Final Report **Project WFD38 (SNIFFER)**

Investigating the effect of changing retention time in natural and artificial water bodies

Simulations using PROTECH-D

May 2006

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EXECUTIVE SUMMARY

WFD38: Phytoplankton Classification Tool: WP4 Artificial & Heavily Modified Water Bodies (May, 2006)

Project funders/partners: SNIFFER & Environment Agency

Background to research

The Environment Agency and SNIFFER have commissioned this R & D project to examine the potential effect that changes in retention times could have on the phytoplankton communities of a range of water body types. In order to investigate the potential effects of varying retention times, four water bodies were modelled by the phytoplankton community model, PROTECH (Phytoplankton RespOnses To Environmental CHange); two were natural lakes and the others were reservoirs.

Objectives of research

1. Gain understanding of the relationship between phytoplankton composition and mean annual biomass and changing retention time.
2. Assess the potential impact a 20% decrease in the inflow would have on the modelled water bodies.

Key findings and recommendations

All the water bodies tested in this study were sensitive to retention times that were shorter than their original time. The general response in lakes was a decrease in both biomass and cyanobacteria abundance because flushing loss processes began to prevail. The response in the reservoirs was more varied and less consistent.

At longer retention times, responses in both total chlorophyll and cyanobacteria abundance were smaller and particularly so beyond a retention time of 100 days for the lakes in this study. This suggests that drought induced effects upon the phytoplankton annual mean biomass are minimal.

Annual chlorophyll a, one measure of ecological status in lakes did not alter significantly as a result of any of the changes in retention time except for the extremely short retention times in the lake simulations.

Mixed reservoirs were relatively insensitive to changes in retention time because the limiting factor for phytoplankton growth under these conditions was light availability, and flushing loss and nutrient supply were less important. Although many months were sensitive to changes in retention times, it appears that June could possibly be the most sensitive month, particularly at longer retention times. Cyanobacteria dominance was most responsive at this time and under those conditions.

When setting MEP and GEP for the phytoplankton biological quality element in reservoirs, the metrics used will be broadly similar to those required to set HES and GES in natural lakes.

Key words: Phytoplankton, retention time, PROTECH, inflow

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1. INTRODUCTION

3.1 The concept of retention time

There are many factors that can influence the growth of phytoplankton, but the importance of the rate of flow through a given water body has been rarely examined. More specifically, there is little understanding how a given lake phytoplankton population may be sensitive to changes in this through flow.

The rate of flow through a water body is usually expressed as its retention time (Reynolds, 1984) and is estimated annually to be the time taken (in days) for the volume of the water body to be completely replaced. The main importance of retention time on a water system is in determining the duration that nutrients are available and consequently, its effect is greatly dependent on the nutrient load entering the water body. A secondary consideration is that if the retention time is low (i.e. flow is high) then there is a flushing effect imposed upon the planktonic flora and fauna. Therefore, given the increasing demands on water use, it is important to understand if changes in through flow may be detrimental to a water body and possibly adversely affect its ecosystem and water quality.

3.1 The PROTECH model

PROTECH (Phytoplankton RespOnses To Environmental CHange); Reynolds et al. 2001) is a process-based computer model with a proven record of capturing the dynamics of phytoplankton populations in a variety of lakes and reservoirs throughout the world (Lewis et al., 2001; Elliott & Thackeray, 2004; Elliott et al., 2005). It simulates the simultaneous daily growth of up to eight different species types that are selected from a parameter library of over 100 species. The net growth of these species is dependent on their resource base (nutrient and light availability) and the water temperature, coupled with loss processes such as zooplankton grazing, flushing and settling out of the water column. For this study, the key factors would be changes in nutrient supply and flushing loss. For a more detailed discussion of the model see Reynolds et al. (2001).

3.1 The study overview

In order to investigate the potential effects of varying retention times, four water bodies were modelled by the phytoplankton community model, PROTECH; two were natural lakes and the others were reservoirs. A number of different years were simulated and validated against observed data for the four water bodies. These simulations were then re-run with their respective inflow/outflow rates modified to cover a range of 0.1 – 5 times the observed daily flows. This flow modification took two forms: firstly, only the flow was increased or decreased which meant that the nutrient loading to the lake/reservoir also changed because the observed nutrient concentration associated with flow was not corrected. Thus a doubling of the inflow rate would lead to twice the nutrients entering the system, a situation representative of a lake with only diffuse nutrient sources. The second method of flow modification corrected this associated nutrient concentration so that the daily nutrient load did not change. Comparisons between these different simulations were then made with particular interest being focused on points of sudden changes in mean annual phytoplankton biomass and relative cyanobacteria abundance.

2. METHODS

2.1 Validation

The lakes studied were Bassenthwaite Lake and Blelham Tarn. Bassenthwaite Lake is shallow, with a mean depth of 5.3 m, a maximum depth of 19.0 m and a volume of 27.9 Mm^3 (Ramsbottom, 1976). According to WFD typology it is a moderate alkalinity, shallow lake. For this study two years were simulated (1996 and 2000). These years represented the highest (28 d, 1996) and lowest (13 d, 2000) annual retention times recorded for the period where data were available. The species simulated in these simulations were: *Chlorella*, *Stephanodiscus hantzschii* Grun., *Stephanodiscus astraea* (Ehrenb.) Grun., *Anabaena*, *Fragilaria*, *Asterionella*, *Planktothrix* and *Aphanizomenon*. The model captured well the seasonal phytoplankton dynamics of the two years (Fig. 1). Further simulations were conducted for Bassenthwaite Lake, using a continuous 20 year scenario driven by observed inflow data where available and meteorological data calibrated from a regional climate model for present day climate (Elliott *et al.* 2005). These data were used because they were readily available for the lake from a previous study and provided more variation in the driving data so that some idea of the uncertainty associated with the single year simulations could be assessed. The simulated mean fortnightly total chlorophyll from this 20 year run validated well with the observed fortnightly means (Fig. 2).

Blelham Tarn is a small lake with a mean depth of 6.8 m, maximum depth of 14.5 m and volume of 0.69 Mm^3 (Ramsbottom, 1976). This is also a moderate alkalinity, shallow lake. A good set of driving data were available for 1974 and were used as the basis for the simulations; the annual retention time was calculated to be 34 d for this year. The species simulated were: *Anabaena*, *Aphanizomenon*, *Ceratium*, *Cryptomonas*, *Asterionella*, *Fragilaria*, *Planktothrix* and *Stephanodiscus hantzschii*. Again, there was good agreement between the simulated and observed total chlorophyll *a* (Fig. 3).

Two reservoirs were also modelled for this study, the first was based on Rutland Water and the second was a generic reservoir system typical of the south-east of England. Rutland Water is a large reservoir that, when filled to capacity, is 32.0 m deep and has a volume of 136.6 Mm^3 and according to WFD typology is a high alkalinity shallow lake. Data were available for 1993 (annual retention time of 827 d) and PROTECH captured well the observed total chlorophyll *a* (Fig. 4). The species simulated were: *Anabaena*, *Rhodomonas*, *Oocystis*, *Asterionella, Microcystis, Aphanizomenon*, *Stephanodiscus astraea* and *Stephanodiscus hantzschii*.

The second reservoir was given the characteristics of a highly eutrophic reservoir in the south-east of England (likely to be typed as a high alkalinity, shallow lake). When this reservoir (herein referred to as SE Reservoir) is at fully capacity, it has a maximum depth of 20.0 m and volume of 55.7 Mm^3, with an annual retention time of 445 d. No validation data were available because the reservoir does not exist but the simulation certainly shows the typical seasonal phytoplankton patterns observed in reservoirs in this area of the UK (Fig. 5). The species simulated were: *Chlorella*, *Stephanodiscus hantzschii*, *Rhodomonas*, *Microcystis*, *Oscillatoria*, *Anabaena*, *Melosira* and *Chlamydomonas*.

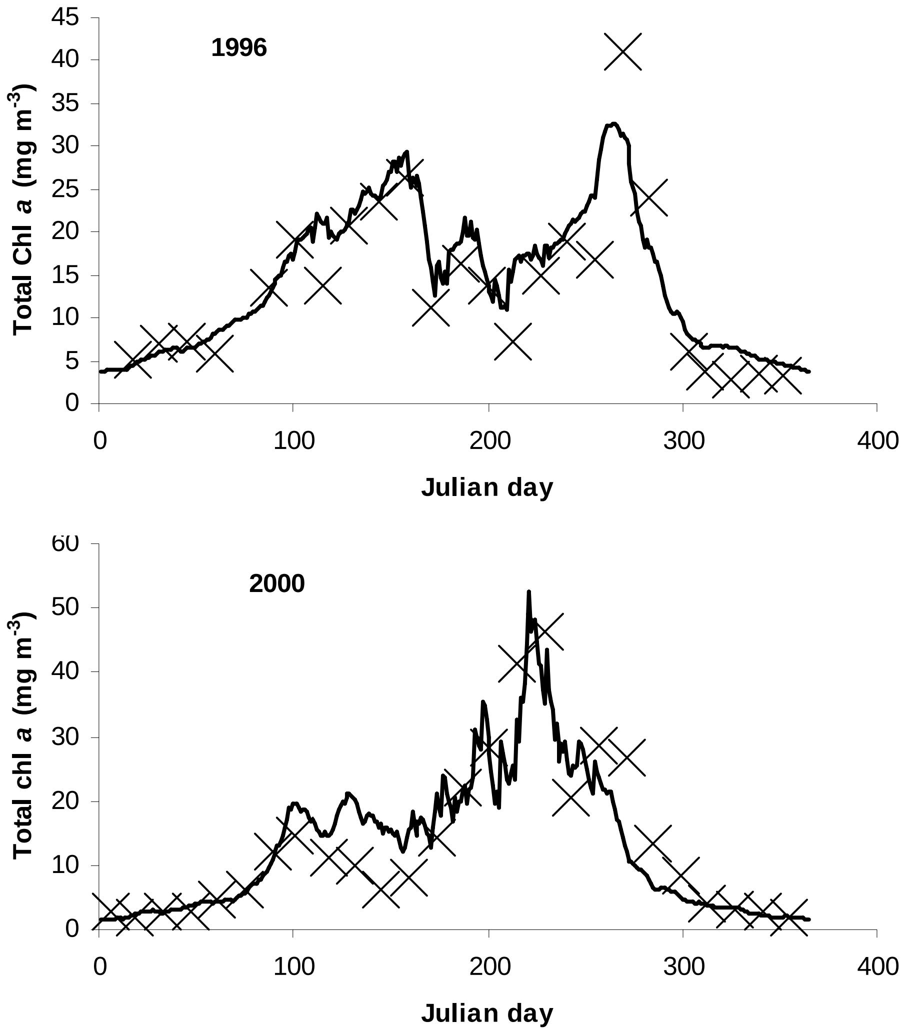


Fig. 1. The total chlorophyll *a* validation runs (solid line) compared to the observed data (crosses) for 1996 and 2000 in Bassenthwaite Lake.

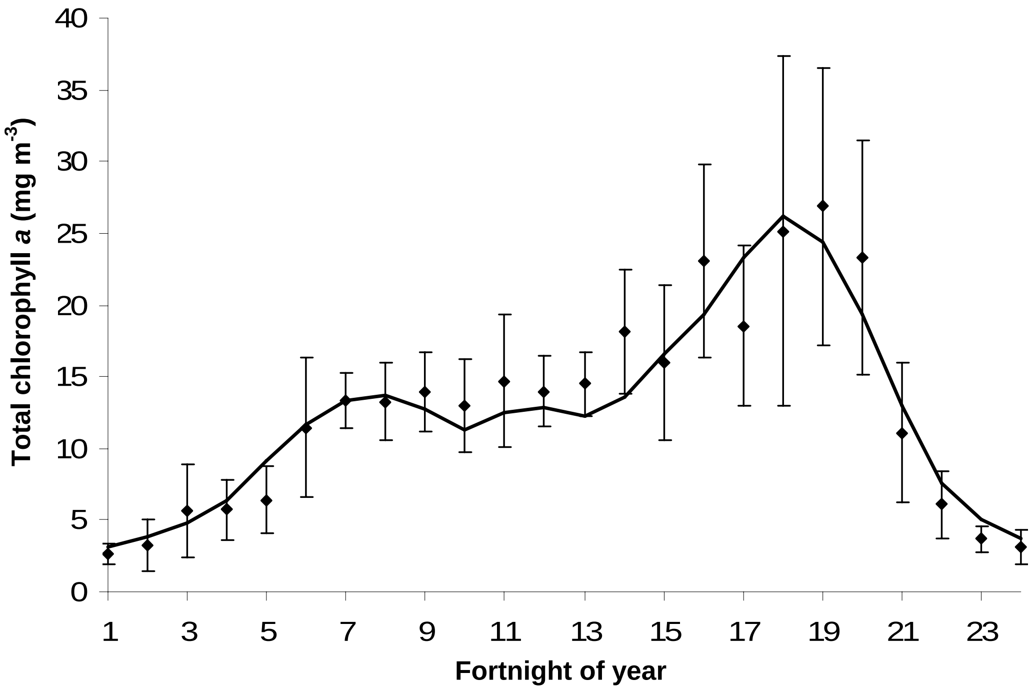


Fig. 2. Comparison between observed mean fortnightly total chlorophyll (1991-2002) and simulated (20 year run) for Bassenthwaite Lake..

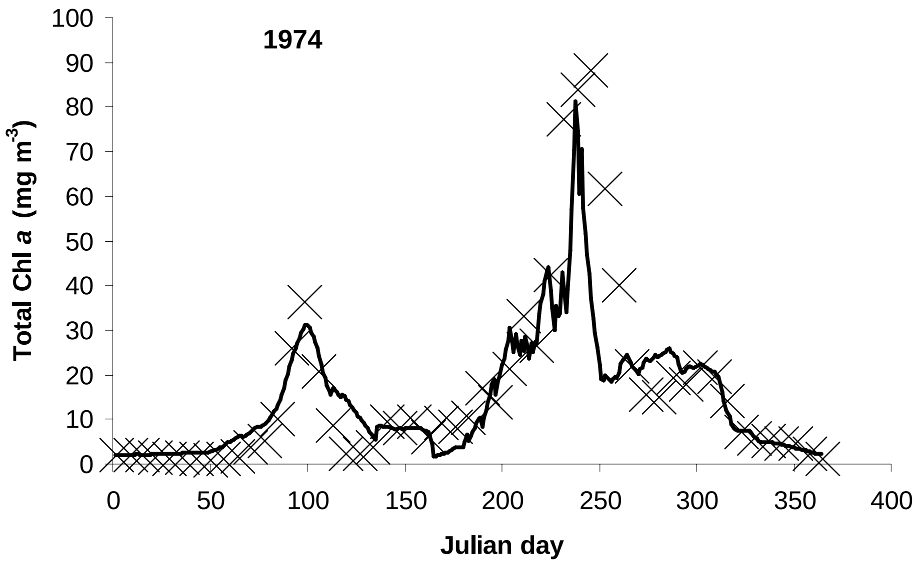


Fig. 3. The total chlorophyll *a* validation run (solid line) compared to the observed data (crosses) for 1974 in Blelham Tarn.

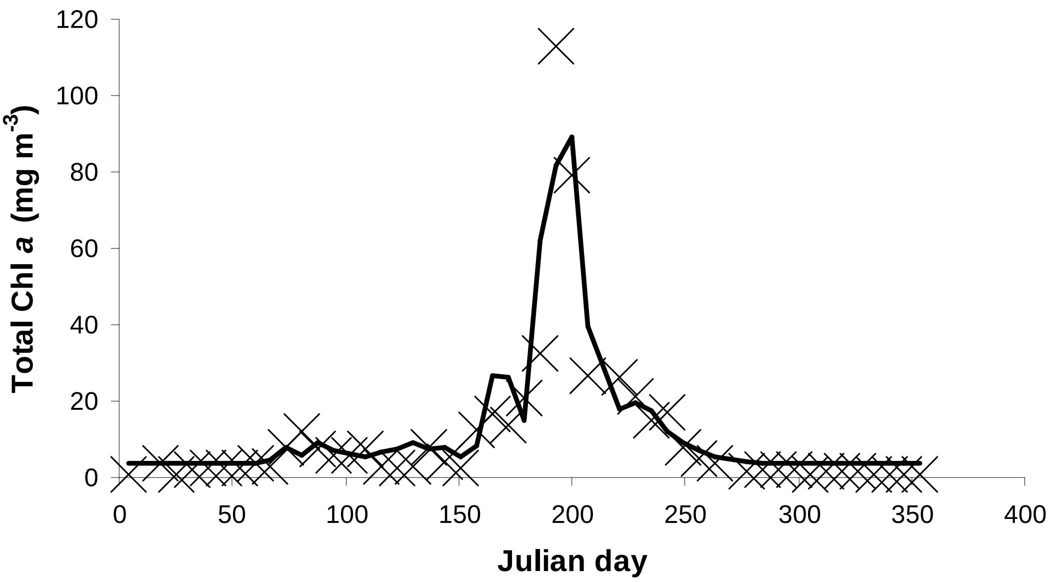


Fig. 4. The total chlorophyll *a* validation run (solid line) compared to the observed data (crosses) for 1993 in Rutland Water.

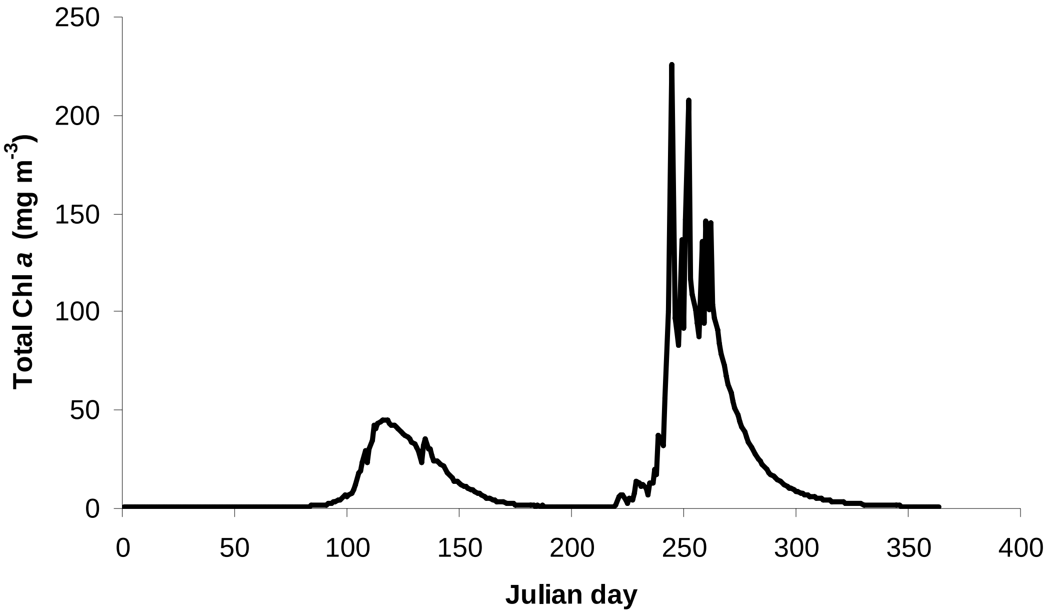


Fig. 5. The total chlorophyll *a* validation run (solid line) for the SE Reservoir.

2.2 Methodology for varying retention time

The validation simulations presented above were perturbed to produce a wide range of retention times (Table 1). The method of flow variation took two forms: either simply multiplying the flow by a factor ("No Corr Load") or by also correcting the nutrient concentration in the inflow ("Corr Load") so that the nutrient load to the water body remained the same as in the original validation run. The natural equivalents of these two methods would be a change in flow for a lake that receives all its nutrients from diffuse inputs ("No Corr Load") or from point sources ("Corr Load"). Clearly most water bodies will lie somewhere between these two extremes. For the two reservoir simulations, additional runs were made with the effect of artificial mixing included.

Table 1. The original retention time and the range produced by the variations to the flow.

Water body	Retention time (d)	Retention time range (d)
Bassenthwaite Lake (1996)	28.3	3-283
Bassenthwaite Lake (2000)	13.1	1-264
Blelham Tarn	33.9	7-338
Rutland Water	826.5	165-8265
SE Reservoir	444.7	89-4447

The simulations were analysed for any changes in annual mean phytoplankton biomass and cyanobacteria abundance and, for the single year simulations, April-September monthly means of phytoplankton biomass and cyanobacteria abundance were also recorded.

3. RESULTS

The results are presented to show the effects of altered retention times on annual chlorophyll and annual cyanobacterial abundance and a breakdown by month showing seasonal sensitivities. On the figures the original retention time is marked and the retention time associated with a 20% decrease in the inflow (equivalent to a 25% increase in residence time). Such a reduction in riverine flow is thought to be able to support good ecological status in river systems (WFD48 report, 2006).

3.1 Bassenthwaite Lake 1996 (low flow year) - No Corr Load

There was little change in total chlorophyll with a change in retention time except with small retention times which caused a slight decrease in the original level of chlorophyll produced by the lake (Fig. 6a). The cyanobacteria abundance also demonstrated the same response (Fig. 6b).

At the monthly resolution, there was a small change in the spring and summer months presented, with a sharp decline at the short retention times and a smooth decline at the higher times. April was the only month that had a slight increase in chlorophyll when retention time ranged between 30 – 110 days (Fig. 7a). Cyanobacteria abundance declined sharply with decreasing retention time for all the months except April and there was little change with longer retention times (Fig. 7c).

3.2 Bassenthwaite Lake 1996 (low flow year) - Corr Load

Under this scenario, increased retention time caused a steady increase in mean annual chlorophyll that levelled off after approximately 70 days (Fig. 6a). A decrease in retention time caused a decrease in chlorophyll and this pattern was also reflected in the abundance of cyanobacteria (Fig. 6b).

All of the spring and summer months had the same general pattern of increasing biomass with longer retention times, with May proving to be the most sensitive month (Fig. 7b). Again, this general pattern was reflected by the cyanobacteria, but both May and June were the most sensitive months (Fig. 7d).

3.3 Bassenthwaite Lake 2000 (high flow year) - No Corr Load

Following a similar pattern to 1996, total chlorophyll and cyanobacteria abundance decreased sharply with short retention times and changed little with longer times (Fig. 8). There was little difference between the monthly patterns except for August which had a marked increase in total chlorophyll between 50-140 days (Fig. 9a); there was no corresponding change in cyanobacteria abundance (Fig. 9c).

3.4 Bassenthwaite Lake 2000 (high flow year) - Corr Load

This variation of the 2000 scenario produced a slight variation to the pattern described above because total chlorophyll did increase with longer retention times (Fig. 8a). The monthly patterns were similar to each other and showed the same trend of increasing total chlorophyll with increasing retention time (Fig. 9d). In terms of cyanobacteria abundance, June had the greatest increase with retention times but all had formed a plateau at times longer then 60 days (Fig. 9d).

3.5 Bassenthwaite Lake 20 year run – No Corr Load

The results of the 20 year run showed a very similar response to the single year runs, with little change at high retention times to the total chlorophyll or cyanobacteria abundance (Fig. 10a and

Fig. 11a). The variation in these results was fairly similar across the range of retention times with a slight increase in the limits of variation with longer retention times.

3.6 Bassenthwaite Lake 20 year run – Corr Load

Like the single year runs, this scenario caused a sharp increase in total chlorophyll with increasing retention time (Fig. 10b and Fig. 11b). Cyanobacteria also showed the same trend but was less severe and similar to the response produced by the No Corr Load scenario. Interestingly, the level of variation in the annual means increased markedly with longer retention times but still followed the upward trend.

3.7 Blelham Tarn - No Corr Load

The general responses of total chlorophyll and cyanobacteria abundance to the changes in retention times were similar to those modelled in Bassenthwaite Lake but were less smooth (Fig. 12). There were abrupt changes in chlorophyll between retention times of 50-110 days, suggesting that this period is particularly sensitive but the overall trend was similar to that in Bassenthwaite Lake.

For total chlorophyll and cyanobacteria, most months showed little response to retention times except for total chlorophyll in August which mimicked the annual trend (Fig. 13).

3.8 Blelham Tarn - Corr Load

This scenario produced very similar trends to the No Corr Loads scenario, although there was dampening of the chlorophyll change between 50-110 days (Fig. 12). Monthly responses were greater in September for total chlorophyll and showed a marked decline in cyanobacteria abundance in July with longer retention times (Fig. 13).

3.9 Rutland Water - No Corr Load

In this very long retention time system, total chlorophyll and cyanobacteria abundance changed little in the middle range times but showed a slight increase with longer times and a very sharp increase of nearly 7 mg m^{-3} at the shortest times (Fig. 14). Monthly analysis showed that the many changes in total chlorophyll were occurring in August and September (Fig. 15a) whereas the largest changes for cyanobacteria happened in June and September (Fig. 15c).

3.10 Rutland Water - Corr Load

There was little difference between the above scenario with the exception of the short retention time simulations which did not produce increased amounts of chlorophyll (Fig. 14). Monthly patterns were similar to the No Corr Load scenario but in July both total chlorophyll and cyanobacteria abundance were more erratic in their response to changing retention times (15b, d).

3.11 Rutland Water with artificial mixing - No Corr Load

Under the influence of artificial mixers, changes in retention time produced little effect of on total chlorophyll and, apart from a slight decline at short retention times, on cyanobacteria abundance (Fig. 16). The monthly responses were generally similar to each other, showing a gradual rise in total chlorophyll with longer retention times (Fig. 17a). Cyanobacteria abundance was generally unresponsive except for in June where short times led to a marked decrease and longer times to a more rapid increase.

3.12 Rutland Water with artificial mixing - Corr Load

The total chlorophyll and cyanobacteria abundance responses were identical to those in the No Corr Load scenario (Fig. 16). Monthly responses were also identical (Fig. 17).

3.13 SE Reservoir - No Corr Load

Total chlorophyll changed little except at short retention times where it spiked in a way similar to that simulated in Rutland Water (Fig. 18a). Cyanobacteria abundance also spiked and, at longer retention times, increased slighty before reaching a plateau (Fig. 18b).

July and August appeared to be the months most sensitive to short retention times and showed the same spiked pattern seen in the total chlorophyll (Fig. 19a). For the cyanobacteria, June and July appeared to be the most sensitive months (Fig. 19c).

3.14 SE Reservoir - Corr Load

The spike in total chlorophyll at short retention times was missing and, in fact, a decrease was simulated along with a slight increase at longer retention times (Fig. 18a). The cyanobacteria did still spike but otherwise showed little change at other retention times (Fig. 18b).

August and September were the most sensitive months for total chlorophyll with both increasing at longer retention times and the latter falling at short times (Fig. 19b). The cyanobacteria abundance followed a pattern similar to the No Corr Load scenario (Fig. 19).

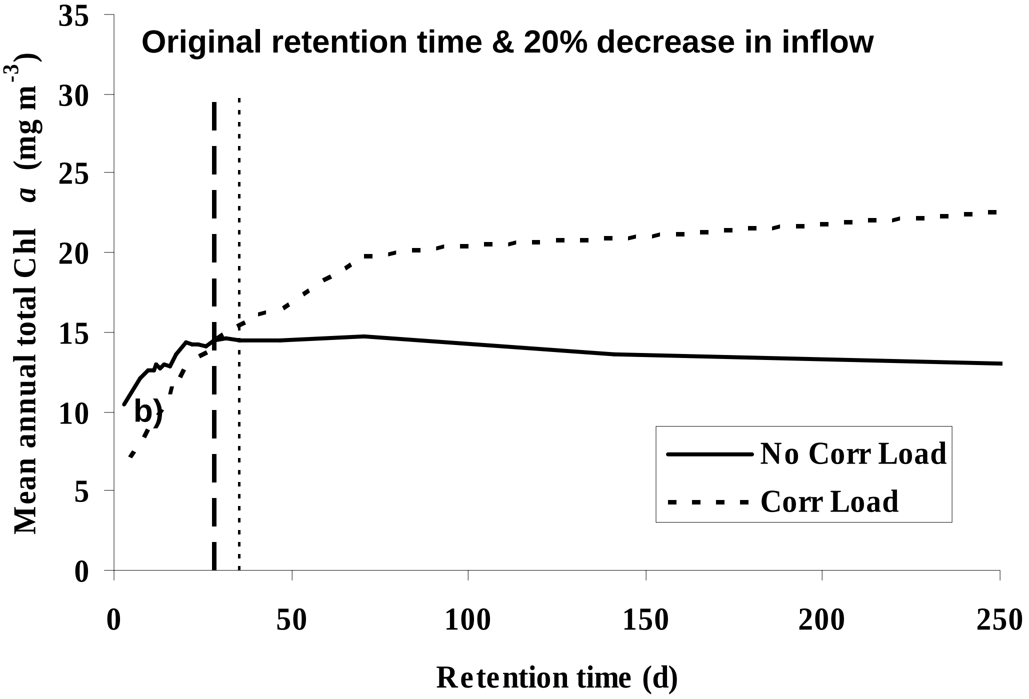
3.15 SE Reservoir with artificial mixing - No Corr Load

There was no response in total chlorophyll to changes in retention time but the abundance of cyanobacteria did decline at short retention times (Fig. 20). There were few months that showed any clear sensitivity and any changes were very small (Fig. 21a). Cyanobacteria abundance was more sensitive with most months showing a decline at short retention times and a slight increase at longer times (Fig. 21c).

3.16 SE Reservoir with artificial mixing - Corr Load

Apart from a slight increase in total chlorophyll at retention times greater than 500 days, there was little change (Fig. 20a). This was contrasted with the marked increase in cyanobacteria dominance at these longer retention times (Fig. 20b). Most months changed little, but June and July showed the greatest sensitivity in total chlorophyll (Fig. 21b). The increase in cyanobacteria with longer retention times was expressed over all six months analysed and expressed similar patterns with May and June being the most responsive (Fig. 21d).

a)



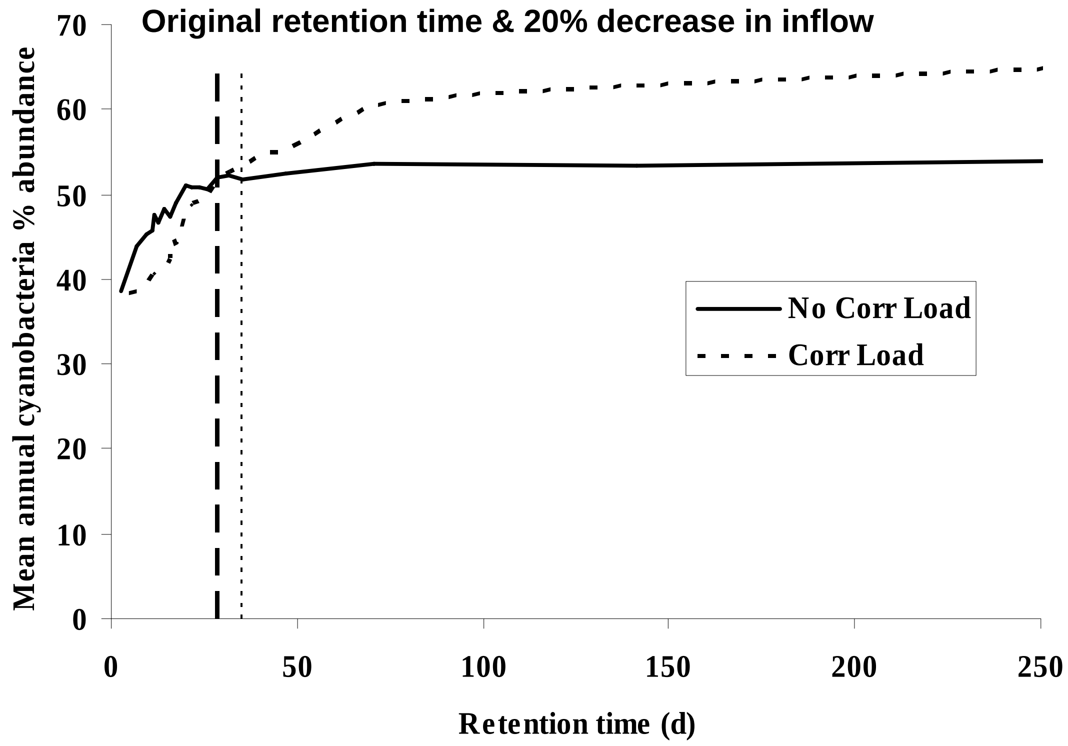


Fig. 6 Comparisons between the two methods of flow change for Bassenthwaite Lake 1996. a) Mean annual chlorophyll *a* biomass (mg m^{-3}); b) Mean annual cyanobacteria percentage abundance. Note: No Corr Load describes the method where flow was simply multiplied and Corr Load the method that corrected the inflow concentrations.

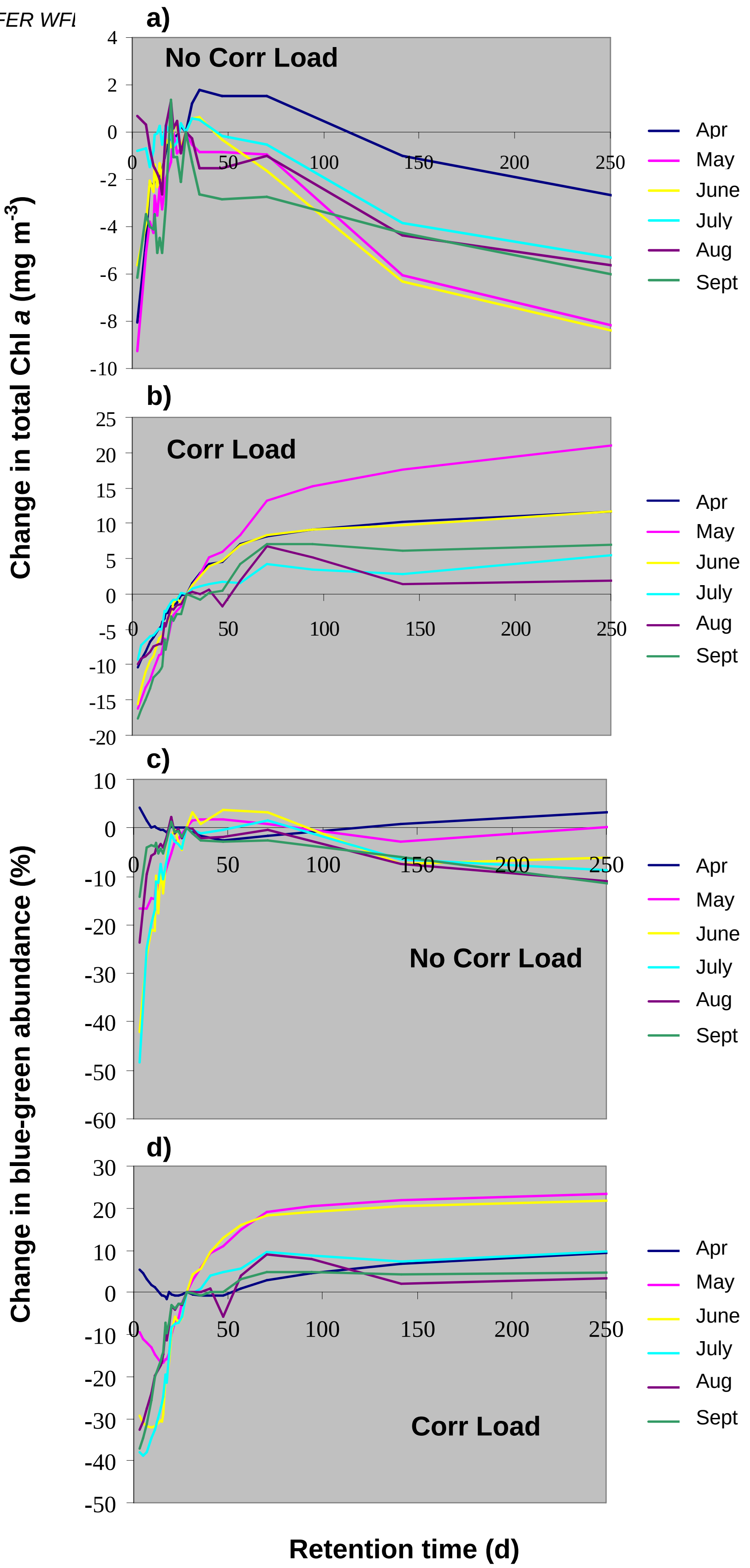


Fig. 7 Comparisons for Bassenthwaite Lake 1996. a & b) Absolute change in monthly mean chlorophyll *a* biomass (mg m^{-3}) and; c & d) monthly cyanobacteria percentage abundance for April-September for both flow methods.

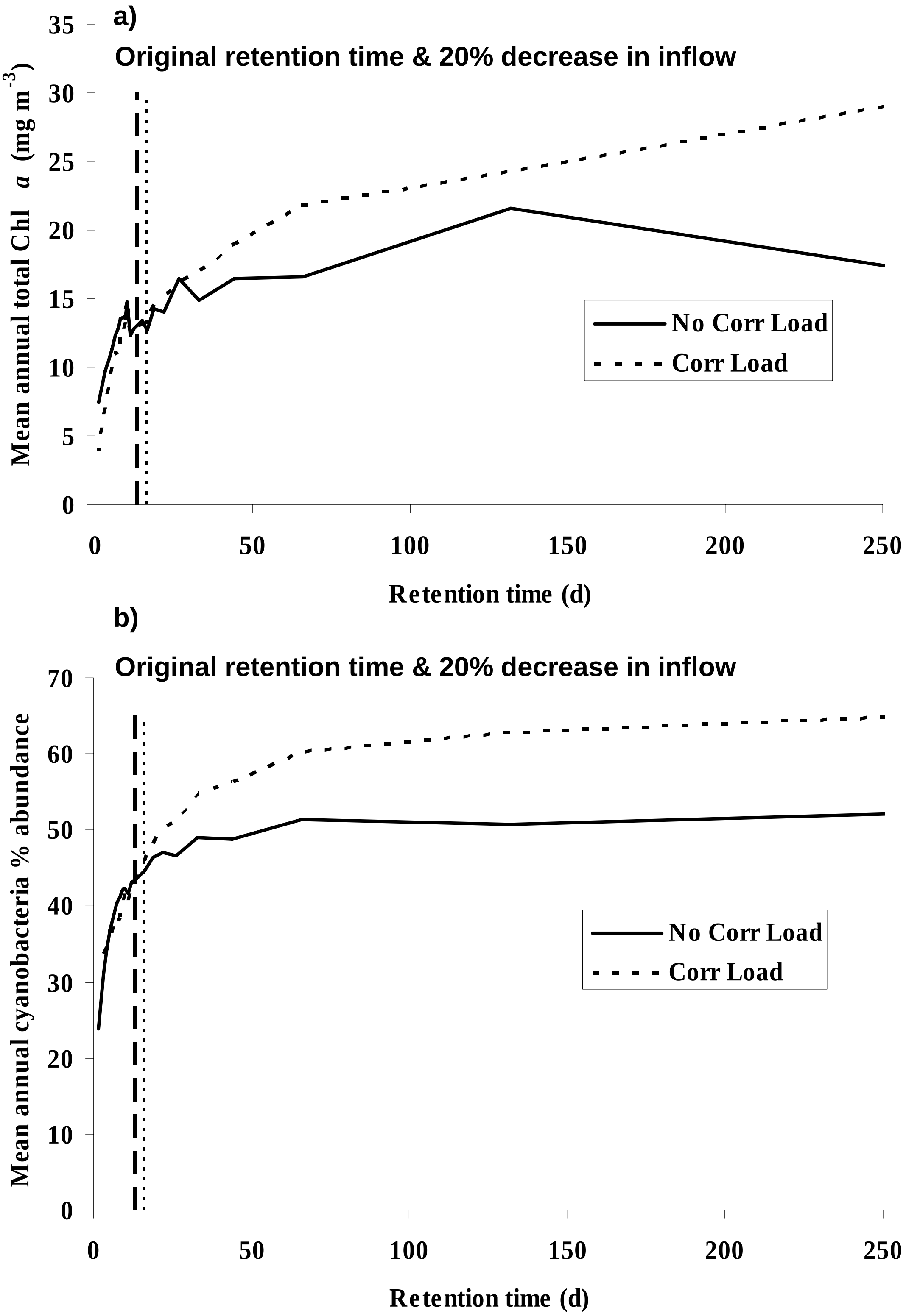


Fig. 8 Comparisons between the two methods of flow change for Bassenthwaite Lake 2000. a) Mean annual chlorophyll *a* biomass (mg m^{-3}); b) Mean annual cyanobacteria percentage abundance. Note: No Corr Load describes the method where flow was simply multiplied and Corr Load the method that corrected the inflow concentrations.

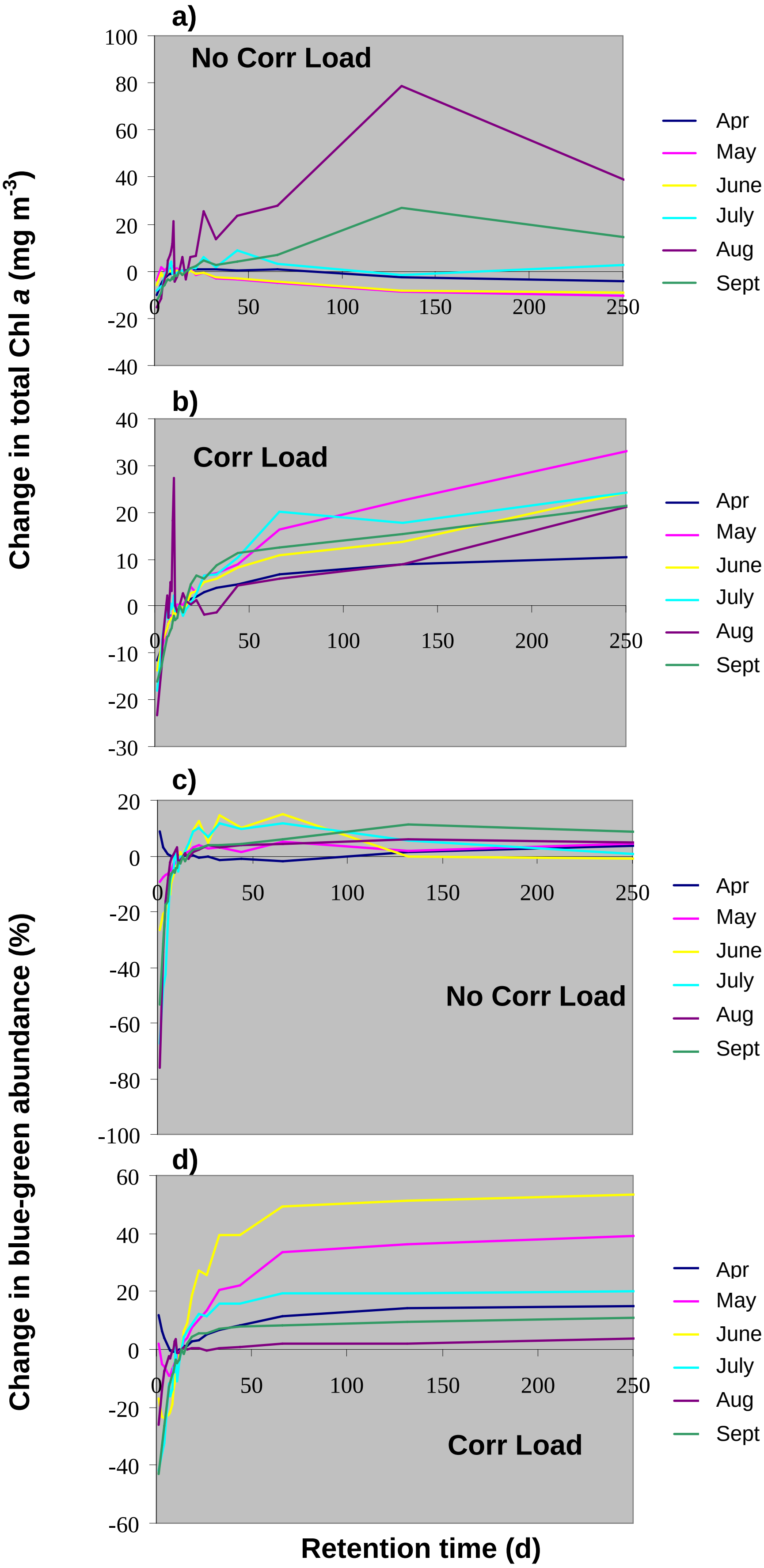
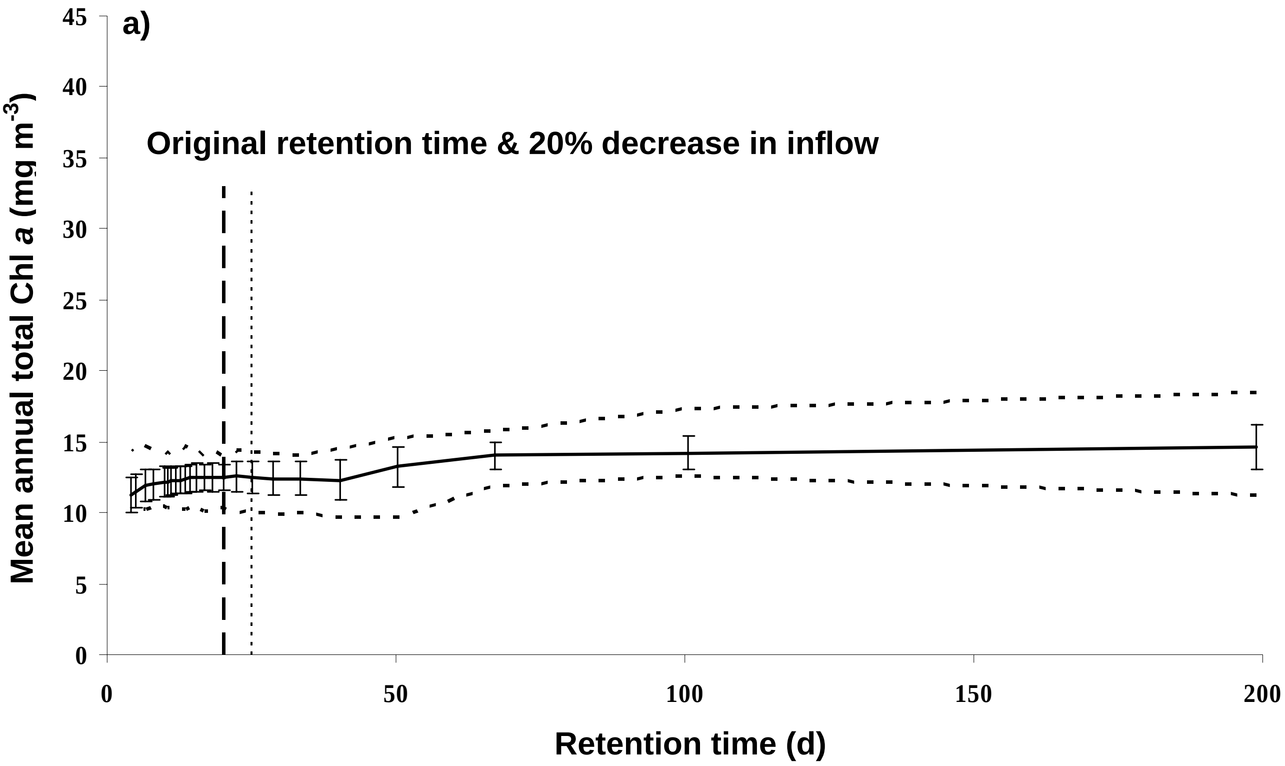


Fig. 9 Comparisons for Bassenthwaite Lake 2000. a & b) Absolute change in monthly mean chlorophyll *a* biomass (mg m^{-3}) and; c & d) monthly cyanobacteria percentage abundance for April-September for both flow methods.



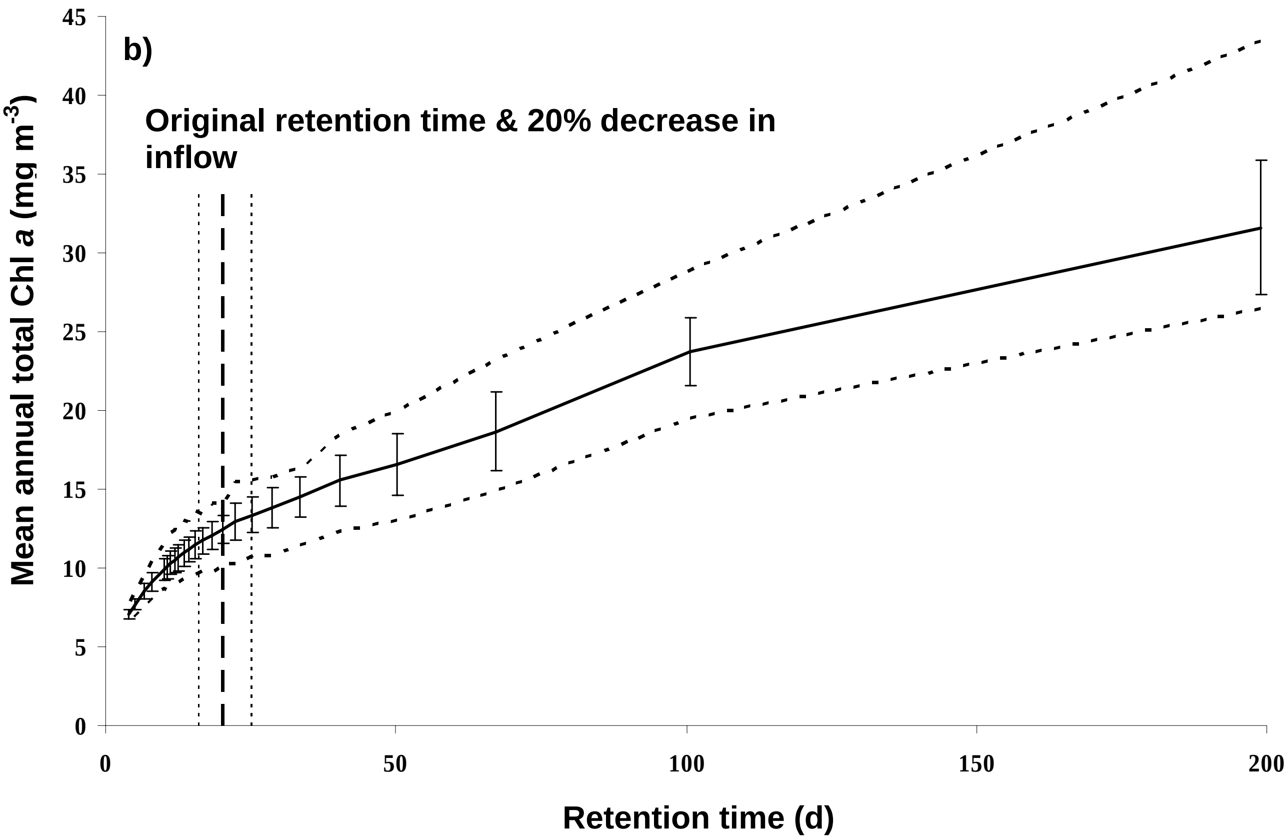


Fig. 10 Comparisons of mean annual chlorophyll *a* biomass (mg m^{-3}) between the two methods of flow change for Bassenthwaite Lake 20 year run. a) No Corr Load; b) Corr Load. Dotted lines indicate maximum and minimum values and bars show 1 standard deviation.

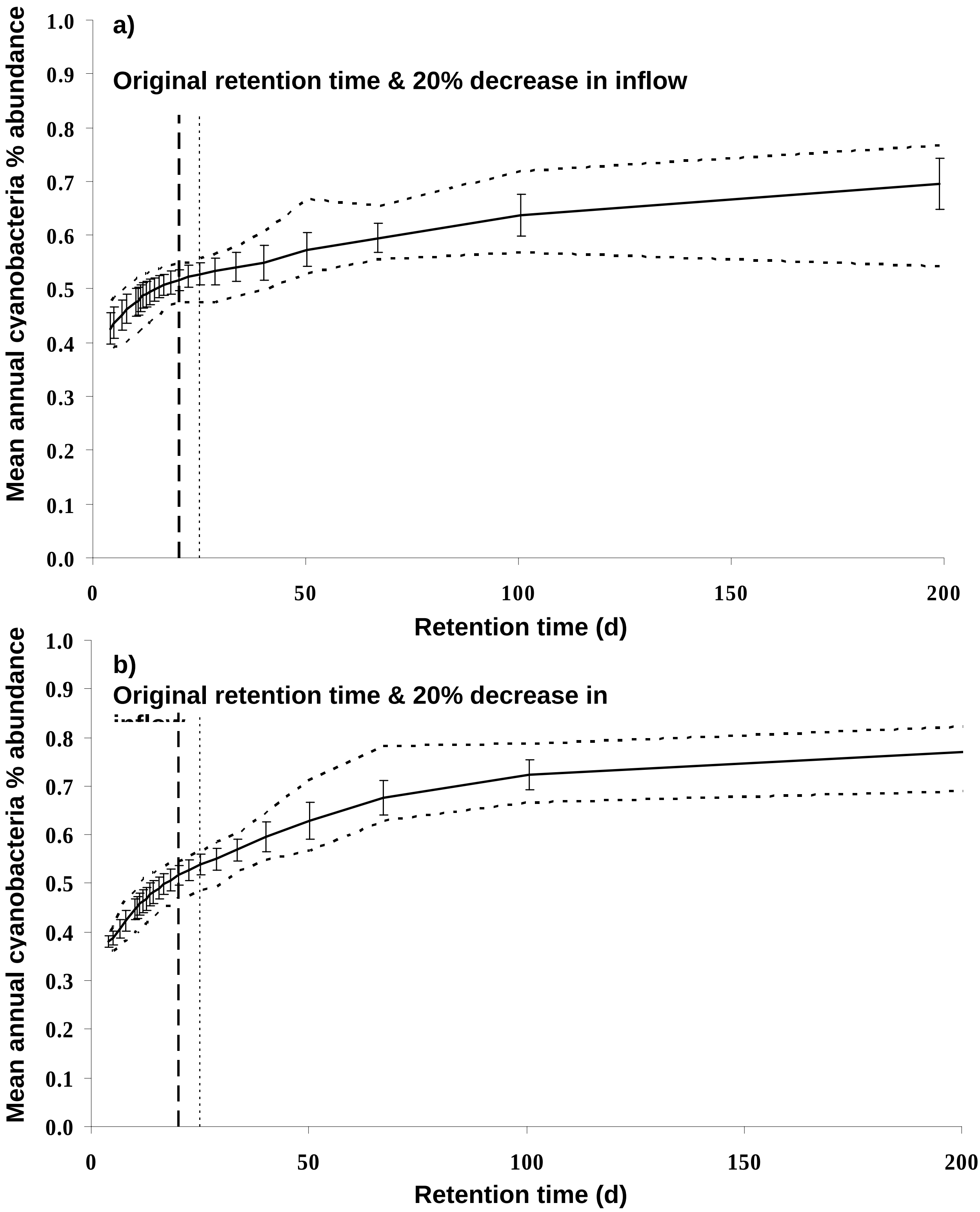
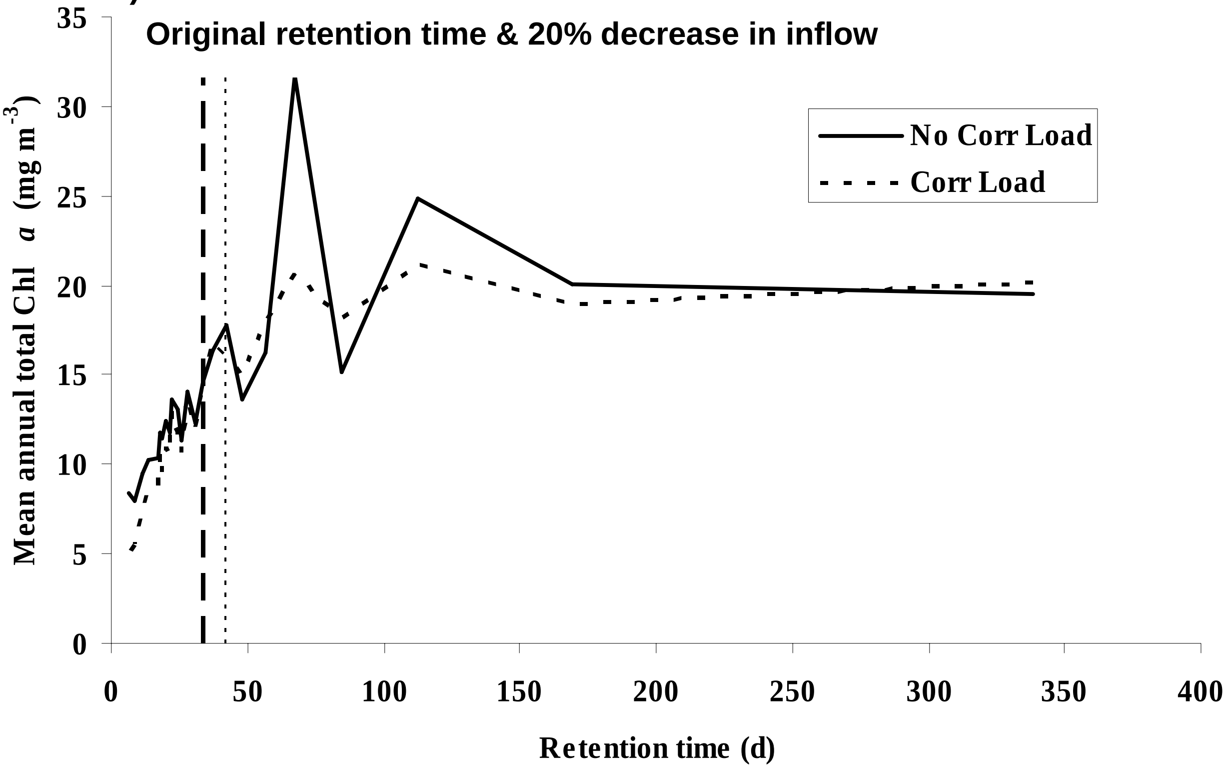


Fig. 11 Comparisons of mean annual cyanobacteria percentage abundance between the two methods of flow change for Bassenthwaite Lake 20 year run. a) No Corr Load; b) Corr Load. Dotted lines indicate maximum and minimum values and bars show 1 standard deviation.

a)



b)

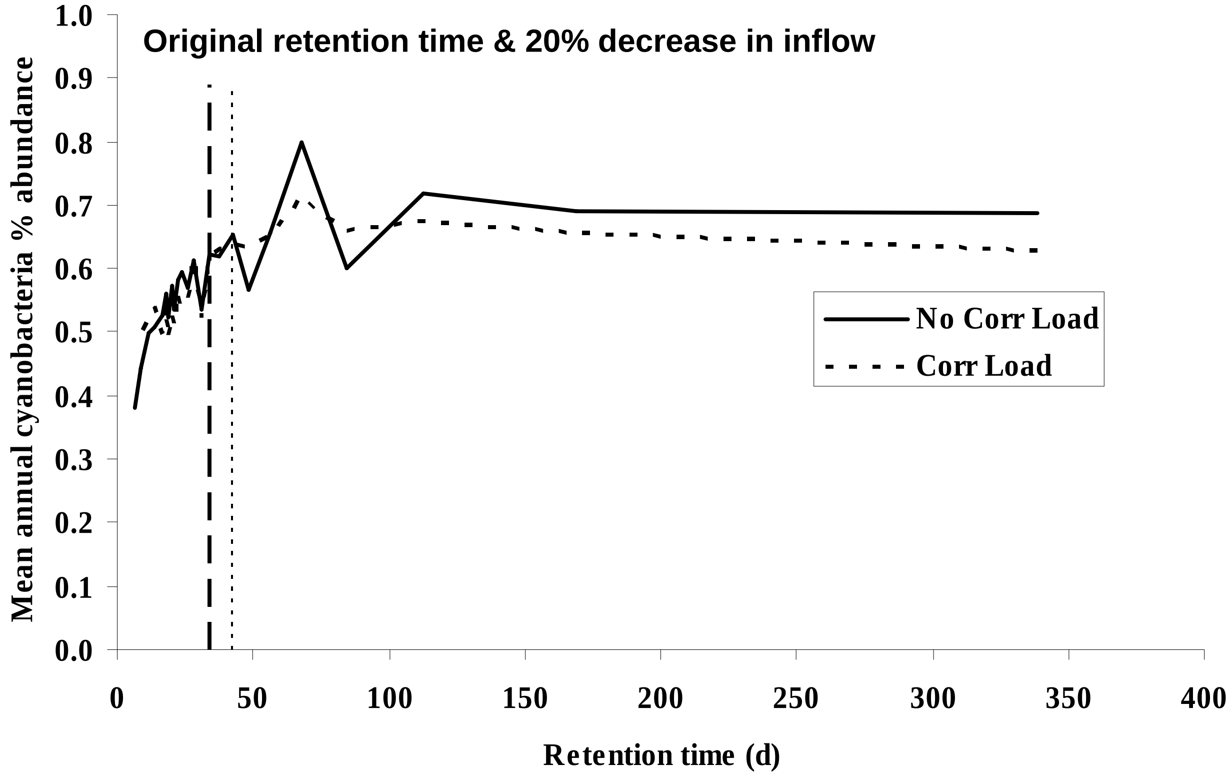


Fig. 12 Comparisons between the two methods of flow change for Blelham Tarn 1974. a) Mean annual chlorophyll *a* biomass (mg m^{-3}); b) Mean annual cyanobacteria percentage abundance. Note: No Corr Load describes the method where flow was simply multiplied and Corr Load the method that corrected the inflow concentrations.

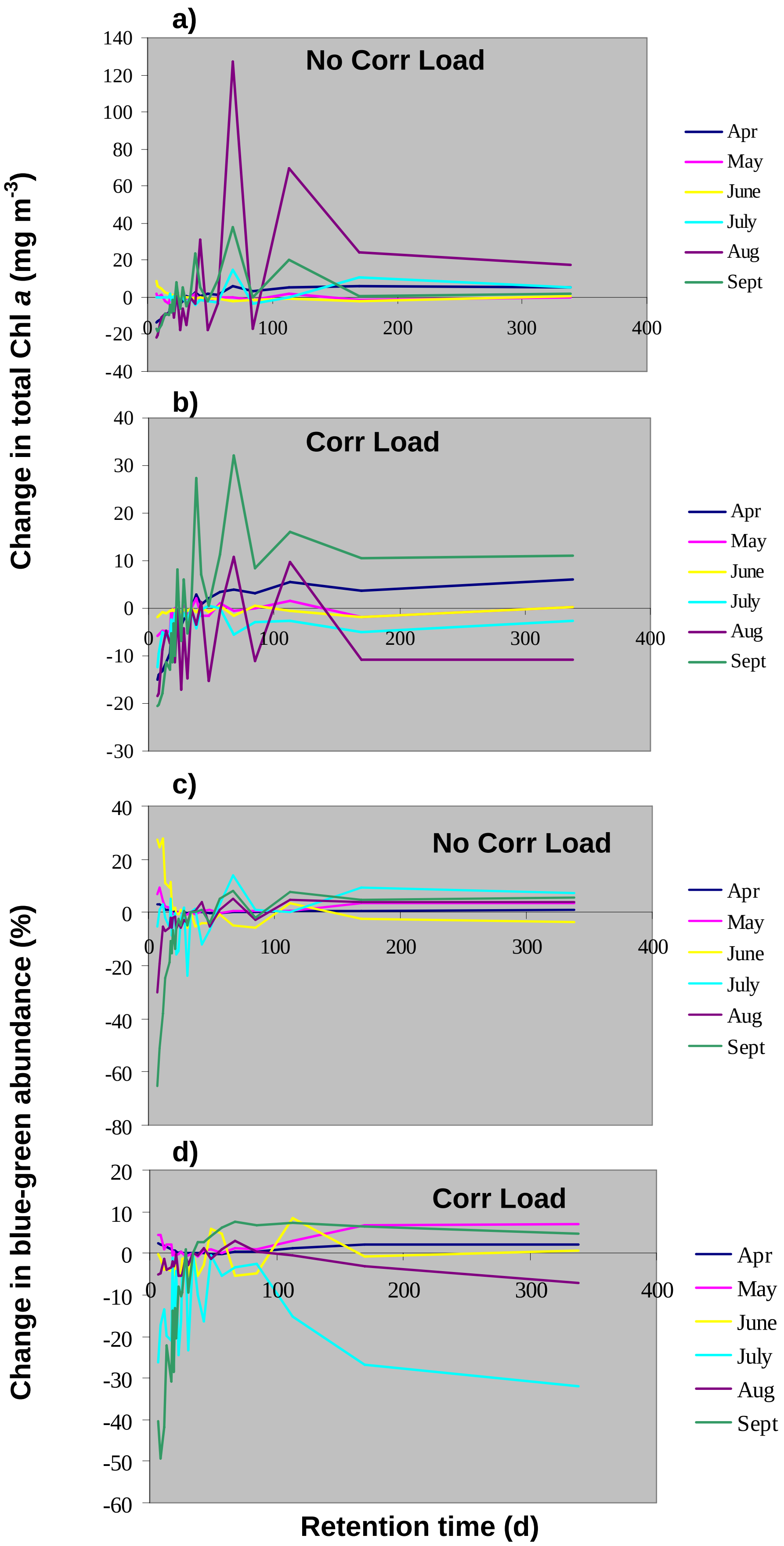
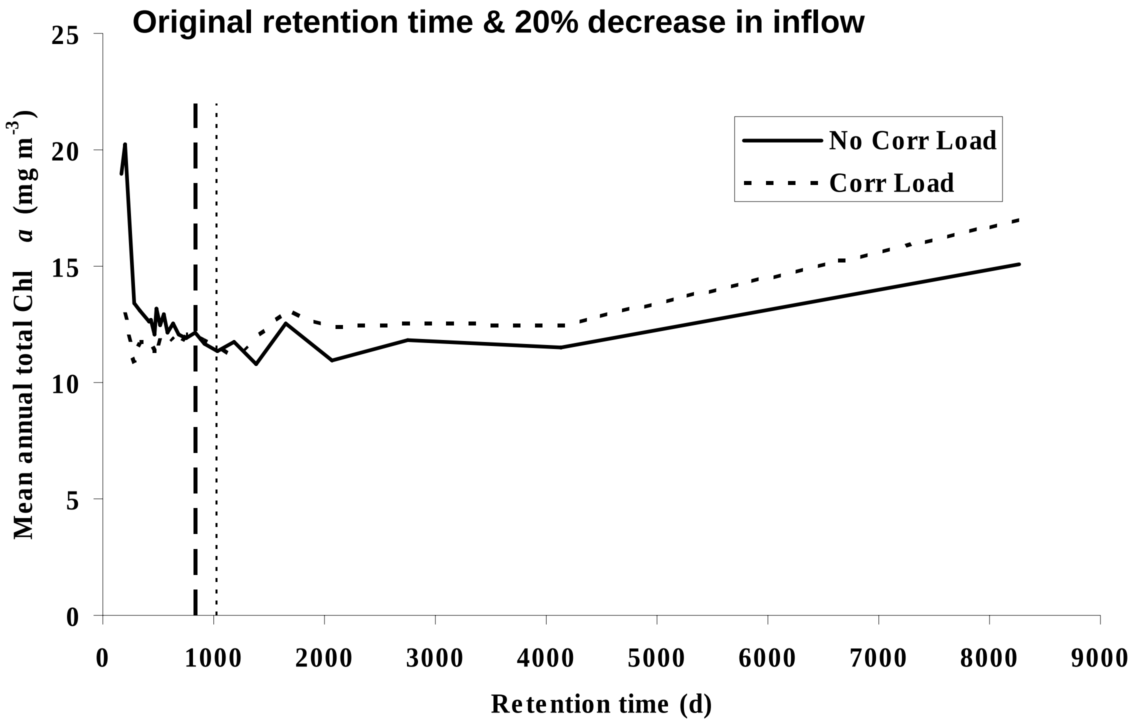


Fig. 13 Comparisons for Blelham Tarn 1974. a & b) Absolute change in monthly mean chlorophyll *a* biomass (mg m^{-3}) and; c & d) monthly cyanobacteria percentage abundance for April-September for both flow methods.

a)



b)

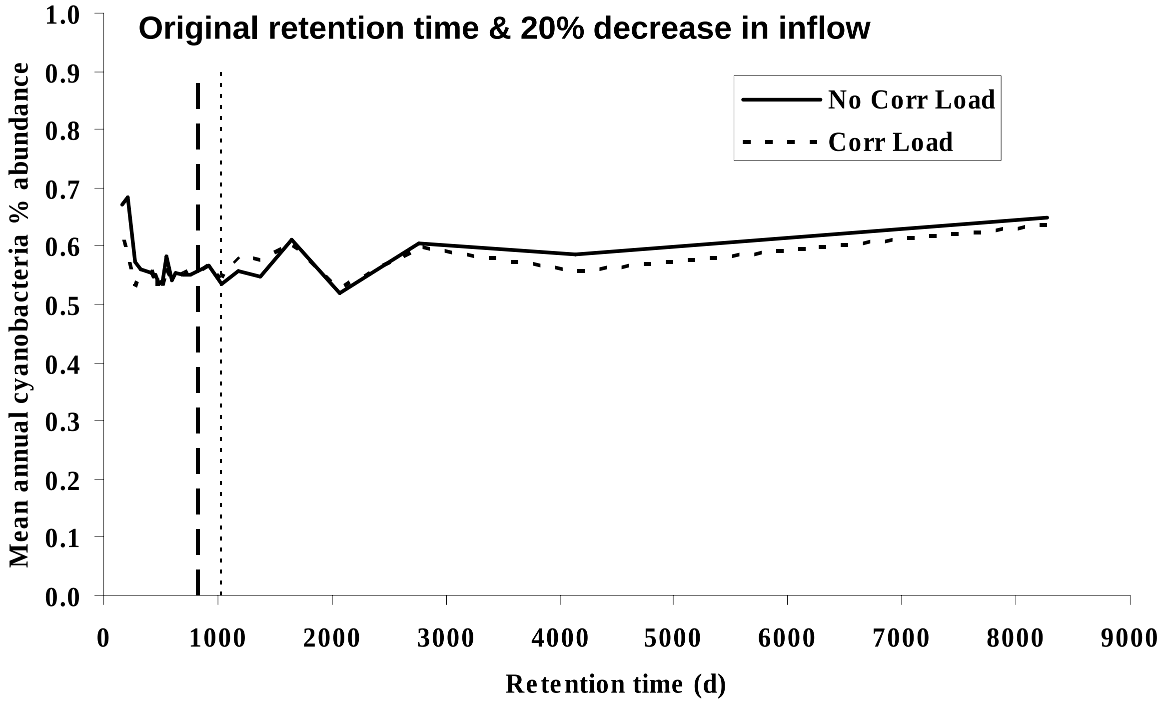


Fig. 14 Comparisons between the two methods of flow change for Rutland Water 1993. a) Mean annual chlorophyll *a* biomass (mg m^{-3}); b) Mean annual cyanobacteria percentage abundance. Note: No Corr Load describes the method where flow was simply multiplied and Corr Load the method that corrected the inflow concentrations.

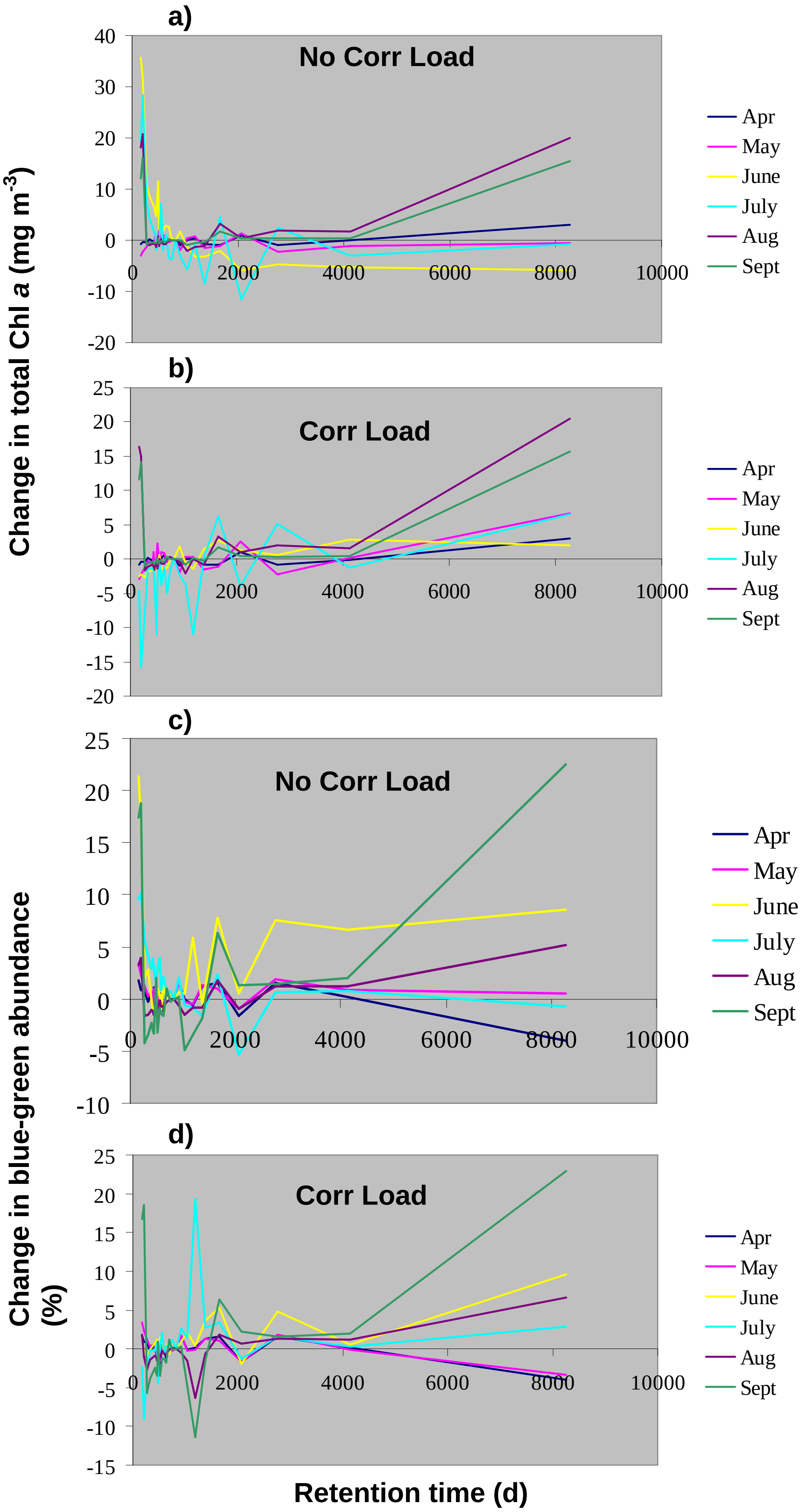


Fig. 15 Comparisons for Rutland Water 1993. a & b) Absolute change in monthly mean chlorophyll *a* biomass (mg m^{-3}) and; c & d) monthly cyanobacteria percentage abundance for April-September for both flow methods.

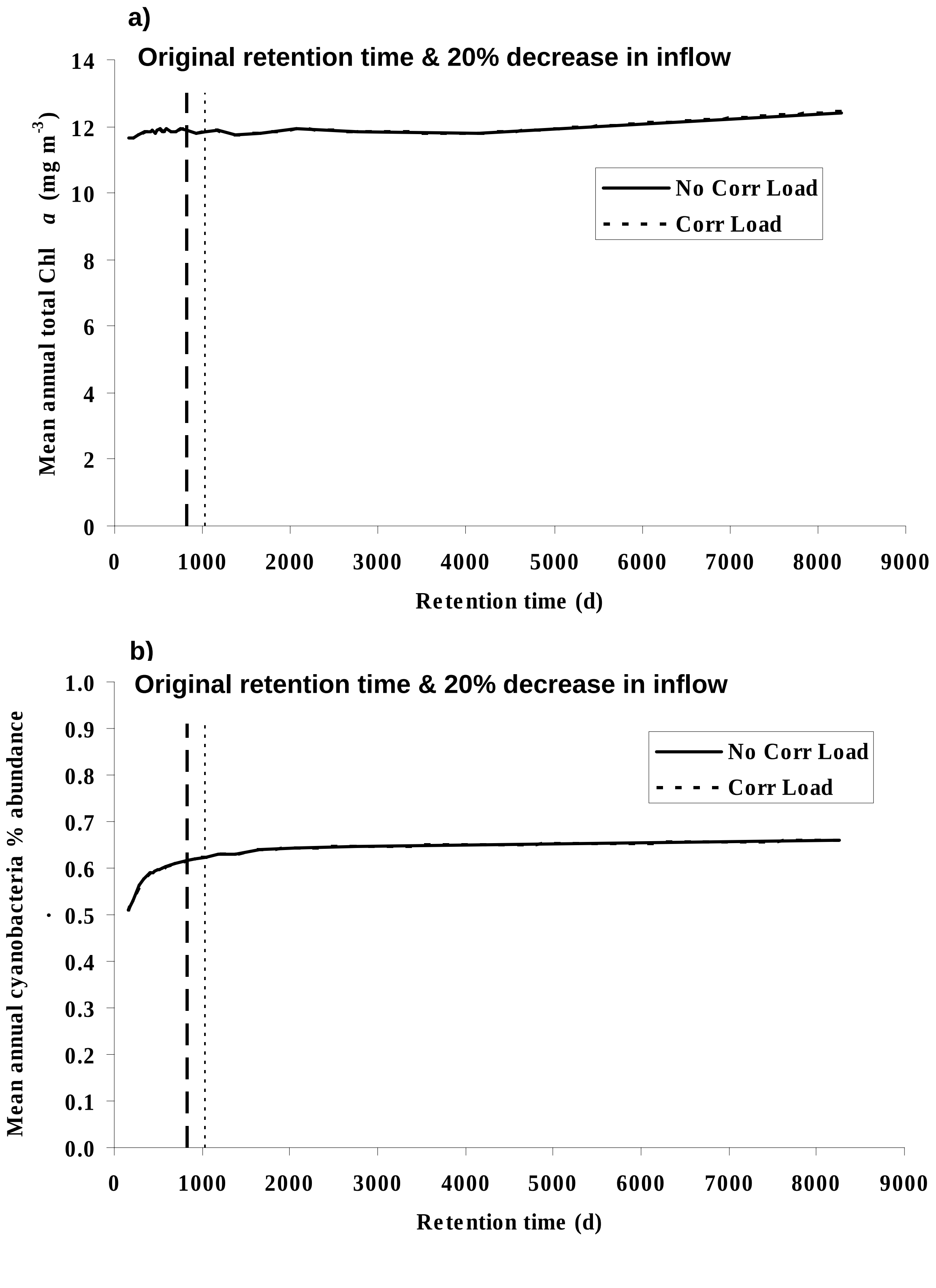


Fig. 16 Comparisons between the two methods of flow change for Rutland Water 1993 with artificial mixing simulated. a) Mean annual chlorophyll *a* biomass (mg m^{-3}); b) Mean annual cyanobacteria percentage abundance. Note: No Corr Load describes the method where flow was simply multiplied and Corr Load the method that corrected the inflow concentrations.

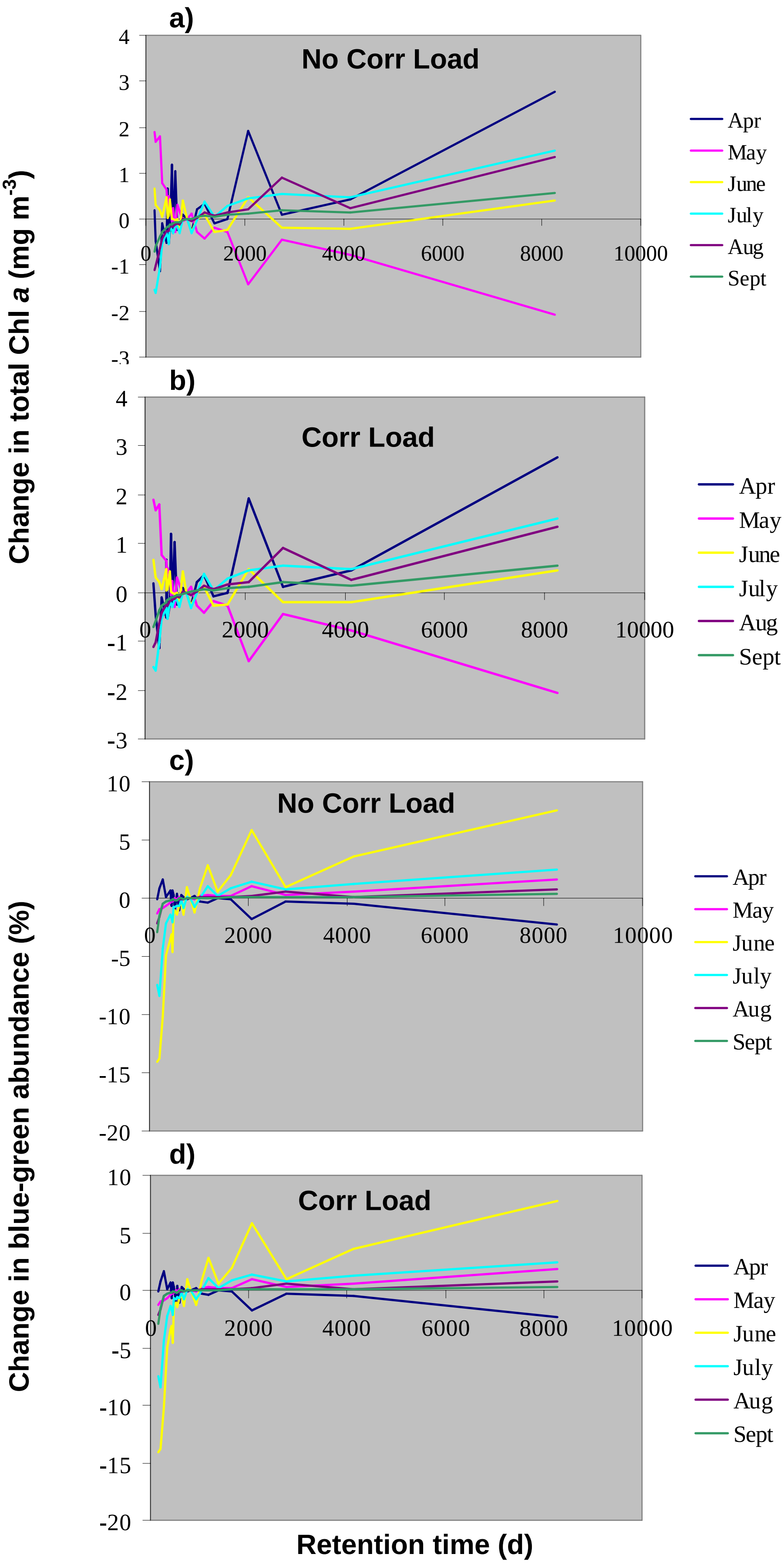


Fig. 17 Comparisons for Rutland Water 1993 with artificial mixing. a & b) Absolute change in monthly mean chlorophyll *a* biomass (mg m^{-3}) and; c & d) monthly cyanobacteria percentage abundance for April-September for both flow methods.

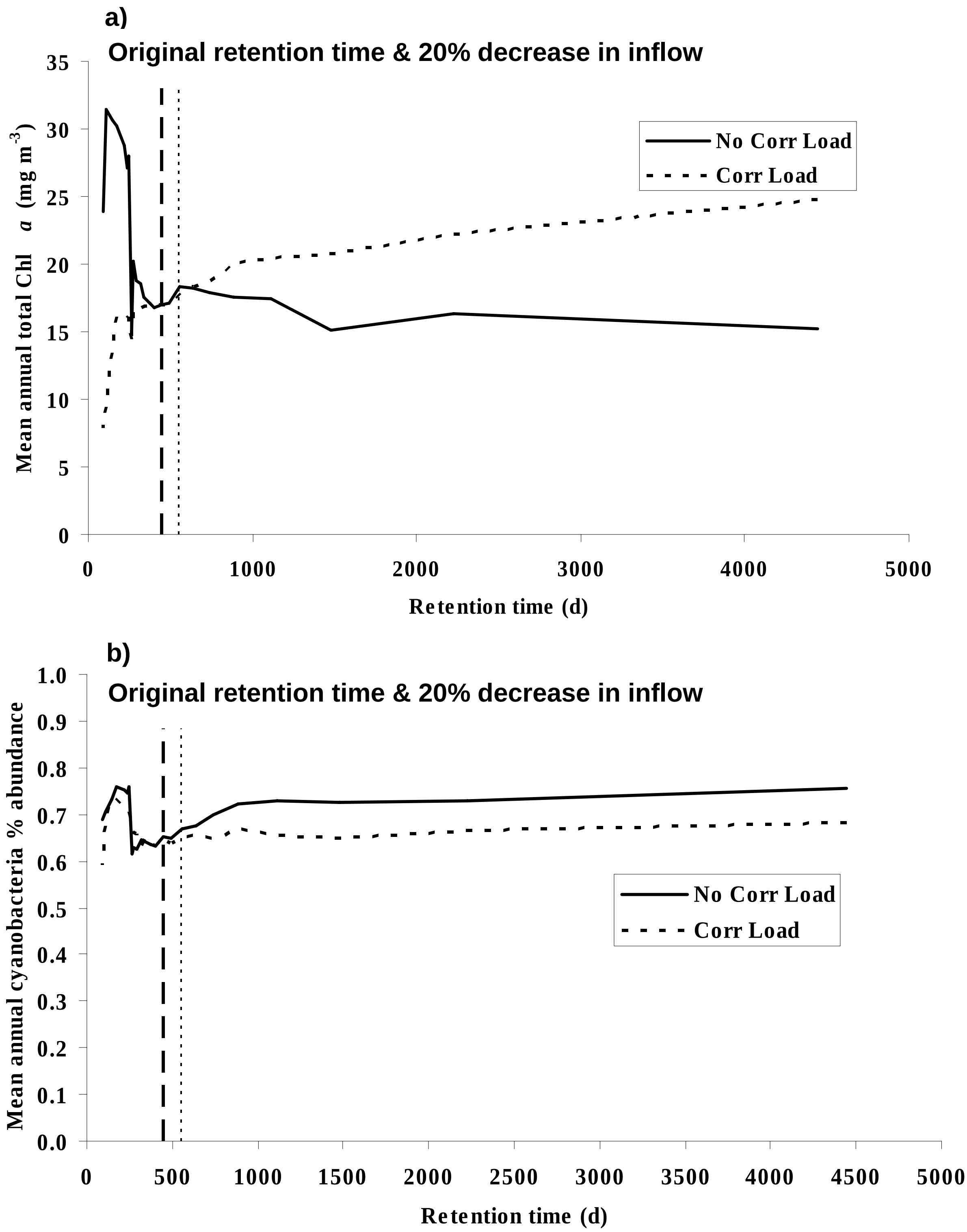


Fig. 18 Comparisons between the two methods of flow change for SE Reservoir. a) Mean annual chlorophyll *a* biomass (mg m^{-3}); b) Mean annual cyanobacteria percentage abundance. Note: No Corr Load describes the method where flow was simply multiplied and Corr Load the method that corrected the inflow concentrations.

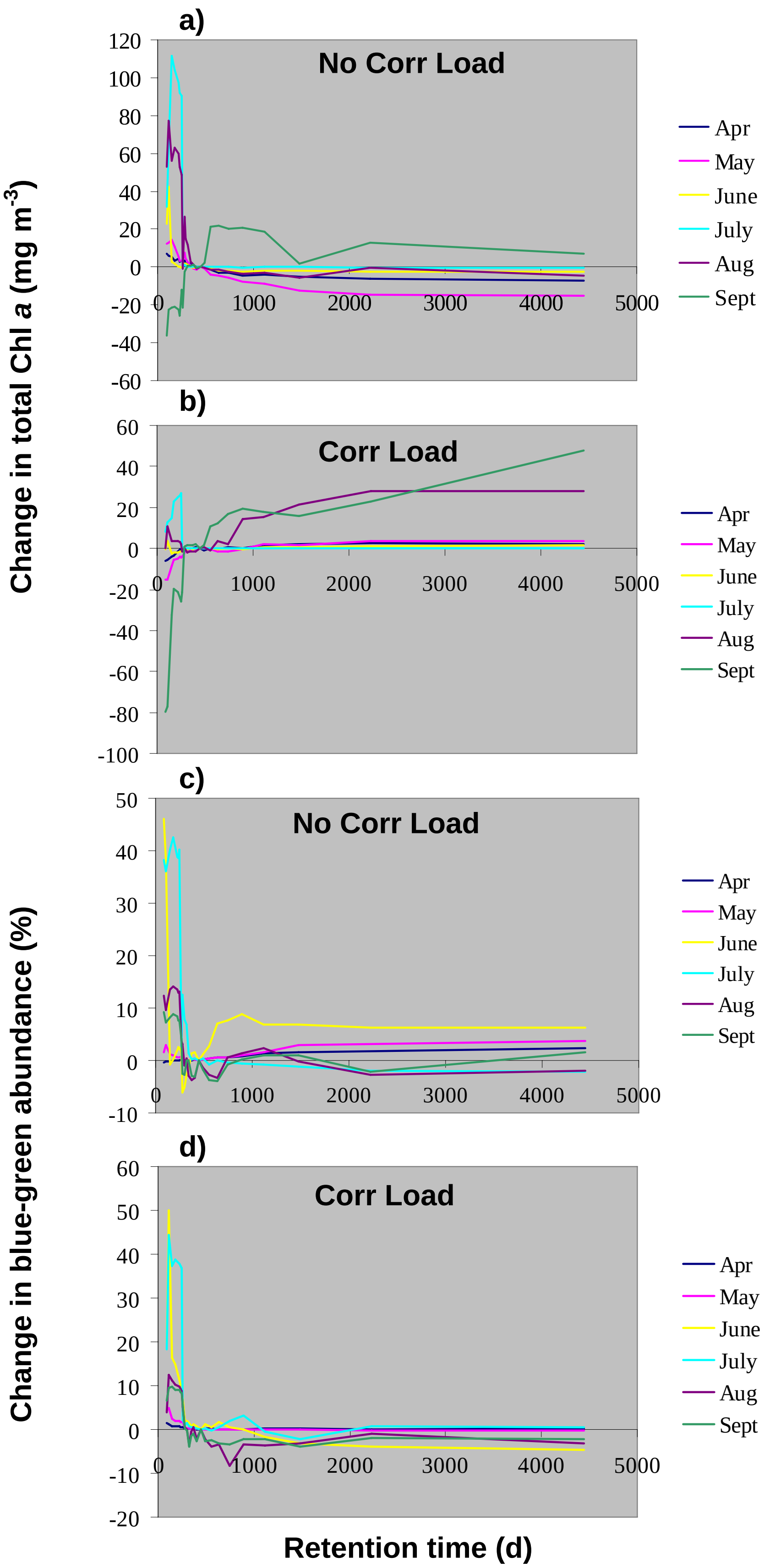


Fig. 19 Comparisons for SE Reservoir. a & b) Absolute change in monthly mean chlorophyll *a* biomass (mg m^{-3}) and; c & d) monthly cyanobacteria percentage abundance for April-September for both flow methods.

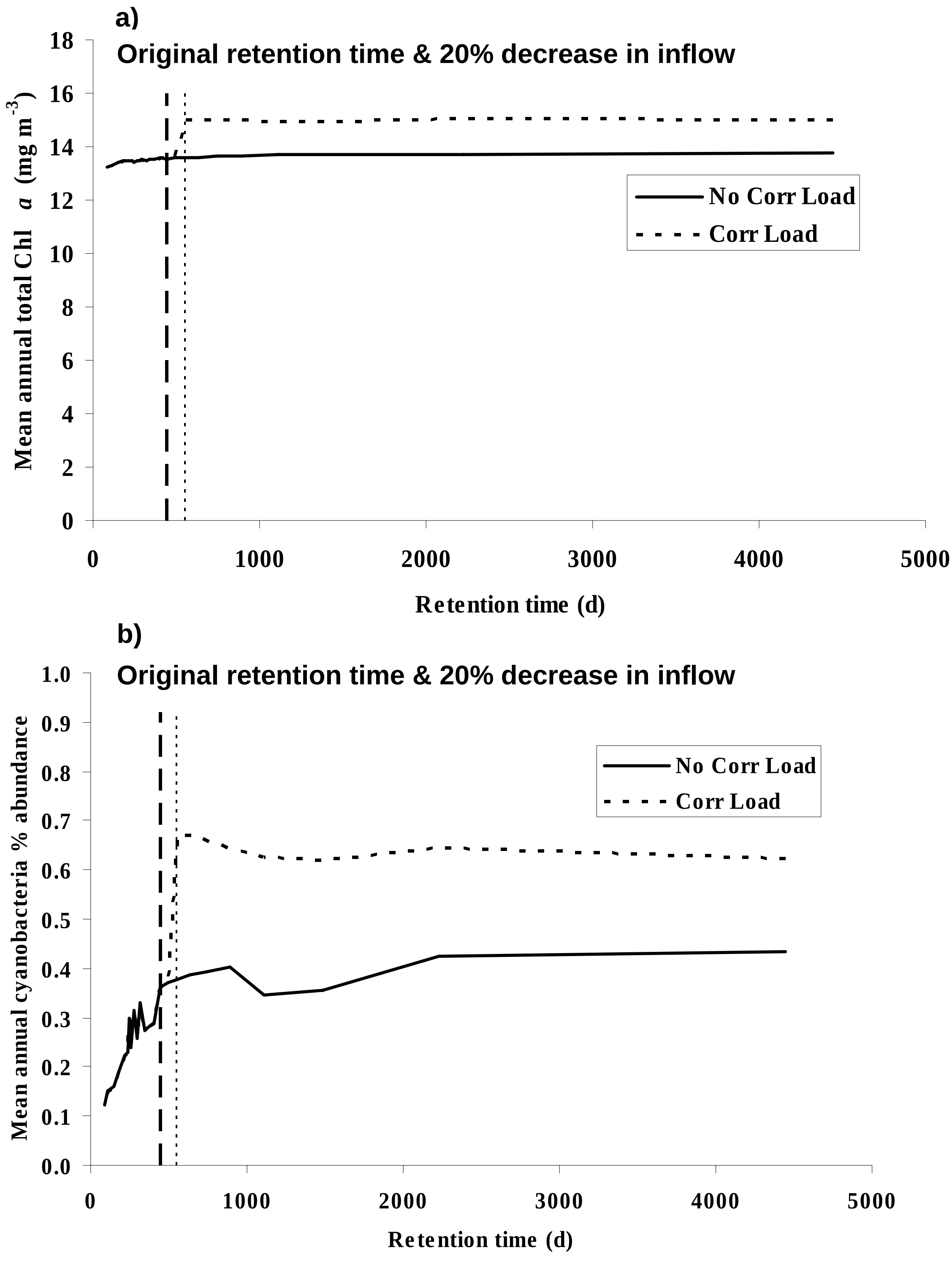


Fig. 20 Comparisons between the two methods of flow change for SE Reservoir with artificial mixing. a) Mean annual chlorophyll *a* biomass (mg m^{-3}); b) Mean annual cyanobacteria percentage abundance. Note: No Corr Load describes the method where flow was simply multiplied and Corr Load the method that corrected the inflow concentrations.

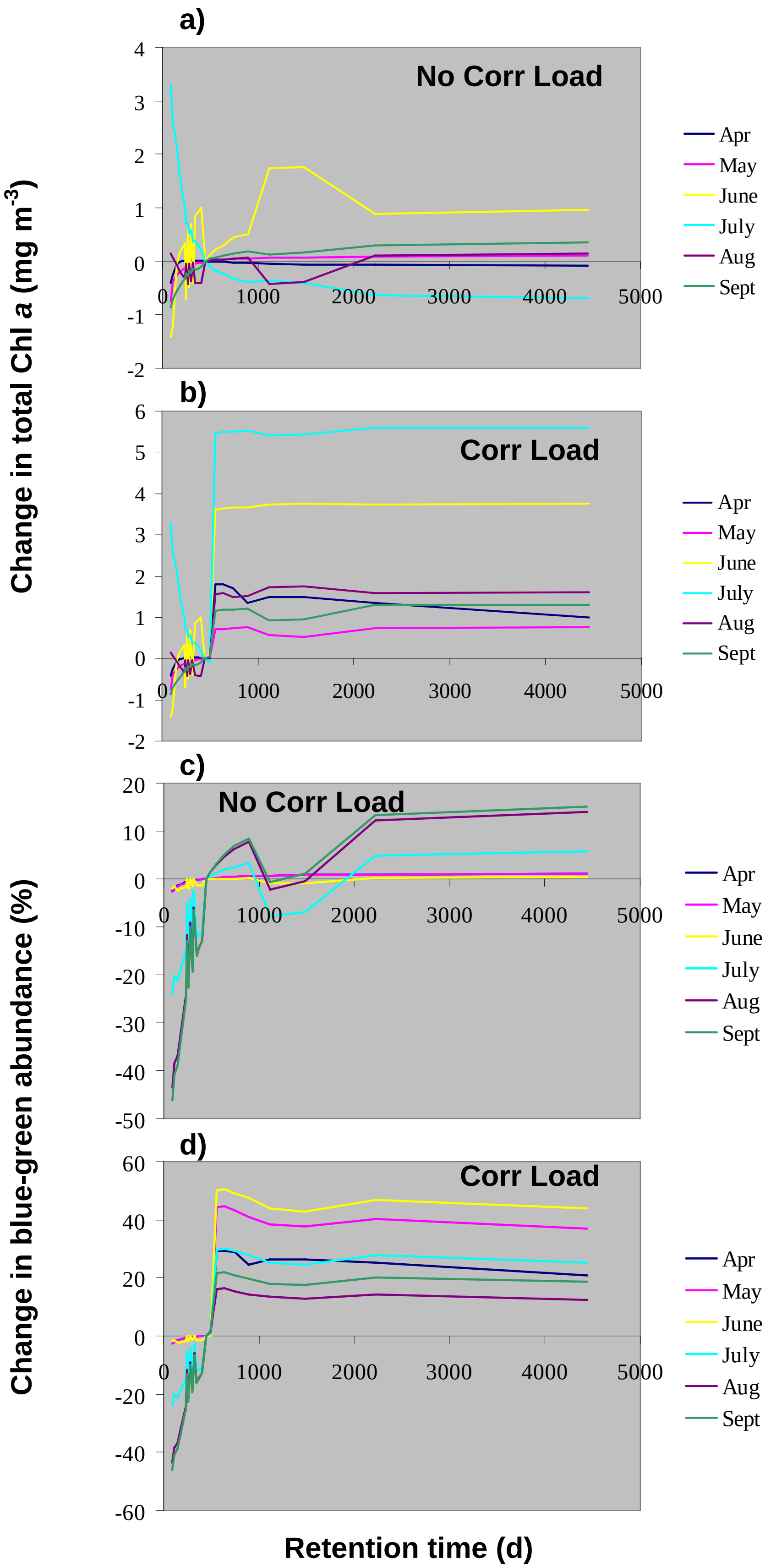


Fig. 21 Comparisons for SE Reservoir with artificial mixing. a & b) Absolute change in monthly mean chlorophyll *a* biomass (mg m^{-3}) and; c & d) monthly cyanobacteria percentage abundance for April-September for both flow methods.

4. DISCUSSION

4.1 Changes in annual total chlorophyll means and cyanobacteria abundance

There was a large variation in the results across the different water bodies simulated. Changes were apparent in chlorophyll concentrations across the wide range of retention times used, even for systems with very long retention times under "normal" conditions e.g. Rutland Water (Fig. 11). The patterns of variation are further complicated depending on whether nutrient load remains constant (Corr Load/Point Source) or whether nutrient concentration remains constant so that load is related to the changes in flow (No Corr Load/Diffuse Source).

This distinction makes a very large difference to some of the water bodies tested (e.g. Bassenthwaite Lake and SE Reservoir) and very little to others. This difference appears to be independent of both trophic status and water body type. Also, many of the water bodies demonstrate sudden switches in biomass and cyanobacteria dominance, possibly indicting a switch from being limited by nutrients to limitation by flushing loss or by changes in the mixed depth. This implies that changes in retention times are capable of causing discrete, discontinuous response behaviours as well as continuous changes.

Finally, it is interesting to consider, that most systems simulated are relatively insensitive to a 20% reduction in inflow, thought sufficient to support good ecological status in rivers (WFD48 report, 2006). These large, long retention time systems appear to be very robust to change within this range of inflow reduction, which will probably fall within the inter-annual variation experienced by most water bodies. Thus, their phytoplankton communities are only likely only to be sensitive to "extreme flow events" i.e. flow regimes that fall outside the typical annual pattern.

Due to a lack of driving data in this study, there was not much scope for modelling different years of the water bodies presented, however, Bassenthwaite Lake did provide some opportunity. Interestingly, there was not much difference between the two years modelled and the continuous 20 year run, suggesting the general pattern of response in the total chlorophyll and cyanobacteria abundance may be robust even in different years, although extrapolations from these data for one lake must be treated with caution.

The short retention time lakes (Blelham Tarn and Bassenthwaite Lake) do have some general responses in common; they both show asymptotic patterns with longer retention times for total chlorophyll and cyanobacteria abundance and the response is relatively similar in pattern under both loading scenarios. Furthermore, at retention times greater than 100 days, these systems exhibited very little reaction in their communities to longer retention times. This suggests that, below the 100 days threshold, these systems are perhaps more sensitive to the changes in flushing than nutrient load. This result also has implications for lakes experiencing drought conditions. Under such conditions, flow would be greatly limited and hence the retention time of the lake would be extended. The simulations suggest that, in terms of retention times alone, the lake will be relative insensitive to such a change beyond the retention time threshold suggested, although an increased dominance by cyanobacteria species would be more likely. However, to achieve annual retention times of over 100 days in these lakes would require a decrease of between 60-70% in the flow which would only occur in severe drought conditions, although similar effects could be achieved seasonally (e.g. a severe summer drought). Such seasonally restricted changes in retention time are beyond the scope of this study.

In the reservoirs with their characteristically longer retention times, the systems were much less responsive except when artificial mixing was absent and retention times low. In fact, the effect of artificial mixing was so dominant that changes in retention times had little effect; however, it should be noted that the mixed scenario for the SE Reservoir induced a sudden step change in both biomass and cyanobacteria abundance with a slight lengthening of retention time and seemed to reflect a change in dynamic between the two dominant species in the turbulent system (*Melosira* and *Planktothrix*).

In terms of impacts upon lake status assessment using phytoplankton abundance (chlorophyll), all the systems simulated currently have annual means of chlorophyll *a* of between 12 and 16 mg m^{-3}, which would classify them as Good or Moderate status (Table 2).

Table 2. Draft GB chlorophyll boundaries for the Water Framework Directive. Ref = reference level, HG = High/Good, GM = Good/Moderate, MP = Moderate/Poor, PB = Poor/Bad.

	Ref	**HG**	**GM**	**MP**	**PB**
	Chlorophyll *a* mg m^{-3}				
Mod Alk Shallow	3.5	7	11	25	47
High Alk Shallow	4	7.5	15	34	74

Using this table as a guide, the simulations suggest that for the lakes and reservoirs tested, their status would be unchanged or slightly deteriorate in low flow conditions. Under high flow conditions, reservoirs had a lower status, according to their chlorophyll concentrations. In the lakes tested, the result suggest that shorter retention times might lead to a slight improvement in their status indicated by the lower simulated chlorophyll concentrations. Using such flow changes as a management tool, however, is an unlikely proposition.

4.2 Changes in monthly total chlorophyll means and cyanobacteria abundance

The effect of retention times on the monthly results varied considerably between the different water bodies. The No Corr Loads scenarios generally produced little differences between the months and the few changes were small. In contrast, the Corr Loads scenarios showed that May, June, July, August and September all proved to be sensitive in at least one scenario but June was often the most sensitive month in terms of cyanobacteria dominance. Thus, in June, the general pattern was for the cyanobacteria dominance to increase with longer retention times, signifying that this month may be particularly important in most water bodies. In the UK, June is often the transition month for phytoplankton succession as the community recovers from the collapse of the spring bloom and the summer bloom begins to develop. This also means that this month is particularly sensitive to drought conditions. Therefore, while all the months mentioned above are sensitive, it may be that June is the most important month in terms of phytoplankton and abstraction of water; further verification of this hypothesis is most definitely merited.

4.3 Recommendations for the setting on MEP and GEP for phytoplankton in Heavily Modified and Artificial Waterbodies

The results of the PROTECH simulations have indicated that at least in the lakes studied, changes in retention time, even large increases, have a relatively small impact on chlorophyll and cyanobacterial abundance. They have also shown that where artificial mixing is used, its impacts on phytoplankton mask any effects caused by changes in retention time.

Under WFD, setting of Maximum ecological potential (MEP) and Good ecological potential (GEP) for biological quality elements should reflect "*as far as possible, those associated with the closest comparable surface water body type, given the physical conditions which result from the artificial or heavily modified characteristics of the waterbody*". Within still water Heavily Modified or Artificial water bodies the modifications are often related to the presence of an impoundment associated with an abstraction, either directly from the water body or from the upstream catchment. Additionally there may be artificial destratification measures in place within reservoirs. Abstractions from within a lake may have the effect of reducing the retention time of the water body. Abstractions from upstream of the lake will have the effect of increasing the retention time of the downstream lake.

The results of this study suggest that decreasing retention times (i.e. faster throughput of water) can decrease the chlorophyll and cyanobacteria concentrations which, when used as a measure of ecological status, will result in a shift towards higher status classes. Increased retention times, even when large, appear to cause a small increase in chlorophyll and cyanobacteria concentrations which result in a tendency towards lower status classes although the changes may be insufficient to change status where the classes are wide. Artificial destratification effects appear to be so prominent as to mask any effects on phytoplankton caused by changes in retention time. The implication therefore is that other factors e.g. water depth, alkalinity and nutrient status will be exerting a greater control on the phytoplankton in reservoirs than the impacts from abstraction.
These conclusions lead to the suggestion that for water bodies where an impoundment and associated abstraction are the modifications giving it a designation of HMWB or AWB, the factors which control phytoplankton in natural lakes will also control phytoplankton in these water bodies and therefore MEP and GEP should be broadly similar to HES and GES in natural lakes of similar type.

5. CONCLUSIONS

If the results of these simulations are considered together and some allowance is made for the two different nutrient loading methods, it would appear that the following broad conclusions can be made:

All the water bodies tested in this study were sensitive to retention times that were shorter than their original time. The general response in lakes was a decrease in both biomass and cyanobacteria abundance because flushing loss processes began to prevail. The response in the reservoirs was more varied and less consistent.

At longer retention times, responses in both total chlorophyll and cyanobacteria abundance were smaller and particularly so beyond a retention time of 100 days for the lakes in this study. This suggests that drought induced effects upon the phytoplankton annual mean biomass are minimal.

Annual chlorophyll a, one measure of ecological status in lakes did not alter significantly as a result of any of the changes in retention time except for the extremely short retention times in the lake simulations.

Mixed reservoirs were relatively insensitive to changes in retention time because the limiting factor for phytoplankton growth under these conditions was light availability, and flushing loss and nutrient supply were less important. Although many months were sensitive to changes in retention times, it appears that June could possibly be the most sensitive month, particularly at longer retention times. Cyanobacteria dominance was most responsive at this time and under those conditions.

When setting MEP and GEP for the phytoplankton biological quality element in reservoirs, the metrics used will be broadly similar to those required to set HES and GES in natural lakes.

6. REFERENCES

Elliott, J.A., Escot, C., Basanta-Alves, A. & Cruz-Pizarro, L. (2005). Simulations of phytoplankton dynamics in El Gergal reservoir, southern Spain (PROTECH). In: *New Tools for the Monitoring, Modelling & Management of Fresh Waters* (eds K. Rouen & G. George). *Freshwater Forum*, **23**, 78-92.

Elliott, J.A. & Thackeray, S.J. (2004). The simulation of phytoplankton in shallow and deep lakes using PROTECH. *Ecological Modelling*, **178**, 357-369.

Elliott, J.A., Thackeray, S.J., Huntingford, C. & Jones, R.G. (2005). Combining a Regional Climate Model with a phytoplankton community model to predict future changes in phytoplankton in lakes. *Freshwater Biology*, **50**, 1404-1411.

Lewis, D.M., Elliott, J.A., Lambert, M.F. & Reynolds, C.S. (2002). The simulation of an Australian reservoir using a phytoplankton community model (PROTECH). *Ecological Modelling*, **150**, 107-116.

Ramsbottom, A.E. (1976). Depth charts of the Cumbrian lakes. *Freshwater Biological Association*.

Reynolds C.S. (1984). *The ecology of freshwater phytoplankton*. Cambridge, Cambridge University Press.

Reynolds C.S., Irish, A.E. & Elliott, J.A. (2001). The ecological basis for simulating phytoplankton responses to environmental change (PROTECH). *Ecological Modelling*, **140**, 271-291.

WFD48 Project: Development Of Environmental Standards (Water Resources) Stage 3: Environmental Standards (2006). SNIFFER Report.