CHAPTER 6
DISCUSSION OF RESULTS

6.1 Influent composition

As stated in Section 4.1, the wastewater at Esholt is around 50% trade waste in origin. This was not ideal for the experimental work as the most likely application of waste stabilisation ponds in the UK will be for rural communities, mainly treating domestic wastewater. The choice of the Esholt site was based on: proximity to Leeds; the availability of sufficient land near the screened sewage line and a manned site (allowing unrestricted access during working hours). These criteria could not be met at the smaller works in the area treating mainly domestic wastewater.

The influent to the ponds was augmented by waste sludge from the works from time to time. The usual practice for the Esholt works was to waste the sludge in the screened sewage line after the take-off point. However, after the construction of the ponds, this practice changed and some sludge was wasted before the take-off point to allow better mixing in the screened sewage line before reaching the primary tanks. Consequently, the influent to the pilot ponds differed from a typical wastewater as shown in Table 6.1

The most obvious difference was the very much higher SS concentration in the influent to the pilot-ponds: over 3 times the concentration of a strong wastewater. Over 80% of these solids were settleable, but the non-settleable fraction (178 mg/l) was well within the range for a domestic sewage. The settleable solids content was 50% higher than a strong wastewater. This augmentation of solids is attributed to the sludge wastage. This is very significant to the interpretation of the results as it improves the measured efficiency of SS removal in the pilot-ponds, and very much increases the solid load to the sludge layer compared to a works without this augmentation. The organic fraction of the settleable solids was slightly lower than the average wastewater, therefore sludge digestion should be at the same rate or less.

Table 6.1 Comparison of typical wastewater parameters with the influent to the pilot-scale ponds

	Typical wastewater range (Metcalf & Eddy, 1991)	Mean value (or range used) for the influent to pilot ponds
SS (mg/l)	100-350	1078
StS (ml/l)	5-20	32.7
BOD (mg O_2 /l)	110-400	485
Settleable BOD (%)	25-40	64
Settleable SS (%)	50-70	83
Organic fraction of StS (%) Ratio of BOD to COD	75 0.4-0.8	57-63 0.33
Ammonia (mg N/l)	12-50	18-40
Nitrate (mg N/l)	0	0.19
Nitrite (mg N/l)	0	0.04
Phosphate (mg P/l)	3-10	3.2
pH		7.2

The average BOD to the pilot-ponds was 85 mg/l above the range given by Metcalf & Eddy (1991) potentially classified as a very strong sewage. Around two thirds of the BOD was settleable, higher than the usual 40 %. This is very likely to improve the BOD removal figures for the ponds, but the extent is uncertain, especially as there was a great potential for the settled BOD to feedback to the pond water during sludge digestion. Comparison of COD to BOD concentrations suggested that the influent was slightly less biodegradable than a typical sewage.

The concentration of nutrients appears to be within the range of a typical domestic wastewater, especially for the inorganic nitrogen species. The concentration of phosphate was at the lower end; this is the nature of the wastewater entering the Esholt works which is naturally very low in phosphorus. The possible consequence of this is phosphorus limitation for algal growth, but as the concentration of phosphate leaving the pilot-ponds was rarely zero, it is considered unlikely.

6.2 The uncertainty of the surface loading

As explained in Appendix A, the BOD concentration in the influent varied greatly and a log transformation was required before estimation of the mean (485 mg O_2 /l). This estimated mean has a level of uncertainty and the 95% confidence interval was 430 – 547 mg O_2 /l. The mean value was used to calculate the surface loading applied to each pond, therefore these values also have a level of uncertainty as shown in Table 6.2

Table 6.2 The surface loading range as calculated from the 95% confidence interval for the mean BOD

Phase	95% loading range for Red pond (kg/ha.d)	95% loading range for Green pond (kg/ha.d)	95% loading range for Blue pond (kg/ha.d)
1	(63) 54-74	(62) 52-74	(51) 43-61
2		(116) 96-139	(169) 100-210
3			(117) 100-136
4		(82) 60-108	(107) 89-128

In Phase 1 all the ponds were set to the same loading so there was complete overlap of the loading range. In Phase 2, there was some overlap between the Green and Blue pond loadings. In Phase 4 there was significant overlap of the loadings to the Green pond and the other two. Table 6.2 illustrates the difficulty of achieving a fixed loading at pilot-scale with the great variation in the influent BOD, and also that it may become a problem to compare performance when the loading range is narrowed as in Phase 4. This uncertainty is far less however, than on a full-scale system where the influent flow also varies and the level of uncertainty is then multiplied by this factor.

6.3 BOD removal

If BOD removal is the primary factor by which facultative ponds are assessed, then the performance of the pilot-scale ponds must be described as excellent. In Section 2.4.7 the BOD removal range quoted by other authors was 50-90 %; the pilot-scale ponds average removal was 91.0 % with a range of 67.5-98.6 %. These values include the contribution of algae in the effluent. With the algal (and other) solids removed from the effluent, the average removal was 97.2 % (with a range of 89.7-99.7 %). These excellent removal efficiencies are no doubt assisted by the high inlet BOD concentration and the very long hydraulic retention time in the ponds. The contribution of chlorophyll-a to the reduction of overall removal efficiency, as given in Section 5.2.2, suggests that combined with algal removal facilities, the primary facultative ponds might be suitable to achieve discharge consent levels of BOD = 20-30 mg O_2 /l without further treatment. The EC UWWTD standard for pond effluents is less than 25 mg filtered BOD /l. If this consent were applied, then data from the pilot-ponds suggests that primary facultative ponds should achieve this without the need for algal removal facilities.

Mara and Pearson (1998) stated that 70-90% of the effluent BOD may be due to algal solids, based on filtered BOD samples, the pilot-pond results suggest that up to 94 % of the outlet BOD was due to algal solids in summer and down to 0 % in winter. Bucksteeg (1987) stated that 100 μg chlorophyll-a ≡ 3 mg BOD. The findings from the pilot-ponds suggest that (from equation 5.1 in Section 5.2.2) 100 μg chlorophyll-a ≡ 5.8 mg BOD. Though it is unlikely that these relationships are linear for all values of chlorophyll-a (which would, according to the pilot-pond data, sometimes contribute more than 100% to the effluent BOD). Bucksteeg's relationship was based on the difference between filtered and unfiltered samples. From the pilot-pond data, no clear relationship could be found between filtered BOD and chlorophyll-a, thus the multiple regression analysis with loading was used to make this estimate.

The pilot-pond data agree with the findings of other authors that BOD removal is independent of season. No correlation could be found with temperature and this is mostly likely as a result of the sludge feedback processes during summer and algal solids in the effluent balancing the higher degradation rates. Although sludge feedback was not measured, sludge mats were observed on the surface of all the ponds during the summer months. These mats usually broke up in the wind and did not usually enter the effluent.

BOD removal was shown to correlate principally with BOD loading more than any other parameter; however, the BOD loading and hydraulic retention time could not be partitioned in the pilot-pond experiment. This is the biggest weakness of the testing regime. Ideally the ponds should have been subject to the same hydraulic retention time and of shorter duration, than the experimental conditions. The extremely long retention times were as a result of the higher-than-expected influent BOD concentration. A Phase 5 was planned for the pilot-ponds, to run between March and August 2002, when the retention time would be reduced to 40 days by the introduction of some tap water to the influent. Although the testing schedule and all the equipment was ready to start this phase, the supply of tap water was interrupted due to a leakage. Yorkshire Water were unable to rectify this problem in time for Phase 5 to be carried out. With the data available therefore, it is difficult to tell whether BOD loading, rather than hydraulic retention time, was in fact the principal parameter for BOD removal.

The data showed excellent agreement with the model of McGarry and Pescod (1970) (Section 5.2.6) which is strong evidence that over the test range (63-169 kg/ha.d) there was no significant loss of performance. Therefore to optimise BOD removal, the ponds should be loaded as highly as possible within this range to achieve a specified effluent quality. However, as explained in Section 5.2.6, the use of this model means the very high influent BOD concentration engulfs the relatively small changes in effluent concentration. The other issue here is that the highest loading of 169 kg/ha.d was confined to one pond over 5 months' duration. During this interval, the effluent concentration steadily worsened and the calculated average for this interval does not take into account this trend. The loading was reduced in March 2001, possibly before the maximum deterioration in performance had occurred.

The agreement found with the complete mix model of Marais and Shaw (1961) which asserts temperature and hydraulic retention time to be the key variables for BOD removal, is complicated by the lack of partitioning between BOD loading and hydraulic retention time. In comparison to the model of McGarry and Pescod, this relationship is much weaker. As the data stand, therefore, BOD loading is the most likely key parameter for BOD removal in the pilot-scale ponds.

The pilot-ponds did not remain facultative during the winter months, yet this had no effect on BOD removal. This supports the findings of Almasi and Pescod (1996) that even in anoxic and anaerobic conditions, pond bacteria can consume large quantities of BOD. Hence, facultative conditions are not required for BOD removal.

6.4 Suspended solids removal

The average effluent suspended solids concentrations were 50-70 mg/l during summer periods and 25-40 mg/l during winter periods. As the influent concentration of suspended solids was extremely high, the removal efficiencies (87.9-99.2%) were also high and the subtle differences in effluent concentration lost. Schneiter *et al.* (1983) had stated that primary ponds in temperate areas may remove 60-70% of influent suspended solids by sedimentation processes; the estimate for the pilot-ponds was 83%. It would be irresponsible to suggest that the suspended solids removal from the pilot-ponds should be expected from other facultative ponds in the UK with typical SS influent concentrations of the range given in Table 6.1. Calculation of the removal from the pilot-ponds based on typical influent concentrations would not be useful either, as this would ignore the contribution the extra solids made to the sludge.

Very occasionally, high concentrations of suspended solids were observed in the effluent samples due to resuspension of sludge. These observations (on about 2-3 occasions) were made principally in the Blue pond effluent, though it was also observed once in the Red pond effluent. These were not necessarily associated with spring over-turn conditions as suggested by US EPA (1983), as they could also occur in mid summer. There is therefore evidence that the sludge plays a very active role in the performance of the ponds. Due to this, it has not been possible to derive models from the data, nor to make any sensible predictions, for SS removal. The most useful data are the suspended solids effluent concentration and the contribution of chlorophyll-a to this. The model given in Section 5.3 suggests that effluent suspended solids concentrations of more than 33 mg/l are most likely due to algae, and that on average 100 μg chlorophyll-a ≡ 6.8 mg of suspended solids.

Mara *et al.* (1992) reported that the algal concentration in facultative pond effluents may reach 1000-1500 μg/l in temperate climates during the summer. The effluent chlorophyll-a from the pilot-ponds reached over 1000 μg / l in around 9 % of samples, whilst around 40 % of samples contained less than 100 μg / l. At the loading range applied to the pilot-ponds therefore, the issue of algal-laden effluents may not be a serious issue for much of the year. The main reasons for this, as given in Section 5.5.4.1, were: algal predation in summer and low algal populations in winter due to light or temperature limitation. The outlet was designed to abstract water from 10 cm below the surface, rather than the 40 cm recommended by Pearson (1990), thus these samples may be considered almost surface samples. While abstracting from a deeper level in the pond may reduce the algal load further, it may also increase the risk of sludge-feedback solids entering the effluent.

Like the BOD, SS removal does not appear to rely on facultative conditions: the ponds were able to produce effluent concentrations of less than 40 mg/l during the winter in anoxic conditions. However, the SS effluent concentrations achieved in the pilot-ponds suggest that some additional form of treatment, such as those given in Section 2.4.8.2, will be necessary to achieve consents of 40 mg/l or less all year round. The EC UWWTD requirement for waste stabilisation pond effluents is < 150 mg SS/l.

6.5 Ammonia removal

The ammonia concentration in the influent to the pilot-ponds was well within the range of a normal wastewater, therefore unlike the BOD and SS, the application of the ammonia removal data is more straightforward. The influent ammonia concentration fluctuated seasonally (higher in summer than winter). These fluctuations were similar to the variations in the crude sewage to the Esholt works as measured by Yorkshire Water. A possible explanation for these fluctuations is that the rate of mineralisation of organic nitrogen is likely to be higher in summer due to the higher temperatures in the sewage line and the inlet tubing. Also, the sewage was likely to be more concentrated in summer due to lower rainfall. The effect was not as marked during the second year, this may have been because, unlike during the first year, the second winter was not significantly wetter than the following summer. Neither the BOD nor SS concentrations fluctuated in this way; perhaps these values were evened out by the sludge wastage. The seasonal fluctuations in influent ammonia concentration increased the calculated removal efficiency in summer compared to winter. However, they did not account for most of the seasonal difference, as the effluent concentration data showed a similar seasonal fluctuation.

The effect of surface BOD loading on ammonia removal efficiency was only noticeable when the loading range was 63-169 kg/ha. d. After the range was reduced in Phase 4 (63-107 kg/ha.d) there appeared to be no difference in performance and the effect of season became more prominent. At loadings of 107 kg BOD/ha.d or less, the ammonia removal exceeded 70% by early summer, but dropped to around zero in mid-winter (January and February 2002). The data support the evidence from other ponds in temperate climates that ammonia removal efficiency in summer is much better than in winter. The seasonal difference was not as noticeable in the first year as the second, the most likely explanation is the much wider range of surface ammonia load applied ($g/m^2.d$) in the first year. Also, during the first year, the applied loading was not constant throughout the year, in fact the lowest loadings were associated with the start-up (Phase 1) during summer and the highest loadings (Phase 2) were during winter.

The overall performance data differed from Year 1 to Year 2 (worse during the second year). It was hypothesed in Section 5.4.2 that the second year's data might be more reliable due to the presence of an established sludge layer and the absence of duckweed. This is a reasonable assumption, especially as the applied loadings were also unrelated to season during the second year. With the data available, however, it is uncertain if even the second years' performance data was the typical case, or just part of the pond "evolution" process. Ammonia data from subsequent years would provide a better idea.

There is the possibility that the sludge layer, augmented as it was by waste sludge solids, fed back much higher quantities of ammonia to the pond water than would happen in an average facultative pond without this problem. However, as no measurements were made on the organic nitrogen content of the sludge, this cannot be established.

The ammonia removal efficiency in the pilot-ponds varied widely (0-88 %), this variability was despite the very long retention times. Ammonia removal in the facultative ponds was shown to have relationships with: temperature, pH and chlorophyll-a; all these values were higher in the summer than winter and correlated with each other. The pH of the influent was usually around 7.2, so only the effect of algal photosynthesis could drive the pH higher than this. Due to the relationship with chlorophyll-a and pH, the data support the hypothesis that algal uptake or volatilisation were the principal pathways for the removal of ammonia. The data from the second year showed a good approximation to the model of Pano and Middlebrooks (1982). The pilot-pond data suggest that under UK climatic conditions, hydraulic retention time is not as important as season, and even very long retention times did not lead to better performance in winter conditions. There was evidence of an effect of hydraulic retention time on ammonia removal during the summer even at very long retention times of more than 60 d. Also, as the hydraulic retention time and loading were not partitioned, it is possible that shorter retention times during the first year were responsible for the variation in performance rather than ammonia loading.

The data suggest that, unlike BOD and SS, ammonia removal does require facultative conditions to take place. The ammonia removal achievable in the pilot-ponds during the winter if facultative conditions could be maintained is not known, but the data from the Eudora ponds in Kansas as reported by Pano and Middlebrooks (1982) suggest ammonia removal could take place, though the efficiency would be lower than during summer.

The pilot pond data suggest that, at the surface loadings applied, very low concentrations of effluent ammonia (eg. < 5 mg N/l) cannot be relied on at any time of the year. If the pond becomes anoxic or devoid or algae during the winter, the ammonia removal is likely to drop to zero. Thus, where ammonia consents are applied, further treatment such as maturation ponds, reed beds or pond upgrading as described in Section 2.4.9.4 will be required.

6.6 Facultative conditions

It was the experience of the pilot-pond experiments that the establishment of a threshold for facultative conditions based on either DO, redox or chlorophyll-a measurements was problematic, and site observations gave an as good (if not better) idea of what was going on. The use of the chlorophyll-a criterion of Pearson (1996) seems sensible as it concurred with the site observations. The correct loading for the maintenance of facultative conditions was not established as none of the ponds maintained a sufficient algal population during the winter. However, reducing the loading further may not be the answer because the Red pond, which was set at the lowest loading of 63 kg/ha.d, experienced widespread predation effects leading to unstable algal communities during the summer. It is not known if these predation effects would still occur at even lower loadings. It is also not known if lower loadings would be able to sustain an algal population during the winter.

The higher loadings led to more stable chlorophyll-a concentrations during the summer, but these were not necessarily associated high DO concentrations. The reason for this is that the chlorophyll-a concentration did not reveal stratification effects: under these conditions, the algae trapped on the surface could contribute >500 µg /l to the chlorophyll-a, but hardly anything to the DO concentration. An example of this is shown in Figure 5.40 in the Blue pond (loaded at 107 kg/ha.d) during June 2001. In the Red pond, at a loading of 63 kg/ha.d, the correlation between DO and chlorophyll-a was closer, suggesting that at this lower loading, the oxygen supply and demand of the system was more equally balanced. The pilot-ponds had lots of excess sludge which increased the solids load to the pond liquid, increasing the pressure on the algae. Another very important factor, apart from loading, was the relative position of the ponds (see Section 6.8).

The unpleasant outcome of winter failure was odour, which was observed at loadings of 107 kg/ha.d and above: the change in pond colour was itself was not really an issue. The extent of the odour problem was not assessed: direct measurements were not taken and the background odour from the Esholt works usually masked anything coming from the relatively small ponds. Night-time odour observations were not made due to health and safety considerations. Interestingly though, the observations of odour were made in December and January: a time of year when the nuisance to nearby populations is likely to be lower than during summer (which is a more typical time for sewage-related odours).

The failure “points” as used in Sections 5.5.6 and 5.5.7 are extremely tentative because they assume that “failure” and recovery were the same phenomena; Section 5.5.8 suggested that was not the case. The recovery data had to be included due to the lack of failure points. The limitation of two-years experimental work on three ponds is that the maximum number of winter failure points was six; three of which were lost due to the duckweed infestation. Thus, the comparisons with McGarry and Pescod’s Envelope of Failure (1970) and Mara’s global design equation (1987) are interesting, but probably not most useful without further data points. If the winter failure threshold for solar intensity (shown in Figure 5.44) is correct, then there is no benefit to loading a facultative pond at less than 80 kg/ha.d to avoid winter failure. This supports the findings of Cauchie *et al.* (2000) that in temperate climates low temperature and short day-length in winter may be insufficient to support an algal population / maintain facultative conditions at all.

The Esholt works is quite exposed and frequently windy, but the pilot-pond area was relatively sheltered by an adjoining row of trees. Consequently, the amount of mixing in the ponds was disappointing, and this was made worse by their physical design which allowed far too much freeboard. From fear of overflowing, the ponds each had a 40 cm freeboard, much more than was in fact required. The excess freeboard gave the water extra shelter from the wind. These observations support the assertion that physical design and location of ponds are very important.

It is clear from the pilot-pond experiments that the surface BOD loading required to maintain facultative conditions was much lower than that to optimise BOD and SS removal. The data suggest that no significant loss of performance for BOD and SS was experienced up to loadings of 116 kg/ha.d, whilst facultative conditions were not maintained during mid-winter even at 63 kg/ha.d. These observations may go some way to explain the discrepancies between the recommendations from Germany and the US. Perhaps in Germany the main consideration is performance, whereas in the US the maintenance of facultative conditions is the deciding factor.

6.7 Sludge accumulation

The sludge height was measured by the white-towel test which was appropriate given the very small size and relatively shallow depth of the ponds. The test was economical, reliable, quite sensitive to the small heights of sludge frequently encountered (around 1-2 cm), and the results were quick and easy to interpret. The major weakness of the test was that it appeared to be not as sensitive to the loose sludges, like those found around the outlets, as it was to the thicker, more sticky ones found around the inlets.

The amazing capacity of the ponds to reduce the volume of sludge is by far their biggest strength. Only 10-20 % of the incoming solids remained after 20 months operation. The total volume of sludge accumulated over 20 months was more or less the same in all the ponds, and apparently independent of the total volume of incoming settleable solids. The mean rate of accumulation was also the same for all ponds at 0.09-0.095 m^3/month. The per capita accumulation rates were within the ranges quoted for French ponds by Carré *et al.* (1990) and Canadian ponds by Schneiter *et al.* (1983).

Although the ponds were subject to much higher concentrations of settleable solids than typical systems, they coped very well indeed. The incoming settleable solids were not higher in % VS than a normal sewage, so the degradation rate should not have necessarily been higher either. The incoming sewage was only one source of solids: the internal production of algal and other cells and the contribution of leaves blown into the ponds also made a contribution. The effect of the latter was most notable in the Red pond.

Most sludge accumulated around the inlets as expected, but also tended to accumulate at the sides and outlet as found by other authors, despite the steep, even sides and flat base. There was also evidence of sludge slides in the Blue pond (as noted by Nelson and Jimenez (2000)).

It would be expected that the sludge volume would reduce over the summer periods and accumulate in winter as predicted by Parker and Skerry (1968) and Scheiter *et al.* (1983). This did not appear to be the case for the Blue and Green ponds where the accumulation in summer was more or less the same as in winter. This may have been due to the sedimentation of algal cells balancing with the increased rate of anaerobic degradation. The ratio of VS to TS increased up to 9 months' operation in these two ponds, then decreased steadily, independent of season. This suggests that the sludge accumulation rate was influenced by the early stages of pond evolution where the sludge stabilises quickly over time and the accumulation rate decreases rapidly as observed by Nelson (2002). This is because the proportions of digesting sludge to digested sludge change over time: the digesting sludge becoming a lower proportion of the total sludge accumulation. These early stabilisation effects possibly out-weighed the seasonal effects. In the Red pond, however a different picture emerged: the sludge tended to accumulate in winter and decrease in summer. It is hypothesed that the contribution of leaves falling in autumn led to the observation of winter accumulation. Strangely, the ratio of VS to TS at start-up was different for the Red pond than the other two, but after 9 months' operation, the ratio was about the same for all ponds.

The estimates for the desludging interval are the most precarious of the interpretation of the sludge data, yet are within the range quoted for French ponds by Carré *et al.* (1990). The predictions based on the accumulations in the Blue and Green ponds are the most appropriate as they do not have excess augmentation from fallen leaves. These predictions are quite stringent as they are based on steady accumulation around the inlet and thus do not take into account sludge-slides from the inlet area to the rest of the pond. The turning points in the graphs occurred within the last interval (15-20 months) and as there are no further points, it is not certain if these turning points were a useful measure of the background trend, or fluctuations. However, it is likely that they do reflect the background trend, given the experience of others that the sludge accumulation rate takes over 2 years to stabilise (Marais, 1970). Data from progressive years are needed, ideally up to the point of desludging. When compared with the data from French ponds, the prediction of more than 7 years to desludging does seem reasonable. If some extra depth were permitted around the inlet, the interval should be even longer.

6.8 The relative position of the ponds

The positioning of the pilot-ponds may well have affected their performance. Although they were very close together, there was still evidence that this had an effect. In an ideal situation, the loadings to the ponds should have been rotated to eliminate the effect of location. This was not done in the interests of continuity: to observe the changes in the pond evolution over time.

The Red pond was nearest the row of trees, and although the trees did not overhang, they still provided an element of shelter and many leaves fell into the Red pond. The Blue pond was furthest away from the trees, thus the most exposed of the three. From the DO profiling data, it appeared was that the Red and Green ponds were not as well mixed as the Blue: the latter experienced deeper penetration of dissolved oxygen than the other two at all times of the year. Although the Blue pond had a higher inflow rate than the other two which may partly explain this, waves were also observed far more frequently on the Blue pond than the other two for the same wind conditions. The wind effects appeared to have a strong impact on sludge feedback: in heavy winds, gas and or sludge solids could be observed quickly rising to the surface in the lee of gusts of wind. The effect of the steep sides most likely exaggerated this effect with convection effects upsetting the sludge. This may be an explanation for the sludge being more evenly distributed across the bottom of the Blue pond. However, the flow rate into the Blue pond was always the greatest, so the water had more energy to spread the solids further.

This suggests that the location of the pond is very important, even by 10 metres: therefore a very exposed pond could work very differently from a sheltered one.

6.9 Application issues

The pilot-ponds were designed to treat population equivalents of 5-10 people. Many facultative ponds in the UK serve this population range, so in terms of size, the pilot-ponds may be considered full-scale. Of course there are much larger systems in the UK, as detailed in Chapter 3, treating populations in excess of 100 p.e.

The physical design of the pilot-ponds differed from a typical full-scale system: for example the sides were much steeper and had very flat bases than would normally be expected which undoubtedly impacted on the mixing. The pilot-ponds had a uniform depth which allowed the sludge to collect more evenly across the bottom, but the frequent observations of feedback suggest that there might have been benefits to keeping the sludge layer more isolated from the water. Hence, the suggestion of a deeper inlet area (perhaps 2 m) has some validity for UK conditions; though it is very unlikely that full-scale ponds would experience the same settleable solid load as the pilot-ponds and therefore would certainly not have the same volume of sludge to contend with.

The pilot-ponds operated on a continuous constant flow basis; this would not be the case for full-scale ponds where the inflow would vary diurnally and from day to day. With such very long hydraulic retention times, the difference is not expected to be significant in terms of performance, but may affect the sludge distribution and general mixing characteristics. If full-scale ponds were connected to a combined drainage system, then the very high variability in flow during high rainfall could theoretically have an impact on pond performance; the pilot-ponds were only subject to direct rainfall. Bucksteeg (1987) stated however, that German facultative ponds with 5 m^2/person could take up to 40 times DWF with no loss in performance.

The small surface area of the pilot-scale ponds meant that they could not take full advantage of the wind. A larger full-scale pond would have a larger fetch thus the energy of the wind could penetrate deeper. Hence a full-scale pond in an exposed location may not suffer stratification effects as frequently as the pilot-ponds and may remain better aerated in winter. As mentioned in Section 6.1, the pilot ponds were dealing with effluent which was 50% trade waste and less biodegradable than normal domestic sewage. It is not known how often industrial chemicals entered the ponds, but it is very likely to have happened during the 2 years as the Esholt works is subject to these inputs. The impact of these chemicals on the biology of the ponds is also not known, though it is anticipated that with the very long hydraulic retention time, the impact would have been minimal. The low concentration of phosphorus may also impacted on the pond performance compared to a system where this nutrient was abundant.

The pilot-ponds were run for only two years, this is a long time in experimental terms, but quite short for waste stabilisation ponds which may take more than 2 years to establish themselves. The performance of the ponds may drop off later in maturity as more sludge accumulates or they might settle down and become less prone to predation and duckweed. Because ponds evolve slowly, it was decided to keep the loadings as constant as possible to measure the evolutionary changes; this was at the expense of experimental randomness.

Although the climatic conditions at Esholt were almost typical for average UK conditions, the two years of operation might not have been typical for UK weather. For example, January 2001 was slightly colder than average and January 2002 was much warmer (Met-Office, 2002). These comparisons were made with data from 1961-1990 and do not take into account any trends in climate change. Anyway, two year's data is not that bad, as many studies on ponds have been conducted over a few months or only up to a year.

6.10 Mosquito breeding

Mosquito breeding was observed predominately in the Red pond; because the ponds were so close together the obvious explanation for this was the BOD loading. As the BOD loading to the Red pond was the lowest, the second obvious explanation was that it was due to the low loading. However, these assumptions may be completely unfounded. Stringham and Watson (2001) stated that the COD and TKN range which *Culex* mosquito larvae prefer is 50-500 mg N/l and 400-2000 mg COD/l. It is unlikely that the COD nor TKN in the Blue and Green ponds rose above these ranges, given the effluent BOD and ammonia concentrations. It is, therefore, unlikely that it was the nutrient level, and hence the loading, to the Red pond that was to blame.

The more likely explanation is the fallen leaves and general debris which entered the Red pond. Sludge feedback brought quantities of leaves back to the surface as shown in Section 5.6.2. These leaves provided the sheltered breeding sites the larvae required. Mosquito larvae also prefer still waters to moving waters: the quiescent waters of the sheltered Red pond probably provided a more comfortable habitat than the more exposed ponds.

Two maturation ponds were constructed on the site in 2001 next to the Blue pond as described by Johnson and Mara (2002). The first maturation pond was at a loading of 5 kg BOD /ha.d and was more exposed to the wind than the facultative ponds; it did not experience mosquito breeding at any time of the year.

To test this hypothesis, it would have been necessary to rotate the loading rates to all the facultative ponds. This was not done during the first two years to test if mosquito breeding was a phenomenon of only the first year of operation; this was not the case as shown in Figure 5.37, Section 5.5.4.

6.11 Duckweed

The duckweed infestation was unfortunate but not inevitable for waste stabilisation pond systems. The removal of the weed increases the maintenance requirements of the ponds, but careful choice of start-up water can reduce the probability of it taking hold. The weed does not like water movement, therefore a larger pond with plenty of wind-action would reduce its growth; also, as given in Appendix B, if the micro-algae are actively photosynthesising, the weed stops growing. This was the observation on the facultative and maturation ponds. If the pond has no isolated areas and an outlet that abstracts from the surface, the duckweed should flow out before it takes hold.

6.12 Optimising facultative ponds in the UK

Pond performance is only one of the issues for the correct sizing of facultative ponds. The main issue has to be the maintenance of facultative conditions if the pond is not to cause a nuisance. It is this factor which will determine the appropriate loading, as the value is much lower than that required to optimise BOD and SS removal. The pilot-pond data suggest that the maintenance of facultative conditions may not be possible during the UK winter at any loading according to the criterion of >300 µg chl-a /l, due to light-limitation. This assumption was based on the failure/solar intensity curves and really should now be tested at loadings below 63 kg/ha.d. If the assumption is correct, however, then there is no benefit to loading a facultative pond at less than 80 kg/ha.d. At this loading the winter failure interval was found to be very short (two weeks at < 1 mg O_2/l) and the pond was dominated by algae more than any other organism at all times of the year. A pond loaded at 80 kg/ha.d would be 3/4 the size of a pond loaded at 60 kg/ha.d and thus the land saving is significant.

It was the experience of the pilot-scale ponds that surface mixing was an issue in summer and winter. This was attributed to their small size and sheltered location. Due to this, some induced surface movement may be useful. This action would ensure that the algae did not become trapped on the surface during summer and would alleviate some of the loss of DO during the winter. More movement on the surface would also dissuade mosquito breeding. To optimise the treatment therefore, it would be sensible to have a some assistance, certainly for small ponds for less than 10 people. The typical UK facultative ponds use flowform cascades to provide this assistance: these look very nice, but might not be the most efficient option. It is not known, without further work, how much movement or surface aeration is needed.