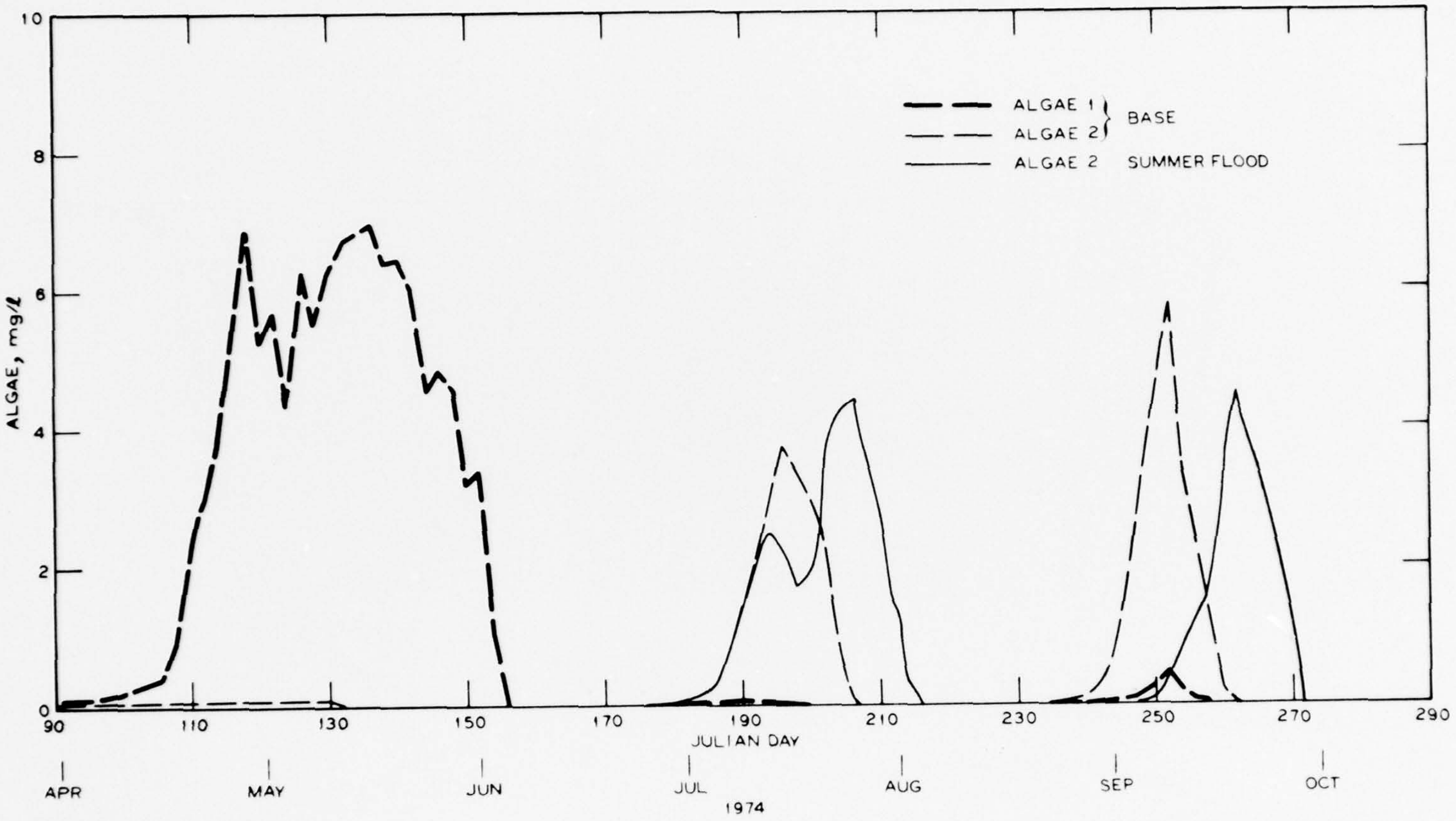

Figure 32. Effect of a summer flood on surface algae concentrations

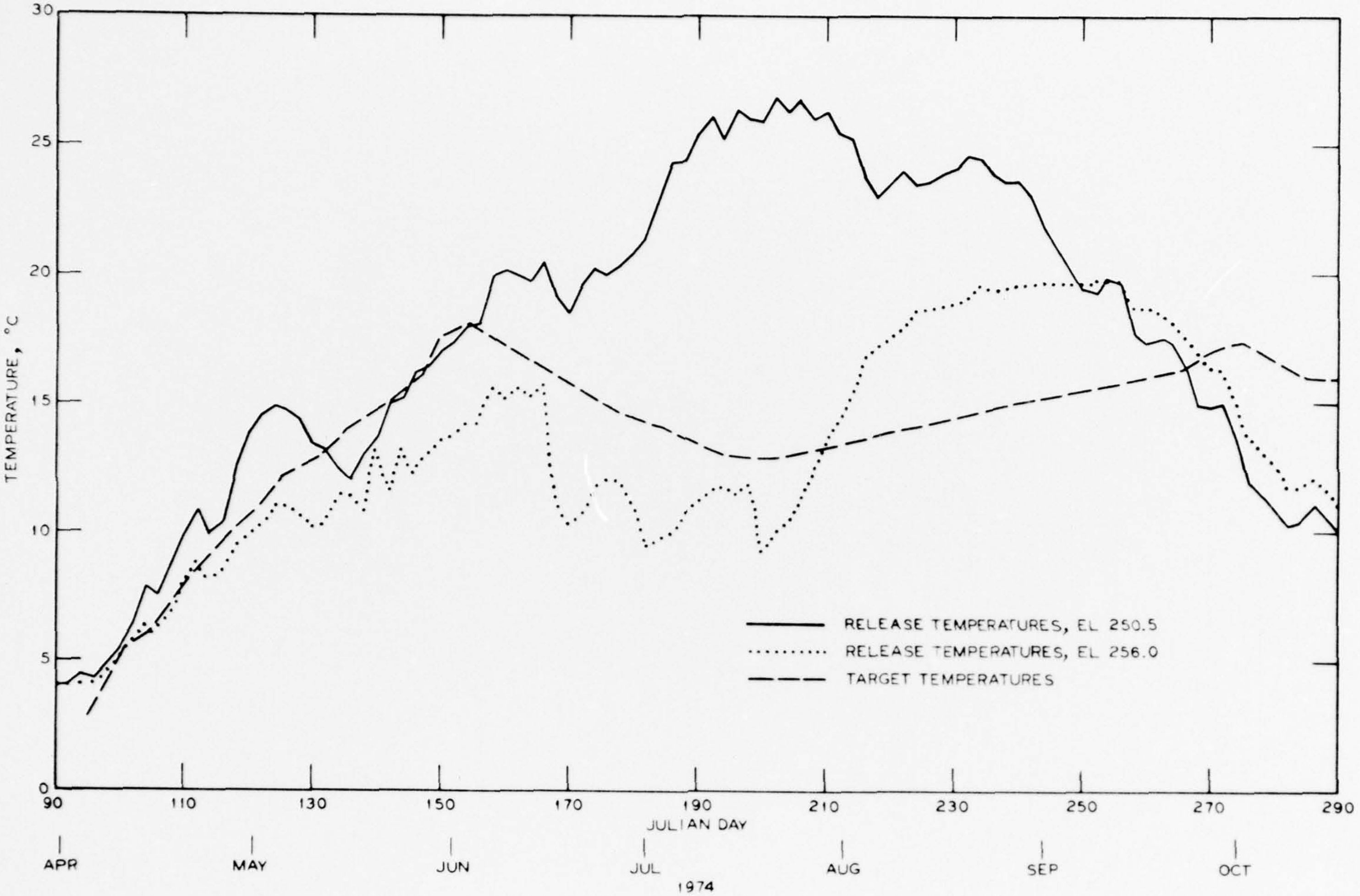

Figure 33. Comparison of release and target temperatures at elevations 250.5 and 256

In accordance with ER 70-2-3, paragraph 6c(1)(b), dated 15 February 1973, a facsimile catalog card in Library of Congress format is reproduced below.

Ford, Dennis E
Preliminary water-quality evaluation of a lower pool elevation for proposed LaFarge Lake, Wisconsin, by Dennis E. Ford, Kent W. Thornton, ⌊and⌋ Donald L. Robey. Vicksburg, U. S. Army Engineer Waterways Experiment Station, 1977.
1 v. (various pagings) illus. 27 cm. (U. S. Waterways Experiment Station. Miscellaneous paper Y-77-2)
Prepared for U. S. Army Engineer District, St. Paul, St. Paul, Minnesota, under Intra-Army Order No. NCS-IA-76-86-EDH.
Includes bibliography.

1. Computerized simulation. 2. Ecological models. 3. LaFarge Lake. 4. Mathematical models. 5. Reservoirs. 6. Water quality. I. Robey, Donald L., joint author. II. Thornton, Kent W., joint author. III. U. S. Army Engineer District, St. Paul. (Series: U. S. Waterways Experiment Station, Vicksburg, Miss. Miscellaneous paper Y-77-2)
TA7.W34m no.Y-77-2